Lower Duwamish Waterway Group

Port of Seattle / City of Seattle / King County / The Boeing Company

Lower Duwamish Waterway Remedial Investigation

REMEDIAL INVESTIGATION REPORT

DRAFT

For submittal to:

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Region 10
Seattle, WA

The Washington State Department of Ecology
Northwest Field Office
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Acronym	Definition
ac	acres
AET	apparent effects threshold
ANOVA	analysis of variance
AOC	Administrative Order on Consent
API	Asian and Pacific Islander
ASIL	acceptable source impact levels
ASTM	American Society for Testing and Materials
ATSDR	Agency for Toxic Substance and Disease Registry
AWQC	ambient water quality criteria
BACT	best available control technology
Battelle	Battelle Marine Research Laboratory
BBP	butyl benzyl phthalate
BEHP	bis(2-ethylhexyl) phthalate
bgs	below ground surface
BHC	benzene hexachloride
BMP	best management practice
BOD	biological oxygen demand
Boeing	The Boeing Company
BTEX	benzene, toluene, ethylbenzene, and xylene compounds
CAA	Clean Air Act
CERCLA	Comprehensive Environmental Response, Compensation, and Liability Act
CERCLIS	Comprehensive Environmental Response, Compensation, and Liability Information System
CFR	Code of Federal Regulations
cfs	cubic feet per second
City	City of Seattle
COC	chemical of concern
COPC	chemical of potential concern
CORRACTS	Corrective Action Site
County	King County
cPAH	carcinogenic PAH
CPM	coarse particulate matter
^{137}Cs	cesium-137
CSCSL	Confirmed and Suspected Contaminated Sites List
CSL	cleanup screening level
CSM	conceptual site model

Acronym	Definition
CSO	combined sewer overflow
CT	central tendency
CWA	Clean Water Act
cy	cubic yards
D_{50}	median particle diameter
DAIS	Dredged Analysis Information System
DARRP	Damage Assessment, Remediation, and Restoration Program
DMMP	Dredged Material Management Program
DNAPL	dense non-aqueous phase liquid
DO	dissolved oxygen
DPD	Department of Planning and Development
DQO	data quality objective
DSOA	Duwamish sediment other area
dw	dry weight
EAA	early action area
EB/DRRP	Elliott Bay/Duwamish River Restoration Program
EBI	Elliott Bay Interceptor
Ecology	Washington State Department of Ecology
EDR	Environmental Data Resource, Inc.
EE/CA	engineering evaluation/cost analysis
EFDC	Environmental Fluid Dynamics Code
EIM	Environmental Information Management
EOF	emergency overflow
EPA	US Environmental Protection Agency
EPC	exposure point concentration
ERA	ecological risk assessment
ERNS	Emergency Response Notification System
ERTS	Environmental Report Tracking System
ESA	Endangered Species Act
ESG	Environmental Solutions Group, Inc.
FIR	food ingestion rate
FPM	fine particulate matter
FR	Federal Register
FS	feasibility study
FWM	food web model
GC/ECD	gas chromatography with electron capture detection
GC/MS	gas chromatography/mass spectrometry
GIS	geographic information system

Acronym	Definition
GSA	General Services Administration
GWI	Great Western International
Herrera	Herrera Environmental Consultants
HHRA	human health risk assessment
HI	hazard index
HPAH	high-molecular-weight polycyclic aromatic hydrocarbon
HpCDD	heptachlorodibenzo-*p*-dioxin
HpCDF	heptachlorodibenzofuran
HPLC/PDA	high-performance liquid chromatography/photodiode array detection
HQ	hazard quotient
HSL	Hazardous Sites List
HWMP	Hazardous Waste Management Program
HxCDD	hexachlorodibenzo-*p*-dioxin
HxCDF	hexachlorodibenzofuran
I-5	Interstate 5
ICP-MS	inductively coupled plasma-mass spectrometry
ID	identification
IDW	inverse distance weighting
IP	intraperitoneal
IRA	independent remedial action
IWP	Industrial Waste Program
J-qualifier	estimated concentration
JN-qualified	tentative identification of estimated concentration
KCIA	King County International Airport
KCDNR	King County Department of Natural Resources
LAET	lowest apparent effects threshold
2LAET	second lowest apparent effects threshold
LDW	Lower Duwamish Waterway
LDWG	Lower Duwamish Waterway Group
LOAEL	lowest-observed-adverse-effect level
LOEC	lowest-observed-effect concentration
LPAH	low-molecular-weight polycyclic aromatic hydrocarbon
LQG	large-quantity generators
LUST	leaking underground storage tank
MDL	method detection limit
mgd	million gallons per day
mgy	million gallons per year

Acronym	Definition
ML	maximum level
MLLW	mean lower low water
MNR	monitored natural recovery
M-qualifier	calculated concentration
MTCA	Model Toxics Control Act
MT/yr	metric tons per year
NAPL	non-aqueous phase liquid
NBF	North Boeing Field
NFA	No Further Action
NMFS	National Marine Fisheries Service
NOAA	National Oceanic and Atmospheric Administration
NOAEL	no-observed-adverse-effect level
NOEC	no-observed-effect concentration
NPDES	National Pollutant Discharge Elimination System
NPL	National Priorities List
N-qualifier	tentative identification
NTCRA	non-time-critical removal action
NW	northwest
NWR	National Wildlife Refuge
O&M	operation and maintenance
OC	organic carbon
OCDD	octachlorodibenzo-*p*-dioxin
OCDF	octachlorodibenzofuran
OIG	Office of the Inspector General
OSWER	Office of Solid Waste and Emergency Response
PAH	polycyclic aromatic hydrocarbon
Pa	pascal
^{210}Pb	lead-210
PCB	polychlorinated biphenyl
PCE	tetrachloroethene
PCT	polychlorinated terphenyl
PeCDD	pentachlorodibenzo-*p*-dioxin
PeCDF	pentachlorodibenzofuran
PEF	potency equivalency factor
PIE	Pacific International Engineering, PLLC
PM	particulate matter
$PM_{2.5}$	particulate matter less than 2.5 micrometers in diameter
PM_{10}	particulate matter less than 10 micrometers in diameter

Acronym	Definition
Port	Port of Seattle
PPA	Potential Priority Area
ppb	parts per billion
ppt	parts per thousand
PRG	preliminary remediation goal
PSAMP	Puget Sound Ambient Monitoring Program
PSC	Phillip Services Corporation
PSCAA	Puget Sound Clean Air Agency
PSDDA	Puget Sound Dredged Disposal Analysis
PSEP	Puget Sound Estuary Program
PSMAF	Puget Sound Maritime Air Forum
PVC	polyvinyl chloride
QA/QC	quality assurance/quality control
QAPP	quality assurance project plan
QEA	Quantitative Environmental Analysis, LLC
RAO	remedial action objective
RBC	risk-based concentration
RBTC	risk-based threshold concentration
RCRA	Resource Conservation and Recovery Act
RCW	Revised Code of Washington
RETEC	RETEC Group, Inc.
RfD	reference dose
RFI	RCRA facility investigation
RI	remedial investigation
RL	reporting limit
RM	river mile
RME	reasonable maximum exposure
ROC	receptor of concern
ROD	Record of Decision
ROW	right-of-way
RPS	relative penis size
SAIC	Science Applications International Corporation
SARA	Superfund Amendments and Reauthorization Act
SCAP	source control action plan
SCWG	Source Control Work Group
SD	storm drain

Acronym	Definition
SEA	Striplin Environmental Associates, Inc.
SF	slope factor
SI	site inspection
SIR	sediment ingestion rate
SL	screening level
SMPD	semi-permeable membrane device
SMS	Washington State Sediment Management Standards
SPMD	semipermeable membrane device
SPU	Seattle Public Utilities
SPWG	Sediment Phthalates Work Group
SQG	small-quantity generator
SQS	sediment quality standard
STAR	sediment transport analysis report
STM	sediment transport model
SU	standard unit
SUF	site use factor
SVOC	semivolatile organic compound
SW	southwest
SWAC	spatially weighted average concentration
SWF/LF	solid waste facility/landfill
T-105	Terminal 105
T-106	Terminal 106
T-107	Terminal 107
T-108	Terminal 108
T-117	Terminal 117
TBT	tributyltin
TCDD	tetrachlorodibenzo-*p*-dioxin
TCDF	tetrachlorodibenzofuran
TCE	trichloroethene
TCRA	time-critical removal action
TE	trapping efficiency
TEF	toxic equivalency factor
TEQ	toxic equivalent
TOC	total organic carbon
TPH	total petroleum hydrocarbons
tpy	tons per year
TRIS	Toxics Release Inventory System
TRV	toxicity reference value

Acronym	Definition
TSD	treatment, storage, and disposal
TTA	Tier Two Area
TSS	total suspended solids
U&A	Usual and Accustomed
UCL	upper confidence limit
U-qualifier	not detected at given concentration
USACE	US Army Corps of Engineers
U.S.C.	United States Code
USCG	United States Coast Guard
USFWS	US Fish and Wildlife Service
USGS	US Geological Survey
UST	underground storage tank
VCP	Voluntary Cleanup Program
VOC	volatile organic compound
WAC	Washington Administrative Code
WARM	Washington Ranking Method
WDF	Washington State Department of Fisheries
WDFW	Washington State Department of Fish and Wildlife
WHO	World Health Organization
Windward	Windward Environmental LLC
WIR	water ingestion rate
WPCC	Washington State Pollution Control Commission
WQA	water quality assessment
WQS	water quality standard
WRCC	Western Regional Climate Center
WRIA	Water Resource Inventory Area
WSDOH	Washington State Department of Health
WSDOT	Washington State Department of Transportation
ww	wet weight
WWTP	wastewater treatment plant

Executive Summary

This document presents the results of the remedial investigation (RI) for the Lower Duwamish Waterway (LDW), located in Seattle, Washington. The RI provides information on the extent of contamination and the risks to humans and the environment posed by the contamination. The RI will be used to help make decisions about necessary sediment cleanup actions and other actions to manage risks.

In 2000, the City of Seattle, King County, the Port of Seattle, and The Boeing Company, working collectively as the Lower Duwamish Waterway Group, agreed to conduct a remedial investigation/feasibility study (RI/FS) for the LDW as a voluntary action with oversight by the US Environmental Protection Agency (EPA) and the Washington State Department of Ecology (Ecology). In September 2001, the LDW was formally listed as a Superfund site; and in February 2002, the LDW was formally listed as a Washington Model Toxics Control Act site.

This RI presents the results of many years of investigations conducted for the LDW study area, which extends from the southern tip of Harbor Island to just south of the Upper Turning Basin (Map ES-1). The RI data will support the analyses to be conducted in the FS to allow EPA and Ecology to make final cleanup decisions for the LDW.

This RI describes what is known about the LDW, including:

- the history of the LDW as well as the environmental setting, habitat, and uses
- the deposition and transport of sediment within the LDW
- the distribution of contamination in the LDW, including concentrations of chemicals in sediment, water, and tissues
- information regarding potential historical and ongoing sources of chemicals to the LDW as well as the source control and identification strategy
- the results of the baseline human health risk assessment (HHRA) and ecological risk assessment (ERA), which assess risks to people and ecological species from contamination within the LDW prior to remedial actions

The specific topics addressed in each section of the RI are summarized below. Details are provided in the RI, its appendices, and attachments.

ES.1 ENVIRONMENTAL SETTING

The LDW is an estuary that has been extensively modified over the past 100 years. Diversion of two major rivers (i.e., White River in 1906 and Black River in 1916) approximately 100 years ago reduced the flow volume and watershed area by about 70%. Today, the Green River is the primary source of water for the Duwamish River.

Many of the natural curves of the estuary were eliminated when the navigation channel was created in the early 20th century. Prior to 1920, the King County Commercial Waterway District No. 1 had dredged the LDW to an Upper Turning Basin at River Mile (RM) 4.6. This Upper Turning Basin serves as a trap for most sediment being carried downstream by the Green/Duwamish River. Frequent dredging is still necessary to remove sediment from this area and thereby keep it from being transported into the lower reaches of the LDW. The construction of the Howard Hansen Dam, approximately 65 mi upstream of the LDW, in 1961 effectively decreased peak flows in the river system, which now rarely exceed 12,000 cubic feet per second.

Today, the LDW is maintained as a federal navigation channel by the US Army Corps of Engineers. The shoreline along the majority of the LDW has been developed for industrial and commercial operations. Common shoreline features include constructed bulkheads, piers, wharves, sheet piling walls, buildings that extend over the water, and steeply sloped banks armored with riprap or other fill material. Despite the highly modified river habitat, the LDW is home to diverse communities of fish, birds, mammals, and invertebrate species.

The LDW corridor is one of Seattle's primary industrial areas. Current land use, zoning requirements, and land ownership within this corridor are consistent with the characteristics of an active industrial waterway. Two Seattle neighborhoods, South Park and Georgetown, are located to the west and east, respectively, of the LDW. These neighborhoods support a mixture of residential, recreational, commercial, and industrial uses. The LDW supports considerable commercial navigation, but is also used for various recreational activities such as boating, kayaking, fishing, and beach play. The LDW is also the site of the Muckleshoot Tribe's commercial fishery for salmon.

Intertidal habitats are dispersed in relatively small patches downstream of RM 3.0, with the exception of Kellogg Island, which represents the largest contiguous area of intertidal habitat remaining in the LDW. There is great interest in restoring and improving the intertidal habitat in the LDW. Several restoration projects have already been completed, some of which included the creation of public parks.

ES.2 SEDIMENT DYNAMICS

Several studies of LDW sediment movement have been conducted since the 1970s, but the most extensive work has been done in the past 3 years as part of the RI. This type of analysis will be used in the FS to estimate the potential for contaminated surface sediment to be buried by cleaner sediment in the future. This work included the collection of new data to be used in conjunction with historical data to develop a sediment transport model (STM) for the LDW. This model, as well as other lines of evidence, was used to develop a physical conceptual site model (CSM) for the LDW. The key components of the CSM are:

- On annual time scales, more sediment is added (deposited) throughout the LDW than is removed (eroded).
- The rate at which sediment is deposited is generally higher in deeper areas, especially the navigation channel, than in shallower areas, such as intertidal benches.
- High-flow events that occur periodically (e.g., one event every 2 years or less frequently), generally as a result of high rainfall events, have the greatest influence on sediment erosion. Net erosion (when the amount of erosion exceeds that of deposition) occurs at 20% or less of the LDW during such events, with estimated depths of 10 cm or less (maximum depths of 20 cm).[1] Over much of the LDW, erosion depths are much less, particularly between RM 0.0 and RM 2.0, where net erosion is predicted to be negligible.
- Tugboats in the navigation channel tend to mix a thin layer of sediment (i.e., 1 to 2 cm) repeatedly over certain areas where these boats travel. Tugs have the potential to erode sediment to a greater depth while maneuvering over relatively small areas outside the navigation channel.

The STM developed for the RI/FS (QEA 2007) is an important tool for predicting the changes, both erosion and deposition, that occur on the sediment bed. Simulations were run for both short (days or weeks) and long (years) time periods. The STM was also used to estimate changes in the physical composition of the sediment surface layer over time as sediment from upstream and lateral sources (e.g., storm drains) becomes incorporated into the sediment bed. Based on the results of the modeling conducted to date, approximately 99% of the total external sediment load to the LDW comes from the Green River, upstream of the LDW, where concentrations of key chemicals (such as polychlorinated biphenyls [PCBs]) are much lower than they are in certain areas of the LDW.

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[1] These depths are similar to the 10-cm (3.9-in.) depth used to characterize surface sediment within the LDW and typically less than the biologically mixed layer.

ES.3 NATURE AND EXTENT OF CHEMICAL CONTAMINATION

The presence of chemical contamination in the LDW has been known for many years, prompting numerous environmental studies. From 1990 to 2001, approximately 1,200 surface sediment samples, 230 subsurface sediment samples, and 90 fish and shellfish tissue samples were collected and analyzed for metals and organic compounds. A Phase 1 RI (Windward 2003a) was initially produced based solely on information available at that time. Additional chemical data needs were subsequently identified (Windward 2003e), and a plan for filling those data needs was prepared (Windward 2004h). Since 2003, approximately 600 additional samples were collected and analyzed to characterize chemical contamination in the following media:

- Juvenile chinook salmon tissue
- Fish, clam, and crab tissue
- Benthic invertebrate tissue
- Seep water (groundwater seeping from banks along the river)
- Surface sediment (top 10 cm or 3.9 in.)
- Subsurface sediment (below the top 10 cm)
- Porewater (water in spaces between sediment particles)

Based on surface sediment data, the LDW can be characterized as having localized areas with relatively high chemical concentrations (hot spots) separated by relatively large areas with lower chemical concentrations (Table ES-1). Many of the highest concentrations of key chemicals are in areas that were identified as candidates for early action (known as early action areas [EAAs] (Maps ES-2a, ES-2b, and ES-2c) (Windward 2003d). PCBs, metals, polycyclic aromatic hydrocarbons (PAHs), and phthalates were frequently detected in surface sediments. Many other organic chemicals, such as volatile organic compounds (VOCs), other semivolatile organic compounds, and pesticides, were less frequently or rarely detected.

Table ES-1. Summary statistics for key chemicals in surface sediment

Chemical	Unit	Number of Locations	Average Concentration	Median Concentration	95TH Percentile Concentration	Maximum Concentration
Total PCBs	µg/kg dw	1,327	1,170	137	4,170	220,000
Arsenic	mg/kg dw	852	17	11	30	1,100
cPAHs	µg/kg dw	828	460	260	1,500	11,000
Dioxins and furans	ng/kg dw	47	92.7	10.2	448	2,100
BEHP	µg/kg dw	832	590	230	2,400	14,000

Note: summary statistics were calculated assuming one-half the reporting limit for non-detect results.

dw – dry weight

BEHP – bis(2-ethylhexyl) phthalate

cPAH – carcinogenic polycyclic aromatic hydrocarbon

PAH – polycyclic aromatic hydrocarbon

PCB – polychlorinated biphenyl

The distribution of chemicals within subsurface sediment has also been characterized. In many areas, the highest chemical concentrations were buried under surface sediment with lower concentrations. These results are consistent with the hypothesis that much of the sediment contamination in the LDW is the result of historical releases that have been buried by cleaner and more recently deposited sediment, as predicted for many areas by the STM.

Some areas have high chemical concentrations in both subsurface and surface sediment. This distribution of contamination generally is found in areas estimated to have low net sedimentation rates by the STM. In a few areas where higher sedimentation was predicted, the presence of high chemical concentrations near the surface could be the result of localized disturbance events or recent or ongoing sources of contamination.

Tissue samples of many different fish and invertebrate species have been collected and analyzed. Most of the tissue data represent English sole, shiner surfperch, Pacific staghorn sculpin, juvenile chinook salmon, Dungeness and slender crabs, soft-shell clams, and small invertebrates that live in the sediment, such as amphipods and marine worms. These species were selected because they were assumed to be representative of species that could be consumed by people, fish, or wildlife within the LDW. Their tissues were analyzed for a wide variety of chemicals.

Total PCB concentrations in clams and benthic invertebrates, as well as in mobile species that may have foraging ranges smaller than the LDW (e.g., Pacific staghorn sculpin and shiner surfperch), were significantly correlated with total PCB concentrations in sediment. This correlation was not evident for English sole and crabs, which likely have larger foraging ranges. Mean total PCB concentrations were highest for English sole and crab hepatopancreas tissue followed by shiner surfperch and lowest for juvenile chinook salmon. Inorganic arsenic concentrations in softshell clams were much higher than in fish and crabs collected in the LDW and in clams from Puget Sound. PAH concentrations were highest in clams, mussels, and benthic invertebrates, and phthalates were frequently detected in clams and benthic invertebrates. Most other organic chemicals were infrequently detected.

The most comprehensive tissue sampling was conducted in 2004, several months after three dredging events were completed in the lower reach of the LDW and near Harbor Island (Duwamish/Diagonal in the LDW, Todd Shipyard in the West Waterway, and Port of Seattle in the East Waterway). Total PCB concentrations were much higher in some fish and crab species in 2004 than in older (1995 to 1998) and more recent (2005 and 2006) samples, suggesting that exposure to total PCBs may have been higher immediately following the dredging events than is typical for the LDW.

Chemical concentrations in surface water, seep water, and porewater were also summarized in the RI. A large number of surface water grab samples were collected by King County in 1996 and 1997. Chemical concentrations in these samples were generally low or undetected. More recently (2005), water samples were collected and analyzed for PCB congeners using methods that are more sensitive than those used previously. Total PCB concentrations in whole water samples ranged from 0.13 to 3.2 ng/L, with the lowest concentrations detected during periods when the flows were highest.

Seep water and porewater data were collected in areas along the banks of the LDW where EPA and Ecology considered the likelihood of contamination to be the greatest. These data were collected instead of additional groundwater data because they are more directly relevant to animals using the LDW. The results of seep water sampling showed detections of most metals, but generally at low concentrations. Elevated concentrations of VOCs in both seep water and porewater were detected in a localized area near one facility with well-documented VOC contamination in groundwater (i.e., Great Western International). Groundwater quality and sediment quality do not appear to be closely connected in the LDW, with a few potential exceptions (e.g., the head of Slip 1). The importance of groundwater pathways should be considered on a site-by-site basis for areas with potential groundwater issues prior to sediment remediation.

ES.4 BASELINE ECOLOGICAL RISK ASSESSMENT

The baseline ERA presents risk estimates for benthic invertebrate, crabs, fish, and wildlife species that may be exposed to chemicals in the LDW. Chemicals of potential concern (COPCs) were first identified through a conservative risk-based screening process so that risk analyses could focus on chemicals more likely to be of concern.

There were 10 receptors of concern (ROCs)[2] evaluated in the ERA (i.e., the benthic invertebrate community, crabs, juvenile chinook salmon, Pacific staghorn sculpin, English sole, spotted sandpiper, great blue heron, osprey, river otter, and harbor seal). ROCs are species selected to represent larger groups of animals because not all species can be individually evaluated in the ERA. The risk estimates were based either on COPC concentrations in sediment, water, and aquatic biota from the LDW or on estimated dietary chemical doses, depending on the species and chemical.

For crabs, fish, and wildlife, the site-related chemical concentrations or doses were compared to concentrations or doses from the scientific literature that have been

[2] Key considerations for selecting ROCs include the potential for direct or indirect (e.g., ingestion of fish or invertebrates) exposure to sediment-associated chemicals, human and ecological significance, site usage, sensitivity to COPCs at the site, susceptibility to biomagnification of COPCs (i.e., higher-trophic-level species), and data availability.

shown to cause specific harmful effects in the same or related species. The effects of primary concern are those that decrease survival, growth, or reproduction.

For benthic invertebrates living in intertidal and subtidal sediments, sediment chemistry and site-specific toxicity test results indicated that harmful effects are not likely in approximately 75% of the LDW area. There is a higher likelihood for adverse effects in approximately 7% of the LDW area, which was designated as having chemical concentrations or biological effects in excess of cleanup screening level (CSL) criteria of the Washington State Sediment Management Standards (SMS). The remaining 18% of the LDW area had chemical concentrations or biological effects between the sediment quality standard (SQS) and CSL criteria, indicating that risks to benthic invertebrate communities are less certain in these areas than in areas with concentrations greater than one or more CSL values.

Forty-one chemicals exceeded the SQS in one or more surface sediment samples. Concentrations of total PCBs, bis(2-ethylhexyl) phthalate (BEHP), and butyl benzyl phthalate exceeded SMS criteria more frequently than any other chemicals. No other chemicals exceeded the SQS in more than 5% of the surface sediment samples. Risks to the benthic invertebrate community from VOCs detected in sediment porewater were very low, with the exception of one small area located near Great Western International at RM 2.4. Because this area was considered to be a worst-case exposure area with respect to the potential for adverse effects for benthic invertebrates from VOCs, and other areas with seep water data had much lower VOC concentrations, it was concluded that there was a very low likelihood of appreciable risks from VOCs in the rest of the LDW. Risks to benthic invertebrates from tributyltin (TBT) were very low.

For the other receptors evaluated as ROCs, hazard quotients (HQs) were calculated based on a range of effects data, including no-observed-adverse-effect levels (NOAELs) and lowest-observed-adverse-effect levels (LOAELs). The highest LOAEL-based HQ for fish, crabs, and wildlife species was 5.5 (for lead in spotted sandpiper), indicating that ecological risks in the LDW for these species were estimated to be relatively low considering the uncertainties in the analyses (Table ES-2).

Table ES-2. Summary of ecological risks for fish and wildlife species

CHEMICAL	RECEPTOR OF CONCERN	NOAEL-BASED HQ	LOAEL-BASED HQ
COCs with LOAEL-Based HQs ≥ 1.0[a]			
Total PCBs	crab	**10**	**1.0**
	English sole	**4.9 – 25**	0.98 – **5.0**
	Pacific staghorn sculpin	**1.5 – 19**	0.30 – **3.8**
	river otter	**5.8**	**2.9**
PCB TEQs	spotted sandpiper	**1.9 – 15**	0.18 – **1.5**
Cadmium	juvenile chinook salmon	**5.0**	**1.0**

CHEMICAL	RECEPTOR OF CONCERN	NOAEL-BASED HQ	LOAEL-BASED HQ
	English sole	**6.1**	**1.2**
	Pacific staghorn sculpin	**3.0 – 5.2**	0.60 – **1.0**
Chromium	spotted sandpiper	**1.3 – 8.8**	0.26 – **1.8**
Copper	spotted sandpiper	0.62 – **1.5**	0.45 – **1.1**
Lead	spotted sandpiper	0.58 – **19**	0.17 – **5.5**
Mercury	spotted sandpiper	**1.1 – 5.3**	0.21 – **1.0**
Vanadium	English sole	**5.9**	**1.2**
	Pacific staghorn sculpin	**3.2 – 5.9**	0.65 **– 1.2**
	spotted sandpiper	**2.0 – 2.7**	**1.0 – 1.4**
COPCs with NOAEL-Based HQs ≥ 1.0 and LOAEL-Based HQs < 1.0[b]			
Total PCBs	spotted sandpiper	0.51 – **2.0**	0.18 – 0.71
PCB TEQs	osprey	**1.6**	0.16
	river otter	**4.5**	0.59
Arsenic	juvenile chinook salmon	**1.1**	0.73
	English sole	**1.2**	0.80
	crab	**3.9**	na
Benzoic acid	English sole	**1.5**	na
	Pacific staghorn sculpin	**2.1**	na
Cadmium	Pacific staghorn sculpin	**3.0 – 4.9**	0.60 – 0.98
Chromium	juvenile chinook salmon	**2.1**	na
	English sole	**1.1**	na
Copper	Juvenile chinook salmon	**1.9**	0.93
	English sole	**1.9**	0.93
	Pacific staghorn sculpin	0.9 – **1.5**	0.45 – 0.77
Mercury	river otter	**2.8**	0.57
TBT	Pacific staghorn sculpin	**1.6** – **2.9**	0.18 – 0.33
Vanadium	juvenile chinook salmon	**4.0**	0.79
Zinc	crab	**2.5**	0.91

Note: HQs for fish are the highest HQs in cases where more than one approach was used. **Bold** identifies NOAEL-based HQs greater than 1.0 or LOAEL-based HQs greater than or equal to 1.0.

COPC – chemical of potential concern
HQ – hazard quotient
LOAEL – low-observed-adverse-effect level
na – not available
NOAEL – no-observed-adverse-effect level
PCB – polychlorinated biphenyl
TBT – tributyltin
TEQ – toxic equivalent

COCs were defined as chemicals with LOAEL-based HQs greater than or equal to 1, which indicates a potential for adverse effects. Seven chemicals were identified as chemicals of concern (COCs) (PCBs, cadmium, chromium, copper, lead, mercury, and vanadium) for at least one fish and wildlife species (Table ES-2). No quantitative ecological risk estimates were calculated for dioxins and furans.

Based on the risk estimates, uncertainties discussed in the ERA, natural background concentrations, and COC concentrations in EAAs, chemicals were identified as risk drivers for ecological receptors in accordance with EPA (1998) and Ecology (Washington Administrative Code [WAC] 173-340-703) guidance. The risk drivers from both the ERA and the HHRA will be the focus of remedial analyses in the FS. In consultation with EPA and Ecology, PCBs were identified as a risk driver for river otter because estimated exposure concentrations for river otter were greater than the LOAEL by a factor of 2.9 and uncertainties in the risk estimate were relatively low. In addition, 41 chemicals were selected as risk drivers for the benthic invertebrate community because concentrations of these 41 chemicals exceeded the SQS at one or more locations. COCs not selected as risk drivers may be addressed through focused evaluation in the FS or included in the post-remedial monitoring program that is part of the 5-yr review that EPA conducts at all Superfund sites after cleanup is completed.

ES.5 BASELINE HUMAN HEALTH RISK ASSESSMENT

The baseline HHRA estimated the risks people could face from exposure to chemicals in LDW sediments and seafood (exposure pathways). Direct contact with sediments during commercial netfishing, clamming, or beach play in the LDW and consumption of seafood from the LDW were identified as primary exposure scenarios. Quantitative risk estimates for other exposure scenarios, such as swimming, were also included in the HHRA but were calculated in a previous risk assessment, which suggested that risks from those scenarios were insignificant.

In keeping with EPA risk assessment guidance, reasonable maximum exposure (RME) estimates were calculated for all exposure scenarios to avoid underestimating risks. Consequently, risk estimates may be overestimated for many individuals. This approach is consistent with EPA's policy of "RME," which uses high-end, but plausible, estimates of exposure for assessing risks.

There are limited data available on the amount of resident seafood organisms currently being harvested and consumed from the LDW. EPA and Ecology required the application of seafood consumption rates that have been developed for adult and child consumers based on seafood harvest from King County or Puget Sound. The tribal seafood consumption rates represent relatively high rates that might occur for harvest from the LDW at some time in the future, and were identified by EPA as the most appropriate RME seafood consumption scenarios for tribal members, although they likely overestimate the current consumption of resident seafood organisms from the LDW. An additional RME scenario required by EPA and Ecology was based on seafood consumption by adult API. There is considerable uncertainty about the applicability of some of the seafood consumption rates to this HHRA, particularly for clams, given the size of the LDW and the availability of suitable habitat.

Other scenarios with exposure assumptions higher or lower than those of the RME scenarios were also evaluated to provide a broad range of risk estimates. For seafood

consumption, these other scenarios included consumption rates estimated for the Suquamish Tribe, "average exposure" scenarios using central tendency consumption rate estimates, and a "unit risk" scenario based on an assumed one seafood meal per month. The one-meal-per-month exposure scenario is not meant to actually describe seafood consumption that is occurring on the LDW (there are few data[3] on current recreational seafood consumption rates for the LDW). Instead, it is intended to allow individuals to evaluate their own exposure depending on how many meals of seafood of different types collected from the LDW are consumed per month. For example, if an individual consumed two meals per month, then the risk for this individual would be two times the one-meal-per-month risk estimate.

There are no LDW-specific data to estimate the degree to which humans may currently be directly exposed to sediments via beach play or clamming activities. The exposure scenarios assumed for these activities were intended to represent exposures for a health-protective estimation of risks. The tribal netfishing scenario, on the other hand, reflects exposure conditions that could occur under current tribal fishing activities within the LDW.

Once the exposure scenarios were selected, chemistry data for each COPC were compiled over the assumed exposure area (e.g., intertidal sediments for clamming). For the seafood consumption scenarios, data for several different species (e.g., English sole, perch, crab, clams, mussels) were used to represent a typical seafood consumer's diet.

Based on health-protective exposure assumptions, estimated cancer risks in the LDW were determined to be highest for the seafood consumption scenarios (Table ES-3). The cumulative risk for all carcinogenic chemicals ranged from 7 in 10,000 (7×10^{-4}) to 3 in 1,000 (3×10^{-3}) for the RME seafood consumption scenarios (adult tribal RME based on Tulalip Tribes' data, child tribal RME based on Tulalip Tribes' data, and adult API RME), with the primary contributors being PCBs, arsenic, and carcinogenic PAHs (cPAHs). Seafood samples from the LDW were not analyzed for dioxins and furans, so risks from these chemicals are not included in seafood consumption risk estimates but were assumed to be unacceptable. The evaluation of non-cancer HQs indicates the potential for adverse effects other than cancer associated with seafood consumption, particularly for arsenic and PCBs (Table ES-3).

[3] In 1997, King County conducted a survey on seafood collection and consumption that included locations in the LDW.

Table ES-3. Summary of risks associated with seafood ingestion scenarios

Scenario	Ingestion Rate (g/day)	Meals per Month[a]	Exposure Duration (years)	Excess Cancer Risk	Non-Cancer HI[b]
Adult tribal RME (Tulalip data)	97.5	13.1	70	3×10^{-3}	47
Adult tribal CT (Tulalip data)	15	2.0	30	1×10^{-4}	5
Adult tribal (Suquamish data)	597.7	80	70	3×10^{-2}	348
Child tribal RME (Tulalip data)	39.0	5.2	6	7×10^{-4}	104
Child tribal CT (Tulalip data)	6.0	0.8	6	7×10^{-5}	10
Adult API RME	51.5	6.9	30	1×10^{-3}	35
Adult API CT	5.3	0.7	9	2×10^{-5}	2
One meal per month – benthic	7.5	1.0	30	1×10^{-4}	6
One meal per month – crab	7.5	1.0	30	4×10^{-5}	1
One meal per month – clam	7.5	1.0	30	2×10^{-4}	4
One meal per month – pelagic fish	7.5	1.0	30	2×10^{-4}	10

[a] It is assumed that one meal is equal to 227 g (8 ounces).

[c] Total across all chemicals. This total is not directly interpretable for risk assessment because it includes hazard quotients across multiple endpoints. The values indicate that the HI may exceed 1 for individual endpoints.

API – Asian and Pacific Islander

CT – central tendency

HI – hazard index (a sum of the HQs for individual chemicals)

RME – reasonable maximum exposure

Cancer risks for direct sediment contact RME scenarios for netfishing, clamming, and beach play were much lower (Table ES-4). Risk estimates ranged from 5 in 1,000,000 (5×10^{-6}) to 1 in 10,000 (1×10^{-4}). COCs for these pathways included PCBs, arsenic, cPAHs, and dioxins and furans. None of the direct sediment RME scenarios had HQs greater than 1 for non-cancer effect, and therefore, non-cancer effects are not expected from direct contact with sediments.

Table ES-4. Summary of risks associated with direct sediment exposure scenarios

Scenario	Exposure Area	Incidental Sediment Ingestion Rate (g/day)	Exposure Frequency (days/yr)	Exposure Duration (years)	Excess Cancer Risk
Netfishing RME	all subtidal and intertidal	0.050	119	44	3×10^{-5}
Netfishing CT	all subtidal and intertidal	0.050	63	29	5×10^{-6}
Beach play RME	Area 1	0.20	65	6	2×10^{-5}
	Area 2				5×10^{-5}
	Area 3				3×10^{-5}

Scenario	Exposure Area	Incidental Sediment Ingestion Rate (g/day)	Exposure Frequency (days/yr)	Exposure Duration (years)	Excess Cancer Risk
	Area 4				3×10^{-5}
	Area 5				8×10^{-6}
	Area 6				9×10^{-6}
	Area 7				5×10^{-6}
	Area 8				7×10^{-6}
Clamming 7 days per year	intertidal area accessible from the shore	0.1	7	30	1×10^{-6}
Tribal clamming RME scenario	intertidal area accessible from the shore or from a boat	0.1	120	64	1×10^{-4}
Tribal clamming 183 days per year	intertidal area accessible from the shore or from a boat	0.1	183	70	2×10^{-4}

Note: Non-cancer hazard quotients did not exceed 1 for any chemical and are therefore not shown in this table.

CT – central tendency

RME – reasonable maximum exposure

Nineteen chemicals were identified as COCs (a COC has a cancer risk estimate greater than 1 in 1,000,000 [1×10^{-6}] or an HQ greater than 1 for an RME scenario) for at least one RME seafood consumption scenario.[4] Five chemicals were also identified as COCs based on a cancer risk estimate greater than 1 in 1,000,000 (1×10^{-6}) for at least one direct sediment RME scenario.[5] Four of these COCs were identified as risk drivers based on the magnitude of their risk estimates and the relative percentage of their contributions to total human health risk. PCBs, arsenic, cPAHs, and dioxins and furans were identified as risk drivers for both seafood consumption and direct sediment exposure scenarios. COCs not selected as risk drivers in the baseline HHRA may be addressed through focused evaluation in the FS or included in the post-remedial monitoring program that is part of the 5-yr review that EPA conducts at all Superfund sites after cleanup is completed.

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[4] The COCs based on seafood consumption scenarios were PCBs, arsenic, cPAHs, dioxins and furans, aldrin, BEHP, alpha-BHC, beta-BHC, carbazole, total chlordane, total DDTs, dieldrin, gamma-BHC, heptachlor, heptachlor epoxide, hexachlorobenzene, pentachlorophenol, TBT, and vanadium. Dioxins and furans were included as a COC for seafood consumption, even though no quantitative risk estimates were made. It was assumed that dioxin and furan excess cancer risk estimates for RME seafood consumption scenarios would have been greater than 1×10^{-6}.

[5] The COCs based on direct sediment exposure were PCBs, arsenic, cPAHs, dioxins and furans, and toxaphene.

These findings do not constitute a definitive characterization of human health risks. There are many uncertainties associated with the site-specific risk estimates for each exposure scenario. In spite of these uncertainties, the baseline risk characterization for the LDW site is considered to be health-protective and sufficient to support risk management decisions.

ES.6 BACKGROUND CONCENTRATIONS

Both EPA and Ecology recognize two types of background, natural and urban (also referred to as anthropogenic or area), although their definitions and uses differ. The most important difference is in the potential application of urban background concentrations in making risk management decisions. EPA generally does not require cleanup levels below urban background concentrations because of the potential for recontamination from sources unrelated to the site, cost effectiveness, and technical practicability (EPA 2002d). Under Ecology's regulations, when area background concentrations would result in recontamination of the site to levels that exceed cleanup levels, that portion of the cleanup action that addresses cleanup below area background concentrations may be delayed until the offsite sources of hazardous substances are controlled. In these cases, the cleanup action will be considered an interim action until cleanup levels are attained.

Preliminary natural background concentrations in sediment for four risk driver chemicals (total PCBs, arsenic, cPAHs, and dioxins and furans) were calculated from 10 Puget Sound reference areas (Table ES-5). The application of these natural background data to remedial decisions will be determined by EPA and Ecology.

Table ES-5. Preliminary background concentrations for risk driver chemicals in surface sediment

Risk Driver Chemical	Unit	Preliminary Natural Background Concentration	Urban Background Concentrations[a]
Total PCBs	µg/kg dw	31	21 – 135
Arsenic	mg/kg dw	12	6 – 37
cPAHs[b]	µg/kg dw	77	101 – 106
Dioxin and furan TEQ	ng/kg dw	0.52	2.41 – 14.9

[a] Range represents mean concentrations from several data sources described in more detail in Section 7.2.

[b] cPAH concentrations are given in terms of benzo(a)pyrene equivalents

cPAH – carcinogenic polycyclic aromatic hydrocarbon

dw – dry weight

PCB – polychlorinated biphenyl

TEQ – toxic equivalent

The RI presents various urban background data from multiple sources (e.g., Green River data upstream of the LDW and sediment data from urban bays) for the risk

driver chemicals but does not recommend specific area background concentrations, which may be developed as part of the FS in consultation with Ecology and EPA.

ES.7 RISK-BASED THRESHOLD CONCENTRATIONS

Sediment risk-based threshold concentrations (RBTCs), which are defined as concentrations of risk driver chemicals in sediment that are associated with specific risks, are helpful in risk management decisions. If sediment were remediated to meet a sediment RBTC, for example, then the exposure conditions following that cleanup action would result in reduced risk. RBTCs were estimated for each of the risk driver chemicals identified in the risk assessments. Sediment RBTCs were calculated for risks associated with direct sediment contact by people, seafood consumption, and protection of the benthic invertebrate community.

For each of the direct sediment contact RME scenarios (beach play, tribal clamming, and tribal netfishing), sediment RBTCs were calculated using the risk equation for three different excess cancer risk thresholds (Table ES-6). Other RBTCs were also calculated for informational purposes. For example, RBTCs associated with the central tendency scenarios were also calculated. Sediment RBTCs derived for human health direct sediment contact scenarios should be applied as spatially weighted concentrations over the exposure areas identified in the HHRA. For the netfishing scenario, the exposure area is the entire LDW. For the clamming and beach play scenarios, the exposure areas include intertidal areas where such activities may occur.

Table ES-6. Summary of sediment RBTCs for human health risk drivers for direct sediment exposure scenarios

RISK DRIVER CHEMICAL	UNIT	1 IN A MILLION RISK LEVEL	1 IN 100,000 RISK LEVEL	1 IN 10,000 RISK LEVEL
Arsenic	mg/kg dw	1.3 – 3.7	13 – 37	130 – 370
cPAHs[a]	µg/kg dw	90 – 380	900 – 3,800	9,000 – 38,000
Dioxin and furan TEQ	ng/kg dw	13 – 37	130 – 370	1,300 – 3,700
Total PCBs	µg/kg dw	500 – 1,700	5,000 – 17,000	50,000 – 170,000

[a] cPAH concentrations are given in terms of benzo(a)pyrene equivalents

cPAH – carcinogenic polycyclic aromatic hydrocarbon

dw – dry weight

na – not applicable

PCB – polychlorinated biphenyl

RBTC – risk-based threshold concentration

TEQ – toxic equivalent

For seafood consumption, sediment RBTCs for total PCBs were estimated using a food web model. A range of RBTCs was calculated. Sediment RBTCs for the 1 in a million (1×10^{-6}) and 1 in 100,000 (1×10^{-5}) risk thresholds for the tribal RME (adult and child) scenario could not be calculated; the contribution of total PCBs from water alone was

high enough to result in a risk level above those risk thresholds even in the absence of any contribution from sediment (Table ES-7).

Table ES-7. Sediment RBTCs for total PCBs based on excess cancer risks in HHRA seafood consumption scenarios

Seafood Consumption Scenario	RBTC (µg/kg ww)		
	1 in a Million Risk Level (1×10^{-6})	1 in 100,000 Risk Level (1×10^{-5})	1 in 10,000 Risk Level (1×10^{-4})
Adult tribal RME (Tulalip data)	nc	nc	19
Child tribal RME (Tulalip data)	nc	nc	320
API RME	nc	nc	180
Adult tribal CT (Tulalip data)[a]	nc	53	> 380[b]
Child tribal CT (Tulalip data)[a]	nc	190	> 380[b]
API CT[a]	25	> 380[b]	> 380[b]
Adult one meal per month of benthic fish[a]	nc	20	> 380[b]
Adult one meal per month of pelagic fish[a]	nc	na	> 380[b]
Adult one meal per month of crabs[a]	nc	> 380[b]	> 380[b]
Adult one meal per month of clams[a]	nc	> 380[b]	> 380[b]

[a] The RBTC for this scenario is presented for information only and will not be used in remedial decision-making.

[b] A sediment RBTC of > 380 µg/kg dw indicates that even under current conditions in the LDW, risks are already estimated to be less than the target risk level.

API – Asian and Pacific Islander

CT – central tendency

dw – dry weight

HHRA – human health risk assessment

nc – not calculated (sediment RBTC could not be calculated because contribution from water led to estimated tissue concentrations greater than the applicable risk level, even in the absence of any contribution from sediment)

PCB – polychlorinated biphenyl

RBTC – risk-based threshold concentration

RME – reasonable maximum exposure

For the 1 in 10,000 (1×10^{-4}) risk level, sediment RBTCs for total PCBs ranged from 19 to 320 µg/kg dw, which are lower than the 1 in a million RBTCs for the direct sediment exposure scenarios (500 to 1,700 µg/kg dw). The sediment RBTCs calculated for river otter ranged from 218 to 250 µg/kg dw. These sediment RBTCs for total PCBs are lower than the current spatially weighted average PCB concentration in the LDW (approximately 380 µg/kg dw[6]). RBTCs corresponding to the 1 in 10,000 (1×10^{-4}) risk level for CT and one-meal-per-month scenarios are higher than 380 µg/kg dw,

[6] Using the IDW method documented in RETEC (2007, in prep.), the SWAC of total PCBs for the LDW from RM 0.0 to 6.0 was 342 µg/kg dw. The SWAC, based on an earlier interpolation used in the FWM, was 380 µg/kg dw.

indicating that the current risks associated with these scenarios are lower than 1 in 10,000 (Table ES-7).

Sediment RBTCs were not calculated for arsenic, cPAHs, or dioxins and furans for the human seafood consumption pathway because either data were not available or the relationship between sediment and tissue for these chemicals is not sufficiently understood. Sediment RBTCs for the protection of benthic invertebrates were the SQS and CSL from the SMS.

A comparison of RBTCs with background concentrations will be important in risk management decisions by EPA and Ecology because cleanup levels cannot be established or maintained below background concentrations because of the potential for recontamination from sources not related to the site (Table ES-8). Sediment RBTCs for the 1×10^{-6} threshold for direct sediment contact are higher than the preliminary natural background concentrations for PCBs, cPAHs, and dioxins and furans. For arsenic, the RBTCs for the 1×10^{-6} threshold for direct contact are well below both preliminary natural and urban background concentrations.

Table ES-8. Comparison of sediment RBTCs and background levels for risk driver chemicals

Risk Driver Chemical	Unit	Risk Level	Sediment RBTCs	Natural Background Concentration	Urban Background Concentration[a]
Arsenic	mg/kg dw	1 in a million (1×10^{-6})	1.3 – 3.7[b]	12	6 – 37
		1 in 100,000 (1×10^{-5})	13 – 37[b]		
		1 in 10,000 (1×10^{-4})	130 – 370[b]		
		HQ = 1	na		
		SQS	57		
		CSL	93		
cPAHs[c]	µg/kg dw	1 in a million (1×10^{-6})	90 – 380[b]	77	101 – 106
		1 in 100,000 (1×10^{-5})	900 – 3,800[b]		
		1 in 10,000 (1×10^{-4})	9,000 – 38,000[b]		
		HQ = 1	na		
Dioxin and furan TEQ	ng/kg dw	1 in a million (1×10^{-6})	13 – 37[b]	0.52	2.41 – 14.9
		1 in 100,000 (1×10^{-5})	130 – 370[b]		
		1 in 10,000 (1×10^{-4})	1,300 – 3,700[b]		
		HQ = 1	na		
Total PCBs	µg/kg dw	1 in a million (1×10^{-6})	500 – 1,700[b]	31	21 – 135
		1 in 100,000 (1×10^{-5})	5,000 – 17,000[b]		
		1 in 10,000 (1×10^{-4})	50,000 – 170,000[b] 19 – 320[d]		
		HQ = 1	218 – 250[e]		

Risk Driver Chemical	Unit	Risk Level	Sediment RBTCs	Natural Background Concentration	Urban Background Concentration[a]
		SQS	130[f]		
		CSL	1,000[g]		

[a] Range represents mean concentrations from several data sources described in more detail in Section 7.2.

[b] RBTCs developed from the human health direct sediment contact RME scenarios.

[c] cPAH concentrations are given in terms of benzo(a)pyrene equivalents

[d] RBTCs developed from the human health seafood consumption RME scenarios.

[e] RBTCs developed from the otter prey ingestion scenario, estimated through the FWM.

[f] Reported value is lowest AET, which is functionally equivalent to the SQS, but in dry weight units.

[g] Report value is second lowest AET, which is functionally equivalent to the CSL, but in dry weight units.

dw – dry weight

na – not available

cPAH – carcinogenic polycyclic aromatic hydrocarbon

PCB – polychlorinated biphenyl

RBTC – risk-based threshold concentration

RME – reasonable maximum exposure

TEQ – toxicity equivalent

For cPAHs and dioxins and furans, direct sediment contact sediment RBTCs at the 1×10^{-6} risk level are also similar to urban background concentrations, while the direct sediment contact sediment RBTC for total PCBs is above urban background concentrations. At higher risk thresholds, RBTCs are well above preliminary natural and urban concentrations, except for arsenic, for which the 1×10^{-5} RBTCs are similar to background concentrations.

Tissue RBTCs for total PCBs were also calculated for a given risk threshold and seafood consumption rate. For example, the total PCB concentration in seafood that would be necessary to achieve an excess cancer risk of 1×10^{-5} would be 4.2 µg/kg ww for the adult tribal RME scenario and 14 µg/kg ww for the API RME scenario. Just as it is important to put sediment RBTCs in the context of sediment background levels, it is important to put tissue RBTCs in the context of the PCB concentrations in other common dietary food items. As described in Section 7.3.1 and in WSDOH (2006), PCBs are frequently detected in many different seafood species throughout Puget Sound at concentrations greater than these tissue RBTCs. These tissue RBTCs are also lower than the PCB concentrations that have been detected in some common food items from a typical diet (WSDOH 2006). Consumption of many of these common food items at the rates assumed under these RME scenarios would subject the consumer to unacceptable risks. Tissue RBTCs for lower consumption rates, such as one meal per month, are much lower than tissue RBTCs for the RME scenarios. However, even a 1×10^{-6} tissue RBTC for the one-meal-per-month scenario (11 µg/kg ww) is low.

ES.8 POTENTIAL CHEMICAL SOURCES AND ONGOING SOURCE CONTROL EFFORTS

Much of the sediment contamination in the LDW is believed to be the result of historical sources. Many of these historical sources, including direct discharges of municipal and industrial wastewater, have largely been controlled through compliance with modern regulatory requirements, improved business practices, elimination of product use, or cleanups of industrial property. Many of the historical direct discharges have been diverted to the sanitary sewer system and are now treated at the West Point wastewater treatment plant. Inspection and compliance programs are in place to monitor and control the handling and disposal of manufacturing wastes produced by industrial operations along the LDW. In addition, the chemical loading from stormwater and combined sewer overflows (CSOs) has been greatly reduced by permit programs that require the monitoring and control of discharges that could otherwise enter the LDW.

Although sources of chemicals to the LDW are much reduced in number and magnitude compared to the past, as chemical profiles in sediment cores demonstrate, sources that could lead to sediment contamination, or recontamination of sediment following cleanup, still exist in some areas of the LDW. Ecology, in its lead role in the Source Control Work Group (SCWG), is responsible for investigating these potential ongoing sources and evaluating whether they have been controlled sufficiently for sediment cleanup to begin.

Potential ongoing sources include stormwater runoff, CSOs, industrial wastewater discharges, deposition from uncontrolled or partially controlled air emissions (from motor vehicles and the burning of wood and fossil fuels), illicit discharges and spills, erosion of contaminated bank material, and upstream contributions from the Green River. Groundwater is not generally considered to be a major source of chemicals to sediment in the LDW. Chemicals identified in upland groundwater or seeps along the LDW have not been shown to accumulate in adjacent sediments or to result in significant exposures to aquatic life, except in a few very localized areas.

The characterization of potential chemical sources to the LDW began in the 1980s. In recent years, considerable effort has been expended by in sampling potential upland sources. The chemicals most often detected in these samples are PCBs, total petroleum hydrocarbons, phthalates, and metals (i.e., copper, mercury, and zinc).

Source control efforts are expected to continue for years to come. Ecology and the SCWG have identified 15 specific areas within the LDW as an organizational tool to prioritize source investigations (Table ES-9). Seven of these areas were previously identified as EAAs based on sediment contamination (Windward 2003d). Cleanup has occurred at two of these seven locations (i.e., Duwamish/Diagonal and Norfolk) and is currently in the planning stages for three more (Slip 4, Boeing Plant 2/Jorgensen Forge, and T-117).

Table ES-9. Areas of interest for source control

EARLY ACTION AREAS	TIER TWO AREAS
Duwamish/Diagonal (RM 0.4 – RM 0.6 E)	Glacier Bay (RM 1.3 – RM 1.5 W)[a]
EAA 2 (RM 2.2 W)	Ash Grove Cement (RM 0.0 – RM 0.2 E)[a]
Slip 4 (RM 2.8 – RM 2.9 E)	Slip 1 (RM 0.9 – RM 1.0 E)
Boeing Plant 2/Jorgensen Forge (RM 2.9 – RM 3.7 E)	Glacier NW to St. Gobain (RM 1.3 – RM 1.7 E)[a]
T-117 (RM 3.5 – RM 3.7 E)	Slip 3 to Slip 2 (RM 1.8 – RM 2.0 E)
RM 3.8 E	Slip 3 (RM 2.0 – RM 2.1 E)
Norfolk (RM 4.8 – RM 5.0 E)	Slip 4 to Seattle Boiler Works (RM 2.4 – RM 2.8 E)[a]
	Slip 6 (RM 3.9 – RM 4.4 E)

[a] The properties owned by these companies are not necessarily included in the designated area, nor are the companies necessarily responsible for contamination in these areas.

CSO – combined sewer overflow

EAA – early action area

NW – northwest

RM – river mile

T-117 – Terminal 117

Initial source control efforts have been focused on source identification at the five EAAs that have been or are being remediated as non-time critical removal actions. Current and subsequent source control efforts are being focused on the other EAAs and Tier Two areas. As part of their source control evaluation process, summary of existing information and identification of data gaps reports and source control action plans are being prepared to help direct future investigation and remediation efforts while assessing potential sources for the chemicals identified. These efforts will continue to evaluate potential sources to the LDW and implement corrective actions to reduce or eliminate identified sources.

ES.9 KEY FINDINGS

Key findings for the RI are summarized below.

- Over the past 100 years, the LDW has been highly modified from its natural configuration to support urban and industrial development. Changes have included reductions and control of water flow, significant shoreline modifications, loss of intertidal habitat, and installation of riprap, pier aprons and sheet pile walls. Some limited areas of natural shoreline still exist within the LDW.
- Industrial and commercial facilities occupy most of the shoreline; one residential community is also located along the shoreline with another community nearby.

- The LDW is currently used primarily as an industrial navigational corridor. Secondarily, it supports recreational uses such as boating, kayaking, fishing and beach play. It also supports the Muckleshoot Tribe's commercial fishery for salmon.
- Despite significant alterations in habitat and areas with elevated chemical concentrations, the LDW contains a diverse assemblage of aquatic and wildlife species and a robust food web that includes top predators.
- Much of the sediment contamination in the LDW comes from historical releases that are now generally buried under cleaner, more recently deposited sediment. The highest levels of chemical contamination are generally detected at depth.
- In general, high concentrations of chemicals, including PCBs, were detected in surface sediment in localized areas separated by larger areas of the LDW with relatively lower concentrations. Relatively high surface sediment contamination is present in some areas as a result of a number of processes, including low net sedimentation rates in a few areas with primarily historical contamination or because of the presence of ongoing, localized sources.
- The highest risks to people are associated with consumption of fish, crabs, and clams, with lower risks associated with activities that involve direct contact with sediment, such as clamming, beach play, and netfishing.
- Most of the human health risk is from PCBs, arsenic, cPAHs, and dioxins and furans.
- Ecological risks to fish and wildlife were relatively low, with the exception of risks to river otter from PCBs.
- Sediment contamination in approximately 75% of the LDW is estimated to have no effect on the benthic invertebrate community; approximately 7% of the surface sediment has chemical concentrations exceeding the higher of the two state standards, associated with potential adverse effects to the benthic invertebrate community. The potential for effects in the remaining 18% of the LDW is more uncertain. Most of the state sediment standard exceedances were for PCBs and phthalates, although 41 different chemicals had at least one exceedance.
- Sediment is depositing within the LDW, resulting in the burial of historical contamination over much of the area. Almost all new sediment (99%) that enters the LDW originates in the Green River.
- A few areas in the LDW may be scoured during high flow events, such as storms. Based on the sediment transport model, the scour depth is relatively shallow and is generally limited to sediment in the biologically active zone; thus, deeper sediments would not be exposed from high flow events. Tug

maneuvering is a potential source of localized erosion that could disturb sediment at greater depths in relatively small areas. Routine boat traffic is expected to only mix the top few cm of sediment, which is part of the biologically active zone mixed by benthic invertebrates.

- The site conceptual model has been confirmed by several lines of evidence, including chemistry profiles in sediment. The depths of most peak PCB concentrations were consistent with the expected rate of burial, with a few exceptions that may be related to localized influences.
- Based on this understanding of sediment transport in the LDW, LDW surface sediment is gradually becoming more similar in character to the sediment being transported by the Green River and will thus become cleaner with time.
- Assuming local sources are controlled and hotspots are remediated, chemical concentrations in surface sediment are estimated to be reduced by up to 50% every 5 to 10 years from transport of cleaner sediment from the Green River.
- A comparison of risk-based sediment concentrations with background concentrations will be important in risk management decisions by EPA and Ecology because cleanup levels cannot be established or maintained below background concentrations because of the potential for recontamination from sources not related to the site.
- Most of the key risk-based goals for total PCB concentrations in sediment associated with the human consumption of seafood are below background concentrations in reference areas in Puget Sound. The only risk-based goal for an RME seafood ingestion scenario that is higher than urban or natural background concentrations is the 1×10^{-4} risk level for the API RME scenario. The risk-based goal for otter is approximately the same as the 1×10^{-4} API RME risk-based goal.
- Risk-based goals for seafood consumption for arsenic, cPAHs, and dioxins and furans could not be calculated because the relationships between concentrations in clam tissue and sediment are complex and involve important variables that may never be well understood.
- The risk-based goals for direct sediment contact for arsenic at the 1×10^{-6} risk level were lower than the natural background concentration.
- The risk-based goals at the 1×10^{-6} risk level for direct sediment contact via netfishing and clamming for cPAHs and dioxins and furans were higher than the natural background concentrations and were similar to the ranges of urban background concentrations.

- The risk-based goals at the 1×10^{-6} risk for direct sediment contact related to beach play for cPAHs were similar to natural background concentrations and lower than typical urban background concentrations.
- The risk-based goals for direct sediment contact for total PCBs were higher than both the natural background concentration and the range of urban background concentrations.
- The control of local sources is critical to the long-term success of specific remedial actions in the LDW. Ecology, as lead agency of a consortium of regulatory authorities, has begun implementing a source control program in parallel to the RI/FS process.
- Continued coordination of cleanup actions and source control will be necessary to ensure any actions taken are not unduly impacted by local sources.

1 Introduction

This document presents the results of the remedial investigation (RI) for the Lower Duwamish Waterway (LDW). Located in Seattle, Washington, the LDW consists of the downstream portion of the Duwamish River, excluding the East and West Waterways around Harbor Island. It extends from river mile (RM) 0.0 near the southern tip of Harbor Island to upstream of the Upper Turning Basin, which is located near RM 4.8 (Map 1-1) and has served as the City's major industrial corridor since about 1911.

The LDW has been the subject of numerous environmental investigations and characterization studies completed under the auspices of various governmental and private entities, including the US Environmental Protection Agency (EPA), the National Oceanic and Atmospheric Administration (NOAA), The Boeing Company (Boeing), King County (County), the Port of Seattle (Port), the Elliott Bay/Duwamish River Restoration Program (EB/DRRP), and the US Army Corps of Engineers (USACE). The LDW was evaluated and proposed for inclusion on the National Priorities List (NPL) pursuant to Section 105 of the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA, or Superfund), 42 US Code 9605, by EPA; and on September 13, 2001, the LDW was formally listed as a Superfund site. In February 2002, the Washington State Department of Ecology (Ecology) listed the LDW under the authority of the Washington Model Toxics Control Act (MTCA).

Superfund regulations require that an RI and a feasibility study (FS) be conducted for all listed sites. In 2000, the City of Seattle (City), the County, the Port, and Boeing, working collectively as the Lower Duwamish Waterway Group (LDWG), agreed in an Administrative Order on Consent (AOC) to conduct an RI/FS for the LDW. Oversight of the LDW RI/FS is being provided by EPA and Ecology. As part of the RI, existing data were synthesized and new data were analyzed to determine the nature and extent of contamination, assess sources of contamination to the LDW, and conduct baseline risk assessments. Information collected during the RI will be used to help identify areas requiring cleanup. The purpose of the FS is to analyze and compare alternatives or approaches to remediate those areas that need cleanup to reduce risks.

1.1 LDW RI Program Overview

To maximize the effective use of existing data, the LDW RI was completed in two phases (Windward 2000). For the Phase 1 RI (Windward 2003a), LDWG compiled and analyzed existing data to support an understanding of the nature and extent of chemical distributions in the sediments of the LDW and presented preliminary risk estimates based on those chemical distributions. Information evaluated during the Phase 1 RI was used to support the identification of candidate early remedial action sites also known as early action areas (EAAs). EAAs were selected using a risk-based approach to identify high-priority areas for which remedial actions could be initiated

prior to completion of the RI/FS process, as described in a technical memorandum submitted to EPA and Ecology in July 2003 (Windward 2003d). The Phase 1 RI also served as the basis for identifying additional data needed to complete the RI and the baseline ecological risk assessment (ERA) and human health risk assessment (HHRA) (Windward 2003e).

The principal objective of Phase 2 of the RI process was to collect and evaluate data necessary to support the RI/FS and future risk management decision making for the site. Phase 2 tasks included:

- Further characterize the physical and biological nature of the LDW area, including human use
- Further characterize the nature and extent of chemical distributions within the LDW area
- Summarize available source information and the status of source control activities
- Estimate risks to both human health and the environment under baseline conditions (prior to early action) and identify risk driver chemicals
- Determine risk-based threshold chemical concentrations in sediment for these risk drivers

To complete these tasks, additional studies and investigations were conducted as follows:

- Tissue studies
  - Juvenile chinook salmon tissue sampling and chemical analyses
  - Fish, clam, and crab tissue sampling and chemical analyses
  - Benthic invertebrate community characterization, including co-located tissue and sediment sampling and chemical analyses
- Clam, crab, and shrimp surveys
- Groundwater/porewater studies
  - Groundwater seep survey and chemical analyses
  - Sediment porewater sampling and chemical analyses
- Sediment studies
  - Sediment transport studies
  - Surface sediment sampling, chemical analyses, and toxicity testing
  - Subsurface sediment sampling and chemical analyses
- Other supporting studies

- Reconnaissance studies to assess site use by sandpiper
- Reconnaissance studies to assess potential human use of the LDW shoreline by means of various public access points
- Gastropod study to assess imposex
- Bathymetry study

The protocols and study design of these investigations were developed in consultation with EPA and Ecology. Results have been discussed in various documents and technical memoranda and reviewed by various public stakeholders; all results are available at www.ldwg.org/rifs_docs.htm. The results of these investigations, along with acceptable Phase 1 RI data and other new data of acceptable quality, have been used in the development of the baseline ERA and HHRA and are incorporated in this RI report.

1.2 DOCUMENT ORGANIZATION

This document is organized as follows:

- Section 1 – Program overview and organization of the RI
- Section 2 – History, physical characteristics, and environmental setting of the LDW and site use
- Section 3 – Sediment dynamics in the LDW
- Section 4 – Nature and extent of chemical contamination within the LDW study area
- Section 5 – Results of the Baseline ERA
- Section 6 – Results of the Baseline HHRA
- Section 7 – Background concentrations of risk driver chemicals
- Section 8 – Risk-based threshold concentrations (RBTCs)
- Section 9 – Potential pathways and sources of chemical contaminants and continuing source control efforts
- Section 10 – Source information for areas of interest
- Section 11 – Key findings of the RI
- Section 12 – References for all cited sources

These sections are supported by the following appendices:

- Appendix A – Baseline ecological risk assessment
- Appendix B – Baseline human health risk assessment
- Appendix C – Risk implications of Round 3 surface sediment data

- Appendix D – Food web model
- Appendix E – Surface sediment baseline dataset selection, data management, and summary data tables
- Appendix F – Subsurface core analysis
- Appendix G – Permitted and listed facilities within the LDW area
- Appendix H – LDW outfall locations
- Appendix I – Groundwater update
- Appendix J – Subsurface Sediment SMS Data

In addition, oversized maps are bound in a separate map folio. A CD containing the complete dataset used in this RI is attached to the back cover.

2 Environmental Setting

This section describes the LDW, its history, and its environmental setting, including physiography, physical characteristics, hydrogeology, ecological habitats, biological communities, and land use.

2.1 SITE DESCRIPTION

The Duwamish River originates at the confluence of the Green and Black Rivers near Tukwila, Washington, and flows northwest for approximately 19 km (12 mi), splitting at the southern end of Harbor Island to form the East and West Waterways, prior to discharging into Elliott Bay (Map 2-1). The LDW study area consists of the downstream portion of the Duwamish River, excluding the East and West Waterways.[7]

The LDW serves as a major shipping route for bulk and containerized cargo, and the shoreline along the majority of the LDW has been developed for industrial and commercial operations. Common shoreline features include constructed bulkheads, piers, wharves, sheet piling walls, buildings that extend over the water, and steeply sloped banks armored with riprap or other fill material (Weston 1999a). A portion of the LDW (i.e., the reach downstream of the Upper Turning Basin) is maintained as a federal navigation channel by USACE (Weston 1999a). Navigation elevations maintained within the LDW generally range from -30 ft mean lower low water (MLLW) from Harbor Island to the 1st Avenue S Bridge to -20 ft MLLW from the 1st Avenue S Bridge to Slip 4 and –15 ft MLLW from Slip 4 to the Upper Turning Basin (Weston 1999a). Intermittent, shallow benches exist in the nearshore, intertidal, and shallow subtidal zones of the LDW, outside the navigation channel. These benches are of various dimensions and elevations, with minimum elevations of less than 3 ft MLLW (Windward and QEA 2007). Shallow sills of sediment are present at the southern ends of the East and West Waterways, where the LDW splits around Harbor Island (Weston 1999b).

Intertidal habitats are dispersed in relatively small patches (i.e., generally less than 1 acre [ac] in size), with the exception of Kellogg Island, which represents the largest contiguous area of intertidal habitat remaining in the Duwamish River (Tanner 1991). Areas of low intertidal mudflats are present below upper bank riprap in the reaches upstream of the 1st Avenue S Bridge.

[7] The East and West Waterways are part of the Harbor Island Superfund site.

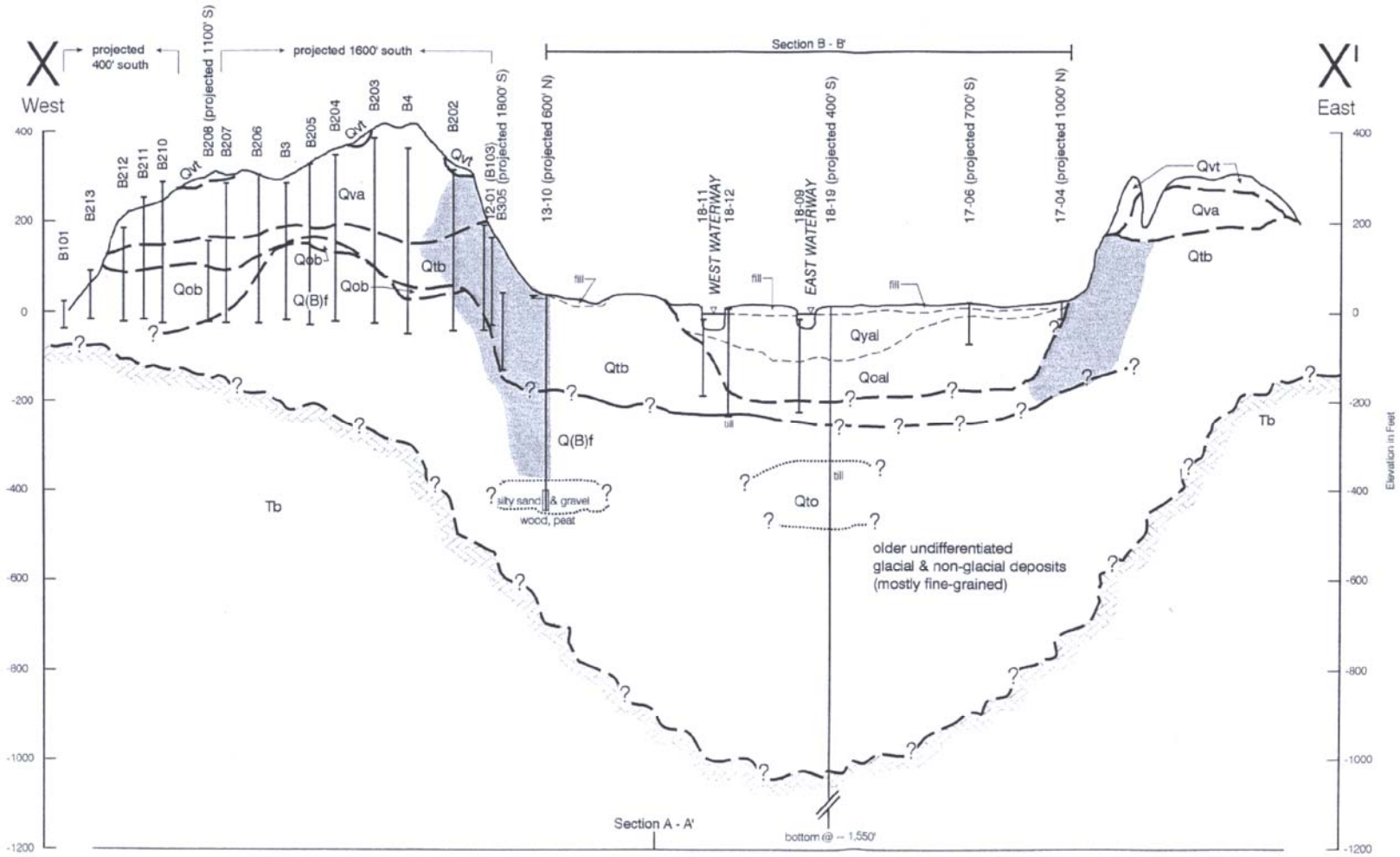
2.2 SITE HISTORY

Prior to the 20th century, the Duwamish River was fed by the Green, Black, and White Rivers, with a combined drainage area of approximately 4,250 km^2 (1,640 mi^2) (Blomberg et al. 1988). The river meandered through a valley of floodplains, freshwater wetlands, and tidal marshes before emptying into Elliott Bay (Figure 2-1). Flooding was a common occurrence in the river valley. Prior to the 1850s, the Duwamish River area was occupied by Native American tribal communities. The primary activities of these native peoples included fishing, hunting, and some limited farming. People of European origin began to settle in the area around the 1850s, clearing the Duwamish shorelines and draining wetlands for logging and agricultural purposes. In the early 1900s, continued issues with flooding in the area precipitated the installation of levees and dams, and the subsequent channelization of the river. As the quality of transportation and roads improved, commercial, industrial, and residential developments were constructed in the cleared lowland forest and farmland areas.

Throughout the 1900s, the watershed area and flow volumes to the Duwamish River were reduced by about 70% as a result of the diversion of the river's tributaries (Kerwin and Nelson 2000). In 1906, the White River was diverted to the Puyallup River to help control flooding (Harper-Owes 1983). In 1916, the Black River, which was fed by the Cedar River and Lake Washington, was reduced to a minor stream when the level of Lake Washington was lowered through the construction of the Lake Washington Ship Canal, and the Cedar River was subsequently diverted to Lake Washington (Harper-Owes 1983). Today, the Green River is the primary source of water for the Duwamish River.

To facilitate navigation and economic development, the LDW was straightened and dredged in many areas by Commercial Waterway District No. 1 (Washington Statute Revised Code of Washington [RCW] 91.04). Dredging between 1903 and 1905 created the East and West Waterways, and dredged material from the river was used to create Harbor Island (Weston 1993a). The river has been dredged and channelized from just upstream of the Upper Turning Basin to the southern tip of Harbor Island since about 1916. Upstream of the Upper Turning Basin, the river is contained by dikes.

Most of the upland areas adjacent to the LDW have been heavily industrialized for many decades. Historical and current commercial and industrial operations include cargo handling and storage, marine construction, boat manufacturing, marina operations, concrete manufacturing, paper and metals fabrication, food processing, and airplane parts manufacturing. Two mixed commercial and residential communities, Georgetown and South Park, are also located near the waterway (Wilma 2001a).



Source: Booth and Herman (1998)

Figure 2-1. Cross section of Greater Duwamish Valley stratigraphy at southern end of Harbor Island

The arrival of Seattle's first railroad in 1874 attracted numerous industrial activities. All major rail connections to Seattle from the south ran through Georgetown. Early tracks crossed the mudflats on pilings; and, in later years, railroad yards were built on fill dredged from the river. Industrial development increased as the mudflats were filled with soil from Seattle's former hills. In 1928, Seattle's first municipal airport, Boeing Field, was opened. Seven years later, Boeing opened its Plant 2 on the west side of Boeing Field (Wilma 2001a).

The neighborhood of South Park is located on the west bank of the LDW. Established as a farming community, South Park grew by more than 66 ac when the Duwamish was straightened, and the new land was claimed by farmers. In the 1920s, Boeing, located just north of South Park, built airplanes for the military. During World War II, South Park changed rapidly as industry from neighboring Georgetown attracted workers, creating a critical housing shortage in the farming community (Wilma 2001b).

2.3 PHYSIOGRAPHY

The LDW is located at the downstream end of the 1,466-km^2 (566-mi^2) Green/Duwamish River watershed (referred to as Water Resource Inventory Area [WRIA] 9), which includes portions of the Cities of Seattle, Tukwila, SeaTac, Renton, Kent, Federal Way, Auburn, Black Diamond, and Enumclaw, plus forested areas in unincorporated south-central King County (Map 2-2).

From the south end of Harbor Island to just south of the Upper Turning Basin, the LDW is approximately 8.8 km (5.5 mi) in length. The depth of the LDW varies from approximately -56 ft MLLW near the mouth to -10 ft MLLW near the head of the navigation channel (RETEC 2006). The average width of the LDW is 134 m (440 ft), widening downstream of the 1st Avenue S Bridge. The width of the navigation channel is 45.7 m (150 ft) (RETEC 2006).

2.4 METEOROLOGY

The climate of the Puget Sound area is characterized as "Pacific marine." The prevailing winds move moist air inland from the Pacific Ocean, moderating both winter and summer temperatures. According to the Western Regional Climate Center (WRCC), which has compiled meteorological data from 1931 to 2005 (WRCC 2007), monthly average winter temperatures at Sea-Tac International Airport (December through February) range from 2 to 8°C (36 to 47°F), and monthly average summer temperatures (June through August) range from 12 to 23°C (54 to 73°F). Over 50% of the annual precipitation in the region falls between the months of October and January. Annual monthly average precipitation at Sea-Tac International Airport ranges from 2 to 15 cm (0.8 to 6.0 in.), with an annual average precipitation of 97 cm (38 in.) (WRCC 2007). Winds in the Duwamish Valley are typically from the south and

southwest in the late fall, winter, and spring and from the north in the summer and early fall (PSCAA 2007c; Battelle et al. 2001). Annual monthly average wind speeds at Sea-Tac International Airport range from 11 to 14 km/hr (7 to 9 mi/hr) (WRCC 2007).

2.5 HYDROGEOLOGY

The hydrogeology of the Duwamish River basin, including the nature of subsurface materials and geologic units, is complex and influences both surface water and groundwater flow systems within the basin. Regional recharge and discharge patterns also play a significant role in defining the groundwater flow system of the Duwamish basin. The following subsections provide an overview of the relevant geologic features that influence the groundwater flow pathways to the LDW.

2.5.1 Geologic history

The Greater Duwamish Valley was formed by the carving action of glaciers that last advanced into this area from British Columbia approximately 15,000 years ago. When the ice sheets began to retreat approximately 5,700 years ago, the waters of Puget Sound extended up the Duwamish Valley as far south as Auburn, about 32 km (19 mi) upstream of the present mouth of the LDW at Elliott Bay. Around that same time, the Osceola Mudflow descended from Mount Rainier, depositing a massive layer of sediment into the then marine waters near present-day Auburn and Kent. The mudflow diverted the historical course of the White River, at that time a tributary of the Puyallup River, to the Green River (Booth and Herman 1998).

The alluvial fill within the Duwamish Valley deepened over time from the deposition of upstream fluvial sediments of the White, Green, and Black Rivers, advancing the mouth of the Duwamish River farther to the north. The fill included beds of fine silts and sands deposited as riverine and floodplain deposits, with coarser sands and gravels deposited near the water's edge. These sediments eventually buried the post-glacial form of the valley so that only a few outcroppings of bedrock remain exposed at the ground surface. As the river flooded and migrated back and forth across the floodplain, these sediments were redeposited by the river and continually intermixed with additional riverine and floodplain deposits (Booth and Herman 1998).

In the late 1800s and early 1900s, the river was extensively modified. Tide flats and floodplains were filled to straighten the river channel, resulting in the abandonment of almost 6 km (3.7 mi) of the original meandering river bed. Current side slips in the LDW are remnants of these old river meanders (Map 2-1).

The channel has been frequently dredged for navigational purposes, and the excavated material was used to fill the old channel areas and the lowlands to bring them above flood levels. Subsequent filling of the lowlands for continued development resulted in a surficial layer of fill over most of the lower Duwamish Valley. Although the sediment types encountered in the LDW are variable (either from

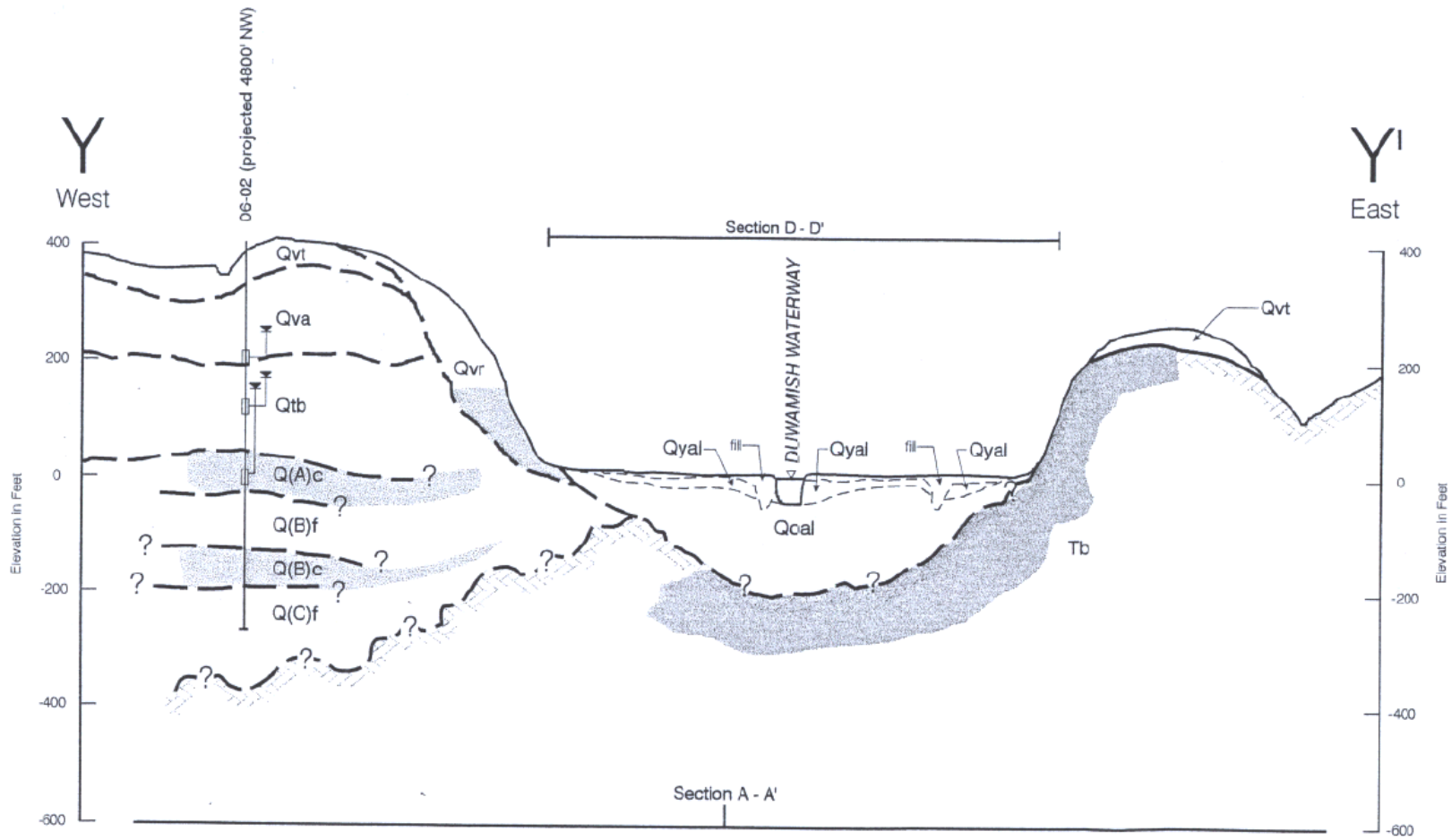
changing regional or local hydrodynamics or anthropogenic disturbances), basic sedimentary patterns of interbedded silts and sands are evident in the LDW. These patterns are controlled by Duwamish drainage basin morphology, precipitation, and flow rates. Both the regional and local aspects of these patterns are discussed in the following subsections.

2.5.2 Regional stratigraphy

The three principal geologic assemblages within the Greater Duwamish Valley that establish the hydrogeologic system, from oldest to youngest, are:

- Bedrock
- Glacial and non-glacial sedimentary units (glacially overridden and dense units that make up the plateaus to the east and west of the Duwamish Valley)
- Undifferentiated quaternary alluvial deposits (principal aquifer and groundwater pathway for the Duwamish basin)

Map 2-3 presents the surficial geology of the Greater Duwamish Valley basin and provides definitions of the surficial geologic units. Figures 2-1 and 2-2 are schematic cross sections of the regional stratigraphy at two locations in the Duwamish Valley as identified on Map 2-3 (cross sections X-X′ and Y-Y′, respectively). Figure 2-1 is a cross section of the LDW near Harbor Island, and Figure 2-2 presents the regional stratigraphy at approximately RM 3.0. Table 2-1 provides subsurface geologic unit nomenclature in Figures 2-1 and 2-2 and discussed in this text (Booth and Herman 1998; US Geological Survey [USGS] 2005). The following subsections describe the assemblages and their influence on groundwater flow patterns in the Greater Duwamish Valley.



Source: Booth and Herman (1998)

Figure 2-2. Cross section of Greater Duwamish Valley stratigraphy near LDW RM 3.0

Table 2-1. Definitions of subsurface geologic nomenclature

GEOLOGIC NOMENCLATURE	DEFINITION
Q(A)c	coarse-grained deposits, presumably immediately underlying Olympia-age (approximately 17,000 to 40,000 years ago) deposits (correlation is speculative)
Q(B)c	coarse-grained deposits, immediately underlying unit Q(B)f
Q(B)f	fine-grained deposits, immediately underlying the Q(A)c unit; speculative correlation would imply an age of at least 100,000 years old
Q(C)f	older undifferentiated, unconsolidated fine-grained deposits
Qoal	older alluvium
Qob	Olympia beds
Qtb	transitional beds, formed as the ice first advanced and laminated silt and clay were deposited; unit is a regionally significant aquitard, allowing very little movement of groundwater
Qva	coarser outwash, deposited by streams derived from the advancing ice sheet, initiated by the Vashon Advance Outwash (approximately 15,000 years ago); primary shallow aquifer in the upland areas throughout the region
Qvt	lodgment till, deposited by the melt-out of debris at the base of the glacier; composed of heterogeneous, compact sediment (glacial deposit)
Qyal	younger alluvium
Tb	the Blakely Formation, which includes marine sandstone, some conglomerate, and minor amounts of siltstone (bedrock)

Source: Booth and Herman (1998) and USGS (2005)

2.5.2.1 Bedrock

Bedrock in the Greater Duwamish Valley provides the lower boundary of the aquifer system and limits groundwater flow in the basin. At the northern end of the Duwamish Valley, the elevation of the bedrock unit ranges from roughly 60 m (200 ft) to over 500 m (1,640 ft) below ground surface (bgs). Exposed bedrock in the eastern and southern areas of the Greater Duwamish Valley (see Figures 2-1 and 2-2) is predominantly marine and continental sedimentary rocks intermixed with isolated areas of igneous rock deposited during the Tertiary period. Sedimentary rock units within the Greater Duwamish Valley are not an important source of groundwater because the predominantly cemented, fine-grained nature of the material precludes rapid groundwater movement. However, igneous rock layers are extensive in the area and can store and move water much more readily (Booth and Herman 1998).

2.5.2.2 Glacial and non-glacial sedimentary deposits

The glacial and non-glacial sedimentary units within the Duwamish basin are complex sequences of interbedded and unconsolidated deposits. In areas where bedrock occurs at significant depths below the river valley, these glacial sedimentary deposits serve as the lower boundary of the alluvial deposits in the Greater Duwamish Valley. The upland plateau areas to the east and west of the valley are formed predominantly of these glacially deposited sedimentary units (see Figures 2-1 and 2-2).

Little information is available on the glacially overridden sedimentary units within the LDW study area. These overridden deposits are mainly fine-grained materials; their maximum depth is unknown (Booth and Herman 1998). Although these deposits provide a geologic boundary to the alluvial deposits, they also provide a potential hydraulic pathway for the flow of upland groundwater to the Duwamish Valley alluvial sediments.

Thick sequences or silt beds (transitional beds) could potentially limit the upland inflow of groundwater where these deposits occur. The presence of saline water in the deeper alluvial sediments outside of current tidal influence areas suggests that there is little influx of fresh water into the original marine delta deposits. The lack of fresh groundwater in these deep alluvial sediments may indicate that the inflow of upland groundwater in this layer is limited (Figures 2-1 and 2-2).

2.5.2.3 Duwamish Valley alluvial deposits

The near-surface alluvial deposits in the Duwamish Valley extend to depths of roughly 60 m (200 ft) bgs within a trough bounded and underlain by either the bedrock unit or the dense glacial deposits and non-glacial sedimentary deposits. The geologic history of this valley suggests that the alluvial deposit sequences include estuarine deposits, typically fine sands and silts (often including shell fragments), which progress upward into more complex, interbedded river-dominated sequences of sand, silt, and gravel. These layers of alluvial deposits delineate the areas of advancing river delta sedimentation that increase in thickness from south to north (Booth and Herman 1998). Channel fill materials, which were deposited during settlement and when the river was straightened, are included in this category because fill material has properties similar to alluvium, as described below.

On a regional scale, the fill and alluvial deposits can be separated into various generalized units as illustrated in Figures 2-1 and 2-2. Geologic unit definitions and nomenclature are provided in Table 2-1. These regional units (from younger to older) include:

- **Fill –** The lower Duwamish River was straightened in the early 1900s into a navigation channel, using fill materials derived mostly from local sources. Much of the fill placed in the old river channels was dredged material, which is similar in hydraulic conductivity to the native younger alluvium. In the vicinity of the LDW, various depths of fill are present, ranging in thickness from 1 to 6 m (3 to 20 ft). Locally, the shallowest aquifer occurs within the lower portion of this fill material, especially in the northern sections of the LDW where upland was created during the last century. The depth of fill varied greatly and generally consisted of sand and silty sand in the saturated zone.
- **Younger alluvium (Qyal) –** This younger alluvium has been identified at the bottom of filled Duwamish River channels. Younger alluvium deposits are

composed predominantly of sand, silt, gravel, and cobbles deposited by streams and running water (USGS 2005). In the central Duwamish Valley, roughly between RM 2.0 and RM 5.2, younger alluvial deposits are of relatively constant thickness and depth, based generally within 1.5 to 3 m (5 to 10 ft) of present-day mean sea level. These deposits are thicker in the south than in the north within the LDW; the thickest deposits on the northern end are estimated to occur at a depth of roughly 30 m (100 ft) bgs. The younger alluvium consists of sands, silts, and clays and includes abundant natural organic material. This layer is often distinguished from the overlying fill by abundant fibrous organic material typical of tide marsh deposits.

- **Older alluvium (Qoal**) – The older alluvium is characterized by estuarine deposits, often including shells at its lower depths, and composed of silts and clays with sandy interbeds (USGS 2005). This unit is commonly identified at depth between 15 and 30 m (50 and 100 ft) bgs in the central Duwamish Valley, increasing in depth toward the mouth of the LDW to between 45 and 60 m (150 and 200 ft) bgs. This unit has been best characterized at the Boeing properties in the central valley, where the older alluvium becomes finer-grained with increasing depth. In this area, the upper two-thirds of the older alluvium typically consist of sand and silty sand, and the lower third consists of sandy silt. The older alluvium also becomes significantly finer toward the north, with the sand almost completely absent near the mouth of the LDW. At this location, the layer is composed almost entirely of silt and clay, representing the farthest extent of the delta deposits into the marine waters and displaying the finest-grained material of the Duwamish Valley alluvial sequence.

2.5.3 LDW sediment stratigraphy

This section provides an overview of the stratigraphy of sediments identified in the LDW. Stratigraphy is the identification of individual sediment layers or groups of layers differentiated above or below by unity of color, texture, or gross appearance (Krumbein and Sloss 1963). The core logs provided in the subsurface sediment data report (Windward and RETEC 2007) characterize the stratigraphy of the units identified in the cores collected from within the LDW during the 2006 study.

The core logs provide the following stratigraphic information:

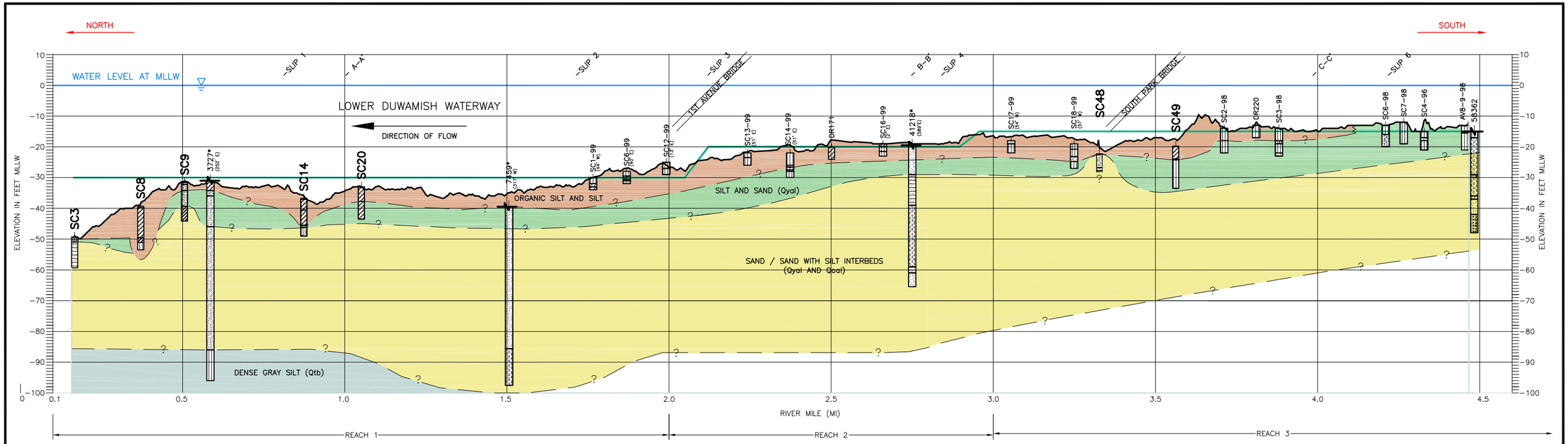
- A stratigraphic profile based on recovered depths, using description categories consistent with American Society for Testing and Materials (ASTM) nomenclature
- Detailed LDW sediment descriptions and comments
- A combined lithologic and stratigraphic profile based on *in situ* depths

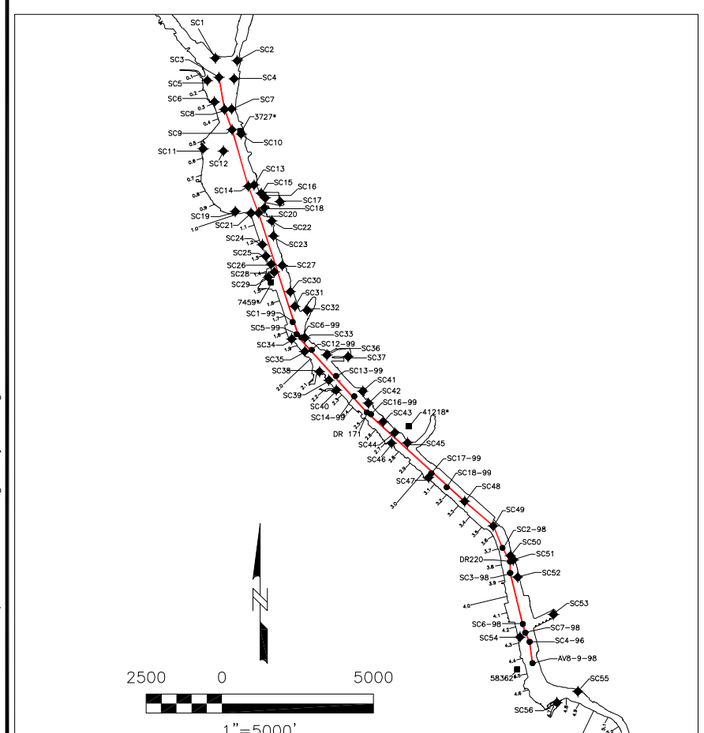
Based on field observations, the LDW sediments were grouped into three stratigraphic units. These units were delineated primarily based on density, color, sediment type, texture, and marker bed horizons; although other information, including the presence of anthropogenic matter, depth in the waterway, and available information on historical dredging events, was also considered.

Vertical stratigraphic profiles were created for the entire LDW by comparing and evaluating sediment stratigraphy information from the LDW subsurface sampling event (Windward and RETEC 2007) and from other regional cores.

The detailed stratigraphic information available from the cores collected in 2006, along with historical data from adjacent upland sites and other in-water boring data, was used to differentiate sedimentary units in the LDW from the larger undifferentiated Quaternary Alluvium unit of the Greater Duwamish Valley as described in Section 2.5.2.3.

To illustrate the distribution of these alluvial units in the LDW, a longitudinal cross section along the navigation channel centerline and three stratigraphic cross sections (A-A′, B-B′, and C-C′) at RM 1.0, RM 2.75, and RM 3.9 were prepared (Figures 2-3 through 2-6, respectively). The cross sections show the LDW at different river widths and together illustrate the change in sediment depositional patterns associated with the change in river morphology.alluvium (Qyal) unit, but may also straddle the older alluvium (Qoal) unit in upstream areas.





800 0 1600
HORIZONTAL SCALE IN FEET
VERTICAL EXAGGERATION = 50:1

LEGEND

DR-171 ● Historical Subsurface Core Locations
- SC4-96 = PSSDA96 study
- SC2-98 = PSDDA98 study
- SC6-99 = PSDDA99 study
- DR171 = EPASI study

3727* ■ Upland Cores
- 7459 = Hart Crowser 1979 study
- 41218 = Yonemitsu Geological Services 1979 study
- 3727 = Dames and Moore 1988 study
- 58362 = Seattle Public Utilities (CPT-D77_01) 1985 study

SC48 ◆ 2006 Subsurface Core Locations

Authorized Navigation Depth (ft MLLW)

Mid-channel mudline elevation (based on 2003 bathymetry survey, ft MLLW LDW)

Water level at MLLW (ft)

KEY

SC14-99 (5.1 ft)
- Core Name
- (Distance from section to Core)
- Top of Core
- Observed Contacts
- Bottom of Core

STRATIGRAPHY**

- RECENT
- UPPER ALLUVIUM/TRANSITION
- LOWER ALLUVIUM
- DENSE POST-GLACIAL AND GLACIAL UNIT

Qtb = TRANSITIONAL SILTS
Qyal = YOUNGER ALLUVIUM
Qoal = OLDER ALLUVIUM

LITHOLOGY

- Clay/Silt
- Organic silt
- Silt
- Sand with silt
- Sand with silt interbeds
- Sand
- Gravel, sandy gravel

NOTES

1) MLLW – mean lower low water

2) 2006 Phase 2 cores (SC3, SC8, SC9, SC14, SC20, SC48, SC49) are labeled with bold font in cross section.

3) AV8-9-98 is an average of Core 8 and 9 from PSDDA (1998).

4) Upland cores from GeoMap NW, Pacific Northwest Center for Geologic Mapping Studies. http://geomapnw.ess.Washington.edu/index.php

* Upland cores projected into Navigation Channel. Lower portions of upland cores used to bound deeper sediment units.

** Stratigraphy terminology is used in subsurface data report (Windward 2007) and sediment transport analysis report (Windward, QEA 2007).

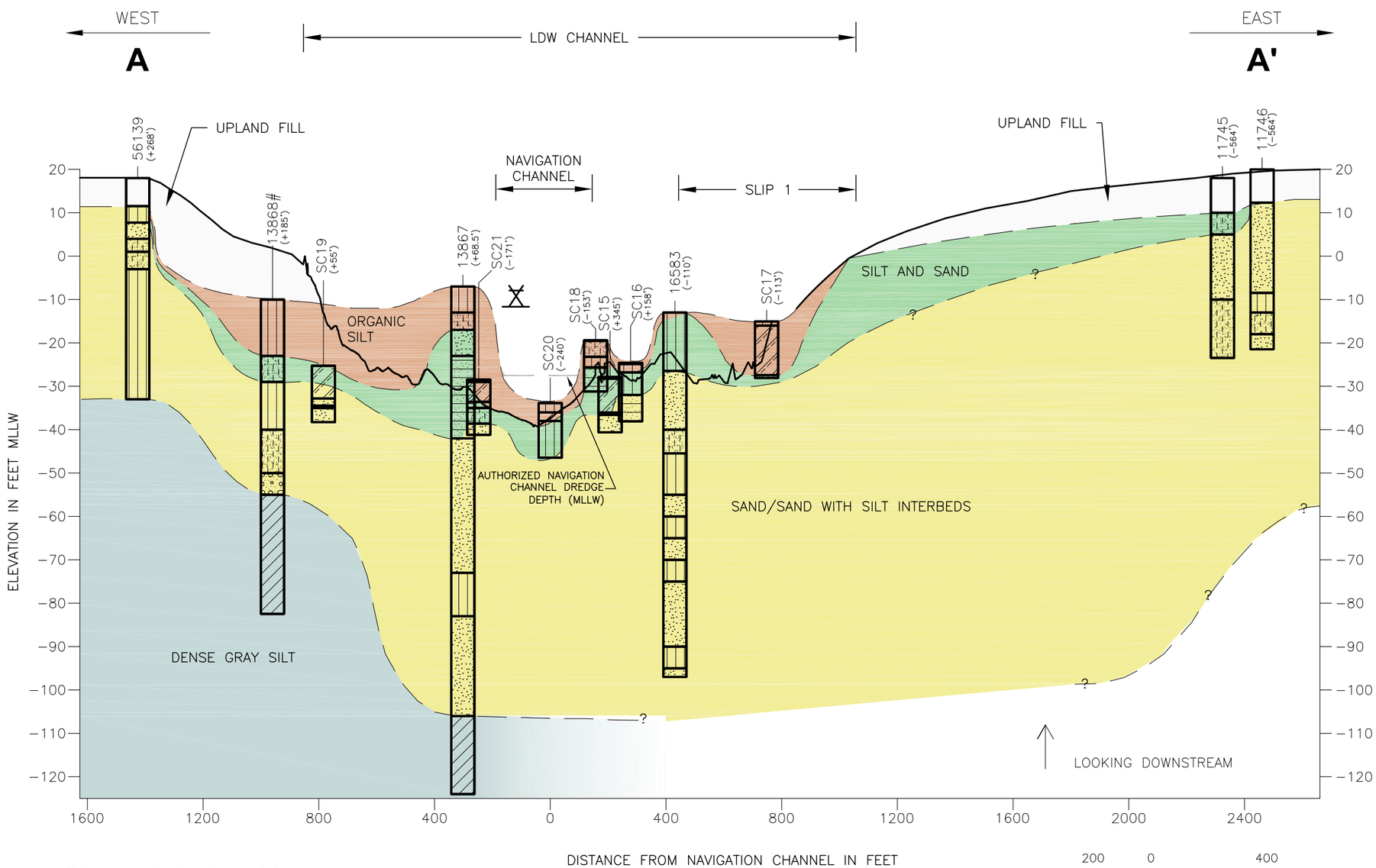
These cross sections are unique and do not correspond with those identified on Map 2-3 or Figures 2-1 and 2-2 for the Greater Duwamish Valley.

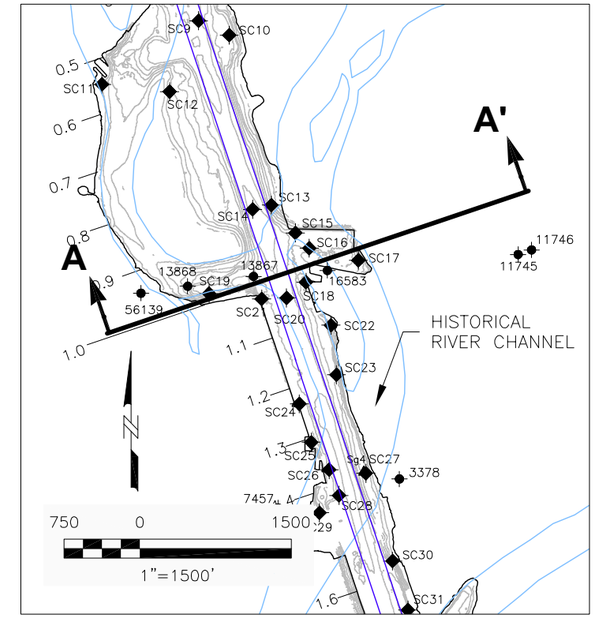
ENSR | AECOM

Merged with ENSR in 2007
RETEC

LOWER DUWAMISH WATERWAY PHASE 2 RI REPORT PORS5-18220-514			LONGITUDINAL CROSS SECTION
DATE: 07/26/07	DRWN: E.M./SEA		FIGURE 2-3

File: H:\18220\X-SECTS 3-13-07\18220-LONG.dwg Layout: fig 2-1 User: emarshall Plotted: Jul 26, 2007 – 8:31am Xref's:





LITHOLOGY

- Fill
- Clay/silt
- Organic silt
- Silt
- Sand with silt
- Sand with silt interbeds
- Sand
- Gravel, sandy gravel

STRATIGRAPHY*

- UPLAND FILL
- RECENT SEDIMENT
- UPPER ALLUVIUM/TRANSITION
- LOWER ALLUVIUM
- DENSE POST-GLACIAL AND GLACIAL UNIT

LEGEND

INTERTIDAL OR SUBTIDAL CORES:

- SC52 – RI SUBSURFACE CORE (2006)

UPLAND BORINGS:

- 13867, 13868 – TWELKER, 1970
- 11745, 11746 – TERRA ASSOC., 1999
- 56139 – GEORESOURCE INC., 1985
- 16583 – GEOENGINEERS, 1995

Approximate mudline surface elevation in ft MLLW (2003 Bathymetry)

Approximate mudline surface elevation (1900 ACOE soundings)

KEY

SC19 (214') – Core Name (Dist. from section to core in feet)

- Top of Core
- Observed Contacts
- Bottom of Core

This cross section is unique and does not correspond with those identified on Map 2-3 or Figures 2-1 and 2-2 for the Greater Duwamish Valley.

NOTES:

1) Boring and core logs shown as in-situ depth. Actual contacts may vary.

2) Upland cores from GeoMap NW, Pacific Northwest Center for Geologic Mapping Studies. http://geomapnw.ess.Washington.edu/index.php

* Stratigraphy terminology is used in subsurface data report (Windward 2007) and sediment transport analysis report (Windward, QEA 2007).

In water core (13868) projected onto upland portion of cross section. Top of core at −10ft MLLW. Mudline elevation line extended to top of upland core to west (56139).

200 0 400
SCALE IN FEET
VERTICAL EXAGGERATION = 15:1

ENSR | AECOM

Merged with ENSR in 2007 RETEC

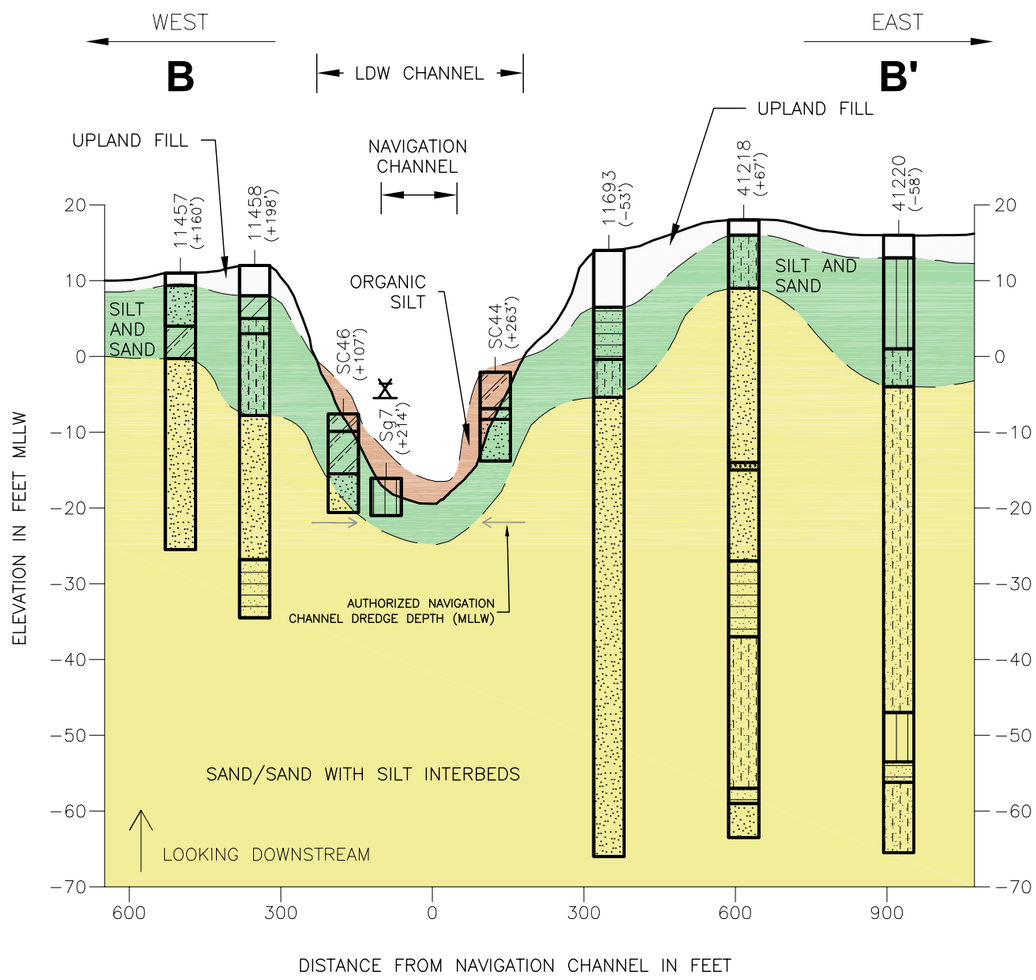
LOWER DUWAMISH WATERWAY
PHASE 2 RI
PORS5-18220-514

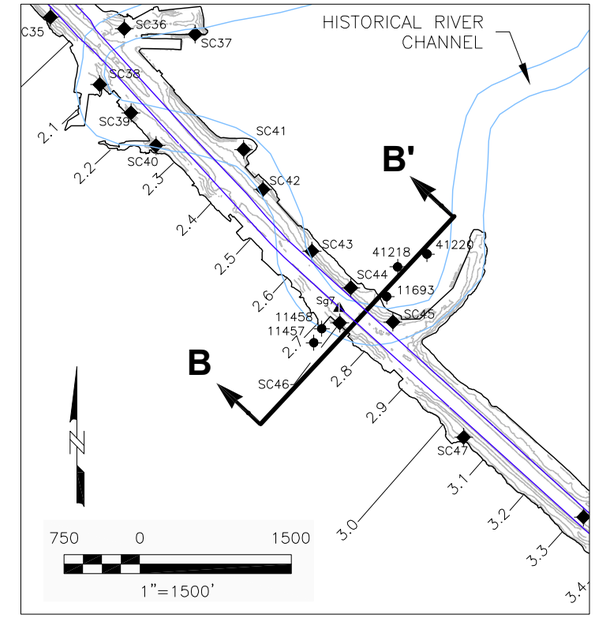
DATE: 8/2/07 | DRWN: E.M./SEA

GENERALIZED CROSS SECTION A-A'
RIVER MILE 1.0

FIGURE 2-4

File: H:\18220\xsect 8-2-07\CROSS-SEC_A-A'.dwg Layout: fig 2-2 User: emarshall Plotted: Aug 03, 2007 – 8:04am Xref's:





This cross section is unique and does not correspond with those identified on Map 2-3 or Figures 2-1 and 2-2 for the Greater Duwamish Valley.

NOTES:

1) Boring and core logs shown as in-situ depth. Actual contacts may vary.

2) Upland cores from GeoMap NW, Pacific Northwest Center for Geologic Mapping Studies. http://geomapnw.ess.Washington.edu/index.php

* Stratigraphy terminology is used in subsurface data report (Windward 2007) and sediment transport analysis report (Windward, QEA 2007).

150 0 300

SCALE IN FEET

VERTICAL EXAGGERATION = 15:1

LITHOLOGY

- Fill
- Clay/silt
- Organic silt
- Silt
- Sand with silt
- Sand with silt interbeds
- Sand
- Gravel, sandy gravel

STRATIGRAPHY*

- UPLAND FILL
- RECENT SEDIMENT
- UPPER ALLUVIUM/TRANSITION
- LOWER ALLUVIUM

LEGEND

INTERTIDAL OR SUBTIDAL CORES:

- SC44, SC46 – RI SUBSURFACE CORE (2006)
- Sg7 – GEOCHRONOLOGY CORE (2004)

UPLAND BORINGS:

- 11457, 11458 – CONVERSE CONSULTANTS 1988
- 11693 – GOLDER ASSOCIATES, 1997
- 41218, 41220 – YONEMITSU, 1979

Approximate mudline surface elevation in ft MLLW (2003 Bathymetry)

Approximate mudline surface elevation (1900 ACOE soundings)

KEY

SC19 (214') – Core Name (Dist. from section to core in feet)

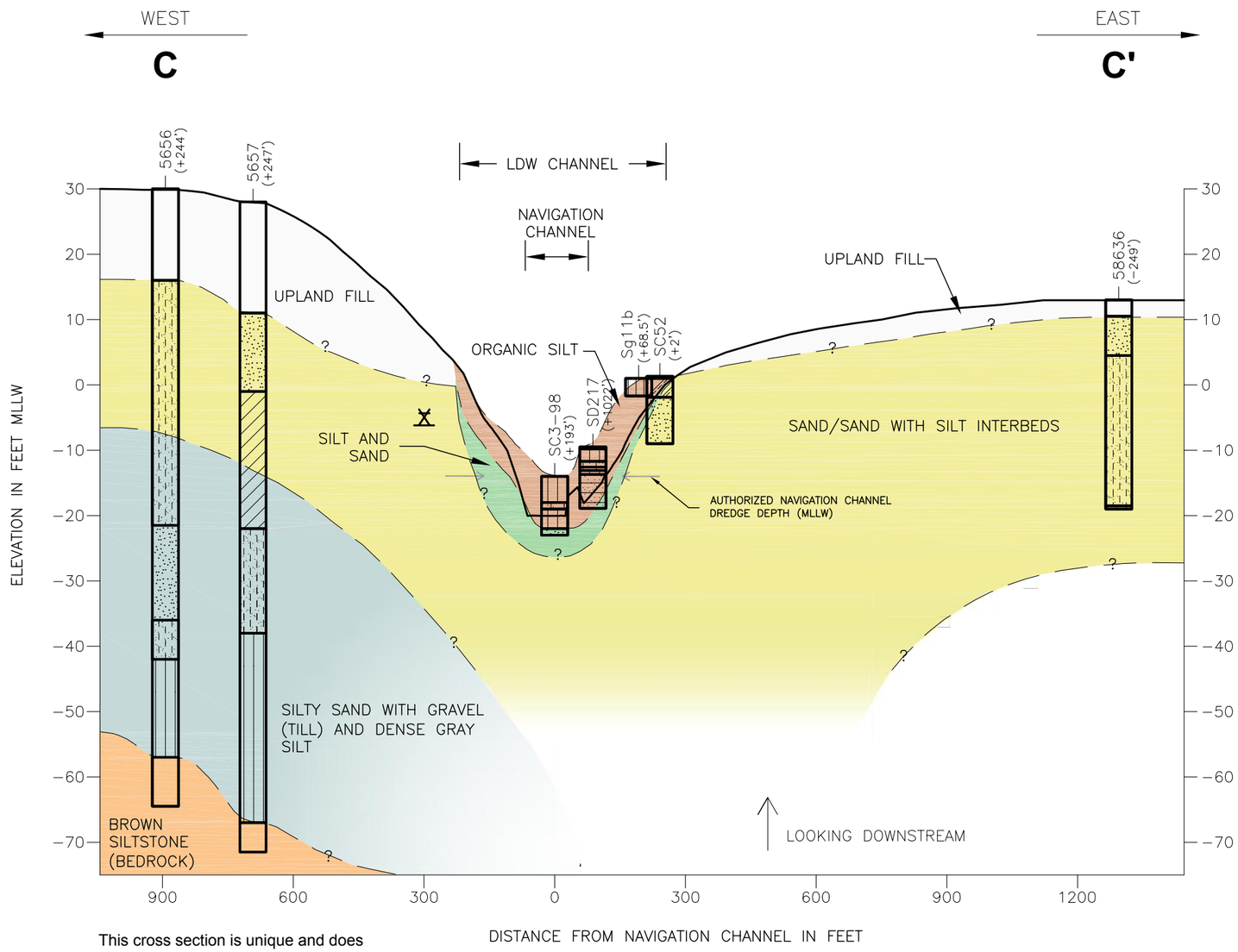
- Top of Core
- Observed Contacts
- Bottom of Core

ENSR | AECOM

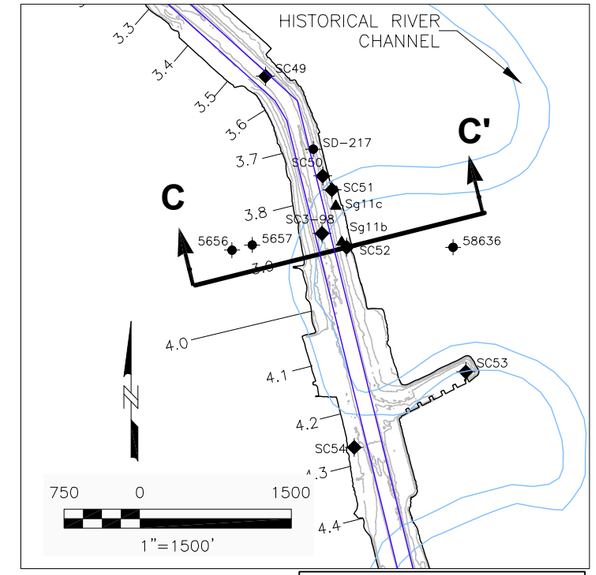
Merged with ENSR in 2007 RETEC

LOWER DUWAMISH WATERWAY PHASE 2 RI PORS5-18220-514		GENERALIZED CROSS SECTION B-B' RIVER MILE 2.75
DATE: 8/2/07	DRWN: E.M./SEA	FIGURE 2-5

File: H:\18220\xsect 8-2-07\CROSS-SEC_B-B'.dwg Layout: fig 2-3 User: emarshall Plotted: Aug 02, 2007 - 10:37am Xref's:



This cross section is unique and does not correspond with those identified on Map 2-3 or Figures 2-1 and 2-2 for the Greater Duwamish Valley.



LITHOLOGY

- Fill
- Clay/silt
- Organic silt
- Silt
- Sand with silt
- Sand with silt interbeds
- Sand
- Gravel, sandy gravel

STRATIGRAPHY*

- UPLAND FILL
- RECENT SEDIMENT
- UPPER ALLUVIUM/TRANSITION
- LOWER ALLUVIUM
- DENSE POST-GLACIAL AND GLACIAL UNIT
- BEDROCK

LEGEND

INTERTIDAL OR SUBTIDAL CORES:

- SC52 – RI SUBSURFACE CORE (2006)
- Sg11b – GEOCHRONOLOGY CORE (2004)

UPLAND BORINGS:

- 5656, 5657 – DAMES & MOORE, 1984
- 58636 – GEOENGINEERS, 1988

Approximate mudline surface elevation in ft MLLW (2003 Bathymetry)

Approximate mudline surface elevation (1900 ACOE soundings)

KEY

SC19 (214')
Core Name –(Dist. from section to core in feet)
–Top of Core
–Observed Contacts
–Bottom of Core

NOTES:

1) Boring and core logs shown as in-situ depth. Actual contacts may vary.

2) Upland cores from GeoMap NW, Pacific Northwest Center for Geologic Mapping Studies. http://geomapnw.ess.Washington.edu/index.php

* Stratigraphy terminology is used in subsurface data report (Windward 2007) and sediment transport analysis report (Windward, QEA 2007).

150 0 300
SCALE IN FEET
VERTICAL EXAGGERATION = 15:1

ENSR | AECOM

Merged with ENSR in 2007 RETEC

LOWER DUWAMISH WATERWAY PHASE 2 RI PORS5-18220-514			GENERALIZED CROSS SECTION C-C' RIVER MILE 3.9
DATE: 07/26/07	DRWN: E.M./SEA		FIGURE 2-6

File: H:\18220\xsect 8-2-07\CROSS-SEC_C-C'.dwg Layout: fig 2-4 User: emarshall Plotted: Aug 02, 2007 – 1:05pm Xref's:

Figure 2-7 provides a pictorial summary of sediment lithology observed in the LDW over the past 120 yrs relative to physical events and urban development.

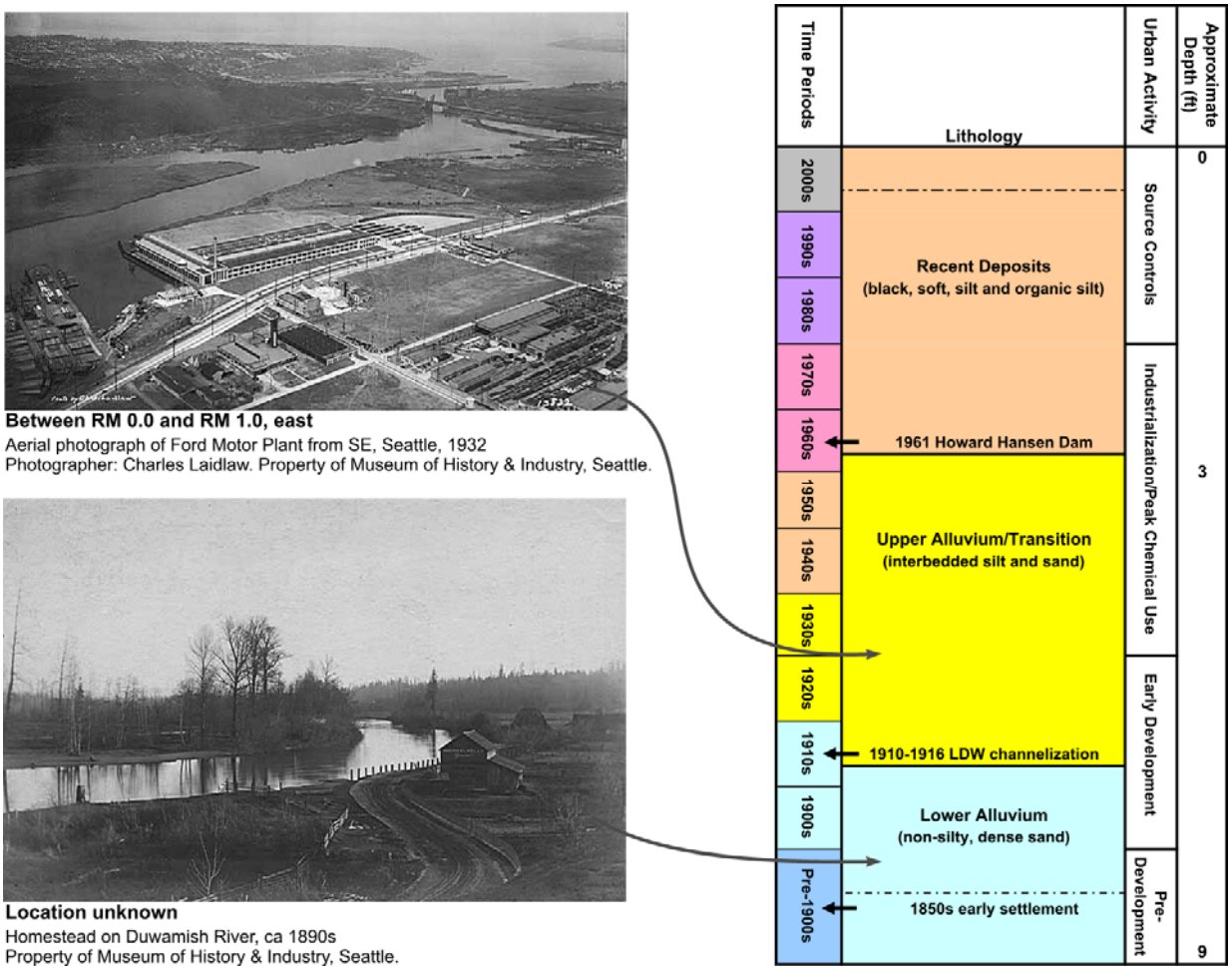


Figure courtesy of RETEC|ENSR

Figure 2-7. Summary of sediment lithology in the LDW relative to physical events and human development

As shown in the cross sections (Figures 2-3 to 2-6), the channel sediment typically consists of softer, recent sediments overlying transitional or upper alluvial sediments, which in turn overlie lower (older) alluvium. The lower alluvial unit represents deposition prior to the industrialization of the area.

In addition to the primary stratigraphic units presented in the cross sections, two other upland layers, fill and glacially–overridden sedimentary units, also have been identified in the study area. LDW sediments were classified as "other" when the sediment matrix was composed of over 50% fill, woody debris, or anthropogenic materials.

2.5.4 Sediment lithology in the LDW

During the subsurface sediment investigations conducted for the RI, the lithology of the sediment cores collected was recorded in the field according to nomenclature established in ASTM D-2488, and a core log field key was used to ensure nomenclature consistency (Windward and RETEC 2007). Lithology is defined as the physical characteristics of the material described in terms of the dominant soil type (e.g., sand or silt), grain size percentages, texture (e.g., fine or medium grain), sorting (e.g., well sorted or poorly sorted), shape, and the color and mineralogy of particles (Krumbein and Sloss 1963).

Geotechnical parameter tests were also performed on a subset of subsurface sediment cores collected in 2006 within the upper 4 ft of the cores (Windward and RETEC 2007). These parameters included moisture content, specific gravity, Atterberg limits (i.e., liquid limit, plastic limit, plastic index), bulk density (dry and wet), and porosity.

Specific gravity, porosity, and wet density sample results did not vary notably with depth, indicating that sediment texture in the upper 4 ft was relatively uniform. The mean specific gravity of all subsurface sediment samples across similar core intervals ranged from 2.64 grams per cubic centimeter (g/cc) to 2.66 g/cc. The mean sediment porosity ranged from 0.59 standard unit (SU) to 0.64 SU, and the mean wet bulk density ranged from 102 pounds per cubic foot (pcf) to 104.4 pcf.

Other geotechnical properties varied with depth. The mean moisture content of all samples was 75% dry weight (dw) at the surface, decreasing to 63% dw below the 2-ft interval, consistent with the decrease in water content with depth as noted on the core logs. The mean dry bulk density across similar core intervals increased with depth from 60.4 to 67.2 pcf.

Atterberg limits testing was performed on fine-grained silt/clay sediments to determine the relationship between moisture content and LDW sediment type. The testing interprets the liquid limit of the sample (when sediment changes from a viscous fluid to a plastic state) and the plastic limit (where sediment behaves as a solid). The plasticity index is the difference between the liquid limit and the plastic limit. The mean liquid limit of all subsurface sediment samples ranged from 61.2% dw to 70.7% dw, and the mean plastic limit ranged from 35.0% dw to 39.3% dw. Sediment samples exhibited medium to high plasticity, with the mean plasticity index varying from 26.2% dw to 32.5% dw (Windward and RETEC 2007).

2.5.5 Sediment physical properties

Sediment composition varies greatly throughout the LDW, ranging from sands to mud (fine-grained silt and clays), depending on the source of the sediments and the local current velocity. The sediment typically consists of slightly sandy to sandy silt with varying amounts of organic material. Finer-grained sediments are typically located in remnant mudflats, along channel sideslopes, and within portions of the navigation

channel. Sediments in the navigation channel near the Upper Turning Basin are predominantly sands, whereas sediments toward the mouth of the river are predominantly fine-grained silts.

Overall, the clay-silt content of surface sediment in the LDW is highly variable, with an average value of 53% (Map 2-4). Sediment in the navigation channel has a higher clay-silt content than other sediment, with an average of 62% clay-silt content (ranging from 40 to 80%). Clay-silt content is more variable outside of the navigation channel, generally ranging from about 15 to 80%, with an average value of 50%.

Three of the five slips along the LDW have high clay-silt content values relative to the overall LDW average. Slips 1, 3, and 6 have average clay-silt values of 79, 71, and 87%, respectively. The clay-silt contents of Slips 2 and 4 are lower, with average values of 41% and 57%, respectively. The area above RM 5.0 has a relatively low average clay-silt content (approximately 11.5%). Coarser sediments are also present in nearshore areas adjacent to combined sewer overflow (CSO) and storm drain (SD) discharges (Weston 1999a).

Total organic carbon (TOC) content in surface sediment is similar throughout the LDW, with an average value of 1.9% and a range of 0.03 to 12% (Map 2-5). The TOC content in the non-channel areas (0.032 to 9.2%) is more variable than the TOC content in the navigation channel (0.31 to 5.3%). No consistent longitudinal trend is evident within the LDW, with the exception of a lower average TOC content of 0.84% upstream of RM 5.0. The TOC content in Slips 1, 3, 4, and 6 is higher than the LDW-wide average, with average TOC contents of 2.3, 2.2, 2.6, and 2.7%, respectively. In Slip 2, the average TOC content (1.5%) is lower than the LDW-wide average.

2.5.6 Groundwater systems, transport, and processes

The previous sections characterized the geologic and stratigraphic composition of the soils and sediments in the Greater Duwamish Valley and the LDW study area. These geological features help define the framework for the aquifer systems and influence the flow and transport of groundwater within the Duwamish basin.

Section 2.5.6.1 provides information on the regional and local groundwater systems within the Duwamish basin, including the spatial flow patterns and characteristics of groundwater within the LDW study area. The following sections then discuss the various physical and biochemical processes that can affect the transport of groundwater as well as influence its chemical content. Section 9.4.6 and Appendix I provide information on the groundwater characteristics and investigation efforts ongoing in the LDW.

2.5.6.1 Valley aquifer and groundwater flow system

In general, groundwater within the Duwamish Valley is unconfined at depths starting at 3 m (10 ft) bgs (Booth and Herman 1998). Regionally, the valley alluvium is a single,

large aquifer system. Locally, the aquifer's flow characteristics vary depending on the nature of the materials making up the local alluvium, the proximity to the river, and local tidal fluctuations. The maximum depth of the alluvial aquifer in the study area extends to roughly 30 m bgs, although deeper areas are present in the central portion of the Duwamish Valley. The thickness of the alluvial aquifer lessens to the north of the valley, where finer-grained silts and clays dominate. The aquifer's thickness also lessens to between 10 and 12 m (30 and 40 ft) bgs toward the eastern and western margins of the valley (Booth and Herman 1998).

The elevation gradient between the glacially overridden deposits in the upland deposits and LDW sediments creates a regional flow system with significant hydraulic potential for the transport of groundwater from the upland areas to the LDW. Groundwater elevations in the uplands range between 30 and 60 m (100 and 200 ft), while elevations in the valley aquifer range between 3 and 6 m (10 and 20 ft) relative to mean sea level. The glacial and non-glacial sedimentary units discussed in the Duwamish groundwater pathways study (Booth and Herman 1998) indicated that upland flow toward the LDW occurs primarily along the western valley within the sandier glacially deposited sedimentary unit. Site-specific studies in the LDW basin often subdivide the aquifer into shallow, intermediate, and deep systems. The shallow aquifer is generally located within the fill and/or younger (upper) alluvium (Qyal), and the deep aquifer is generally located within the older (lower) alluvium (Qoal). The aquifer systems are separated by silt layers. These silt layers are rarely continuous throughout the valley and constrain local groundwater flow. The discontinuous nature of these silt layers allows for a hydraulic connection between the shallower and deeper alluvium layers.

High tides entering the LDW can cause temporary groundwater flow reversals; however, available water level data show that the overall net flow direction is toward the LDW. Areas of tidal influence were generally within 100 to 150 m (300 to 500 ft) of the LDW shoreline (Booth and Herman 1998).

Appendix G of the Phase 1 RI (Windward 2003a) included a detailed groundwater pathways assessment. Map 2-1 of that appendix presented groundwater flow patterns obtained from relevant site-specific investigations of the LDW study area. The groundwater elevation data presented in the appendix was for the shallowest aquifer system, where available data were the most reliable. Within the greater LDW basin, groundwater flows in various directions. Information on vertical gradients provided in the Duwamish groundwater pathways study (Booth and Herman 1998) indicated that downward flow gradients occur predominantly in the shallower alluvium layer, which extends to roughly 10 to 15 m (30 to 50 ft) bgs. Upward groundwater flow gradients are typically present below these depths (Booth and Herman 1998).

Upward groundwater flow rates are at their lowest (between 0.002 and 0.07 m/day) beneath the eastern side of the Duwamish Valley (roughly between RM 2.0 and

RM 5.2), where inflow from the adjacent uplands is limited by layers of bedrock and silt deposits. In general, higher flow rates, ranging from 0.02 to 0.3 m/day, occur in the upland areas to the west of the LDW. The downward gradient within the shallower alluvium can cause a downward migration of dissolved constituents to depths where upward flow gradients limit further downward movement, generally forcing the shallower flow toward the LDW.

As a result of extensive areas of impervious surfaces, areas in the eastern portion of the Duwamish Valley have limited freshwater recharge, and the presence of brackish groundwater suggests zones of stagnant groundwater (Booth and Herman 1998). The presence of brackish and saline water can affect groundwater flow because the less-dense, fresh groundwater tends to migrate above the higher-density saline water. This density variance minimizes the potential for shallower groundwater to mix with the more brackish or saline zones. The density difference between the freshwater aquifer system and the salt water of the LDW tends to concentrate the outflow of the surficial aquifer into the intertidal areas.

2.5.6.2 Oxidation, reduction, and tidally influenced conditions

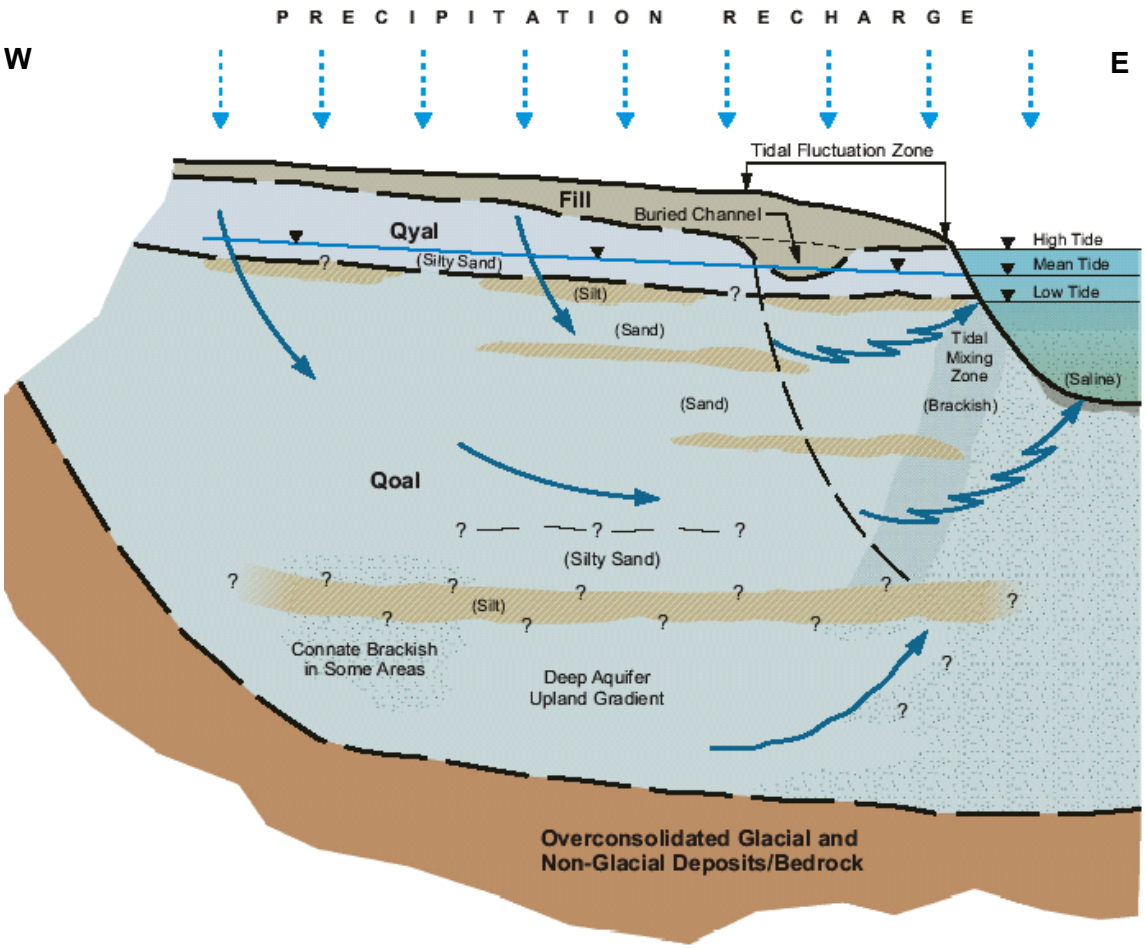
In general, the shallowest water-bearing aquifer receives fresh, oxygenated water through precipitation that penetrates the ground's surface (i.e., infiltration). Much of the LDW study area is paved, greatly limiting the amount of rainwater that can infiltrate directly into the soil.

Shallow aquifer zones in the LDW are located in the silty layers within sandier aquifer soils. Where sand layers are shallow and vertically continuous, more fresh, oxygenated water is expected (e.g., rainwater intrusion). In alluvial soils interbedded with organic-rich silts and clays, the biological oxygen demand (BOD) creates a reducing groundwater environment. Areas contaminated by chemicals that biodegrade readily (i.e., petroleum hydrocarbons) also have the potential to create localized areas characterized by reducing conditions.

Tidal action in the LDW greatly affects groundwater flow direction, flow rate, and overall water quality. Under typical conditions, the hydraulic gradient is toward the LDW. At low tide, the hydraulic gradient between the groundwater system and the LDW is typically at its highest, causing the flow of local groundwater into the LDW. At high tide, the hydraulic gradient can reverse direction, causing LDW surface waters to flow into the adjacent LDW sediments and soil. The amount of LDW water intrusion into and out of the area aquifer depends on tidal patterns, site-specific aquifer conditions and soil stratigraphy (i.e., permeability), and rainfall recharge conditions within the groundwater system.

Tidal action causes a continuous oscillation of groundwater within the effective zone of influence. Site-specific studies have shown that the average zone of oscillation ranged from 91 to 180 m (300 to 600 ft) inland of the LDW shoreline (Booth and

Herman 1998). The periodic reversal in flow direction caused by tidal influences can greatly slow the migration of dissolved chemicals in groundwater. Retarded groundwater movement provides more time for dissolved chemicals to transform and/or decay during transport. Figure 2-8 provides a conceptual diagram of tidal influence affecting the greater Duwamish Valley alluvial aquifer system.



Source: Aspect (2003)

Figure 2-8. Conceptual diagram of aquifer tidal mixing

Water intrusion into LDW groundwater during high tide events can greatly influence nearshore groundwater quality. For example, the infusion of LDW water, which has dissolved oxygen (DO) concentrations up to 10 mg/L, into groundwater, which generally has DO concentrations less than 2 mg/L, can serve as a catalyst for biochemical reactions. The net effect of the exchange between saline and fresh water in the tidal mixing zone is that many chemicals exhibit enhanced natural attenuation as a result of active biogeochemical processes (Boatman and Hotchkiss 1997), as discussed in the following section.

2.5.6.3 Biogeochemical processes and chemical transport

Advective flow rates and the partitioning of chemicals between flowing groundwater and soil particles influence the transport of chemicals in groundwater. Advection, the movement of groundwater through soil pores, is primarily dependent on the specific characteristics of the aquifer, such as porosity, hydraulic conductivity, and discontinuities within the aquifer system. Chemical concentrations and transport within groundwater systems are influenced by numerous factors, including, but not limited to, their aqueous solubility, salinity, pH, reduction-oxidation potential, and the duration of contact between groundwater and contaminated soil.

The migration of organic chemicals and metals is groundwater is slowed greatly through chemical partitioning to the solid materials within the aquifer. The rate at which chemical transport is slowed (the retardation factor) is a function of the dry bulk density and porosity of the aquifer and the soil-water partitioning coefficient of the chemical (Fetter 1993). A soil-water partitioning coefficient for organic compounds can be estimated as the product of the organic carbon (OC) content of the aquifer unit and the specific chemical's water-OC partitioning coefficient. For organic compounds, the retardation factor indicates the compounds mobility relative to groundwater flow. Low-molecular-weight organic compounds are the most mobile (retardation factors near 1), implying that chemicals with low soil-to-water partitioning coefficients travel close to the same rate as the groundwater. Higher-molecular-weight organic compounds, such as polychlorinated biphenyls (PCBs), have low mobility in groundwater (retardation factors of over 6,000 times slower than the movement of groundwater) (Montgomery 1996).

The movement of dissolved metals in groundwater is influenced by their respective solubility (based on pH), salinity, reduction-oxidation potential, and the presence of hydroxides and sulfide minerals in aquifer solids. The reducing environments that are present in many areas of the LDW produce a more soluble and mobile state of some metals (especially iron, manganese, and arsenic) (Hinkle and Polette 1999). The solubility of these same metals can be limited by chemical precipitation reactions with groundwater anions (e.g., sulfate, hydroxides) that are also influenced by the same properties. Low pH can bring metals into solution; however, this condition is rare in the subsurface (Hinkle and Polette 1999).

In addition to advection and partitioning, several other processes can influence chemical concentrations in groundwater downgradient of a contaminated area, as well as concentrations in waters that discharge to the LDW. These processes include:

- **Volatilization –** The transition of a chemical from an aqueous phase into a vapor phase in unsaturated soil above the groundwater table that can occur when low molecular-weight compounds migrate in groundwater.

- **Diffusion –** The movement of molecules along chemical concentration gradients; generally more significant at low groundwater velocities (e.g., through silt and clay layers).
- **Dispersion –** A small-scale mixing process that results from the various paths individual chemical molecules take while traversing through porous soil or sediment. A substantial amount of dispersion can occur in fine silty sand sediment from constant tidal oscillation.
- **Dilution –** The decrease in concentration that occurs along the groundwater pathway from an area with elevated groundwater chemical concentrations as a result of recharge from precipitation. Dilution also occurs at the point of discharge into a tidally influenced water body as a result of water exchange between the surface water body and the discharging aquifer. Tidal monitoring studies have indicated that water intrudes from a surface water body into the adjacent aquifer during high tides and that groundwater discharges to surface water at low tides. This exchange can reduce the dissolved chemical concentrations discharging into the LDW (Yim and Mohsen 1992).
- **Biological degradation –** Non-chlorinated hydrocarbon compounds, including total petroleum hydrocarbons (TPH), low-molecular-weight polycyclic aromatic hydrocarbons (LPAHs), and benzene, toluene, ethylbenzene, and xylene compounds (BTEX), degrade easily under aerobic conditions (Wiedemeir et al. 1994). Hydrocarbon compounds can also be degraded by sulfate- and nitrate-reducing bacteria, although this process is slower than aerobic degradation (Wiedemeir et al. 1994). In general, organic compounds must come into contact with the cell membranes of micro-organisms to be degraded, so degradation rates are generally higher for more soluble compounds (Mackay et al. 1992). Because micro-organisms can also influence the reduction-oxidation potential in the groundwater environment (e.g., through microbial oxidation of organic matter) (Moore et al. 1988), they can alter the speciation and mobility of metals. Reducing conditions can mobilize iron, manganese, arsenic, and other metals present in or sorbed onto sediment particles. Under aerobic conditions, the mobility of these metals is decreased as a result of sorption to or co-precipitation with iron and manganese (hydr)oxides (Moore et al. 1988). The combination of these processes can produce a transition zone between the deeper anaerobic sediment layers and the surface aerobic sediment layers.

The presence of dense non-aqueous phase liquid (DNAPL) may also affect groundwater quality. Upon release to the soil surface, free-phase DNAPL can move downward (via gravity) or laterally along the surface of soil layers. The DNAPL can become trapped in pores and fractures, leaving residual concentrations that may persist in the subsurface for many years. These residuals may slowly release chemicals into groundwater (EPA 1992a).

2.5.6.4 Potential implications for LDW sediment and surface water quality

The most mobile organic compounds in groundwater have the lowest tendency for sorption to sediment materials. Thus, compounds that are relatively mobile in groundwater (e.g., tetrachloroethene, 1,1,1-trichloroethane, BTEX) are not likely to contaminate sediment if present in groundwater discharge to the LDW. These compounds may still present risks to benthic organisms through aqueous exposure if concentrations are sufficiently high, partially because typical sediment remediation techniques (i.e., dredging, capping) do not readily remove these chemicals from the environment. Remediation, removal, and control of these chemicals are addressed through the installation of treatment systems and the implementation of remediation programs and source control measures in upland source areas.

In contrast, PCB compounds have a high affinity for sediment solids and may pose some potential for sediment contamination if present in groundwater in sufficient concentrations. However, the flux of PCB compounds in groundwater is likely to be very low unless the contaminated groundwater is located immediately adjacent to the LDW or co-solvents are present (Montgomery 1996). PCB compounds move over 6,000 times slower through groundwater (e.g., Aroclor 1260 has a retardation factor of 6,840) (Washington Administrative Code [WAC] 173-340). As a result, source control actions, such as contaminated soil removal, would be the most effective way to reduce the potential for PCBs to contaminate groundwater or to migrate to the LDW.

Metals in groundwater, such as copper and zinc, have low mobility because of precipitation or sorption to oxides and clays in the tidal mixing zone. Mixing of oxygenated estuarine water with groundwater within the aquifer results in higher concentrations of chlorides, sulfates, and carbonates in groundwater near the LDW. Reactions with these anions can lower the mobility of some metals (such as lead). Aerobic conditions in the tidally influenced zone also result in the formation of iron and manganese oxides (Boatman and Hotchkiss 1997), which provide an effective surface for sorption and coprecipitation with other metals in groundwater, thus limiting their discharge into estuarine systems (Moore et al. 1988).

The solubility and mobility of arsenic in groundwater is controlled by adsorption and desorption reactions, along with solid-phase precipitation and dissolution reactions (USGS 1996), which are a function of reduction-oxidation conditions and pH. An increase in dissolved arsenic concentrations might occur where natural organic matter has created a reducing environment. Releases of organic chemicals such as TPH also cause reducing conditions as biodegradation processes consume available oxygen, and cause the dissolution of natural arsenic (Moore et al. 1988). Arsenic is commonly found in groundwater within the Duwamish Valley as a result of a combination of these natural and anthropogenic reducing conditions. As groundwater becomes more aerobic (i.e., in the tidal mixing zone), arsenic may be oxidized to arsenate, which tends to sorb onto iron oxide coatings and clay minerals (USGS 1996). Increased

oxygen concentrations in the tidal mixing zone can further attenuate arsenic prior to discharge into the LDW.

2.6 SURFACE WATER HYDROLOGY

The Green River, which is the main freshwater source for the LDW, originates in the Cascade Mountains near Stampede Pass and flows by the Howard Hanson Dam at RM 65.0 and the Tacoma Headworks Dam at RM 61.0 (Culhane et al. 1995). Major tributaries to the Green River include Sunday Creek, Smay Creek, and the North Fork upstream of the Howard Hanson Dam and Newaukum Creek, Soos Creek, and Mill Creek downstream of the Howard Hanson Dam. In addition, the Black River discharges fresh water to the Duwamish River in Tukwila, several km upstream of the LDW. Flow from the Black River is normally minimal (approximately 2.6 m^3/s [92 cubic feet per second (cfs)]) but increases substantially during storm runoff events.

The lower 10 km (6.2 mi) of the river originally opened into a tidal marsh and a broad expanse of intertidal mudflats. In the mid-1800s, estimated annual average discharge from the Duwamish River ranged between 70 and 250 m^3/s (2,500 to 9,000 cfs) (Blomberg et al. 1988), although large floods, between 424.8 m^3/s and 849.6 m^3/s (15,000 and 30,000 cfs), used to down trees along the river's edge, deposit sediment throughout the river basin, and move river channels (Fuerstenberg et al. [1996], as cited in ACOE 1997). The Howard Hanson Dam was installed in 1961 in the upper part of the Green River primarily for flood control and low-flow augmentation to preserve fish life when river flows were naturally low (Sato 1997). The dam effectively decreased peak flows, which now rarely exceed 340 m^3/s (12,000 cfs) but increased moderate flows from 85 to 140 m^3/s (3,920 to 6,460 cfs) because of the periodic metered release of water stored behind the dam (King County 2000b).

Flow has decreased 78% from historical levels, mostly as a result of the diversion of the White River to the Puyallup River basin and the diversion of the Cedar River into southern Lake Washington in 1912. The former diversion was natural; the latter diversion was man-made in support of the construction of the Ballard Locks and the Lake Washington Ship Canal. These changes lowered Lake Washington by over 6 ft and caused increased drainage through the locks rather than through the Black River channel. Collectively, these irreversible changes have resulted in the present LDW hydrologic landscape.

Average downstream flow for the Duwamish River as measured at the Tukwila gaging station was 43.4 m^3/s (1,533 cfs) during 2003-2004 and ranged from 9.3 m^3/s (327 cfs) in August to 93.2 m^3/s (3,290 cfs) in June (Clemens 2007). Flow at the Auburn gaging station ranged from 4.3 to 329 m^3/s (152 to 11,600 cfs [the record high]) between 1962 and 2004 (Clemens 2007). Between 2000 and 2006, the annual average flow rate measured at the Auburn gaging station was 33.7 m^3/s (1,190 cfs) (ranging between 24.1 m^3/s and 40 m^3/s [850.6 and 1,413 cfs]) (USGS 2007). Approximately 80% of the water in the Duwamish River eventually flows through the West Waterway

because of a sill at the south end of the East Waterway, which deters flow (Weston 1999a). Flow rates are greatest during the winter months because of seasonal precipitation and lowest throughout the late summer dry season.

Surface water runoff within the LDW area also contributes to flow within the LDW, from sources such as SDs, tributary creeks, CSOs, and non-point inputs; these sources are expected to be less than 1% of the total discharge, even during peak flow events. During the 1980s SD discharges to the LDW were estimated at 1,868 million gal. per year (mgy) (0.2 m^3/s) (Tetra Tech 1988c). Annual CSO outflow to the LDW, derived from monitoring data collected between 2000 and 2005, averaged 75.7 mgy (0.008 m^3/s) (King County 2006a). The recent average SD inputs are small (approximately 0.02%) relative to the average flow of 43.4 m^3/s at the Tukwila gaging station during the 2003-2004 season.

Stream flow in the LDW is also influenced by water diversion, particularly by the City of Tacoma's Headworks Dam, which diverts at least 3.2 m^3/s (110 cfs) of freshwater daily for municipal use. Discharge of treated effluent from the King County South Wastewater Treatment Plant in Renton to the Duwamish River was eliminated in 1986, decreasing summer flows by as much as 25% (~1.6 m^3/s [56 cfs]) (Harper-Owes 1981).

EPA, the US Geological Survey (USGS), USACE, King County, Metro (now part of the King County Department of Natural Resources and Parks), the University of Washington, and other organizations have measured current velocities within the LDW as part of numerous environmental investigations (Harper-Owes 1983; King County 1999b; Prych et al. 1976; Santos and Stoner 1972; Stevens Thompson & Runyan 1972; Weston 1993b). The most extensive measurements within the LDW have been collected by King County. In 1996, the County deployed current velocity meters at two locations in the LDW (RM 1.1 and RM 3.5) for a 3-month period beginning in August, and recorded currents at 15-minute intervals along a vertical profile (King County 1999b). The two deployment stations were selected to represent the LDW study area with respect to channel width. The LDW is widest at RM 1.1 and is at its narrowest at RM 3.5. Measured current velocities within the LDW during this study rarely exceeded 40 cm/s (1.3 ft/s) (Lavelle et al. 1985).

In the winter of 2003-2004, the County conducted another study of current velocities, deploying two current velocity meters at RM 1.1 for two 4-week periods (King County 2005a). During both monitoring periods, one meter was placed near the center of the navigation channel, while the other was placed on a shallower channel side slope. Reported mean net current speeds for meters placed in the center of the channel ranged from 2.5 cm/s (at 25% channel depth) to 18 cm/s (at 10% channel depth). Mean net current speeds for meters placed at channel side slope locations ranged from 1.3 cm/s (at 25% depth) to 8.9 cm/s (at 10% depth). Currents were predominately oriented along the channel, and velocities were generally slower along the side slopes. Current patterns corresponded to the semi-diurnal tide.

2.7 ESTUARINE FEATURES

Water circulation within the LDW, a well-stratified estuary, is driven by tidal actions and river flow; the relative influence of each is highly dependent on seasonal river discharge volumes. Freshwater moving downstream overlies the tidally influenced saltwater entering the system. Typical of tidally influenced estuaries, the LDW has a relatively sharp interface between the freshwater outflow at the surface and saltwater inflow at depth.

Santos and Stoner (1972) characterized the circulation patterns within the tidally influenced water (or salt wedge) area of the LDW, which typically extends from Harbor Island to near the head of the navigation channel. When freshwater inflow is greater than 28.3 m^3/sec (1,000 cfs), the saltwater wedge does not extend upstream beyond the E Marginal Way S Bridge (RM 6.3), regardless of the tide height (Stoner 1967). During high-tide stages and periods of low freshwater inflow, the saltwater wedge has been documented as extending as far upstream as the Foster Bridge (RM 8.7) (Stoner 1967). At the river's mouth at the northern end of Harbor Island, a salinity of 25 parts per thousand (ppt) is typical for the entire water column; salinity decreases toward the upriver portion of the estuary. The thickness of the freshwater layer increases throughout the LDW as the river flow rate increases.

Dye studies indicate that downward vertical mixing over the length of the saltwater wedge is almost non-existent (Schock et al. 1998). Santos (1972) described how the upstream location or "toe" of the saltwater wedge, typically located between Slip 4 and the head of the navigation channel, is determined by both tidal elevation and freshwater inflow. Fluctuations in tidal elevation also influence flow in the upper freshwater layer, which varies over the tidal cycle.

USGS measured the average net upstream transport of saltwater below the Spokane Street Bridge and reported it as approximately 5.4 m^3/s (190 cfs) (Clemens 2007). This average net upstream flow was about 12% of the average downstream flow measured at the Tukwila gaging station as noted in Section 2.6. During seasonal low-flow conditions, saltwater inputs from the West Waterway were more than one-third of the total discharge from the LDW (Harper-Owes 1983).

2.8 ECOLOGICAL HABITATS AND BIOLOGICAL COMMUNITIES

This section briefly summarizes the types of ecological habitats in the LDW, as well as the species that use these habitats, including benthic invertebrates, fish, wildlife, and plants. Additional information on the ecological habitats and biological communities of the LDW study area is provided in the ERA (Appendix A, Sections A.2.1 and A.2.2).

2.8.1 Ecological habitats and restoration activities

Ecological habitats of the LDW have been modified extensively since the late 1800s through hydraulic changes, channel dredging, the filling of surrounding floodplains,

and the construction of overwater structures, levees, dikes, and other bank stabilization structures. The remnants of natural meanders along the LDW, several of which are now used as slips, and the area west of Kellogg Island are the only evidence of the river's original winding course.

The dominant natural habitat types in the LDW are intertidal mudflats, tidal marshes, and subtidal areas (Table 2-2). Most (around 98%) of the approximately 510 hectares (1,270 ac) of tidal marsh, 590 hectares (1,450 ac) of mudflats and shallows, and all of approximately 500 hectares (1,230 ac) of tidal wetland historically present in the LDW have been either filled or dredged (Blomberg et al. 1988). Remnant tidal marsh areas account for only 2 hectares (5 ac) of the LDW, while mudflats account for only 22 hectares (54 ac) (Leon 1980).

Table 2-2. Ecological habitat types represented in the LDW

Habitat Type	Description	Influential Physical Parameters	Extent and Condition in LDW
Intertidal marshes	Intertidal areas between -4 and +14 ft MLLW; exposed at low tides; marsh soils generally fine-textured and nutrient-rich, supporting grasses, sedges, rushes, and various other plants	salinity gradients, tidal variation, freshwater stream flow, wave action, water temperature, sediment characteristics, and oxygen content	limited extent; various marsh habitat classifications: emergent marsh (e.g., Herring's House, Hamm Creek, and Upper Turning Basin restoration areas), tidal marsh (5 ac in LDW); high and low marsh on Kellogg Island (Blomberg et al. 1988)
Intertidal mudflats	Intertidal areas roughly between -4 and +14 ft MLLW, remnant mudflats isolated from upland riparian vegetation, exposed at low tides, sometimes shaded by overwater structures	salinity gradients, wave action, water temperature, sediment characteristics, and oxygen content	flats and shallows (approximately 30 to 50 ac in LDW) (Blomberg et al. 1988); approximately 8.6 miles of exposed sand/mud substrate (Battelle et al. 2001); largest remnant on Kellogg Island
Intertidal riprap	Armored shoreline consisting of large rocks and rubble (riprap) or vertical wood or metal structures (sheet pile); can be anywhere from -4 to +14 ft MLLW.	salinity gradients, wave action, water temperature, and oxygen content	riprap-covered areas comprise approximately 17% of intertidal areas in the LDW (USFWS 2000b)
Subtidal areas	Area deeper than -4 ft MLLW, never exposed by low tide; includes navigation channel and transition areas; sediment composition ranges from sand to mud	salinity, sediment composition, grain size, OC content, water depth, and temperature	throughout LDW, including navigation channel

LDW – Lower Duwamish Waterway

MLLW – mean lower low water

OC – organic carbon

USFWS – US Fish and Wildlife Service

The majority of the LDW shoreline consists of riprap, pier aprons, or sheet piling (Tanner 1991). Shoreline armoring is usually present at the top of the intertidal zone; areas of sloping mud, mudflats, and hard surfaces exist in the lower intertidal zone (Battelle et al. 2001). These hard surfaces support populations of encrusting organisms, such as barnacles, and burrowing organisms, such as shipworms (Leon 1980). Because of the shoreline armoring, the intertidal mudflats are partially isolated from inputs of sediment, nutrients, and organic matter (i.e., woody debris) from upland riparian vegetation zones; this isolation degrades the habitat quality of these mudflats (Battelle et al. 2001). In addition, overwater structures, which are common throughout the LDW, often shade shallow and intertidal habitats, alter microclimates, and inhibit growth of aquatic plant communities, further degrading the value of nearshore habitats for native fauna (Battelle et al. 2001).

Intertidal marshes are characterized by emergent aquatic plants and low shrubs, are tidally inundated, and occupy regional areas of the floodplain from approximately -4 to +14 ft MLLW (Blomberg et al. 1988; Bravo et al., undated draft). Marsh soils are generally fine-textured and nutrient-rich, supporting grasses, sedges, rushes, and various other plants. Intertidal marshes remain in the LDW to a limited extent, with some emergent marsh present at the Herring's House, Hamm Creek, and Upper Turning Basin restoration areas; approximately 5 ac of tidal marsh remain in the LDW, much of it on Kellogg Island (Blomberg et al. 1988). The shallow water and dense vegetation of intertidal marshes provide refuge as well as foraging and rearing habitat for benthic invertebrates and fish, including juvenile salmonids (Battelle et al. 2001). In addition, intertidal marshes provide important foraging and rearing habitat for many bird species, including great blue heron, killdeer, and marsh wrens. Taller vegetation, such as shrubs growing in intertidal marshes, can also provide nesting habitat for some bird species. In estuaries, intertidal marshes can also serve as wave buffers and provide flood attenuation (Battelle et al. 2001).

Intertidal mudflats are generally defined as the gently sloping areas from MLLW up to the edge of intertidal marsh vegetation (Blomberg et al. 1988). Intertidal mudflats in the LDW study area are located between -4 ft and +14 ft MLLW. These habitats can include mudflats consisting of unconsolidated silts and clays and sand flats consisting of unconsolidated sandy sediments (Simenstad et al. 1991). Approximately 30 to 50 ac of remnant intertidal mudflats and shallows (Blomberg et al. 1988), and approximately 8.6 miles of exposed sand/mud substrate exist in the LDW (Battelle et al. 2001). These areas occur on the northern portion of Kellogg Island and in occasional patches throughout the LDW, where they are exposed at low tides and sometimes shaded by overwater structures. Intertidal mudflats serve as sources of nutrients for primary producers, and provide food and habitat for benthic invertebrates, fish, shorebirds, and aquatic mammals. Intertidal mudflats also attenuate boat wakes for up-slope tidal marshes (Battelle et al. 2001).

Subtidal sediment composition ranges from sand to mud, depending on the sediment source and local current velocity (Windward 2003b). The sediments in the upstream portion of the LDW, near the head of the main channel at the Upper Turning Basin, are predominantly sand. The sediments in the subtidal habitat further downstream (e.g., near Kellogg Island) are characterized as brown or brown-gray sandy mud overlying darker, more clayey mud.

Kellogg Island, located to the south of Harbor Island, is the largest original habitat area remaining in the LDW and is currently designated as a wildlife refuge. Habitat associated with the island comprises high and low marshes, intertidal mudflats, and filled uplands. Kellogg Island is highly altered from its historical size, shape, and function as the result of dredge, fill, and diking activities that occurred from the late 1940s or early 1950s through the 1970s, which greatly altered its interior (Canning et al. 1979). The upland portion of Kellogg Island was created in 1974, when the Port of Seattle deposited 1,700 m^3 (2,200 cubic yards [cy]) of dredged material on the island (Sato 1997). Both introduced and native plant species rapidly colonized the 7-hectare (17-ac) island. Remnants of natural intertidal habitat occur on the northern portion of Kellogg Island and in occasional patches throughout the LDW.

Small intertidal areas of marsh and unvegetated marsh habitat in the LDW have become the focus of habitat restoration activities. The Hamm Creek restoration area, located on the western shoreline at RM 4.3, restored 1 ac of emergent salt marsh, 2 ac of freshwater wetlands, and nearly 2,000 ft of the Hamm Creek stream bed in order to provide suitable habitat for salmon and other fish and wildlife species (EPA 2006d). Habitat restoration projects are also located on Port property at Terminal 107 (T-107) (RM 0.4 W) and Terminal 105 (T-105) (RM 0.1 W) (Port of Seattle 2007). At the Coastal America site in the Upper Turning Basin (RM 4.7 W), 5 ac of intertidal habitat was restored. At Herring's House Park, located on the west side of the LDW near Kellogg Island (RM 0.3 to RM 0.5 W), intertidal habitat has been restored at a former lumberyard.

Additional restoration activities are also taking place at the GSA marsh area (RM 0.8 E), the Diagonal Ave S/Terminal 108 (T-108) restoration area (RM 0.6 E), the 8th Ave S street end/Gateway North restoration area (RM 2.7 E), the S Portland street end park (RM 2.8 W), the Muckleshoot Tribe restoration area at Kenco Marine near the Upper Turning Basin (RM 4.6 W), the S 112th St mitigation site (RM 5.7 E), and the County's Cecil B. Moses Park (North Winds Weir, RM 5.7 W). These restoration areas are presented on Map 2-6. Information on several additional restoration sites within the LDW is available through NOAA's Damage Assessment, Remediation, and Restoration Program (DARRP) (NOAA 2007b). The objectives of these projects include the removal of rock riprap and over-water wharf structures, restoration of natural tidal flows, and natural re-colonization by native wetland plants (Cordell et al. 1996).

2.8.2 Benthic invertebrates

Benthic invertebrate species are important components of the LDW ecosystem because they serve as a major food resource for commercially and recreationally important fish and wildlife and play a critical role in the overall nutrient cycling of the system. This section briefly describes studies that have investigated site use by benthic invertebrates. Additional details are presented in Appendix A, Section A.2.2.2.

2.8.2.1 Summary of LDW studies

Numerous studies have investigated site use of the LDW by benthic invertebrates. Over the course of several years, Cordell et al. (1996; 1997; 1999) conducted studies of the benthic community at several restoration sites throughout the LDW. Other studies of the benthic community were primarily conducted near Kellogg Island (NOAA and Ecology 2000; King County 1999a; Leon 1980; West et al. 2001; Williams 1990). Surveys of larger benthic invertebrates were conducted throughout the LDW in 2003 and 2004, including a survey of crabs and shrimp (Windward 2004a) and a survey of clams (Windward 2004c). In addition, sampling of the benthic community was conducted at both intertidal and subtidal locations throughout the LDW in 2004 as part of the RI (Windward 2005e). A detailed list of the taxa identified in these studies is included in Appendix A, Attachment A1.

2.8.2.2 Summary of site use

Benthic invertebrate communities observed in the LDW consisted of 670 taxa, representing 178 families in 13 phyla. Typical of estuarine environments, the benthic invertebrate community was dominated by annelid worms, mollusks, and crustaceans. Crustaceans were the most diverse of these three groups in the LDW, including more than 250 taxa. Mollusks included various bivalves and snails. The most abundant large epibenthic invertebrates included slender crabs, crangon shrimp, and coonstripe shrimp. Dungeness crabs were also common, although their distribution was generally limited to the portions of the LDW with higher salinity.

Benthic invertebrates in the LDW form two distinct communities. The infaunal community is typified by burrowing polychaetes and bivalves. King County (1999e) noted that at most sampling locations, the infaunal community was dominated by surface detrital/surface-deposit feeding organisms. The epibenthic community (invertebrates living on top of the sediment), consisted mainly of larger crustaceans and mussels, and was dominated by surface detrital and surface filter-feeding organisms.

Intertidal Community

Surveys of the benthic invertebrate communities (including infauna and smaller epibenthic species) were conducted at 12 intertidal locations throughout the LDW in 2004 (Windward 2005e). The total number of organisms ranged from 500 to 16,233 per 0.1 m^2 (Table 2-3). A total of 61 invertebrate taxa were identified in the sediment core

and transect frame samples. The most abundant organisms were annelids and crustaceans.

Table 2-3. Phase 2 intertidal benthic invertebrate community survey results

River Mile	Location ID[a]	Total Abundance[b]	Swartz's Dominance Index
0.2	B1a	4,842	3
0.6	BCA-1[c]	11,667	6
0.8	B3a	11,958	4
0.9	B2a	16,233	8
1.4	B4a	8,858	4
2.1	B6a	1,475	5
2.3	B5a-2	500	3
2.9	BCA-3[c]	9,050	2
3.1	B7a	8,600	4
3.5	B8a	14,100	6
4.5	B9a	5,875	5
4.8	B10a	6,600	6

[a] Sampling locations are shown on Map A.2-1 in Appendix A.

[b] Total number of individual organisms in a standard 0.1-m^2 area determined by extrapolating the number in the composite sample representing a total area of 0.012 m^2 to the number expected in the larger area (0.1 m^2) by multiplying by 8.33.

[c] BCA locations were sampled only for the benthic invertebrate community analysis.

BCA – benthic community analysis

ID – identification

A Swartz's Dominance Index was calculated for each location. The Swartz's Dominance Index is defined as the minimum number of taxa that make up 75% of the total abundance (Swartz et al. 1985; as cited in PTI 1993). Separate samples were also collected at each location and analyzed for biomass. Total biomass ranged from 0.1 g ww at location B5a-2 to 2.7 g ww at location B6a. Average biomass for intertidal locations was 1.4 g ww (Windward 2005e). Total weights at each location represented an area of 0.012 m^2.

Cordell et al. (2001) conducted both epibenthic and infaunal surveys at seven restoration sites throughout the LDW from 1993 through 1999. The surveys demonstrated that the diversity and abundance of intertidal organisms varied both seasonally and among locations in the LDW. The greatest diversity of organisms (i.e., species richness) occurred in the downstream portion of the LDW; diversity was lower in the area of the Upper Turning Basin.

Two other benthic invertebrate surveys were conducted in the LDW by Williams (1990) and Leon (1980). Williams (1990) identified 80 invertebrate taxa inhabiting intertidal habitats at Kellogg Island. Nematodes, oligochaetes, small harpacticoid

copepods, ostracods, and sabellid polychaetes were the dominant invertebrates. Leon (1980) found 43 different benthic taxa in sediment cores from the intertidal mudflats at Kellogg Island. Most taxa occurred infrequently; nine taxa accounted for 97% of all individuals. Small marine worms of the genus *Manayunkia*, oligochaetes, and harpacticoid copepods made up nearly 80% of all individuals (Leon 1980). In comparison, there were very few organisms at a mudflat site with anoxic sediments near the Duwamish Shipyard, (RM 1.35 W) and there was a greater degree of seasonal variability in the benthic community at a mudflat site at the Riverside Marina, then located near RM 0.2 W (Leon 1980).

In 2003, a clam survey was conducted at 11 intertidal locations between RM 0.0 and RM 4.0 (Windward 2004c). A random sampling design, based on Washington State Department of Fish and Wildlife (WDFW) guidance (Campbell 1996), was employed to survey each of the 11 locations for clam abundance. The mean density of clams at those 11 locations ranged from 0.18 to 0.94 per ft^2. The majority of clams collected were identified as *Macoma balthica* (60%), followed by *Mya arenaria* (20%) and *Macoma nasuta* (18%). *M. balthica* were the smallest clams collected on all beaches with a mean individual wet weight of 0.7 g (range of < 0.1 g to 3.4 g), and *M. arenaria* were the largest clams collected (mean individual wet weight of 15.2 g); however, this species also had the widest range in weight (0.3 g to 54 g) (Windward 2004c).

Potential clam catch rates by people were assessed during the clam survey conducted in 2003 (Windward 2004c). Catch rate exercises were conducted at four beaches identified during the intertidal survey as having high clam abundance. The catch rate exercise used visible signs of clam presence such as holes or siphons to target where to dig, and also focused on collecting larger clams to simulate clamming behaviors that would most likely be conducted by harvesters. *M. arenaria* was the most common clam collected during the assessment of potential catch rates, comprising more than 98% of the total biomass.[8] Catch rates (i.e., total number of clams caught per hour) for all beaches combined were 652 for *M. arenaria* and 189 for all other species.

Subtidal Community

Surveys of the benthic communities (including infauna and smaller epibenthic species) were conducted at 14 subtidal locations throughout the LDW in 2004 (Windward 2005e). The total number of organisms ranged from 72 to 2,300 per 0.1 m^2 (Table 2-4). A total of 246 invertebrate taxa were identified in the van Veen grab samples. In general, annelids, crustaceans, and mollusks were the most abundant organisms at the subtidal locations.

[8] The results of the catch rate exercise indicated that although *M. balthica* was the most abundant clam found in the quantitative survey, *M. arenaria* was the clam most likely to be harvested because of their larger size and presence of shows.

Table 2-4. Phase 2 subtidal benthic invertebrate community survey results

River Mile	Location ID[a]	Total Abundance[b]	Swartz's Dominance Index
0.1	B1b	326	15
0.6	BCA-4	937	13
0.9	B2b	537	7
1.0	B3b	559	12
1.4	B4b	521	8
1.5	B5b	643	8
1.5	BCA-5	72	15
1.7	BCA-2	328	6
2.2	B6b	1,137	3
2.7	B7b	497	5
3.9	B9b	935	6
4.2	B8b	2,300	4
4.3	B10b	1,541	2
4.6	BCA-6	1,689	3

[a] Sampling locations are shown on Map A.2-1 in Appendix A.

[b] Total number of individual organisms retained on a 1-mm sieve in a standard 0.1-m^2 area determined by extrapolating what was enumerated in three composite van Veen grab samples representing a total area of 0.3 m^2 to the smaller area (0.1 m^2) by dividing by 3.

BCA – benthic community analysis (BCA locations were sampled only for the benthic invertebrate community analysis)

ID – identification

Swartz's Dominance Index was calculated for each location. Similar to the intertidal locations, samples were collected and analyzed for biomass. Total biomass ranged from < 0.1 g ww at locations B7b, BCA-4, and BCA-5 to 1.7 g ww at location B4b. Average biomass for subtidal locations was approximately 0.34 g ww (Windward 2005e). Total weights at each location represented an area of 0.0072 m^2.

Ecology (2000) evaluated the benthic community at three subtidal locations in the LDW as part of a sediment quality reconnaissance study for central Puget Sound. The benthic communities at the three locations were dominated by annelids. Mollusks were also common, whereas crustaceans and echinoderms were present in low abundances.

King County evaluated abundance of benthic infauna and epibenthos as a component of their assessment of CSO discharges to the LDW and Elliott Bay (King County 1999e). Subtidal sampling sites included transects located at Kellogg Island as well as downgradient from the Diagonal Avenue S CSO/SD and Duwamish CSO. Polychaetes were abundant in all samples and were the dominant organisms at all locations except at two locations downstream of the Diagonal Avenue S CSO/SD and

Duwamish CSO, where oligochaetes and mollusks were dominant. A Kellogg Island location also had relatively abundant mollusks. Arthropods tended to be more abundant in deeper waters (King County 1999e).

Williams (1990) sampled epibenthic biota to a depth of –8 ft MLLW near Kellogg Island and reported that nematodes, oligochaetes, small harpacticoids, and cumaceans dominated the subtidal epibenthos. As with the intertidal benthos, stations with finer sediments generally had a greater abundance of epibenthic biota (Williams 1990).

Leon (1980) used van Veen grab samplers to characterize the subtidal epibenthic and infaunal sediment biota at five locations downstream of RM 1.5. The study identified more than 60 different taxa, greater than the number found in the intertidal habitat from the same survey. Most subtidal species were deposit-feeding polychaete worms characteristic of deeper, turbid waters of the LDW. Small deposit-feeding clams (*Macoma* sp., *Axinopsida* sp., and *Psephidia* sp.) and the amphipod *Anisogammarus* sp., which feeds on diatoms and green algae, were also present.

Larger Epibenthic Invertebrates

Larger epibenthic invertebrates identified in the LDW include crabs, shrimp, sea stars, anemones, and mussels. Numerous larger epibenthic invertebrate species were caught during two fish and crab surveys conducted throughout the LDW for the RI (Windward 2005b, 2006c). The most abundant epibenthic invertebrates were slender crabs, crangon shrimp, and coonstripe shrimp. Dungeness crabs were also caught in both surveys, although their distribution was generally limited to the portions of the LDW with greater salinity.

A pilot survey and three quarterly surveys were performed in 2003 and 2004 to estimate the abundance of crabs and shrimp within the LDW (Windward 2004a). Three crab species and one shrimp species were caught during the surveys. Slender crabs and Dungeness crabs were the most abundant species, with red rock crabs and dock shrimp being less abundant (Windward 2004a). The majority of these crabs were caught in the downstream, more saline part of the LDW. Red rock crabs and dock shrimp were also caught, primarily in the downstream portion of the LDW, with a few adults caught between RM 1.6 and RM 2.2. During a reconnaissance study conducted by Environmental Solutions Group, Inc. (ESG) (1999), Dungeness crabs and red rock crabs were found at multiple locations near Kellogg Island; no adults were collected upstream of this point, whereas juveniles were found up to the 1st Avenue S Bridge.

During the Puget Sound Ambient Monitoring Program (PSAMP) (1989 to 1999), invertebrates were collected throughout Puget Sound, including locations in the LDW (West et al. 2001). Epibenthic invertebrate species, similar to those caught in the surveys conducted for the RI, were collected with otter trawls. The most common invertebrates were slender crabs and crangon shrimp. Other species caught during the

PSAMP survey, but not in the RI surveys, included porcelain crabs (family Porcellanidae), chitons (class Polyplacophora), and several sea stars.

In September of 2006, King County collected fish for tissue analyses using a high-rise otter trawl in the downstream portion of the LDW (Anchor and King County 2007). Non-target invertebrate species were recorded and released. Coonstripe shrimp was the most common invertebrate species, followed by graceful crab and crangon shrimp. Other crab species included kelp crab, decorator crab, red rock crab, black-clawed crab and pygmy rock crab. Sea star species include short spined sea star, sunflower sea star, and false ochre star. Plumose anemone was also found during this study.

2.8.3 Fish

A diverse fish community inhabits the LDW. This section briefly summarizes what is known about the LDW fish community. Further details regarding life history characteristics and dietary preferences are presented in Appendix A, Section A.2.2.

2.8.3.1 Summary of LDW fish studies

Data on site use by fish are available from 14 studies that have been conducted or are ongoing in the LDW (see Appendix A, Table A.2-6, for a complete list of these studies). In combination, these studies have used various capture techniques to assess the diversity and abundance of fish in the LDW.

2.8.3.2 Summary of site use by fish

The LDW is inhabited by numerous anadromous and resident fish species (see Appendix A, Attachment A2, for a complete list of identified species). Fifty-three resident and non-resident fish species were captured in the LDW during sampling events conducted for the RI (Windward 2004d, 2005b, 2006c). During previous sampling events, Warner and Fritz (1995) recorded 33 resident and seasonal species of fish in the LDW, Miller et al. (1977a) observed 29 species, and Matsuda et al. (1968) recorded 28 species.

Based on the LDW fish studies reviewed in the ERA, shiner surfperch, snake prickleback, Pacific sandlance, Pacific staghorn sculpin, longfin smelt, English sole, juvenile Pacific tomcod, pile perch, rock sole, surf smelt, three-spine stickleback, Pacific herring, and starry flounder were identified as abundant at the time of the sampling events, as were chinook, chum, and coho salmon (see Appendix A, Attachment A2). Fish abundance in the LDW reaches its maximum in late summer to early fall and is generally lowest in winter (Dexter et al. 1981; Miller et al. 1977a).

Anadromous Salmonids—Pacific Salmon

Five species of Pacific salmon (i.e., coho, chinook, chum, sockeye, and pink) have been identified in the LDW study area (Kerwin and Nelson 2000). These anadromous fish use the estuary for rearing and as a migration corridor for adults and juveniles.

Among the numerous beneficial uses identified for the LDW study area, use as habitat for outmigrating juvenile salmonids was listed as the most important (Harper-Owes 1983).

Salmonid residence time in the LDW study area is limited and species-specific. Juvenile chinook and chum salmon have been shown to be present from several days to up to 2 months within the LDW, whereas coho salmon pass through the LDW in a few days (Grette and Salo 1986). Pink salmon and sockeye salmon are rare in the LDW. Salmon found in the LDW spawn mainly in the middle reaches of the Green River and its tributaries (Grette and Salo 1986). The juvenile outmigration of all five species generally commences during the high-flow months of March to June. Outmigration usually lasts through mid-July to early August (Nelson et al. 2004; Warner and Fritz 1995). During these months, salmonids use the estuary to feed and begin their physiological adaptation to higher salinity waters.

Of the salmonid species, chinook salmon have been studied the most extensively in the Green-Duwamish system. Puget Sound chinook salmon were listed as threatened under the federal Endangered Species Act (ESA) on March 24, 1999. The decline of chinook salmon has been attributed primarily to habitat degradation and fragmentation, blockage of migratory corridors, impact from hatchery fish, and commercial and local harvesting practices (Myers et al. 1998a).

Other Salmonids

The coastal Puget Sound bull trout population was listed as threatened under the federal ESA on November 1, 1999. Bull trout were historically found in the LDW because it originally included a much larger upper elevation drainage area, including the White River, which still maintains a bull trout run. Current stock status in the Green-Duwamish system is unknown (WDFW 2000). The decline of bull trout has been primarily attributed to habitat degradation and fragmentation, blockage of migratory corridors, poor water quality, historical fisheries practices, and the introduction of non-native species (64 FR 210: 58910-58933).

Bull trout typically spawn in the upper headwaters of a river system, generally requiring high oxygen concentrations and water colder than 10° C. Although sub-adult bull trout have been captured in the LDW (Shannon 2001; Warner and Fritz 1995), there is no evidence that bull trout are spawned or reared within the LDW.

Summer steelhead (*Oncorhynchus mykiss*) is a non-native stock sustained by wild production of hatchery origin fish (Washington Department of Fisheries et al. 1993). The run size is unknown but approximated to be a few hundred fish. Winter steelhead consist of both wild and hatchery fish with annual returns ranging from 944 to 2,378 fish. Grette and Salo (1986) reported that repeat spawners make up approximately 19% or fewer of returning wild adults in the Green River system. Both winter and summer runs migrate quickly through the LDW (Grette and Salo 1986).

Sea-run cutthroat trout exist in the LDW, but very little is known about this population or its use of the system. Only 11 cutthroat trout were captured in beach seines at nine stations sampled approximately 30 times each throughout the LDW from February through June 1994 (Warner and Fritz 1995).

Non-Salmonid Fish

Based upon the LDW fish studies reviewed in the ERA, the most abundant non-salmonid fish in the LDW are shiner surfperch, snake prickleback, starry flounder, English sole, Pacific staghorn sculpin, Pacific sandlance, juvenile Pacific tomcod, pile perch, rock sole, surf smelt, three-spine stickleback, Pacific herring and longfin smelt (see Appendix A, Attachment A2). In 2006, King County also found tubesnout to be relatively abundant in the downstream portion of the LDW (Anchor and King County 2007). Shiner surfperch, longfin smelt, and Pacific herring are seasonally abundant in the LDW. Pacific sandlance and longfin smelt, though known to be abundant in the LDW, were encountered infrequently in recent beach seine and trawling efforts, as were Pacific herring and surf smelt. Though these species were not encountered frequently during recent sampling, they are occasionally found in large numbers in the LDW (Shannon 2006; Windward 2005b, 2006c). In all previous studies, Pacific staghorn sculpin were consistently one of the most abundant fish captured in the LDW, particularly during the fall months. Pacific staghorn sculpin are opportunistic feeders, and the adults feed at a higher trophic level than other common resident fish species.

Shiner surfperch abundance in the LDW peaks in summer during spawning (Miller et al. 1975). English sole are common in the LDW over all seasons, with peak abundance in spring (Miller et al. 1977b). Adult English sole migrate seasonally to their spawning grounds in Puget Sound in winter (Forrester 1969) and typically spawn in Puget Sound during February and March (Smith 1936). In central Puget Sound, adult populations of English sole spawn in Elliott Bay and Port Gardner but disperse quickly after spawning (Pallson 2001). Starry flounder are also noted to migrate seasonally between very shallow water and in estuaries during the summer, moving into deeper waters in the winter (Morrow 1980).

2.8.4 Wildlife

The aquatic and semi-aquatic habitats of the LDW support a diversity of wildlife species. Formal studies, field observations, and anecdotal reports indicate that up to 87 species of birds and 6 species of mammals use the LDW at least part of the year to feed, rest, or reproduce. This section provides a brief overview of these bird and mammal species. Additional information, including summaries of what is known about how these species use LDW habitats, is provided in Appendix A, Section A.2.2.4.

2.8.4.1 Birds

Many birds have been identified in the LDW area including: passerine/upland birds, raptors, shorebirds/waders, waterfowl, and seabirds; the species are listed in Appendix A (Table A.2-7). Canning et al. (1979) conducted extensive surveys of Kellogg Island, as well as occasional surveys of the entire LDW, from September 1977 to July 1978. They recorded a total of 70 species, including 26 passerines/upland birds, 3 raptors, 11 shorebirds/waders, 17 waterfowl, and 13 seabirds. The study showed that Kellogg Island had a much greater diversity of birds than did the rest of the LDW study area because of its seclusion and greater variety of habitats. Cordell et al. (2001) reported 75 species of birds, including 32 passerine/upland birds, 7 raptors, 8 shorebirds/waders, 16 waterfowl, and 12 seabirds. Diversity and abundance were highest at the Kellogg Island site, but other areas of the LDW were also consistently used by a wide variety of birds. Birds were most abundant in the spring and least abundant in the summer. During the sandpiper presence and habitat survey conducted for the RI in 2004 over the period of four days, 21 bird species were observed while assessing preferred spotted sandpiper habitat areas (Windward 2004i).

Passerines/Upland Birds

Thirty-two species of passerine/upland birds have been documented along the LDW (Canning et al. 1979; Cordell et al. 1999) (see Appendix A, Table A.2-7). Although generally associated with upland habitats, these birds occasionally forage in the exposed mudflats or use freshwater habitats along the LDW for bathing (Canning et al. 1979). During the sandpiper presence and habitat survey, 10 passerine species (starlings; crows; violet green, barn, and tree swallows; house, song, and white-crowned sparrows; rock pigeons, and American robins) were frequently observed in the LDW area (Windward 2004i).

Raptors

Eight species of raptors have been reported to use the LDW: bald eagle, osprey, peregrine falcon, merlin, and four species of hawks (Cordell et al. 1999). During the sandpiper presence and habitat survey bald eagles were frequently observed in the LDW area (Windward 2004i). The bald eagle is listed under ESA as a threatened species but is currently under review for delisting. In Washington, the bald eagle is also listed as a state threatened species (WDFW 2006a, b, 2007). Five bald eagle nests were occupied within 8 km (5 mi) of the LDW study area in 1999 (King County 1999e). One or two pairs of resident eagles can be found in the LDW vicinity during the summer (King County 1999e), and overwintering migrant eagles are routinely observed from the beginning of October through late March.

In 2006, WDFW reported 10 osprey nest sites located along the LDW or nearby, with nests also observed on Harbor Island and along the Duwamish and Green Rivers within about 6.4 km (4 mi) of the Upper Turning Basin (Thompson 2006). Four osprey

nest platforms were identified throughout the LDW during the sandpiper presence and habitat survey, with young observed in each nest (Windward 2004i). Adult ospreys were also frequently observed on tall structures near the nest platforms during this survey.

Two pairs of peregrine falcons currently nest along the LDW. Two peregrine falcons were observed in the LDW area during the sandpiper presence and habitat survey (Windward 2004i). The peregrine falcon is listed as a species of concern under ESA and was downlisted from a state endangered species to a state sensitive species in April 2002 (WDFW 2007).

Cooper's hawks and sharp-shinned hawks have been observed overwintering in the vicinity of the LDW. Red-tailed hawks are a resident species and are commonly observed along grassland/woodland margins of the LDW. These species are not heavily reliant on the LDW for food; they prey on songbirds and rodents but will also eat waterfowl. Swainson's hawks and merlin are rare in the LDW area (Canning et al. 1979; Cordell et al. 1999).

Shorebirds/Waders

Ten species of shorebirds and wading birds have been documented in the LDW: great blue heron, green heron, three species of sandpipers, dowitcher, dunlin, killdeer, sanderling, and lesser yellowlegs (Cordell et al. 1999). Of these species, great blue heron are year-round residents and comprise the only sizeable and consistent population recorded by Cordell et al. (1996) on the Duwamish River. During the sandpiper presence and habitat survey, great blue heron were frequently observed throughout the LDW (Windward 2004i). One nesting great blue heron colony is located 11 km (6.8 mi) to the northwest of the LDW (Kiwanis Ravine colony); another colony is located 12 km (7.5 mi) to the southeast, in Renton (Black River colony) (Butler 1997).

The two most common shorebirds observed in the LDW are killdeer and various species of sandpipers. The spotted, least, and western sandpipers are reported to use the LDW, and spotted sandpipers have been observed nesting on Kellogg Island (Canning et al. 1979). Spotted sandpipers are present in the Northwest region year-round; whereas least and Western sandpipers are present during the winter (Sibley 2003). During the sandpiper presence and habitat survey, small numbers of spotted sandpipers were observed in the LDW area, particularly on and in the vicinity of Kellogg Island and Herring's House Park (Windward 2004i). Killdeer were also observed during the survey in the same areas as spotted sandpiper, but in larger numbers. Though no spotted sandpiper nests were observed during the survey, a killdeer nest was observed on Kellogg Island. Least and western sandpipers occur in mixed flocks and are difficult to distinguish. These species nest primarily in northern Canada and Alaska in the summer months but are reported to frequent Kellogg Island

from September through May. Most are thought to be migrants, though some may reside in the LDW area throughout the winter.

Waterfowl

Seventeen species of waterfowl, including 13 species of ducks, 3 species of geese, and the American coot, use the LDW (Cordell et al. 1999). During the sandpiper presence and habitat survey, Canada geese and mallard ducks were commonly observed, and gadwall and common mergansers were also observed though less frequently (Windward 2004i). In general, these migratory birds overwinter in the Puget Sound area (and further south) and migrate north in the summer. A small resident population of mallards lives year-round in the LDW area, and as many as 290 migratory mallards have been reported to move through the LDW.

Canvasback, greater scaup, gadwalls, bufflehead, and both common and Barrow's goldeneye are reported to use the LDW. Feeding by all diving duck species is centered around Kellogg Island (Canning et al. 1979).

All three species of North American mergansers use the LDW study area. Approximately 30 red-breasted mergansers reportedly overwinter in the LDW area between December and March.

A population of approximately 1,000 Canada geese resides in the vicinity of Lake Washington and includes the LDW population (Canning et al. 1979).

Seabirds

Fourteen species of seabirds were recorded in the LDW during surveys completed in 1979 and 1999 (Canning et al. 1979; Cordell et al. 1997; 1999), including two species of cormorants, three species of grebes, three species of gulls, three species of loons, pigeon guillemot, common murre, and Caspian tern. Wintering cormorants (pelagic and double-crested) use the LDW study area between November and May, with large numbers present between December and April. During the sandpiper presence and habitat survey, double-crested cormorants and gull species were commonly observed in the LDW area (Windward 2004i).

Glaucous-winged gulls and mew gulls are the only gull species reported to use the LDW study area in large numbers. Glaucous-winged gulls are reported to use the area year-round; mew gulls frequent the area from September through May (Canning et al. 1979). Caspian terns have been seen on Kellogg Island (Luxon 2004). Pigeon guillemots and common murres have also been reported in the LDW; however, their use of the LDW is infrequent. The common loon is a state sensitive species; they are reported to be rare in the LDW area (Canning et al. 1979). Three species of grebe have been reported in the LDW area. The Western Grebe is present in the Northwest region during the winter (Sibley 2003). Grebes and other marine bird species have been

declining throughout the greater Puget Sound region in recent years (Nysewander et al. 2001).

2.8.4.2 Mammals

Three species of terrestrial mammals use the LDW (i.e., raccoons, muskrats, and river otters), and three marine mammal species may enter the LDW (i.e., harbor seal, California sea lion, and harbor porpoise) (Dexter et al. 1981; Tanner 1991; WDFW 1999).

Anecdotal information indicates that a river otter family lives year-round on Kellogg Island in the LDW, although otters have not been observed during wildlife surveys (Cordell et al. 2001). Raccoons are reported to be common along the forested ridge slopes to the west of the LDW. Muskrat populations are reported to exist at T-107 and the Upper Turning Basin (Canning et al. 1979).

Harbor seals and sea lions are commonly seen in Elliott Bay and have occasionally been observed in the LDW. During a survey conducted by WDFW from December 1998 to June 1999, sea lions were observed in the LDW on 16 occasions and seals on 17 occasions (WDFW 1999), with most observations of both species occurring below the 1st Avenue S Bridge. Harbor seals have been shown to forage over large distances ranging from 5 km (3.1 mi) (Stewart et al. 1989) to 55 km (34.2 mi) (Beach et al. 1985). Recent information on use of the LDW by harbor porpoises was not available, although it has been noted that they occasionally enter the LDW (Dexter et al. 1981).

2.8.5 Plants

Few studies have investigated the plant communities present in the LDW (Canning et al. 1979; Cordell et al. 2001; Tanner 1991; USFWS 2000a). The methods used to assess plant communities have ranged from analysis of aerial photos to field surveys. Most recently, Cordell et al. (2001) monitored the vegetation of wetland restoration and reference sites in the LDW by conducting surveys during the growing season at six restoration and six reference sites throughout the LDW from 1993 to 1999.

In the LDW, there is a total of 0.0175 km^2 (0.0068 mi^2) of habitat with estuarine macrophytes, primarily limited to portions of Kellogg Island and other small intertidal areas with vegetated intertidal habitat (USFWS 2000b, c). *Carex* (sedges) and *Scirpus* (bulrushes) are the predominant marsh vegetation types between the Upper Turning Basin and Kellogg Island. Downstream from Kellogg Island are more marine plants such as *Salicornia* (grassworts), *Distichlis* (salt grass), and *Atriplex* (salt bush). The interior high marsh plant community of Kellogg Island, which is flooded only by higher spring tides, includes *Carex lyngbyei*, *Distichlis spicata*, *Juncus balticus* (Baltic rush), and *Phragmites* sp., a non-native species (Battelle et al. 2001). The naturally occurring *Carex* patches surveyed in 1993 occurred between elevations of 1.6 and 3.0 m (5.2 and 9.8 ft) above MLLW, and the single patch of naturally occurring *Scirpus* was at

3.7 m (12 ft) above MLLW, indicating that these plants are seldom underwater (Cordell et al. 2001).

2.8.6 Threatened and endangered species

Fifteen species identified in the LDW study area are listed under either the ESA or by WDFW, as candidate species, threatened species, or species of concern (see Table 2-5). Information on the presence and observed frequency of these species in the LDW area is provided in the ERA (Appendix A).

Table 2-5. LDW species listed under ESA or by the Washington State Department of Fish and Wildlife

Common Name	Scientific Name	Status
Chinook salmon	*Oncorhynchus tshawytscha*	federal threatened species, state candidate species
Coho salmon	*Oncorhynchus kisutch*	federal candidate species
Puget Sound steelhead	*Oncorhynchus mykiss*	federal threatened species
River lamprey	*Lampetra ayresi*	federal species of concern, state candidate species
Bull trout	*Salvelinus confluentes*	federal threatened species, state candidate species
Pacific herring	*Clupea herengus pallasi*	federal candidate species, state candidate species
Pacific cod	*Gadus macrocephalus*	federal species of concern, state candidate species
Walleye pollock	*Theragra chalcogrammus*	federal candidate species, state candidate species
Rockfish species	*Sebastes* spp.	state candidate species
Bald eagle	*Haliaeetus leucocephalus*	federal threatened species,[a] state threatened species
Peregrine falcon	*Falco peregrinus*	federal species of concern, state sensitive species[b]
Merlin	*Falco columbarius*	state candidate species
Common murre	*Uria aalge*	state candidate species
Common loon	*Gavia immer*	state sensitive species
Western grebe	*Aechmophorus occidentalis*	state candidate species

Source: (WDFW 2007)

Note: The Puget Sound southern resident orca distinct population segment is listed on both the federal and Washington State endangered species lists. Orca do not use the LDW but are occasionally present in Elliott Bay and may be exposed to chemicals from the LDW through the consumption of prey (such as chum salmon) that spend part of their time in the LDW.

[a] Listing currently under review for removal.

[b] Downlisted from state endangered to state sensitive on April 2002 (WDFW 2002).

2.9 HUMAN USE

This section provides a brief overview of the demography, land use, and site use in the LDW area.

2.9.1 Demography

Although the area surrounding the LDW is often regarded as an industrial corridor, residential neighborhoods (i.e., South Park and Georgetown) are present in the LDW area. Population densities within these mixed residential- and industrial-zoned communities are lower than those of other Seattle neighborhoods, reflecting the significant ratio of industrial land use. The racial diversity of the residential population in the immediate vicinity of the LDW is greater than the racial diversity of residents within the City of Seattle overall or the nearby City of Tukwila (Table 2-6).

Table 2-6. Population data for Duwamish neighborhoods, City of Seattle, and City of Tukwila

Race	Population		
	Duwamish Neighborhoods[a]	City of Seattle[b]	City of Tukwila[c]
Caucasian	11,871 (53.5%)	394,889 (70.1%)	10,032 (58.3%)
African-American	2,143 (9.6%)	47,541 (8.4%)	2,310 (13.5%)
Native American and Alaska Native	397 (1.8%)	5,659 (1.0%)	196 (1.1%)
Asian	4,221 (19.0%)	73,910 (13.1%)	1,788 (10.5%)
Native Hawaiian and other Pacific Islander	333 (1.5%)	2,804 (0.5%)	332 (1.9%)
Other single race	1,868 (8.4%)	13,423 (2.4%)	1,348 (7.8%)
Two or more races	1,379 (6.2%)	25,148 (4.5%)	1,198 (6.9%)
Total	**22,192 (100%)**	**563,374 (100%)**	**17,204 (100%)**

Note: Percentage of total population is given in parentheses.

[a] Population estimates from City of Seattle LDW neighborhoods, including Tract 93, Duwamish/SODO; Tract 99, North Delridge; Tract 108, Riverview; Tract 109, Georgetown; Tract 112, South Park; and Tracts 112, 264, and 265, Highland Park (City of Seattle 2006).

[b] Data from Census 2000 (City of Seattle 2006).

[c] Data from Washington State Office of Financial Management (2002).

2.9.2 Land use and ownership

The LDW study area is part of WRIA 9 (Map 2-2). WRIA 9 extends from the headwaters of the Green River in the Cascade Mountains to the mouth of the Duwamish River at Elliott Bay. Map 2-7 presents current land ownership of properties adjacent to the LDW. Tables 2-7 and 2-8 provide information on land ownership and land use, respectively, within WRIA 9. Over 77% of the land in WRIA 9 is under private ownership, with approximately 8% owned by the federal government and 8% owned by the State of Washington. Approximately 6% of land in WRIA 9 is owned by

local jurisdictions, and a very small percentage of the river basin is owned by tribal entities.

Table 2-7. Land ownership in the Duwamish and Green River Basin (WRIA 9)

Owner	No. of Acres	Proportion
Federal	30,634	8.2%
State	29,512	8.0%
Local	23,980	6.4%
Tribal	319	0.1%
Private	287,911	77.3%
Total	**372,356**	**100.0%**

Source: Ecology (2006c).

Table 2-8. Land use in the Duwamish and Green River Basin (WRIA 9)

Land Use Category	Proportion
Forested	54%
Rangeland	5%
Agricultural	5%
Urban	18%
Barren	10%
Open water	8%
Total	**100.0%**

Source: Ecology (2006c).

Land use within the greater Duwamish drainage basin has changed considerably since the construction of Harbor Island and the LDW nearly 100 years ago. Table 2-9 identifies the changes in land use in the lower 11 miles of the Duwamish River watershed since that time.

Table 2-9. Historical and current land cover from RM 0.0 to RM 11.0

Land Cover Type	Historical Coverage (ac)	Current Coverage (ac)
Forested floodplain	3,385	51.9
Forest terrace	885	na
Shrub	na	27.2
Upland bedrock	72	na
Bare earth and mudflats	na	39.5
Herbaceous vegetation	1.5	157.9
Wetlands	3,252	51.4
Built/developed	0	6,315.9
Open water	699.8	766.5
Lakes/ponds	12.4	0.5

Land Cover Type	Historical Coverage (ac)	Current Coverage (ac)
Mainstream river channel	687.4	766

Source: King County (2005f)

ac – acres

na – not available

RM – river mile

The LDW shoreline is zoned predominantly for recreation, conservancy preservation, and urban industrial land use (ATSDR 2003). A small area on the west side of the LDW between RM 3.1 and RM 3.3 is zoned for single-family residential use (SPU 2007). A shoreline survey of the LDW was completed for the WRIA 9 Steering Committee and SPU in 2004 (Terralogic and Landau 2004). The survey assessed shoreline conditions from RM 6.0 to the northern tip of Harbor Island (including both the East and West Waterway shoreline), and survey results are presented in Table 2-10.

Table 2-10. Results of the shoreline survey conducted for the LDW and the East and West Waterways

Designation	% of Shoreline Surveyed	Total Miles
Shoreline/Bank Structures		
Shoreline armoring – bulkheads	31.93%	7.04
Shoreline armoring – riprap	56.05%	12.36
Concrete boat ramps	0.68%	0.15
Unarmored bank area	11.34%	2.50
Total	**100%**	**22.05**

Source: WRIA 9 Subcommittee and SPU (Terralogic and Landau 2004).

Note: The Lower Duwamish study area for the shoreline survey was defined as the LDW from RM 0.0 to RM 6.0 and the East and West Waterways to the northern tip of Harbor Island. Therefore, this study area includes more than the LDW Superfund site, which extends only to the southern tip of Harbor Island.

Over 85% of the shoreline in the surveyed area was armored (with either bulkheads or riprap). Just over 10% of the shoreline was not armored, and a very small portion (less than 1%) was covered by concrete boat ramps.

2.9.3 Site use

Approximately 43% of the Duwamish estuary subwatershed (extending from RM 11.0 to Elliott Bay (Map 2-2) is used for commercial/industrial purposes, and approximately 39% is residential (King County 2005f). Recreational areas (such as parks), vacant land, and road and utility rights-of-way also exist within the Duwamish estuary subwatershed. Although recreational use may increase at some point in the future, this area is anticipated to remain primarily commercial, industrial, and residential in use.

2.9.3.1 Commercial and industrial site use

Land use, zoning, and land ownership within the LDW corridor are consistent with an active industrial waterway. The LDW provides a navigational corridor for movement of raw materials and products used by local facilities. Many of the industrial and commercial facilities on the LDW operate year-round vessel schedules.

The Muckleshoot Tribe currently conducts seasonal commercial netfishing operations on the LDW as well as within the East and West Waterways, operating out of a facility located south of the 1st Avenue S Bridge on the east side of the LDW. The LDW is part of the Muckleshoot Tribe's Usual and Accustomed (U&A) fishing grounds; consequently, they are permitted by federal law to harvest seafood from this area for tribal use and commercial sale (St. Amant 2007).

2.9.3.2 Recreational site use

The LDW is not a major area for recreational use compared to other water bodies in and around Seattle (King County 1999e). However, several public access points from which people may enter the LDW for recreational purposes exist. In the human access survey conducted along the LDW shoreline in 2005 (Windward 2005h), 93 commercial/industrial, residential, and public properties along the LDW were surveyed for human access and use potential. Of these 93 properties, 27 sites were identified as potential human access sites: 7 on the eastern shore of the LDW, 18 on the western shore, and 2 at the southern end of Harbor Island. Most of these access points were on County or City public access grounds or located at the ends of streets. Many of the access points were elevated above the water surface and separated from the sediment by steep banks covered by riprap or blackberry bushes.

Three marinas are located in the LDW study area. The human access survey (Windward 2005h) identified 17 additional locations used for the launching or pullout of hand-powered boats or motorboats. In addition, 8 sites along the LDW have been used for swimming, and 10 have been used for picnicking. The frequency of recreational activities at these sites may increase in the future as ongoing remedial efforts and habitat restoration projects are completed, but such activities are likely to remain limited as a result of the active industrial and commercial use of the LDW and the availability of other, nearby areas that provide superior recreational opportunities.

Fishing, crabbing, and shellfishing activities have been reported at 34 sites along the LDW shoreline, and individuals have also been observed fishing off the sides of barges moored in the LDW (Windward 2005h). The access survey (Windward 2005h) and a clam abundance survey (Windward 2004c) also discussed the potential for fishing and clamming activities to occur throughout the LDW study area. However, the clam survey did not assess areas farther upstream than RM 3.5; the potential for clamming (and the relative abundance of clams) is currently unknown upstream of this point (Windward 2004c). Through treaty rights, tribal members have access to both public and private property and can harvest clams and other shellfish along the

entire shoreline of the LDW (Windward 2007b). In addition, the navigable waters of the LDW are generally owned by the Port and are accessible to the public by boat.

Several surveys have documented relatively high seafood consumption among several Asian and Pacific Islander (EPA 1999a) and Native American (Suquamish Tribe 2000; Toy et al. 1996) populations within the Puget Sound region, some of whom may fish within the LDW for recreational or subsistence purposes. A review of seafood consumption surveys for the Elliott Bay/Duwamish and larger Puget Sound area was conducted by ESG (1999). The review indicated that the potentially exposed population of fish consumers in the LDW study area consists of a diverse group of people that fish both from shore and boat. In 1997, a fishing survey was also completed by King County (1999c). This intensive survey was conducted for 2 months in the summer of 1997, the period of highest use, and showed that use in the LDW was significantly lower than in other Puget Sound sites. However, people fish seasonally for salmon in the LDW, and a relatively high number of individuals fished/crabbed from Spokane Street Bridge, just north of the LDW. Specifically, seafood has been harvested from T-105 within the LDW and from the Spokane Street Bridge just north of the LDW adjacent to Harbor Island. Seafood consumption rates are discussed in greater detail in the baseline HHRA (Appendix B).

The Washington State Department of Health (WSDOH) issued a seafood consumption advisory for the LDW in July 2002, recommending that people limit their consumption of resident fish and shellfish from the LDW. In 2005, WSDOH updated the advisory with the recommendation that no resident fish or shellfish be consumed from the LDW (WSDOH 2005), and this advisory is still in effect. WDFW is responsible for educating anglers and others who harvest seafood from the LDW of the advisory; however, WDFW does not enforce the advisory with citations or penalties (Windward 2007b). WSDOH and the Public Health – Seattle & King County also recommend that neither fish nor shellfish be collected from any urban shorelines along Elliott Bay, and commercial shellfish harvesting is restricted because of high counts of fecal coliform bacteria (King County 1999d).

2.9.3.3 Residential site use

Two mixed residential/commercial neighborhoods are located adjacent to the LDW. The South Park neighborhood, at the southern edge of the Seattle city limit, borders the west bank of the LDW and encompasses approximately 300 m (984 ft) of residential shoreline (Green-Duwamish Watershed Alliance 1998). The Georgetown neighborhood, located east of the LDW and E Marginal Way S, is separated from the LDW by several commercial facilities, although access to the LDW by foot from this neighborhood is possible. Based on data from the 2000 census, the combined residential population of the South Park and Georgetown neighborhoods is 4,900 (US Census Bureau 2002; ATSDR 2003). During the 15 years prior to 2003, the population

of the Georgetown neighborhood decreased slightly, while the population of the South Park neighborhood increased by over 30% (ATSDR 2003).

2.9.3.4 Miscellaneous site use

Some activities in the LDW do not clearly fall within the use categories previously discussed. At least 12 habitat restoration projects are ongoing along the LDW, with several more planned for the near future (Windward 2005h; NOAA 2007a; EPA 2006d; WRIA 9 Steering Committee 2002). These activities include projects at Herring's House Park, T-107, T-105, the Coastal America restoration site, the Diagonal Avenue S/T-108 public shoreline access area, Hamm Creek, Cecil B. Moses Park, the GSA marsh area, the Muckleshoot Tribe restoration area near the Upper Turning Basin, the 8th Ave S street end/Gateway North restoration area, the S Portland street end park, and the S 112th St mitigation site.

Other activities include the Duwamish Tribe's frequent use of Herring's House Park to launch canoes and for various cultural events (Windward 2005h). The Duwamish Tribe also uses T-107 for monthly gatherings, children's dance group practices, and large annual cultural gatherings. There are plans for the Duwamish Tribe to build a longhouse and cultural center near T-107 and Herring's House Park, which would likely increase their use of these two areas along the LDW.

2.10 SUMMARY

The LDW study area consists of the downstream portion of the Duwamish River, excluding the East and West Waterways. The LDW is a well-stratified estuary that has been extensively modified over the last 100 years.

The LDW serves as a major shipping route for bulk and containerized cargo, and the shoreline along the majority of the LDW has been developed for industrial and commercial operations. Common shoreline features include constructed bulkheads, piers, wharves, sheet piling walls, buildings that extend over the water, and steeply sloped banks armored with riprap or other fill material.

To facilitate navigation and economic development, the LDW was straightened and has been frequently dredged in many areas. The excavated material was used to fill the old channel areas and the lowlands to bring them above flood levels. Subsequent filling of the lowlands for continued development resulted in a surficial layer of fill over most of the lower Duwamish Valley. Most of the upland areas adjacent to the LDW have been heavily industrialized for many decades.

Throughout the 1900s, the watershed area and flow volumes to the Duwamish River were reduced by about 70% as a result of the diversion of the river's tributaries. Today, the Green River is the primary source of water for the Duwamish River. Surface water runoff within the LDW area also contributes to flow within the LDW, from sources such as SDs, tributary creeks, CSOs, and non-point inputs; these sources

are expected to be less than 1% of the total discharge, even during peak flow events. Annual average downstream flow for the Duwamish River, measured at the Auburn gaging station, ranged from 4.3 to 329 m^3/s (152 to 11,600 cfs [the record high]) between 1962 and 2004.

Water circulation within the LDW is driven by tidal actions and river flow. Freshwater moving downstream overlies the tidally influenced saltwater entering the system. Typical of tidally influenced estuaries, the LDW has a relatively sharp interface between the freshwater outflow at the surface and saltwater inflow at depth.

Tidal action in the LDW greatly affects groundwater flow direction, flow rate, and overall water quality. Under typical conditions, the hydraulic gradient is toward the LDW. At low tide, the hydraulic gradient between the groundwater system and the LDW is typically at its highest, causing the flow of local groundwater into the LDW. At high tide, the hydraulic gradient can reverse. Tidal action causes a continuous oscillation of groundwater within the effective zone of influence. The periodic reversal in flow direction caused by tidal influences can decrease the migration of dissolved chemicals in groundwater.

Sediment composition varies greatly throughout the LDW, ranging from sands to mud (fine-grained silt and clays), depending on the source of the sediments and the local current velocity. Sediments in the navigation channel near the Upper Turning Basin are predominantly sands, whereas sediments toward the mouth of the river are predominantly fine-grained silts.

Ecological habitats of the LDW have been modified extensively since the late 1800s through hydraulic changes, channel dredging, the filling of surrounding floodplains, and the construction of overwater structures, levees, dikes, and other bank stabilization structures. The remnants of natural meanders along the LDW, several of which are now used as slips, and the area west of Kellogg Island are the only evidence of the river's original winding course. Small intertidal areas of marsh and unvegetated marsh habitat in the LDW have become the focus of habitat restoration activities.

Benthic invertebrate communities observed in the LDW consisted of 670 taxa, representing 178 families in 13 phyla. Typical of estuarine environments, the benthic invertebrate community was dominated by annelid worms, mollusks, and crustaceans. The most abundant large epibenthic invertebrates crabs and shrimp. The LDW is inhabited by numerous anadromous and resident fish species. Fifty-three resident and non-resident fish species were captured in the LDW during sampling events conducted for the RI. The aquatic and semi-aquatic habitats of the LDW support a diversity of wildlife species. Formal studies, field observations, and anecdotal reports indicate that up to 87 species of birds and 6 species of mammals use the LDW at least part of the year to feed, rest, or reproduce (i.e., raccoons, muskrats, and river otters, harbor seal, California sea lion, and harbor porpoise). Fifteen species

identified in the LDW study area are listed under either the ESA or by WDFW, as candidate species, threatened species, or species of concern (see Table 2-5).

Approximately 43% of the Duwamish estuary subwatershed is used for commercial/industrial purposes, and approximately 39% is residential. Two residential neighborhoods, South Park and Georgetown, are located adjacent to the LDW. The Muckleshoot tribe currently conducts seasonal commercial netfishing operations on the LDW, which is part of their U&A fishing grounds.

The LDW is not a major area for recreational use compared to other water bodies in and around Seattle. However, several public access points from which people may enter the LDW for recreational purposes exist. Although recreational use may increase at some point in the future, this area is anticipated to remain primarily commercial, industrial, and residential in use.

In 2005, WSDOH recommended that no resident fish or shellfish be consumed from the LDW (WSDOH 2005); this advisory is still in effect.

3 Sediment Dynamics

The Phase 1 RI (Windward 2003) summarized existing information on sediment dynamics in the LDW and yielded a basic understanding of the stability of bedded sediments and sediment transport in the system, which was used to develop a preliminary conceptual site model (CSM). This CSM was considered to be preliminary because sufficient site-specific information and data were not available during the Phase 1 RI to confirm it. Based on the data available at the time of the Phase 1 RI, the preliminary CSM postulated:

- The LDW is net depositional on a site-wide scale.
- On a local scale, the sediment bed is either aggrading (i.e., sediment bed elevation is increasing as a result of sediment deposition) or in dynamic equilibrium (i.e., sediment bed elevation is neither increasing nor decreasing).
- Bed erosion occurs only episodically and over small spatial scales.

As described in the Phase 2 work plan (Windward 2004h) and the sediment transport data report (Windward and QEA 2005), additional data were collected in 2004 and 2005, consistent with EPA (2005b) guidance, to support development of a concise description of LDW sediment transport processes that affect sediment stability in the system and to refine the Phase 1 CSM. These additional data and supporting analyses are summarized in Section 3.1 and presented in detail in the sediment transport analysis report (STAR) (Windward and QEA 2007).

Based on information presented in the STAR (Windward and QEA 2007), LDWG, EPA, and Ecology concluded that additional sediment transport modeling (STM) was warranted because some of the key questions about sediment dynamics could not be satisfactorily answered by the empirical approach described in Section 3.1. This modeling has been completed and is summarized in Section 3.2 and presented in detail in the STM report (QEA 2007).

The analyses conducted during the STM, in conjunction with empirical and ship-induced bed scour analyses from the STAR, have resulted in an improved understanding of sediment transport processes in the LDW. Results and findings from the major components of these analyses were integrated and synthesized to produce a clear and concise picture of sediment transport in the LDW, as summarized in Section 3.3. This information was also used to update the CSM, as summarized in Section 3.4, and will be used in subsequent investigations that are aimed specifically at assessing the effectiveness of various remedial alternatives in the FS.

3.1 EVALUATION OF SEDIMENT DYNAMICS

Several types of empirical data were evaluated in the STAR (Windward and QEA 2007) to assess sediment dynamics using a weight-of-evidence approach. Those data are discussed in the sections that follow:

- Sediment physical properties (e.g., sediment particle size) (Section 3.1.1)
- Sediment cores used for age dating (Section 3.1.2)
- Bathymetry (Section 3.1.3)
- Physical and chemical markers in sediment cores for estimating net sedimentation rates (Section 3.1.4)
- Sediment cores used to evaluate erosion properties (Section 3.1.5)

The sediment cores discussed in Sections 3.1.2 and 3.1.5 were collected specifically for the evaluation of sediment dynamics in the RI; the data described in the other sections were collected for other purposes. These data were used in modeling analyses that are described in Sections 3.1.6 (hydrodynamic model) and 3.1.7 (ship-induced bed scour model). These modeling analyses are not the same as the STM application discussed in Section 3.2.

3.1.1 Sediment physical properties

The physical characteristics of bedded sediment affect sediment transport processes that occur in a given area. A large amount of historical data exists that describe physical properties of sediment in the LDW (i.e., sediment particle size, TOC). Additional physical property data were derived from the sediment cores collected as part the STAR (Windward and QEA 2007).

Historical surface sediment particle size data, which were available from approximately 1,250 locations sampled since 1990, were evaluated for spatial trends in the STAR (Windward and QEA 2007). The data were variable in the bench areas and navigation channel, with no apparent spatial structure (e.g., downstream fining). Sediment in the navigation channel tended to have a higher clay-silt content than did sediment in the bench areas. Typical clay-silt content in the navigation channel ranged from about 60 to 80%. Clay-silt content was more variable in the bench areas, ranging from about 30 to 80%.

Additional sediment particle size data were collected in December 2004 as part of the STAR. These data were generated using a laser particle size counter, in contrast to the sieve and pipette method used to generate the historical data. The laser particle size counter generates an almost continuous particle size distribution, using approximately 100 different particle size categories, rather than the 15 particle size categories generated from the sieve and pipette method. Consequently, the data from the 19 sediment cores collected in December 2004 are discussed separately from the historical

data because different summary statistics can be generated from the much larger number of particle size categories.

The particle size distributions in the top 5 cm of the Sedflume[9] cores collected at 19 locations in the LDW (Map 3-1) are summarized in Table 3-1. Eighteen cores were classified as sandy silt, and one core (Sf-5) was classified as silty sand (Windward and QEA 2005, 2006). A statistical analysis of the 18 sandy silt cores indicated that the average median particle diameter (D_{50}) was 30 µm, and the average contents of clay, silt, and sand were 6%, 69%, and 28%, respectively (Table 3-1). In general, variability in particle size in these cores was relatively low. For example, about 90% of the D_{50} values for individual samples varied between 10 and 50 µm (silt). The single core classified as silty sand (Sf-5) had a D_{50} of 329 µm and a sand content of 79%.

In general, the 18 sandy silt cores had more silt and less sand than surface sediment samples contained in the surface sediment dataset, which averaged approximately 41% silt and 41% sand. These differences reflect one of the specific objectives of the Sedflume data collection, which was to target bench areas expected to be depositional. The LDW-wide surface sediment grain size dataset is much more comprehensive and larger than the Sedflume dataset and reflects many different types of sediments in the LDW.

Table 3-1. Particle size data for surface layer (0 to 5 cm) of Sedflume cores

Sediment Core ID	D_{10} (µm)[a]	D_{50} (µm)[a]	D_{90} (µm)[a]	Clay Content (%)	Silt Content (%)	Sand Content (%)
Sf-1	2	12	60	9	82	9
Sf-2	3	24	115	6	72	22
Sf-3	3	19	117	7	75	18
Sf-4	3	19	223	7	71	22
Sf-5	11	329	739	2	19	79
Sf-6-R1	3	21	122	7	73	20
Sf-6-R2	4	39	184	5	60	35
Sf-7	4	41	1,112	5	55	40
Sf-8	3	28	235	6	68	26
Sf-9	5	79	402	4	43	53
Sf-10	3	23	89	6	78	16
Sf-11	4	94	653	5	41	54
Sf-12	4	24	139	5	73	22
Sf-13	3	21	97	7	76	17
Sf-14	3	22	161	7	70	23

[9] The particle size data discussed in this section were collected as supporting data from cores tested in a device called a Sedflume, which was used to measure gross erosion rates over a range of shear stresses, as discussed in greater detail in Section 3.1.5.

Sediment Core ID	D_{10} (μm)[a]	D_{50} (μm)[a]	D_{90} (μm)[a]	Clay Content (%)	Silt Content (%)	Sand Content (%)
Sf-15	2	16	73	8	81	11
Sf-16-R1	3	20	93	7	76	17
Sf-16-R2	2	18	82	8	78	14
Sf-17	3	19	88	7	78	15
Average[b]	3	30	225	6	69	25
Standard deviation[b]	0.8	22	265	1.3	12	13
95% confidence interval[b]	2 – 4	19 – 41	93 – 357	5 – 7	63 – 75	17 – 31
Minimum[b]	2	12	60	4	43	9
Maximum[b]	5	94	112	9	82	54

Source: Windward and QEA (2005; 2006)

[a] D_{10}, D_{50}, and D_{90} values are the 10th, 50th (median), and 90th percentiles of the particle diameter data for that core.

[b] Statistics exclude data for Sf-5, which was classified as silty sand; the other 18 cores were classified as sandy silt.

ID – identification

Wet density data do not exist in the historical baseline surface sediment dataset, but this parameter was analyzed in the Sedflume cores. Wet densities of the 18 sandy silt cores (excluding Sf-5) ranged from about 1.2 to 1.7 g/cm^3. The single silty sand core (Sf-5) had a higher wet density (1.8 to 1.9 g/cm^3). In general, wet density increased with depth within a core because of consolidation effects. Similar to the particle size distributions, wet densities in the Sedflume cores were relatively uniform throughout the LDW. A correlation analysis was conducted between wet density and particle size. Minimal correlation existed between these two bulk bed properties for the 18 sandy silt cores. For example, the correlation coefficient (r) between wet density and D_{50} values for the 18 sandy silt cores was 0.42.

TOC content has been analyzed in over 1,300 surface sediment samples collected throughout the LDW since 1990. In general, the data were spatially variable without an evident longitudinal trend. The median TOC content in the LDW was 1.8% and the range was 0.1 to 6.0%. The TOC content was elevated (i.e., > 4%) in a few isolated areas (e.g., western bench area near RM 0.5 and eastern bench area near RM 3.5).

3.1.2 Geochronology analysis

To facilitate a better understanding of how net sedimentation rates in bench areas compared to rates previously estimated for the navigation channel, a geochronology study was conducted as part of the STAR (Windward and QEA 2007). This study consisted of an age-dating analysis of sediment cores collected from the LDW in December 2004 using the radioisotopes cesium-137 (^{137}Cs) and lead-210 (^{210}Pb).

Fourteen sediment cores were collected from bench areas within the LDW in 2005 (Map 3-2). Each core was segmented in 1-cm intervals, and every fifth segment within the top 81 cm (i.e., 0 to 1 cm, 5 to 6 cm, etc., to 80 to 81 cm) was submitted for ^{137}Cs, ^{210}Pb, TOC, and total solids content analyses; remaining segments were archived. After the initial results were reviewed, additional intervals, some as deep as 111 cm, were analyzed in four cores (Sg-3, Sg-6, Sg-7, and Sg-13) in an attempt to detect a ^{137}Cs peak at a depth greater than 81 cm.

Estimated net sedimentation rates based on the geochronology analysis are presented in Table 3-2. A detailed explanation of the calculation methods is provided in Windward and QEA (2007). Direct comparisons of the net sedimentation rates generated from the two different radioisotopes were possible for four cores: Sg-2, Sg-5a, Sg-7, and Sg-10. For two of the four cores (Sg-2 and Sg-5a), the ^{137}Cs net sedimentation rate fell within the range of the ^{210}Pb net sedimentation rate. In the other two cores (Sg-7 and Sg-10), net sedimentation rates determined from the ^{210}Pb data were lower than those estimated using the ^{137}Cs data.

Table 3-2. Comparison of net sedimentation rates from ^{137}Cs and ^{210}Pb analyses

Sediment Core ID	Mud Line Elevation (ft MLLW)	Range of Estimated Net Sedimentation Rates from ^{137}Cs Analysis (cm/yr)		Range of Estimated Net Sedimentation Rates from ^{210}Pb Analysis (cm/yr)
		Via ^{137}Cs Peak	Via First Presence of Detectable ^{137}Cs	
Sg-1a	-17.7	0.9 – 1.1	1.2 – 1.3	ne
Sg-2	-3.5	ne	0.5 – 0.6	0.4 – 1.1
Sg-3	-21.4	1.9 – 2.1	nd	ne
Sg-4	-12.8	1.6 – 2.0	nd	ne
Sg-5a	-16.5	1.4 – 1.6	nd	0.7 – 9.3
Sg-6	-17.9	2.5 – 2.7	nd	ne
Sg-7	-16.1	1.9 – 2.1	nd	0.5 – 1.1
Sg-8	-10.3	ne	ne	ne
Sg-9	+0.4	0.3 – 0.5	0.8 – 0.9	ne
Sg-10	-11.0	1.6 – 1.8	nd	0.2 – 1.0
Sg-11c	-1.3	ne	ne	ne
Sg-11b	+0.6	ne	0.6 – 0.7	ne
Sg-12	-11.1	> 2.0	nd	ne
Sg-13	-8.3	2.3 – 2.6	nd	ne

Cs – cesium

ID – identification

MLLW – mean lower low water

nd – not detected

ne – not estimated

Pb – lead

yr – year

Net sedimentation rates in the bench areas were estimated to range from 0.2 to > 2.0 cm/yr. In general, net sedimentation rates were greater than 1 cm/yr in the subtidal bench areas and less than 1 cm/yr in the intertidal areas. Thus, net sedimentation rates in the subtidal bench and intertidal areas were lower than those previously estimated for the LDW navigation channel (i.e., 1 to 15 cm/yr between RM 0.0 and RM 1.7, 10 to 25 cm/yr between RM 1.7 and RM 3.4, and 20 to 110 cm/yr between RM 3.4 and RM 4.7) (Windward 2003a). No other consistent spatial trend was apparent in the estimated net sedimentation rates calculated from radioisotope age-dating methods, likely because the data were too sparse to develop spatial relationships.

Several physical, chemical, and biological factors introduce uncertainty into the net sedimentation rates estimated from the radioisotope profiles. These factors include: 1) natural variability in the radioisotope data, 2) variations in sediment physical characteristics, 3) depths to which benthic invertebrates burrow into the LDW sediments (e.g., mixing through bioturbation), 4) physical disturbances of the sediments (e.g., erosion, historical dredging), 5) compression and/or mixing of sediments during core collection/extrusion, and 6) poor sediment recovery rates in core samples. The first four factors are likely the greatest contributors to uncertainty in net sedimentation rates estimated during the geochronology study, primarily because of uncertainty in the extent and magnitude to which these factors vary in the LDW.

3.1.3 Bathymetric analysis

The results of bathymetric surveys conducted in different years were also compared in the STAR (Windward and QEA 2007) to evaluate sediment dynamics. Sediment bed (i.e., mud line) elevations at specific locations were compared to identify areas of potential scour or deposition. A similar analysis was conducted for the Phase 1 RI (Windward 2003a), but additional bathymetric data had been collected since that time.

The bathymetric analysis conducted for the STAR consisted of a comparison of two existing bathymetric datasets collected from a portion of the LDW (between RM 2.6 and RM 4.7) in 2000 and from the entire LDW in 2003, resulting in a qualitative assessment of potential scour and depositional areas. Sediment surface elevations in 2000 were defined using a combination of bathymetric measurements from surveys conducted for USACE[10] and Boeing (Pentec et al. 2001). Sediment surface elevations in 2003 were defined using the acoustic multi-beam bathymetric data collected by David Evans and Associates (Windward and DEA 2004). Because of the higher density of measurements in 2003 (relative to 2000), each measurement from 2000 was paired with the nearest measurement from 2003 and a change in sediment elevation at each paired

[10] Bathymetry charts as contained in USACE file number E-12-2.1-12.3, dated September 15, 2000.

location was computed. Linear interpolation was performed between measurement locations to estimate areas of scour and deposition.

There were several uncertainties associated with positioning and measurement accuracy for the bathymetric surveys (Windward 2003a). The imprecision associated with these uncertainties has the greatest potential effect on estimates for the steep sloped areas of the LDW or at the outermost limits of the survey transects. These uncertainties are further compounded by the fact that the three surveys were performed by different contractors, using different equipment and different methods. Moreover, sediment elevation measurements from the USACE and Boeing surveys in 2000, which were used to define year 2000 conditions in the LDW, were collected along transects spaced at approximately 200-ft intervals, making it difficult to find matching points in the 2003 survey. Based on the uncertainties summarized above, any change in elevation less than 12 in. (30 cm) at a particular point was not considered significant.

Measurement inaccuracies and other uncertainties described above made it difficult to make reliable conclusions regarding fine-scale sediment transport based on this analysis. Nonetheless, areas of scour and deposition in excess of 12 in. (30 cm) were noted in these comparisons. However, most of these areas were located along the edge of the navigation channel (which has been routinely dredged), in the Upper Turning Basin, or at the outermost limits of the survey data; these areas are likely associated with measurement inaccuracies that are in excess of the criterion used in this analysis (i.e., 12 in. [30 cm]).

3.1.4 Additional analysis of net sedimentation rates

In the STAR (Windward and QEA 2007), net sedimentation rates were estimated from the geochronology cores, as described in Section 3.1.2, and evaluated along with several independent chemical and physical lines of evidence including historical information and sediment chemistry cores collected in 2006. The purpose of these additional analyses was to estimate net sedimentation rates for areas other than those where geochronology cores were collected and to validate the net sedimentation rates calculated from the geochronology study.

The site data used in the additional analyses included chemistry and stratigraphy data from 56 subsurface sediment cores collected in 2006 and from historical subsurface sediment cores; grain size distribution data; dredging records; chemical spill, industrial, and regional discharge records; and bathymetric data. These data provided a set of time markers that were found at different depths in the sediment bed at various locations in the LDW. After assigning a date or time period and specific depth for a particular marker, the net sedimentation rate was estimated from that marker. This estimate represented the average rate of net deposition for the time period between the time marker and core collection.

Nine different physical and chemical time markers were evaluated, although not all markers could be evaluated for a given core (Table 3-3). The conclusion of this time marker analysis was that net sedimentation rates were spatially variable, with the highest rates in the navigation channel (greater than 2 cm/yr), moderate rates in the subtidal bench areas (less than 2 cm/yr), and the lowest rates in the intertidal bench areas (less than 0.5 cm/yr). Although geochronology cores, as summarized in Section 3.1.2, were not co-located with the sediment cores for which the empirical sedimentation rates were estimated, the geochronology-based estimates generally agreed with the rates estimated from the physical and chemical markers, particularly at locations where the two core locations were close together. Additional details on how each marker was evaluated are provided in Attachment F of the STAR (Windward and QEA 2007).

Because of the consistency in the findings of the analysis of geochronology cores and the evaluation of other empirical lines of evidence, it was possible to combine the net sedimentation rates estimated from the two methods to develop a large-scale view of LDW net sedimentation, including a map showing the spatial distribution of estimated net sedimentation rates in the LDW within specific "sedimentation areas" (Map 3-3). This map delineated sedimentation areas within the LDW using five categories of net sedimentation rates: < 0.5 cm/yr, 0.5 to 1.0 cm/yr, 1.0 to 1.5 cm/yr, 1.5 to 2.0 cm/yr, and > 2.0 cm/yr. Results of the analysis provided a consistent picture of large-scale net sedimentation processes in the LDW.

Table 3-3. Net sedimentation rates in the LDW estimated from physical and chemical time markers

Subsurface Core ID	Approx. River Mile	Estimated Net Sedimentation Rates (cm/yr) Determined From Time Markers and Event Horizons[a]								
		Physical[b]			Chemical (1-ft, 2-ft Intervals)			Chemical (6-in. Intervals) from a Subset of 2006 Cores		
		Lower (Native) Alluvium[c]	Hanson Dam Const.[d]	Dredge Horizon[e]	Lead/ PCB/ Phthalate Introduction	PCB Peak Usage/ Spill	Source Control Efforts	PCB Introduction	PCB Peak Usage/ Spill	Source Control Efforts
		1916	1961	Variable	1920/1935/ 1950	1960/1974	1980	1935	1960/1974	1980
RI 2006 Cores										
SC-1	0.0	0.9			1.7			1.1	0.9	0.9
SC-2	**0.1**									
SC-3	0.1	0.4								
SC-4	0.2	1.1			1.7		1.2			
SC-5	0.2	0.7	0.5		0.5, 0.9					
SC-6	0.3	2.6	3.0		2.3		2.3	2.6	2.3	2.7
SC-7	0.3				0.7					
SC-8	0.4					3.3	1.2			
SC-9	0.5		1.8	1.5						
SC-10	0.5	2.4	2.7	2.4		2.9				
SC-11	0.5		0.5		0.4					
SC-12	0.6	2.3	1.8		2.9	2.0	2.3, 1.2	2.6	2.0	2.1
SC-13	0.9				1.1			1.1 to 2.1		
SC-14	0.9	2.9			4.0					
SC-15	0.9	2.5	1.4		3.0	4.8				
SC-16	0.9	2.4			3.8	2.9	2.3, 1.2			
SC-17	1.0					2.9				
SC-18	1.0	1.9		1.5, 1.9	0.7, 0.9, 1.1					
SC-19	1.0	3.0	4.7		3.4	4.3				
SC-20	**1.0**									
SC-21	1.0	3.3	3.4		3.5	4.9	2.3			
SC-22	**1.1**									
SC-23	1.3		3.3		4.3	4.8	4.7	3.4	4.8	3.3
SC-24	1.2	1.1	0.7		0.7, 0.9					
SC-25	1.3	2.0 to 2.5			2.5, 3.0		1.2			
SC-26	1.4									

Subsurface Core ID	Approx. River Mile	Estimated Net Sedimentation Rates (cm/yr) Determined From Time Markers and Event Horizons[a]								
		Physical[b]			Chemical (1-ft, 2-ft Intervals)			Chemical (6-in. Intervals) from a Subset of 2006 Cores		
		Lower (Native) Alluvium[c]	Hanson Dam Const.[d]	Dredge Horizon[e]	Lead/ PCB/ Phthalate Introduction	PCB Peak Usage/ Spill	Source Control Efforts	PCB Introduction	PCB Peak Usage/ Spill	Source Control Efforts
		1916	1961	Variable	1920/1935/ 1950	1960/1974	1980	1935	1960/1974	1980
SC-27	1.4	1.5 to 2 .6						1.4	1.2	0.9
SC-28	**1.4**									
SC-29	1.4	0.6	0.4							
SC-30	1.6	1.1								
SC-31	1.7			12.2	1.0, 1.2, 1.5					
SC-32	1.7	1.7 to 2.4	1.9		2.0, 2.5					
SC-33	1.9	2.9			3.0, 3.8			2.6	0.8 to 1.7	0.9, 1.4
SC-34	1.9		2.2							
SC-35	2.0		3.5	2.8, 3.7						
SC-36	2.1	2.8	2.2							
SC-37	2.1	1.8	1.8		2.0, 2.6	1.0	2.3			
SC-38	**2.1**									
SC-39	2.2	2.9								
SC-40	2.2	0.7								
SC-41	2.4	2.6								
SC-42	2.5		2.7							
SC-43	2.6	3.0	0.5							
SC-44	2.7				1.4			1.3	0.5	0.3
SC-45	**2.8**									
SC-46	2.7	2.3		7.6, 1.8						
SC-47	3.1	1.0			1.3, 1.4, 2.2	1.0	1.2			
SC-48	3.3				0.4 to 0.5					
SC-49	3.5		2.4		4.3					
SC-50	3.8	0.9			1.0, 1.2, 1.5					
SC-51	3.8	1.3			1.6			1.2 to 1.5, 0.7		0.6
SC-52	3.9				0.5, 0.7, 0.9					
SC-53	4.2	3.1	3.3							
SC-54	4.3	1.8	2.7							
SC-55	4.9	1.0	0.3							

Subsurface Core ID	Approx. River Mile	Estimated Net Sedimentation Rates (cm/yr) Determined From Time Markers and Event Horizons[a]								
		Physical[b]			Chemical (1-ft, 2-ft Intervals)			Chemical (6-in. Intervals) from a Subset of 2006 Cores		
		Lower (Native) Alluvium[c]	Hanson Dam Const.[d]	Dredge Horizon[e]	Lead/ PCB/ Phthalate Introduction	PCB Peak Usage/ Spill	Source Control Efforts	PCB Introduction	PCB Peak Usage/ Spill	Source Control Efforts
		1916	1961	Variable	1920/1935/ 1950	1960/1974	1980	1935	1960/1974	1980
SC-56	4.7				0.8 to 1.0					
Historical Cores										
B3 (T105 1985)	0.2	4.9	5.1							
DUD006 (D/D 1994)	0.5					1.9; 3.1	2.7			
DR18 (PSDDA99)	1.8	2.2	3.2							
DR39 (PSDDA99)	2.2	1.5								
SC11 (Slip 4 2004)	2.8	1.5	2.2							
S3 (PSDDA98)	3.8	3.0	3.3							
HI-NR-01	< 0						1.4			
HI-NR-02	< 0						1.3			
HI-NR-03	< 0						2.5			
HI-NR-04	< 0						1.3			

Source: Analysis conducted by RETEC, as reported in Windward and QEA (2007), Appendix F.

Notes: See subsurface data report (Windward and RETEC 2007) for core logs. Ranges or multiple estimates are given in some table cells where chemical markers for different chemicals could be observed.

[a] All net sedimentation rate estimates are based on recovered core depths and do not include uncertainty.

[b] Sediments were grouped into three stratigraphic units identified for the LDW, primarily based on density, color, sediment type, texture, and marker bed horizons. The three sediment stratigraphy units were identified as follows: Recent, Upper Alluvium, Lower (Native) Alluvium.

[c] Lower (Native) Alluvium is defined by top of dense sand unit.

[d] Hanson Dam construction is defined by the presence of organic silt.

[e] Dredge event rates show rate for event to top of core and intra-marker rate from stratigraphic marker to dredge effects marker.

ID – identification LDW – Lower Duwamish Waterway PCB – polychlorinated biphenyl

Blank cells indicate that markers were not present or core was not clearly indicative of a strong time marker.

Bold text, gray shading indicate no strong marker; therefore, no calculation was made for the core.

3.1.5 Site-specific erosion properties

Erosion rates as a function of shear stress and depth in the sediment bed were evaluated using 19 sediment cores collected from 17 locations in the LDW (Map 3-1) in December 2004 as part of the STAR (Windward and QEA 2004, 2005). Fourteen of these cores were collected from bench areas, and five cores were collected from the navigation channel. Many of the cores collected from the bench areas for erosion property analysis were located relatively close to the geochronology cores discussed in Section 3.1.2. Duplicate cores were collected at two locations (Sf-6 and Sf-16), one inform a bench area and one from the navigation channel.

Erosion rates as a function of depth in the bed and shear stress were measured in the laboratory over the top 30 cm of each core using a Sedflume (Windward and QEA 2004), which has proven useful for measuring the erosion properties of sediments at many other sites (McNeil et al. 1996). The principal purpose of the Sedflume is to measure gross erosion rates (E_{gross}) over a range of applied shear stresses (τ)[11] at various depths in a sediment core.
Jones (2000) presented the following equation, which was used to quantify the gross erosion rate:

$$E_{gross} = A\,\tau^{n} \text{ when } \tau > \tau_{cr} \qquad \textbf{Equation 3-1}$$
$$E_{gross} = 0 \text{ when } \tau \leq \tau_{cr}$$

where:

E_{gross}	=	gross erosion rate (cm/s)
A	=	site-specific parameter that may be dependent on the sediment dry density (ρ), as well as on other sediment bed properties
τ	=	bottom shear stress (Pa)
n	=	site-specific exponent (unitless)
τ_{cr}	=	critical shear stress (Pa)

The critical shear stress (τ_{cr}), defined as the shear stress at which a small but measurable rate of erosion is observed, was estimated from Sedflume erosion rate data. For Sedflume studies performed at other sites (Jepsen et al. 2001; Jones 2000; McNeil et al. 1996), the critical erosion rate was set at 10^{-4} cm/s, a value that consistently corresponded to the initiation of erosion. Thus, a critical erosion rate of 10^{-4} cm/s was used in this study (Windward and QEA 2007). The critical shear stress was calculated by rearranging Equation 3-1:

$$\tau_{cr} = \left(\frac{E_{gross}}{A}\right)^{\frac{1}{n}} \qquad \textbf{Equation 3-2}$$

where E_{gross} was set equal to 10^{-4} cm/s.

[11] Shear stress is the force that acts parallel to a plane. In this application, the force is hydrodynamic (from moving water) and the plane is the sediment bed surface.

The data generated from the Sedflume analysis was used to assess the spatial variability (i.e., horizontal and vertical) of the erosion rate parameters (A, *n*, and τ_{cr}). The erosion rate relationship (i.e., Equation 3-1) was similar among various cores within a specific 5-cm depth layer (Windward and QEA 2007). This similarity suggested that the subsections of cores could be separated into different groups for a given 5-cm depth layer. This grouping was conducted in an attempt to evaluate whether LDW-wide generalizations could be made using these data. Sixteen core groups were created, each consisting of two to nine core segments. Two core segments were considered outliers and were not included in any core group.

The results for the individual cores within a group were combined to determine average erosion parameter values for that group. The average exponent (n) value for a group was the arithmetic average of the n values for the cores within the group. The average proportionality constant (A_{ave}) was determined by calculating the log-average value:

$$\log(A_{ave}) = (1/K) \sum \log(A_k) \qquad \textbf{Equation 3-3}$$

where K was the total number of cores in the group.

The vertical distribution of τ_{cr} values for all of the core groups is shown in Figure 3-1. In general, critical shear stress increased with depth in the sediment bed, as expected because of bed consolidation. The group-averaged A, n, and τ_{cr} values determined from this analysis were used in developing the horizontal and vertical spatial distributions of these parameters for input to the LDW STM (Section 3.2).

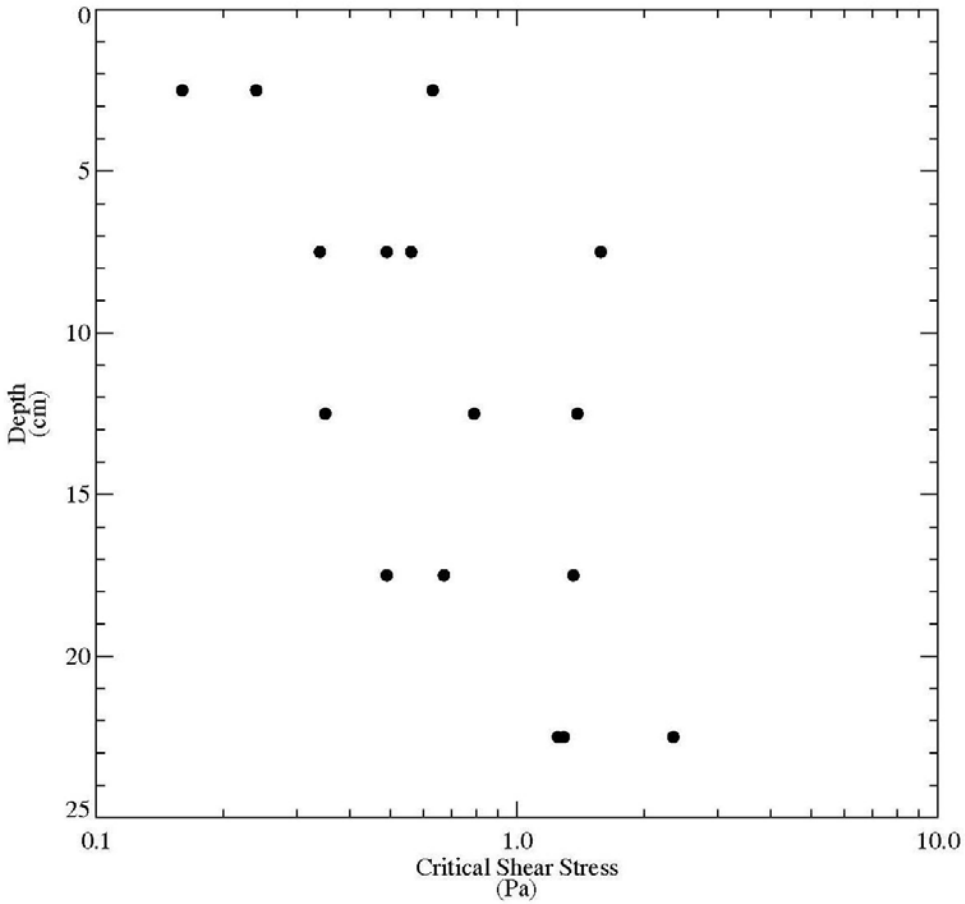


Figure 3-1. Vertical distribution of group-averaged critical shear stress values

3.1.6 Potential effects of natural events

The potential effects of an episodic high-flow event on bed stability in the LDW were evaluated through the use of a hydrodynamic model in the STAR (Windward and QEA 2007). The hydrodynamic model was used to simulate circulation in the LDW for a range of high-flow events (e.g., events with return periods of 2, 10, and 100 yrs) and a range of tidal conditions (i.e., spring and neap tides). Results of the hydrodynamic simulations were used to develop inferences and hypotheses about bed scour in the LDW during high-flow events and the spatial distribution of excess shear stress (i.e., bed shear stress minus critical shear stress).

High-flow events in the LDW are episodic in nature, with typical durations of hours to days. These episodic events may cause bed scour or deposition to occur at specific locations in the LDW. Although it is possible that a particular high-flow event may have measurable effects on bed elevation (i.e., either net deposition or erosion) at some locations, the long-term effects of such an event on the depositional or erosional environment at those locations may be relatively minor. Thus, the estimated effects of

high-flow events are not necessarily related to the magnitude of long-term depositional/erosional effects over annual time scales.

Sediment transport during an episodic high-flow event in the LDW is affected by a range of factors, including hydrodynamic circulation, spatial and temporal variations in bed shear stress, deposition, erosion (including bed armoring and consolidation effects), and sediment loads from upstream sources. These factors are the primary determinants of the effects of a high-flow event on bed stability and sediment transport in the LDW. Thus, a quantitative evaluation of sediment transport during a high-flow event requires the application of hydrodynamic and sediment transport models that incorporate all of the processes discussed above.

A three-dimensional hydrodynamic model of the LDW and Elliott Bay was developed during a water quality study of Elliott Bay, the East and West Waterways, and the LDW (King County 1999e). This model was created using the Environmental Fluid Dynamics Code (EFDC). Calibration and validation results indicated that the model simulated hydrodynamic processes in the LDW and Elliott Bay with reasonable accuracy. Dr. Earl Hayter of EPA increased the resolution of the King County model by developing a finer grid structure (Arega and Hayter 2004). The modified model was re-calibrated, as presented in Attachment B of the STAR (Windward and QEA 2007).

The LDW is a saltwater wedge estuary, and its hydrography has been studied as part of various investigations (Harper-Owes 1981; Santos and Stoner 1972). The location of the toe of the saltwater wedge varies depending on the rate of freshwater flow moving downstream and the stage and magnitude of the saltwater tidal flow moving underneath the freshwater. The location of the saltwater wedge not only affects circulation in the LDW but also has an effect on bed shear stress distributions and bed scour during high-flow events. Results from hydrodynamic simulations of various high-flow events have suggested that the LDW may be broadly separated into three hydrodynamic reaches during high-flow conditions (Windward and QEA 2007):

- **Reach 1.** Located downstream of RM 2.0 and includes the saltwater wedge during all flow and tidal conditions.
- **Reach 2.** Extends from RM 2.0 to RM 3.0 and includes the saltwater wedge toe.
- **Reach 3.** Located upstream of RM 3.0, with flow in portions of this reach corresponding to a freshwater tidal river.

Various modeling analyses were conducted in the STAR (Windward and QEA 2007) to evaluate potential bed scour in the LDW during high-flow events (i.e., discharge greater than or equal to the 2-yr high-flow event). The results of these analyses are presented in the STAR (Maps 3-4 and 3-5) and summarized below.

- Over all flow conditions, bed shear stress tended to be higher in the navigation channel than in the bench areas.

- During high-flow events in Reach 1, negligible bed scour was estimated to occur in most of the area downstream of RM 1.8. Between RM 1.8 and RM 2.0, excess shear stress values of 0.4 pascal (Pa) or less were estimated to occur during high-flow events.
- During high-flow events in Reaches 2 and 3 (i.e., upstream of the saltwater wedge):
  - Minor differences were estimated in the general spatial pattern of excess shear stress during ebb and flood tides. Higher bed shear stresses were estimated during spring tide than during neap tides.
  - In general, lower excess shear stresses were estimated in the bench areas than in the navigation channel for a given high-flow event and tidal condition.
  - Within the portions of the bench areas where erosion was estimated to occur, the potential for erosion tended to be highest near the navigation channel and tended to decrease toward the shoreline.
  - Reach 3 was estimated to have higher excess shear stress values than did Reach 2.
- The hydrodynamic energy regime (represented by excess shear stress) during high-flow events was not estimated to have a significant effect on the erosion rate properties or geochronology at a particular location within the bench areas of the LDW. No correlation was apparent between the structure (i.e., vertical distribution of erosion rate or physical properties) of either a Sedflume core or a geochronology core and the hydrodynamic energy regime at the core location.

Although areas of potential erosion were identified through this analysis, the depth of bed scour in those areas during a specific high-flow event could not be reliably estimated without the application of an STM (see Section 3.2).

3.1.7 Potential effects of ship-induced bed scour

The gross bed scour that could potentially result from ship traffic was also analyzed in the STAR using a numerical model (Windward and QEA 2007). The focus of this analysis was on the upstream and downstream movements of ships within the navigation channel. This analysis was necessarily limited to estimating gross bed scour depth because sediment transport and deposition processes were not explicitly accounted for in the model. Consequently, bed scour results, expressed on a per ship passage basis, were considered to be upper-bound estimates, with the actual bed erosion resulting from ship traffic in the navigation channel likely to be less than that estimated because in a typical event, sediment is suspended for a brief period of time but is then re-deposited shortly thereafter once the propeller-driven forces are no longer present. This analysis was considered to be a screening-level evaluation, with

order-of-magnitude accuracy at best. The model was not designed to assess cumulative effects of ship traffic. Furthermore, the model did not address effects from vessel maneuvering outside of the navigation channel. Such movements could result in localized scour. The STM discussed in Section 3.2 describes long-term simulations that incorporate cumulative effects of all ship traffic.

Results of this modeling analysis included:

- Within the navigation channel, ship movement was estimated to result in an average bed scour depth of less than 1 cm per ship passage in Reach 1, and less than 0.1 cm per ship passage in Reaches 2 and 3. Within the bench areas, average bed scour depths of about 1 to 2 cm per ship passage were estimated in Reach 1, and less than 1 cm per ship passage was estimated in Reaches 2 and 3.
- A sensitivity analysis indicated that reducing ship speed from the LDW speed limit of 5 knots to a speed of 2.5 knots would likely significantly reduce bed scour, with estimated bed scour of less than 1 cm throughout the LDW for all conditions. Doubling the applied ship power (i.e., horsepower) had minimal effect on estimated scour depth.
- The reworked (i.e., mixed) sediment layer was equated with the depth of gross bed scour, based on the assumption that the same layer is continually reworked. Reach 1 was estimated to have an upper-bound average scour thickness of less than about 1 cm in the navigation channel and about 1 to 2 cm in bench areas. In Reaches 2 and 3, the reworked sediment layer was estimated to have an upper-bound average thickness of less than 0.1 cm in the navigation channel and less than 1 cm in bench areas. The frequency of mixing is about 100 to 250 events per year.
- Ship-induced bed scour depths were not found to be significantly related to erosion rate properties or geochronology results at any particular location in the bench areas or navigation channel of the LDW. No correlation was apparent between the structure of a geochronology or Sedflume core and the amount of ship-induced bed scour at the core location.
- The present-day structure of the LDW sediment bed reflects the effects of ship-induced bed scour that has occurred for at least 40 yrs.[12] The lack of evidence in the bathymetric record of large depressions in areas with frequent ship traffic validates the assumption that the upper sediment layer is continually reworked and suggests that ship activity is not a major cause of sediment transport in the LDW.

[12] Many of the businesses that rely on barge or tug traffic, or their predecessors, have existed for at least 40 years.

3.2 SEDIMENT TRANSPORT MODELING

The analyses presented in the STAR (Windward and QEA 2007) produced a significant amount of information on LDW sediment characteristics. However, a limitation of those analyses is the inability to predict erosion, deposition, and net sedimentation throughout the LDW during high-flow events and over multi-year periods. It was recognized by LDWG, Ecology, and EPA that development of an STM could potentially enhance the efficacy of various analyses during the FS process, including monitored natural recovery (MNR) analyses.

Discussions and meetings among LDWG, Ecology, and EPA during July and August 2006 concerning an STM resulted in the formation of an STM group,[13] which worked collaboratively and provided advice on the development, calibration, and application of the model. The STM group held meetings, conference calls, and informal discussions among members at various times between August 2006 and May 2007 to review model status and discuss the next steps to be taken in model development, calibration, and application. This collaborative effort was instrumental in producing a consensus modeling framework and in achieving the study objectives described in Section 3.2.1, which summarizes the results of the STM effort (QEA 2007).

3.2.1 Modeling objectives

The overall objectives of the STM effort were to:

- Develop a quantitative tool to evaluate short- and long-term sediment transport processes in the LDW
- Refine the CSM for the LDW that was developed in the STAR
- Provide information to support FS analyses and inform remedial decision-making

Multi-year simulations were conducted to predict long-term changes in bed elevation (i.e., net sedimentation rate) as well as changes in surface layer sediment composition. These results will be used in the FS to help evaluate remedial alternatives and estimate the rate of natural recovery in the LDW attributable to sediment transport processes. Through the use of long-term (e.g., 30 year) simulations, the STM answered specific questions, including:

- What areas in the LDW are net depositional, net erosional, or in dynamic equilibrium?

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[13] The members of this group have included: Shane Cherry (Cherry Creek Environmental), Karl Eriksen (USACE), Joe Gailani (USACE), Earl Hayter (EPA), Brad Helland (Ecology), Bruce Nairn (King County), Mike Riley (S.S. Papadopulos & Associates), Peter Rude (City of Seattle), Beth Schmoyer (City of Seattle), David Schuchardt (City of Seattle), Jeff Stern (King County), Kym Takasaki (USACE), John Ryan (RETEC|ENSR), Anne Fitzpatrick (RETEC|ENSR), and Kirk Ziegler (QEA).

- How does the composition of the surface layer sediment change over time as external sediment loads (from upstream and lateral sources, such as SDs) become incorporated into the sediment bed?
- What is the effect of high-flow events on episodic scour?
- What is the potential depth of scour during high-flow events?

In addition, the effects of high-flow events on bed scour and the potential for re-exposing buried sediments were evaluated with the STM. The high-flow event simulations were short-term, covering several-day periods equivalent to the storm length and the time necessary for flows to return to normal. For episodic high-flow events, questions of interest included:

- What areas experience erosion during a high-flow event?
- In the areas that experience erosion during high-flow events, what is the potential depth of scour?
- What is the potential for re-exposing buried sediments?

Note that the STM models the movement and fate of suspended and bed load sediment (as physical particles) but does not address chemical transport and fate.

3.2.2 Development of sediment transport model inputs

Inputs and parameters for the STM were separated into three broad categories: 1) sediment characteristics, 2) bed characteristics, and 3) boundary conditions. Sediment characteristics include physical parameters, such as effective particle diameter and settling speed. Bed characteristics include physical parameters of the bulk sediment bed, such as dry density and sediment particle size, and calculated parameters such as erosion rates. Boundary conditions refer to characteristics at the edge of the modeling area (e.g., sediment loads at different inflow locations adjacent to the LDW).

Sediment particles in the LDW have a range of sizes, from less than 1 μm for clays to greater than 2,000 μm for gravel. Simulation of the entire particle size spectrum was impractical. Therefore, particles were separated into four size classes: 1) clay and fine silt (less than 10 μm), 2) medium and coarse silt (10 to 62 μm), 3) fine sand (62 to 250 μm), and 4) medium and coarse sand (250 to 2,000 μm) (Table 3-4). Each sediment size class was represented as an effective particle diameter. The particle size distributions in the sediment bed and incoming loads were evaluated to select the appropriate average effective particle diameters in each general size class to represent particle transport throughout the LDW. Note that effective particle diameters for Classes 1A and 1B were treated as adjustable calibration parameters (Section 3.2.2).

Table 3-4. Characteristics of sediment particle size classes

Sediment Size Class	Particle Size Range (µm)	Effective Particle Diameter (µm)	Effective Settling Speed (m/day)
1A. clay, fine silt	< 10	5	1.3
1B. medium, coarse silt	10 – 62	20	21
2. fine sand	62 – 250	130	770
3. medium, coarse sand	250 – 2,000	1,100	10,200

The settling speeds of sediment particles are related to the effective particle diameter, with settling speed increasing as effective diameter increases. The effective settling speeds for the four sediment size classes had a large range, from about 1 m/day for Class 1A to about 10,000 m/day for Class 3 (Table 3-4). This large range (i.e., factor of 10,000) had a significant effect on the transport characteristics of the different sediment size classes.

The specification of bed properties within the LDW began with the separation of the sediment bed into two distinct types of sediment: cohesive and non-cohesive. Cohesive sediment corresponded to a muddy bed composed of a mixture of clay, silt, sand, and organic matter. Non-cohesive sediment corresponded to a sandy bed with a relatively low amount of clay, silt, and organic matter. The STM required specification of the following bed property inputs within the cohesive and non-cohesive bed areas: 1) dry (bulk) density; 2) initial sediment particle size composition (i.e., relative amounts of Classes 1A, 1B, 2, 3), and 3) effective bed roughness. The median diameter (D_{50}) of particles in the non-cohesive bed areas needed to be specified. The spatial distribution of erosion rate parameters, both horizontally and vertically, for the cohesive bed areas also had to be determined and was derived from an analysis of the Sedflume core data (Section 3.1.5).

Sediment loads from upstream (i.e., Green River) and lateral sources (i.e., SDs, CSOs, streams), including both the magnitude and composition of the loads (i.e., relative amounts of Classes 1A, 1B, 2, 3), were needed to establish boundary conditions for the STM. Loads from the lateral sources were not included during the calibration period simulation (see Section 3.2.3) because the annual average load from lateral sources totaled 0.6% of the upstream load (QEA 2007).

3.2.3 Model calibration and validation

The hydrodynamic model[14] that was originally refined and calibrated during the sediment transport analysis (Windward and QEA 2007), as described in Section 3.1.6, was re-calibrated because modifications were made to the numerical grid.[15] These modifications were necessary so that long-term, multi-year simulations could be completed in a timely manner. In addition, grid cells were added for the LDW slips.

The re-calibration results showed that the hydrodynamic model realistically simulated all of the major characteristics of estuarine circulation in the LDW. The vertical structure of tidal current velocity was realistically simulated, and the STM was able to reproduce the two-layer flow in the region occupied by the saltwater wedge. Strong vertical stratification of salinity has been documented and the model was able to simulate that stratification with acceptable accuracy. In addition, the model realistically simulated the dynamic nature of the saltwater wedge and the location of the toe of the wedge as it varies over the course of a tidal cycle, as well as variations that result from changes in freshwater inflow and the phase of the tidal cycle (i.e., spring and neap tide conditions). Overall, re-calibration of the hydrodynamic model demonstrated that the model is sufficiently accurate and reliable for the objectives of the STM study.

The STM was built on a foundation of mechanistic formulations that are used to simulate the erosion and deposition of cohesive and non-cohesive sediment. The erosion and deposition formulations used in the model were based on results from a large number of laboratory and field studies. Site-specific data were used to determine model inputs and parameters, with robust datasets used to specify the two major inputs: sediment loading from the Green River and erosion properties of LDW sediments. The significant amount of site-specific data, in conjunction with the mechanistic nature of the model formulations, provided an opportunity to develop and calibrate a reliable STM.

Calibration and validation of the STM was a process that involved comparing model estimates to LDW data. These comparisons provided a "reality check" for the STM and were used to evaluate the accuracy and reliability of the model. The first step in the process was calibration of the STM, which involved adjusting model inputs and parameters to optimize the agreement between model results and empirical data. The

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[14] This hydrodynamic model (EFDC) was an important component of the STM. The STM differs from EFDC in that it includes sophisticated algorithms for tracking sediment particles, whereas the hydrodynamic portion of EFDC includes only water movement.

[15] As summarized in Section 3.1.6, additional model cells were added to the hydrodynamic model originally constructed by King County to improve the resolution. Because model results are created independently for each grid cell, smaller grid cells allow the evaluation of smaller scale variability in model results.

second step in the process was model validation, which involved a comparison of model predictions to an independent dataset (i.e., data not used to calibrate the model), with no adjustment of calibration inputs/parameters. Taken together, the results of the model calibration and validation results provided an objective method for "ground-truthing" the STM and determining its reliability as a diagnostic tool to evaluate sediment transport processes in the LDW.

The STM was calibrated and validated over a 21-yr period (1960 through 1980) to evaluate the accuracy and reliability of a model. This time period tested the model's capabilities because of the wide range of tidal and river flow conditions during that period, including several high flow events. This time period was selected because the data were readily available in a format that was compatible with the STM. Flows in the Green River during the 1960-to-1980 time period were very similar to flows during 1981-to-2006 time period, based on a comparison of flow duration curves for the two periods. Thus, using the 1960-to-1980 period for model calibration provided results that were consistent with current conditions. For the STM, the calibration targets (i.e., the parameters used to assess model accuracy or agreement) were net sedimentation rates in the navigation channel and bench areas of the LDW (QEA 2007). Four model parameters were adjusted to achieve the optimum agreement between estimated and empirically derived estimates of net sedimentation rates in the navigation channel and bench areas (Table 3-5) (QEA 2007).

Table 3-5. Sediment transport model calibration parameter values

Adjusted Parameter	Calibration Value
Effective particle diameter of Class 1A sediment	5 µm
Effective particle diameter of Class 1B sediment	20 µm
Average percentage of Class 1A sediment in incoming suspended load[a]	70%
Average percentage of Class 1B sediment in incoming suspended load[a]	18%

[a] The other 12% of the incoming suspended load consisted of Class 2 and 3 sediments.

In addition to the parameters listed in Table 3-5, two additional adjustments were made to the STM during calibration. First, a particle-shielding factor was incorporated into the calculation of bed erosion flux. This factor was used to reduce the erosion flux of smaller particles that can be sheltered by larger particles within a graded bed (i.e., a bed with a wide range of particle sizes) (QEA 2007). The second adjustment to the model was to specify that the sediment bed upstream of RM 4.8 be treated as a "hard-bottom," which means that the bed in that region (i.e., in the river upstream of the Upper Turning Basin) was assumed to not experience erosion or deposition. Bed load transport of sediment was allowed in that region (QEA 2007).

Validation of the model was achieved through comparison of the estimated composition of sediment deposited in the LDW to the observed composition of

surface-layer sediment. The relative amounts of the four size classes (i.e., Classes 1A, 1B, 2, 3) in the sediment estimated to be deposited between RM 0.0 and RM 4.3 (i.e., region downstream of the Upper Turning Basin) during the 21-year calibration period were compared to the relative amounts of the four sediment size classes measured in the Sedflume cores, which represent the upper 30 cm of the sediment bed. No model inputs/parameters were adjusted to optimize the agreement between observed and estimated composition. The satisfactory model results provided additional support for the capability of the model to realistically and accurately simulate erosion and deposition processes in the LDW.

Results of the calibration and validation exercises indicated that the STM is able to realistically and accurately simulate erosion and deposition processes in the navigation channel and bench areas of the LDW. The calibration and validation results, coupled with constraints on the model by mechanistic formulations and site-specific data, indicated that the model is a reliable tool for evaluating sediment transport processes in the LDW over a wide range of spatial and temporal scales.

3.2.4 Sediment bed stability results

The STM was used to evaluate the effects of high-flow events on bed stability in the LDW (QEA 2007). High-flow events with return periods of 2, 10, and 100 yrs were simulated. Boundary conditions for the high-flow simulations were developed using the hydrograph of a high-flow event that occurred during November-December 1975. During this high flow event, spring tide conditions occurred during peak flow conditions. Thus, the high-flow simulations produced conservative results that represented upper-bound estimates of bed scour depths during high-flow events.

Two quantities related to bed scour were derived from the simulation results: net erosion and maximum bed scour depth (QEA 2007). Net erosion at a specific location is the total decrease in bed elevation over the course of a high-flow event (i.e., difference in bed elevation between start and end of 26-day simulation). Net erosion thus incorporates the effects of erosion and deposition during the entire event. Maximum bed scour depth is the maximum depth of erosion that occurs at a specific location during the high-flow event. Typically, maximum bed scour occurs near the time of the peak flow rate, with deposition occurring after the peak flow rate, as river discharge declines.

Spatial distributions of estimated net erosion at the end of high-flow events with return periods of 2, 10, and 100 yrs are presented on Maps 3-6, 3-7, and 3-8, respectively. The spatial distribution of maximum bed scour during the 100-yr event is shown on Map 3-9. The locations of the maximum net erosion depths of bed scour in cohesive and non-cohesive bed areas are denoted on these maps.

Net erosion was estimated to occur over approximately 15 to 20% of the area of the LDW sediment bed during a high-flow event. Net erosion of 10 cm or greater was

estimated to occur over about 0.4% (1.7 ac) of the bed area in the LDW during the 100-yr high-flow event. Maximum net erosion depths during a 100-yr event were estimated to be about 14 cm. Maximum bed scour depths were about 1 cm greater than net erosion depths. Thus, based on the model, scour was estimated to be limited to approximately the upper 15-cm layer of the sediment bed within a few small areas during the 100-yr high-flow event upper-bound estimate. In most areas, the maximum scour depth was estimated to be far less. Even during a 100-yr high-flow event, negligible bed scour was estimated to occur over large areas of the LDW, including nearly all of Reach 1 (Map 3-9).

The model results showed that net erosion primarily occurred between RM 2.2 and RM 4.0 during the high-flow event simulations in the LDW, including the 100-yr event (QEA 2007). Most of the net erosion occurred in this reach of the LDW because most of the region upstream of approximately RM 2.2 behaved as a tidal freshwater river during high-flow events, with the saltwater wedge toe located downstream of approximately RM 2.2 during ebb tides. This is consistent with the hydrodynamic model simulation, which showed that relatively high bed shear stresses occurred upstream of the saltwater wedge (Windward and QEA 2007). Less net erosion was estimated to occur in the reach upstream of RM 4.0 than in the reach from RM 2.2 to RM 4.0 for two reasons. First, bed shear stresses were generally lower upstream of RM 4.0 than in the reach from RM 2.2 to RM 4.0, which is mainly the result of differences in cross-sectional area between the two reaches (i.e., larger cross-sectional area upstream of RM 4.0). Second, more deposition and less net erosion were estimated to occur upstream of RM 4.0 because a large of portion of sand (i.e., sediment Classes 2 and 3) transported from the Green River to the LDW was estimated to be deposited in this reach; the Upper Turning Basin was designed to be an effective sediment trap.

Net erosion was estimated to occur in only a few relatively small areas in the reach downstream of RM 2.2, even during a 100-yr event, with typical net erosion depths of 2 cm or less (Map 3-8). The cause for the significant difference in estimated erosion depths between the reaches upstream and downstream of RM 2.2 is the presence of the saltwater wedge downstream of RM 2.2 during high-flow events, which resulted in relatively low bed shear stresses and a small amount of erosion in that reach.

The effects of deposition on net erosion after the peak flow rate, as river discharge declines, are illustrated by a comparison of the spatial distributions shown on Maps 3-8 and 3-9. These two maps show that: 1) the areal extent of maximum bed scour was estimated to be larger than the areal extent of net erosion at the end of the 100-yr event, 2) net erosion was estimated to be less than maximum bed scour at most locations, and 3) net deposition over the course of the event was estimated to occur at some locations that experienced bed scour at some time during the event. In areas that experienced deposition after the maximum bed scour occurred (i.e., after the peak flow during the event), the amount of deposition was typically relatively small.

The relative amounts of sediment eroded from surface (0 to 10 cm) and subsurface (deeper than 10 cm) layers during the 100-yr event were evaluated by "tagging" sediments within those two layers at the start of the simulation and tracking the sediment from those two bed sources separately (QEA 2007). Results of a mass balance analysis for the 100-yr event indicated that 6% of the mass of sediment eroded from the sediment bed in the LDW was from the subsurface layer (i.e., deeper than 10 cm), with the remaining 94% of eroded sediment originating from the surface layer (0 to 10 cm). The mass balance analysis also showed that about 3 and 0.1% of the total sediment transported downstream of RM 0.0 during the 100-yr event was eroded from the surface and subsurface layers, respectively (the remainder coming from the Green River). These results indicated that subsurface contamination (i.e., >10 cm depth) is unlikely to be significantly eroded and transported within and outside the LDW during a high-flow event.

A sensitivity analysis was conducted to evaluate uncertainty in model predictions that may have resulted from uncertainty in model inputs and parameters. The model was most sensitive to the parameters that control erosion rates, which were specified using Sedflume core data collected from the LDW. Uncertainty in the areal extent of net erosion during a 100-yr event ranged from approximately –35% to 30%, with about -30% to 50% uncertainty in the total mass of eroded sediment. The upper-bound estimate of net erosion depth during a 100-yr event was about 40% greater than during the base-case simulation. These results showed that, overall, the bed stability results had an uncertainty level of about 40% because of uncertainty in model parameters and inputs.

3.2.5 External sediment load analysis and changes in bed composition

A 30-yr simulation was conducted so that the effects of external sediment loads on surface layer bed composition over long-term, multi-year periods could be evaluated (QEA 2007). Boundary conditions for the model were specified using river flow rate and tidal elevation data collected during the 30-yr period from 1960 through 1979. External sediment loads were specified from two sources: 1) upstream loads (i.e., Green River), and 2) lateral loads (i.e., SDs, CSOs, and streams). Original bed sediment (i.e., sediment at the beginning of the 30-yr simulation) was treated as a third source of sediment. The effects of the external loads on the relative amounts of sediment from the three sources (i.e., bed, upstream, and lateral) in the surface layer of the sediment bed over the 30-yr period were estimated by the STM. The surface layer was defined as the top 10 cm of the bed.

Sediment from each of the three sources was separated into four particle size classes (i.e., Classes 1A, 1B, 2, 3), with the sediment transport characteristics of the four particle size classes being the same for all three sediment sources (QEA 2007). For example, the erosion, deposition, and transport of Class 1A sediment was treated the same way for sediment originating from the bed, upstream, and lateral sources. Thus,

the model simulated the erosion, deposition, and transport of 12 sediment classes (i.e., four particle size classes from each of the three sediment sources) during the 30-yr period.

Changes in the composition of the 10-cm layer at the surface of the sediment bed were of particular importance in this analysis. At beginning of the 30-yr simulation, the composition of the 10-cm surface layer was 100% bed-source sediment, with no sediment from upstream and lateral sources. As the 30-yr simulation progressed, upstream- and lateral-source sediments were transported in the LDW and deposited on the 10-cm surface layer, which reduced the relative amount of bed-source sediment in that layer. The model tracked spatial and temporal changes in the relative amounts of sediment from the three sources over the course of the 30-yr period that resulted from erosion, deposition, and transport processes in the LDW.

Sediment loads from lateral sources (i.e., SDs, CSOs, and streams) were estimated using an approach that combined watershed modeling and empirically derived estimates (QEA 2007; SPU 2007).[16] SDs, streams, and CSOs discharge sediment into the LDW at over 200 locations; incorporating each individual discharge location into the model was not practical. Thus, the lateral sources were aggregated and represented by 21 point sources that discharged into the LDW at 16 representative locations (Map 3-10). The total average annual load from lateral sources was estimated to be approximately 1,200 metric tons per year (MT/yr), with 76%, 3%, and 21% of the total lateral load from SDs, CSOs, and streams, respectively (QEA 2007). The total annual load from lateral sources was 0.6% of the average annual total load from the upstream (river) source, which was more than 6,000,000 MT/yr (Figure 3-2).

[16] King County is currently preparing a document that presents loading estimates from King County CSOs.

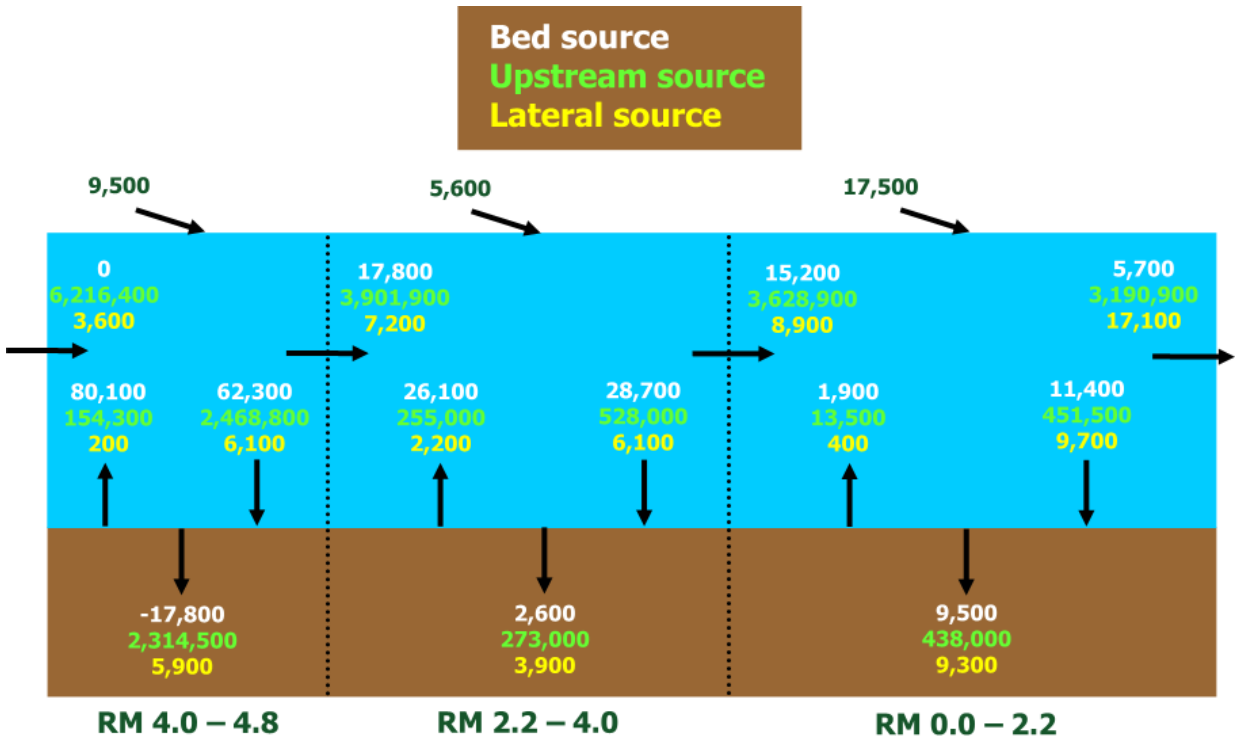

Figure 3-2. Sediment mass balances (in metric tons) for bed-, upstream-, and lateral-source sediment in the LDW (RM 0.0 to RM 4.8) for 30-year period

The spatial distribution of average net sedimentation rates estimated for the 30-yr period is presented on Map 3-11. These model results illustrate several characteristics of sediment transport processes in the LDW (QEA 2007). Reach 1 (RM 0.0 to RM 2.2) was estimated to be net depositional over multi-year periods with net sedimentation occurring everywhere. Within Reach 1, relatively high net sedimentation rates (i.e., greater than 2 cm/yr) were estimated between RM 1.4 and RM 2.2, with lower rates (i.e., typically 0.5 to 2 cm/yr) estimated downstream of RM 1.4. The model estimated a relatively low net sedimentation rate (i.e., less than 0.1 cm/yr) in a small area near RM 0.8 to RM 0.9 over the 30-yr period, with net erosion during high-flow events (Section 3.2.4). Within Reach 2 (RM 2.2 to RM 4.0), net sedimentation rates were estimated to be generally lower than those in Reaches 1 and 3 (RM 4.0 to RM 4.8), with one small area of estimated net erosion (i.e., one grid cell) near RM 3.6. This small area of net erosion over the 30-yr period appeared to be in a state of dynamic equilibrium. Relatively high variability in net sedimentation rates, from less than 0.5 cm/yr to greater than 3 cm/yr, in Reach 2 reflected the dynamic nature of this reach, which was estimated to experience episodic bed scour during high-flow events (Section 3.2.4). Reach 3 was estimated to be net depositional and had the highest net sedimentation rates within the LDW, which is not surprising because the Upper Turning Basin, which is designed to function as a sediment trap, is located in this reach.

Large-scale comparisons of net sedimentation rates among the three reaches were made by calculating reach-averaged values (Table 3-6). Reach 2 was separated into two sub-reaches to reflect significant spatial variations in net sedimentation rates within that reach (QEA 2007). The reach designations discussed in this section, and in the STM, represent a refinement, based on an improved understanding of LDW sediment transport processes, from the three reaches originally discussed in the STAR. The value for Reach 3 corresponds to the average in the area excluding the Upper Turning Basin; average net sedimentation in the Upper Turning Basin was over 40 cm/yr.

Table 3-6. Reach-average sedimentation rates and composition at end of 30-yr period

Reach	Net Sedimentation Rate (cm/yr)	Bed-Source Content (%)	Upstream-Source Content (%)	Lateral-Source Content (%)
1 (RM 0.0 to RM 2.2)	1.9	3	94	3
2 (RM 2.2 to RM 4.0)	2.2	22	76	2
2A (RM 2.2 to RM 2.6)	3.6	2	97	1
2B (RM 2.6 to RM 4.0)	1.8	29	69	2
3 (RM 4.0 to RM 4.8, excluding the Upper Turning Basin)	13	2	96	2

A primary objective of this analysis was to evaluate the effects of external sediment loads on the composition of the surface layer (0 to 10 cm) sediment. This type of analysis will be used in the FS to estimate the rate and magnitude of sediment quality improvements in the future. Large-scale comparisons of the three reaches provided an assessment of the relative effects of upstream and lateral sediment loads on surface layer composition over the course of the 30-yr period (Table 3-6). A large majority (69 to 96%, Table 3-6) of the reduction in bed-source content was the result of the deposition of sediment from the upstream source.

The spatial distribution of bed-source content in the surface layer at the end of the 30-yr simulation period is shown on Map 3-12. The relatively high net sedimentation rates estimated in Reaches 1 and 3 resulted in significant estimates of decreases in bed-source content (QEA 2007). The spatial distribution of upstream-source content in surface layer sediment at the end of the 30-yr period was a mirror-image of the bed-source content; upstream-source content was high at locations where bed-source content was low and vice versa (Map 3-13). In general, the estimated effects of sediment loads from lateral sources were greatest in the immediate vicinity of the discharge points of SDs or CSOs (Map 3-14). However, it is important to note that over 200 SDs and CSOs were represented by only 21 point sources that discharged into the LDW at 16 discrete locations in the model. Therefore, the true lateral-source content in the surficial bed layer is expected to be more widely distributed.

In Reaches 2 and 3, lateral-source contributions were greater than 1% in surface layer sediment in the slips near the SD/CSO discharge points. This effect was seen in Slips 4 and 6, located at approximately RM 2.9 and RM 4.2, as well as in the bench areas that extend downstream of near-shore discharge locations. The effects of lateral sources on surface layer composition were more widely distributed in Reach 1, with lateral-source contributions greater than 1% in a large portion of this reach. Tidal effects within this reach were estimated to spread lateral-source sediment upstream and downstream of the discharge locations. Relatively high lateral-source content values (i.e., greater than 5%) were estimated in the grid cells where the lateral-load discharges were specified. These contributions are likely lower than the estimates because of the aggregation of inputs.

The spatial distributions of surface layer composition shown on Maps 3-12 through 3-14 represent conditions at the end of the 30-yr simulation. Temporal variations in surface layer composition over the entire 30-yr period indicated that bed-source content is estimated to generally decrease at an approximately exponential rate (QEA 2007). For quantities that decrease exponentially with time, a measure of the rate of decline is the half-time, which is the time period over which the quantity decreases by 50%. The half-time metric is useful for expressing the relative rate of decrease in bed-source content resulting from the deposition of sediment from external sources.

The spatial distribution of half-time for bed-source content in the surface layer (i.e., top 10 cm) of the bed is shown on Map 3-15. Half-times for bed-source content in surface layer sediment were 10 yrs or less in about 90% of the LDW bed area. Approximately 6% of the bed area, mostly in intertidal benches between RM 2.7 and RM 3.9, had half-times greater than 30 yrs. In Reach 1, half-times were generally less than 10 yrs, except for a small area near RM 0.8 to RM 0.9, where the half-time was greater than 30 yrs. In Reach 2, the area between RM 2.6 and RM 3.9 had spatially variable half-times that ranged from less than 5 yrs to greater than 30 yrs. The high sedimentation rates in Reach 3 yielded half-times of 10 yrs or less.

Sediment mass balances were constructed for the 30-yr period to gain additional insights about the movement within the LDW of sediment from bed, upstream, and lateral sources. The details of the mass balance calculations were presented in QEA (2007) and summarized briefly here. The overall trapping efficiency (TE) for the LDW (RM 0.0 to RM 4.8) was 49% (Figure 3-3). TE is the portion of the incoming sediment load that is deposited within a particular region. TE varied among the three reaches because of differences in the hydrodynamic and sediment transport characteristics of those reaches. The TE of Reach 3 was 37%, which was higher than TE values of 12% and 7% for Reaches 1 and 2, respectively (Figure 3-3). The high TE value for Reach 3 was attributable to the presence of the Upper Turning Basin, which is designed to be an efficient sediment trap to capture a large portion of the sand that is transported from the river into Reach 3. The lowest TE value was in Reach 2 because of the hydrodynamic characteristics of this reach during high-flow events. Reach 1 had a

higher TE (12%) than did Reach 2 because of minimal erosion during high-flow events. Reach 1 had a substantially lower TE than did Reach 3 because of the relatively small amount of sand transported into Reach 1 compared to the amount of sand transported into and deposited within Reach 3.

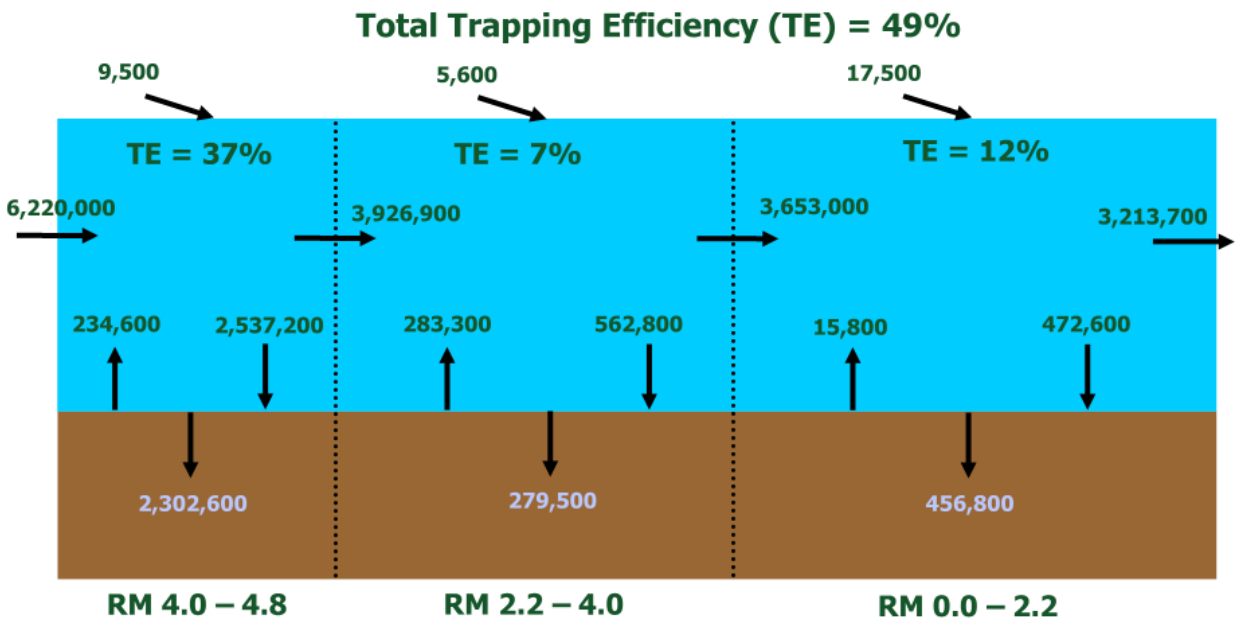

Figure 3-3. Sediment mass balance (in metric tons) in the LDW (RM 0.0 to RM 4.8) for 30-year period

A sensitivity analysis was used to evaluate uncertainty in model predictions. The sensitivity of the model to changes in the magnitude and composition of external sediment loads was investigated [see Section 4.4 of the STM report (QEA 2007)]. The results of the analysis showed that uncertainty in the lateral-source content was about ±40% with respect to the base-case value. The bed-source content varied approximately linearly with respect to changes in the upstream load magnitude; doubling the upstream load caused the bed-source content to decrease by about 50%, while decreasing the load by 50% caused the bed-source content to approximately double. Changes in the particle size composition of the upstream load resulted in variations that were within 10% of the base-case value.

3.3 Integration and Synthesis of Sediment Transport Analysis Results

The analyses conducted for the STM, in conjunction with empirical and ship-induced bed scour analyses, have resulted in an improved understanding of sediment transport processes in the LDW. A large amount of information on LDW hydrodynamics and sediment transport is presented in the STAR (Windward and QEA 2007) and the STM report (QEA 2007), respectively. Results and findings from the major components of the analyses are integrated and synthesized in this section to

produce a clear and concise picture of sediment transport in the LDW. Additional integration of sediment transport analysis results is provided in Section 4.3, where temporal trends in chemical concentrations are discussed, and in Section 11.3.2, where sediment chemistry data are discussed in the context of a chemical CSM.

The first step in understanding sediment transport in the LDW was to understand the hydrodynamics of this saltwater wedge estuary. During low-flow conditions in the Green River, the saltwater wedge extends up to or beyond the upstream portion of Reach 3 (i.e., RM 4.5 to RM 4.8), see Figure 3-4. The saltwater wedge is dominated by two-layer estuarine circulation, with saltier and denser water transported upstream in the lower layer of the water column and fresher water transported downstream in the upper layer. Near-bed velocities and bed shear stresses within the saltwater wedge are tidally driven and relatively low, which results in minimal bed scour within the saltwater wedge during low-flow conditions. During a high-flow event in the Green River, the toe of the saltwater wedge is pushed downstream to the vicinity of the boundary between Reaches 1 and 2 (i.e., RM 2.0 to RM 2.5). Two-layer estuarine circulation exists in Reach 1 during high-flow events (Figure 3-5). High freshwater inflow causes the hydrodynamic characteristics of Reaches 2 and 3 to change from two-layer estuarine circulation (low-flow conditions) to a freshwater tidal river. This change in the hydrodynamic characteristics of these two reaches results in significant increases in near-bed current velocities and bed shear stresses in Reaches 2 and 3 during a high-flow event.

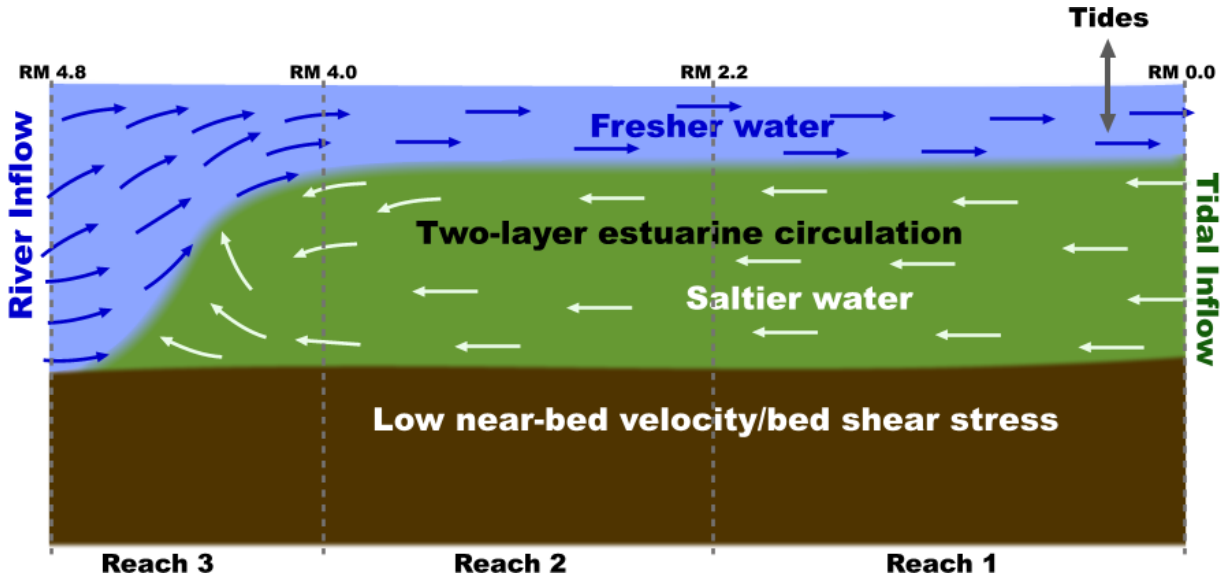

Figure 3-4. Schematic of LDW hydrodynamics during low-flow conditions

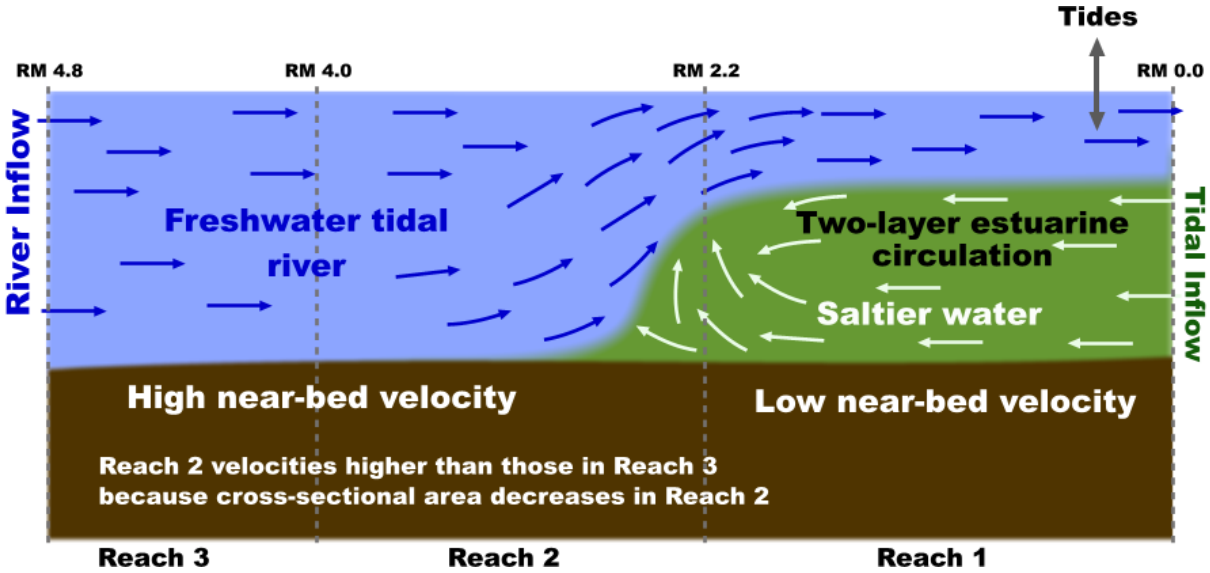

Figure 3-5. Schematic of LDW hydrodynamics during high-flow conditions

3.3.1 Sediment bed stability

Sediment bed stability analyses conducted to date included evaluations of high-flow events, with durations on the order of days to weeks, and long-term simulations over 30 years. Overall, model results from both high-flow and long-term simulations suggested that Reaches 1, 2, and 3 have different sediment transport characteristics, primarily as a result of differences in the hydrodynamic characteristics of each reach. Reach 1 was estimated to have relatively high net sedimentation rate estimates, but these estimates were generally lower than the estimates in Reach 3 because less sand is transported into this portion of the LDW than into the upstream reach. Reach 2 was estimated to have spatially variable net sedimentation rates, with areas of relatively low net sedimentation. This reach experiences the most erosion during a high-flow event as a result of relatively high bed shear stresses caused by changes in hydrodynamics during an event (i.e., transition from estuarine circulation with a saltwater wedge during low-flow conditions to a tidal freshwater river during high-flow conditions). Reach 3 has the highest net sedimentation rate estimates in the LDW, because of the presence of the Upper Turning Basin, which acts as a sediment "sink" and captures a large portion of the load of sand from the Green River.

During a high-flow event, most of the bed scour is estimated to occur in Reach 2, with Reach 1 having minimal erosion, except in a small area near RM 0.8 to RM 0.9. Limited net erosion is estimated in Reach 3. During a high-flow event with a return period of 100 yrs, about 20% of the total bed area in the LDW (i.e., about 80 ac) was estimated to be net erosional, with most of the bed scour occurring in Reach 2. The remaining 80% of the total bed area in the LDW was estimated to be net deposition during a 100-yr event (Figure 3-6). A large majority of the net erosion was estimated for the surface layer (i.e., 0 to 10 cm) of the bed, with less than 1% of the total bed area (about 2 ac)

having net erosion greater than about 10 cm. The maximum depth of bed scour during a 100-yr event was estimated to be about 15 cm, although this depth of bed scour was estimated for very small areas. Approximately 80% of the sediment mass eroded from the bed during a 100-yr event was estimated to be redeposited within the LDW, with the remaining 20% transported out of the LDW.

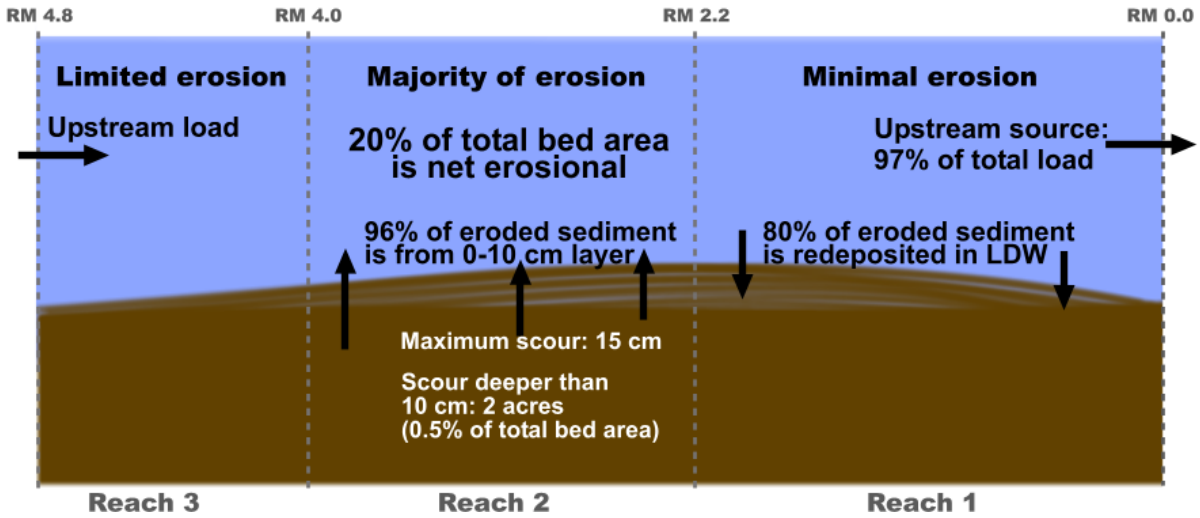

Figure 3-6. Schematic of LDW sediment transport processes during a 100-yr event

Reaches 1, 2, and 3, and thus the entire LDW, were estimated to be net depositional on annual timescales. The general effect of erosion during high-flow events was to reduce the net sedimentation rate in locations where bed scour occurs. Net sedimentation rates on a reach-average basis varied from about 2 cm/yr in Reaches 1 and 2 to over 10 cm/yr in Reach 3 (not including the Upper Turning Basin). Reach 2 was separated into two sub-reaches based on differences in net sedimentation rates, with the spatially averaged rate in Reach 2A (RM 2.2 to RM 2.6) being a factor of 2 higher than the spatially averaged rate in Reach 2B (RM 2.6 to RM 4.0) (Figure 3-7). In general, spatially averaged net sedimentation rates were similar in the navigation channel and bench areas, with differences between the zones being less than a factor of 2 in a particular reach.

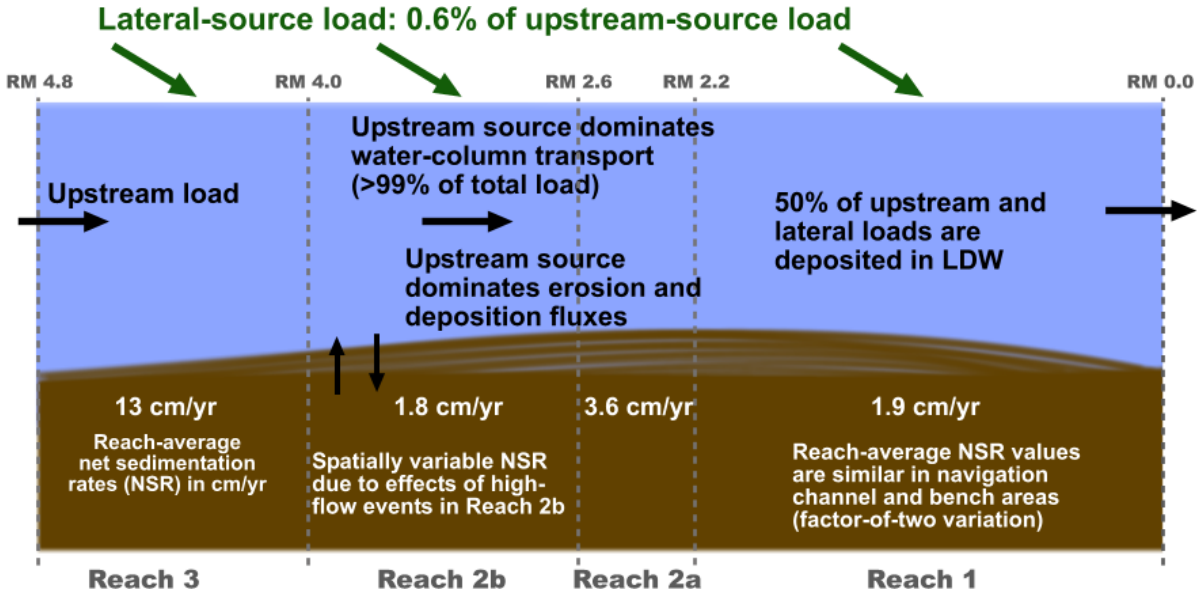

Figure 3-7. Schematic of net LDW sediment transport processes over a 30-yr period

Three of the four questions that were established for the STM (see Section 3.2.1) were related to sediment bed stability. The questions and answers are provided below.

- **What areas in the LDW are net depositional, net erosional, or in dynamic equilibrium?** Over the 30-yr period, one grid cell near RM 3.6 was estimated to be net erosional, but that location should be considered to be in a state of approximate dynamic equilibrium. Other than that one location, the sediment bed in the rest of the LDW was net depositional over the 30-yr period, with average net sedimentation rates ranging from less than 0.1 cm/yr to greater than 3 cm/yr.
- **What is the effect of high-flow events on episodic scour in otherwise net depositional areas?** Within Reach 1, negligible scour was estimated to occur during high-flow events in most of this reach, with only a minor effect on long-term deposition. A small area near RM 0.8 to RM 0.9 was estimated to experience bed scour during high-flow events, which tended to reduce long-term deposition and resulted in a relatively slow reduction in bed-source content. A small, localized area near RM 0.1 may also experience bed scour during high-flow events. Bed scour estimates were spatially variable in Reach 2, and episodic erosion during rare events tended to cause relatively low long-term net sedimentation rates at some locations. The relatively high net sedimentation rate estimates in Reach 3 (i.e., greater than 3 cm/yr) minimized the effects of bed scour during high-flow events over multi-year periods.
- **In areas that are net depositional, what is the potential depth of scour during high-flow events?** The depth of bed scour was dependent on river flow and

tidal conditions during the high-flow event, as well as on the location in the LDW. For a 100-yr event, approximately 20% of the LDW bed area was estimated to experience net erosion, with most of the erosion occurring in Reach 2. A large portion of the net erosion was estimated for the surface layer (i.e., 0 to 10 cm), with only 0.4% of the LDW bed area having net erosion depths greater than 10 cm. Maximum net erosion depth estimates during a 100-yr event were about 14 cm, although such high net erosion occurred over only very small areas.

3.3.2 Changes in sediment bed composition

Based on the modeling, approximately 99% of the total external sediment load to the LDW comes from the upstream Green River (Figures 3-2 and 3-7). The annual average sediment load from lateral sources combined was estimated to be 0.6% of the annual average load from the Green River. Approximately 50% of the external sediment load, from upstream and lateral sources, was estimated to be deposited (or trapped) in the LDW (Figure 3-7). TE (i.e., portion of incoming sediment load deposited within a reach) varied among the three reaches as a result of differences in the hydrodynamic and sediment transport characteristics of these reaches. TE was highest for Reach 3 (37%) and lowest for Reach 2 (7%) (Figure 3-3).

Of the sediment load leaving the LDW, sediment originating from the upstream source also represents over 99% of the total suspended sediment load transported downstream of the LDW (i.e., past RM 0.0), with sediment originating from lateral and bed sources composing about 0.5% and less than 0.2% of the total load exiting the LDW at RM 0.0, respectively (Figure 3-2). The contribution from lateral sources to the total suspended sediment load exiting the LDW at RM 0.0 was estimated to be about three times greater than the contribution from the bed-source sediment.

The bed-source content of the surface layer (0 to 10 cm) was estimated to decrease with time at an approximately exponential rate, primarily because of the deposition of upstream source sediment. The rate of decrease was spatially variable within the LDW as a result of variations in net sedimentation rate. Approximately 90% of the total bed area in the LDW had a half-time[17] of 10 yrs or less. About 6% of the LDW bed area had a half-time of 30 yrs or more; net sedimentation rates in these areas were less than about 0.3 cm/yr. Estimated decreases in reach-averaged bed-source content, and corresponding half-times, over a 30-yr period are shown in Figure 3-8.

[17] Half-time is defined as the time needed for a 50% reduction for a particular sediment source (i.e., bed, lateral, upstream).

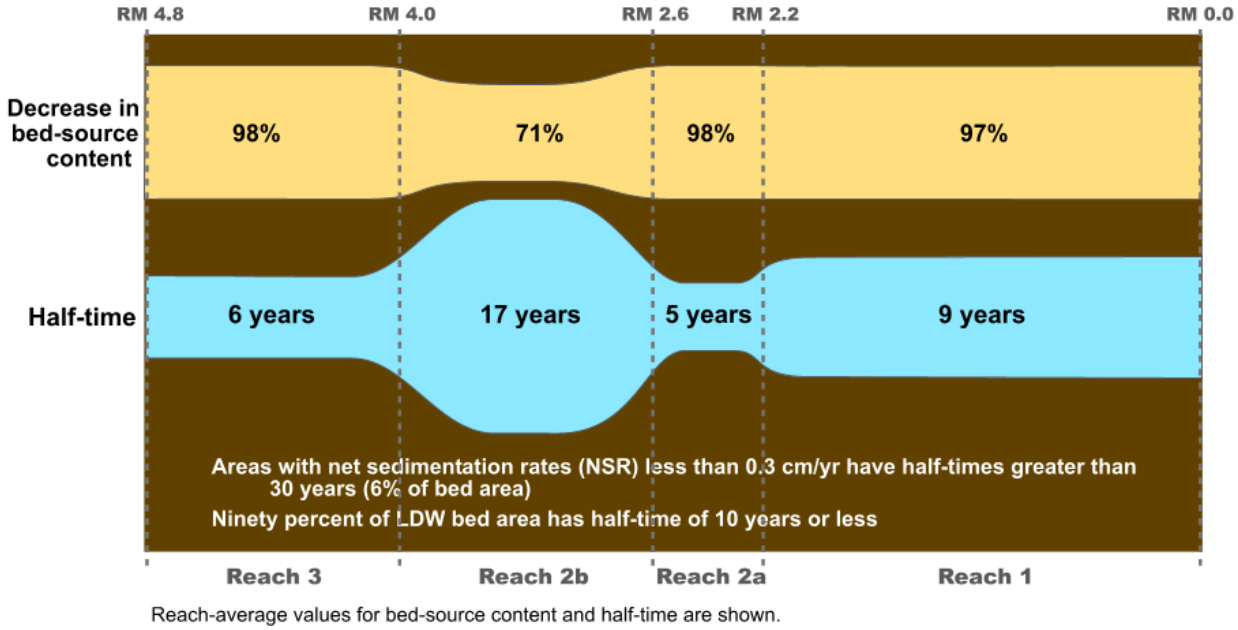

Figure 3-8. Bed-source content in surface layer (0 to 10 cm) and estimated half-times

In Reaches 2 and 3, elevated lateral-source contents (i.e., greater than 1%) in surface layer sediment were estimated to generally occur near the SD, CSO, and/or stream discharge points. The effects of lateral sources on surface layer composition were more widely distributed in Reach 1, with elevated lateral-source content values (greater than 1%) occurring over a large portion of this reach. At the end of a 30-yr period, reach average values of lateral-source content in the surface layer of the bed were about 1 to 2% (Figure 3-9).

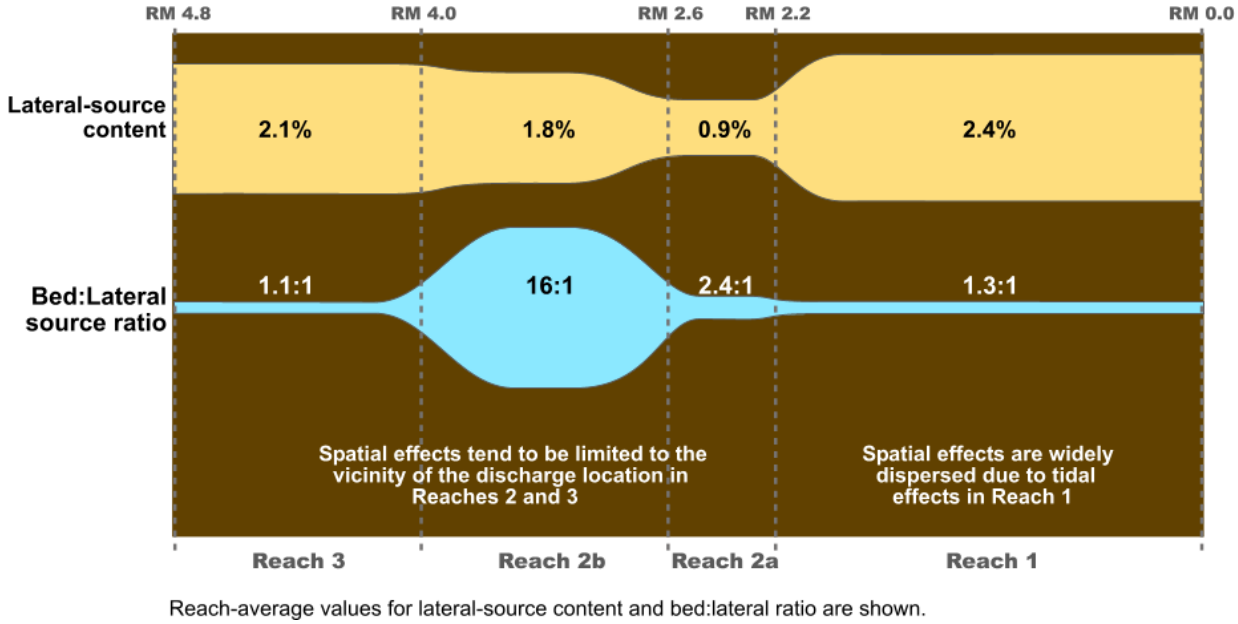

Figure 3-9. Estimated lateral-source content in surface layer (0 to 10 cm)

The fourth question that was established for the STM (see Section 3.2.1) concerns the change in the composition of the surface sediment layer over time as external sediment loads become incorporated into the sediment bed. The average bed-source content of the surface layer (i.e., 0 to 10 cm deep) sediment decreased by 97, 78, and 98% in Reaches 1, 2, and 3, respectively, during the 30-yr period. In Reaches 2A and 2B, bed-source content decreased by 98 and 71%, respectively. A large majority of the reduction in bed-source content was attributable to the deposition of sediment from the upstream sediment source. In general, bed-source content decreased at an approximately exponential rate, which means that a representative measure of the rate of dilution was the half-time (i.e., time for bed-source content was estimated to decrease by 50%). For areas with net sedimentation rates less than about 0.2 to 0.3 cm/yr, bed-source content in the surface layer decreased at a relatively low rate, with half-times of 30 yrs or greater. If the net sedimentation rate was greater than about 0.3 cm/yr, then the half-time of bed-source content was less than 30 yrs. Reach-average half-times for bed-source content in Reaches 1, 2, and 3 were 6, 14, and 9 yrs, respectively. There are localized areas with little change in estimated original bed composition, but most of these areas have already been designated as early action areas, where surface sediment concentrations exceeded state screening levels and net burial by cleaner sediment from the Green River is not occurring at significant rates.

3.4 SUMMARY OF CONCEPTUAL SITE MODEL FOR SEDIMENT TRANSPORT

A range of empirical and modeling analyses was conducted by Windward and QEA (2007) and QEA (2007), with each analysis focusing on a specific component of sediment transport and bed stability in the LDW. The models developed in the STAR and STM provided several lines of evidence with similar conclusions; however, the uncertainty associated with the results and limitations in the models must be acknowledged. The qualitative conclusions derived from these analyses have a relatively low level of uncertainty, whereas the quantitative results have a higher level of uncertainty. The sensitivity analyses demonstrated that model results have an uncertainty level of a factor of 2 or less that is associated with uncertainty in model parameters and inputs. This level of uncertainty is acceptable for the intended uses of the STM.

The results of the empirical analyses and modeling presented in Section 3.1 and the STM presented in Section 3.2 were used to refine the CSM presented in the STAR (in which the CSM was primarily based on the hydrodynamics and geomorphology), and even earlier versions presented in the Phase 1 RI (in which the CSM was primarily based on sediment properties). There have been step-wise refinements in the CSM developed over the past 4 years based on additional studies and modeling analyses that have been conducted for the RI/FS; the CSM may continue to be refined

throughout the FS. The key concepts that define the revised CSM include the hydrodynamics of the saltwater estuary, sediment transport characteristics, erosion properties, and bed composition changes, each of which is summarized below.

3.4.1 Waterway-wide characteristics

- The entire LDW was estimated to be net depositional over annual time scales.
- Net sedimentation rates were generally higher in the navigation channel than in the bench areas. For the navigation channel, the estimated net sedimentation rate decreased from the Upper Turning Basin (near RM 4.5) to downstream areas. For the bench areas, estimated net sedimentation rates were higher in Reaches 1, 2A, and 3 than in Reach 2B. Net sedimentation rate estimates tended to be lower in the intertidal areas than in the subtidal areas.
- Bed erosion is an episodic process that can be expected to be most pronounced during high-flow events. Episodic bed scour was estimated to occur to the greatest extent in Reach 2, was lower in Reach 3 than in Reach 2, and was minimal in Reach 1.[18] Net erosion is estimated to occur over about 20% or less of the LDW bed area during high-flow events, with most of the net erosion depth estimates being less than 10 cm (maximum of 15 cm or less).
- Ship-induced bed scour tends to behave as a mixing process for surficial sediment. In the scour model, the reworked surficial layer had an upper-bound average thickness of less than about 1 cm in the navigation channel and less than about 1 to 2 cm in the bench areas, with the frequency of such mixing being about 100 to 250 events per year.

The CSM states that the LDW is net depositional over annual time scales, with a variable rate of net deposition (i.e., net sedimentation rate). The best estimate of the spatial distribution of net sedimentation rates in the LDW is presented on Map 3-11. Net depositional areas can be categorized as follows:

- **Lower net depositional areas**: areas with net sedimentation rates less than 0.5 cm/yr. In small, isolated areas within this category, the net sedimentation rate is minimal (e.g., less than 0.1 cm/yr) and the bed may approach a state of dynamic equilibrium (i.e., minimal changes in bed elevation over annual time scales).
- **Intermediate net depositional areas**: areas with net sedimentation rates ranging from 0.5 to 2.0 cm/yr.

[18] Similar conclusions were presented in the STAR, but the reach designations were different in the STAR compared to the current CSM. For example, Reach 3 was defined as upstream of RM 3.0; it is now defined as upstream of RM 4.0.

- **Higher net depositional areas**: areas with net sedimentation rates greater than 2.0 cm/yr.

The CSM is discussed below for each of the three LDW reaches separately, based on results of the various modeling exercises. Viewing these three reaches separately provides a more comprehensive understanding of sediment dynamics and bed stability within the LDW.

3.4.2 Reach 1: RM 0.0 to RM 2.2

Reach 1 is net depositional on annual time scales in both the navigation channel and the adjacent bench areas. Based on net sedimentation rates estimated by the model, the navigation channel is classified as intermediate and higher net depositional, with a small area near RM 0.8 to RM 0.9 with lower net deposition. The bench areas range from intermediate to higher net depositional, with two small areas classified as lower net depositional. With respect to episodic erosion, this reach is always within the saltwater wedge, even during a 100-yr high-flow event. The permanent presence of the saltwater wedge serves as a protective barrier for the bed within this reach. Consequently, bed shear stresses (i.e., near-bed current velocities) are dominated by tidally driven flows, which are relatively low for all flow conditions. As a result, relatively low bed scour (less than 2 cm) is limited to a small area near RM 0.8 to RM 0.9. The potential for re-exposing buried sediments as a result of scour during high-flow events is minimal in this reach. Ship-induced mixing of the surficial bed layer potentially extends to average depths of about 1 to 2 cm in the bench areas and less than 1 cm in the navigation channel.

3.4.3 Reach 2: RM 2.2 to RM 4.0

Reach 2 is net depositional on annual time scales; however, estimated net sedimentation is spatially variable in this reach, ranging from lower to higher net depositional. Reach 2 includes the saltwater wedge toe during high-flow events; the saltwater wedge extends even farther upstream during average-flow conditions. More net erosion is estimated for this reach during high-flow events than for Reaches 1 and 3, but erosion is generally limited to the upper 10 cm of the sediment bed with maximum net erosion depths of approximately 15 cm or less in small areas. Ship-induced mixing of the surficial bed layer potentially extends to average depths of less than 1 cm in the bench areas and less than 0.1 cm in the navigation channel.

3.4.4 Reach 3: RM 4.0 to RM 4.8

Reach 3 is net depositional on annual time scales. This reach includes the Upper Turning Basin, which traps much of the coarse-grained sediment entering the LDW. The relatively high net sedimentation rates estimated for this reach (average of 13 cm/yr, not including the Upper Turning Basin) classify the navigation channel and bench areas as higher net depositional. Modeling results indicate that episodic erosion

may occur during high-flow events in Reach 3, but the areal extent of net erosion is significantly less than the areal extent in Reach 2. During high-flow events, flow in this reach is characteristic of a freshwater tidal river in that the saltwater wedge is not present at all. Bed scour during high-flow events is limited to the upper 15 cm or less of the sediment bed. Ship-induced mixing of the surficial bed layer potentially extends to average depths of less than 1 cm in the bench areas and less than 0.1 cm in the navigation channel.

4 Nature and Extent

This section presents a discussion of the nature and extent of chemical contamination in the LDW based on available data for sediment, tissue, and water. Section 4.1 discusses how data were selected for use in the RI. Section 4.2 presents summaries of chemical distributions in sediment, tissue, and water, with a focus on four risk driver chemicals (PCBs, arsenic, carcinogenic PAHs [cPAHs], and dioxins and furans), which are discussed in separate subsections. Section 4.3 discusses total PCB concentrations in subsurface sediment in combination with information on sediment transport to determine whether vertical patterns of PCBs are consistent with the predictions of the STM and the CSM.

4.1 DATA SELECTION, SUITABILITY, AND REDUCTION

This section presents the data quality objectives (DQOs) for available tissue, sediment and water chemistry data for the LDW and describes how data were selected for use in the RI. It also describes how raw data from the laboratories were managed for use in the evaluation of the nature and extent of contamination.

4.1.1 Data quality objectives

DQOs were established to determine whether chemistry data were acceptable for all uses in the RI/FS. The DQO process used to identify acceptable datasets was reviewed and approved by EPA and Ecology (Windward 2001, 2005i). Table 4-1 lists the DQOs that must be satisfied for chemistry data to be considered acceptable for all uses in the RI/FS, categorizing them according to the level at which each DQO would be applied: event, station, sample, or result. A DQO applied at the result level could cause a result record to be qualified for a particular chemical but not for other chemicals analyzed during the same study.

Table 4-1. Data quality objectives for chemistry data to be considered acceptable for all uses in the RI/FS

LEVEL	DATA QUALITY OBJECTIVE
Event	Hard copy or original electronic copy of data report must be available.
	Field coordinates must be available.
	Data must have been collected since 1990.
	Data must have been collected using appropriate sampling methods.
	Existence and location of supporting documentation (i.e., analytical raw data, chain-of-custody forms, and sample handling descriptions) must be known.[a]
Station	Stations located within dredge prisms or remediated areas should be identified.
Sample	Sediment sample depth must be identified.
	Sample type must be clearly identified.

LEVEL	DATA QUALITY OBJECTIVE
Result	Data validation qualifiers must be present, or derivable from laboratory qualifiers or QA information, and must be applied in a manner consistent with EPA functional guidelines (EPA 1999c, 2002e).[a]
	Each result must have a laboratory-generated form (usually referred to as a Form 1).[a]
	For non-detects, reporting limits and appropriate qualifiers must be given.
	Calculated values must be recalculated.
	Analytical methods must be identified.

[a] The existence of supporting documentation, the availability of Form 1s, and a review of the presence and application of data validation qualifiers were not considered necessary if the data were previously approved by EPA for all uses in the RCRA or CERCLA programs.

CERCLA – Comprehensive Environmental Response, Compensation, and Liability Act
EPA – US Environmental Protection Agency
FS – feasibility study
RCRA – Resource Conservation and Recovery Act
RI – remedial investigation

EPA has not established definitive guidelines specifying the level of data validation required for Superfund investigations. However, EPA Order 5360.1 and Office of Solid Waste and Emergency Response (OSWER) Directive 9355.9-01 (EPA 1993a) requires that environmental measurements be of known quality, verifiable, and defensible. The Office of the Inspector General (OIG) concluded in an audit of Region 9 Superfund sites (EPA 1995) that data used for cleanup decision-making should be validated using EPA functional guidelines (EPA 1999c, 2002e). According to these guidelines, two different levels of data validation are generally recognized for chemistry data. A summary data validation, referred to as QA1, represents a lower level of effort compared to a full validation, referred to as QA2. The elements of summary and full data validations for environmental chemistry data are presented in Table 4-2 (EPA 1999c, 2002e).

Table 4-2. Elements of summary and full data validations for environmental chemistry data

ELEMENT	APPLICABLE ANALYSES	SUMMARY DATA VALIDATION (QA1)	FULL DATA VALIDATION (QA2)
Quality control analysis frequencies	all	X	X
Analysis holding times	all	X	X
Instrument performance check	organic compounds, ICP-MS metals		X
Initial instrument calibration	all		X
Continuing instrument calibration	all		X
Laboratory blanks	all	X	X
ICP interference check sample	metals		X
System monitoring compounds (surrogates)	organic compounds	X	X
Matrix spikes/matrix spike duplicates	all	X	X

Element	Applicable Analyses	Summary Data Validation (QA1)	Full Data Validation (QA2)
Laboratory control samples	all	X	X
ICP serial dilution	metals		X
Field QA/QC (field blanks, field duplicates)	all	X	X
Internal standards	VOCs, SVOCs, ICP-MS metals		X
Pesticide cleanup checks	pesticides/PCBs		X
Target compound identification and quantitation (requires verification of reported results with raw data)	organic compounds		X
RLs	all	X	X

ICP-MS – inductively coupled plasma-mass spectrometry

PCB – polychlorinated biphenyl

QA/QC – quality assurance/quality control

RL – reporting limit

SVOC – semivolatile organic compound

VOC – volatile organic compound

Data of acceptable quality may still be associated with uncertainty in the RI. For example, a chemical not detected in a sample may actually be present, but its concentration below the reporting limit (RL) is unknown. This uncertainty applies to all cases in which chemicals are reported as undetected; the magnitude of this uncertainty increases with increasing RLs. None of the sampling events evaluated for inclusion in the RI were excluded in their entirety because of elevated RLs. The uncertainties associated with data quality and that are relevant to conclusions of the risk assessments are discussed in both the ERA (Appendix A) and HHRA (Appendix B).

4.1.2 Data selection

Many environmental investigations conducted within the LDW have included the collection of samples of surface sediment, subsurface sediment, fish or shellfish tissue, surface water, sediment porewater, or seep water for chemical analysis. This section describes the datasets selected for defining the nature and extent of contamination.

4.1.2.1 Surface sediment

Surface sediment samples have been collected during numerous sampling events since 1990 (Table 4-3). A baseline surface sediment chemistry dataset was compiled to characterize sediment from the LDW prior to any remedial actions (i.e., excluding data collected after removal actions at the Duwamish/Diagonal EEA and Boeing Developmental Center south storm drain outfall at the Norfolk EAA). This baseline dataset, which was used in the ERA and HHRA, included surface sediment data collected during the first two rounds of sampling conducted for the RI. A third round of surface sediment data was collected in 2006; these data were combined with the

baseline dataset used in the ERA and HHRA to characterize the nature and extent of contamination. This combined baseline/Round 3 dataset is referred to in the remainder of this document as the RI baseline dataset. The criteria for including or excluding surface sediment data in the RI baseline dataset were determined in agreement with EPA and Ecology, as follows (Windward 2006d):

- Surface sediment data were excluded if they did not meet the DQOs, as described in Section 4.1.1.
- Surface sediment samples were excluded if the sampled areas were dredged as part of maintenance activities after the sample had been collected.
- Older data were excluded if a sediment sampling location was resampled at a later date within 10 ft of the original location, either as part of a monitoring program or to characterize temporal trends.
- Surface sediment samples were excluded if they were collected after removal actions at two EAAs (i.e., Duwamish/Diagonal and Boeing Developmental Center south storm drain outfall near RM 3.9, so that the dataset would represent baseline (pre-early action) conditions.19
- Sediment samples near the Duwamish/Diagonal EAA were included only if they had been collected outside of a 200-ft perimeter of the EAA after the removal action (completed in 2004) because samples within 200 ft could have been influenced by the action.
- Sediment samples were categorized as surface sediment samples if they were collected to a depth of 15 cm or less. At 24 locations in the Norfolk EAA, samples were collected from the 0-to-2-cm depth in addition to the 0-to-10-cm depth on either the same date or at a later date. Only data from the 0-to-10-cm depth were included in the RI baseline dataset.

Table 4-3. Selection of surface sediment samples for the RI baseline dataset

Final Determination	Number of Samples
Included in RI baseline surface sediment dataset.	1,484[a]
Excluded because of subsequent dredging of those sediments.	58
Excluded because of more recent sampling outside of monitoring program.	95
Excluded because of more recent sampling as part of monitoring program.	30
Excluded because of data quality considerations.	23

[19] There is no EPA policy or guidance on whether baseline risk assessments should take into account the risk reduction achieved by removal actions that occur during the RI/FS. The risk reduction achieved at these two areas will be evaluated in the FS. Surface sediment data collected after dredging or other remedial activities at the Duwamish/Diagonal and Boeing Developmental Center south storm drain areas are described in Sections 4.2.1.1, 10.3.1, and 10.3.15.

Final Determination	Number of Samples
Excluded because 0-to-10-cm samples were preferred over 0-to-2-cm samples at these locations.	24
Excluded because post-removal sampling was not appropriate for the RI baseline surface sediment dataset at this location (Duwamish/Diagonal).	50
Total number of surface sediment samples	**1,764**

[a] Samples were collected from 1,365 discrete locations.

Both intertidal and subtidal sediment chemistry data were represented in the RI baseline dataset. An elevation of -2 ft MLLW was used to differentiate between intertidal and subtidal locations; this elevation corresponds to the shoreline (i.e., land/water interface) elevation defined using aerial photographs taken in 1999 by the US Fish and Wildlife Service (USFWS) (2000c). Surface sediment samples were collected from 593 intertidal locations and 772 subtidal locations in the RI baseline dataset (Maps 4-1 and 4-2).

Data quality reviews were conducted on each dataset that met the above criteria and was thus considered for use in the RI/FS. These reviews culminated in a determination of whether an individual dataset was considered suitable for all uses in the RI/FS (Windward 2005i, j, 2007a, d).

Appendix E.1 presents a more detailed description of the process for selecting data collected from dredged areas, from resampled locations less than 10 ft apart, and from the Duwamish/Diagonal EAA and the Boeing Developmental Center south storm drain area. Table E.2-1 in Appendix E, Section E.2, identifies the surface sediment samples that were excluded from the RI baseline surface sediment dataset and the rationale for their exclusion. Approximately 85% of the surface sediment samples (1,484 of 1,764) were included in the RI baseline dataset, as shown in Table 4-3.

The surface sediment sampling events included in the RI baseline dataset are listed in Table 4-4. The locations of the RI baseline surface sediment samples collected during site-specific or LDW-wide studies are shown on Maps 4-1 and 4-2, respectively, and sampling locations with their sample IDs are shown on Maps 4-3 and 4-4a through 4-4j.

Table 4-4. Summary of sampling events included in the RI baseline surface sediment dataset

Sampling Event	Event Code	Year of Sampling	Chemicals	Number of Sampling Locations[a]	Source
LDW RI: surface sediment sampling for chemical analyses	LDWRI-SurfaceSediment Round3	2006	metals, SVOCs, PCB Aroclors, dioxins and furans on subset of samples, butyltins, conventionals[b]	44	Windward (2007c)

Sampling Event	Event Code	Year of Sampling	Chemicals	Number of Sampling Locations[a]	Source
LDW RI: chemical analyses of benthic invertebrate and clam tissue samples and co-located sediment samples	LDWRI-Benthic	2005	metals, SVOCs, PCB Aroclors, selected PCB congeners on subset of samples, butyltins, conventionals[b]	35	Windward (2005a)
LDW RI: surface sediment sampling for chemical analyses and toxicity testing	LDWRI-SurfaceSediment Round1 & Round2	2005	metals, SVOCs, PCB Aroclors, selected PCB congeners and dioxins and furans on subset of samples, butyltins, conventionals[b]	160	Windward (2005c; 2005d)
Boyer Towing dock replacement	Boyer Towing	2004	metals, SVOCs, PCB Aroclors, TBT, conventionals[b]	3	WR Consulting (2004)
Slip 4 EAA site characterization	Slip4-EarlyAction	2004	PCB Aroclors, mercury	30	Integral (2004)
Rhône-Poulenc surface/subsurface sampling	RhônePoulenc 2004	2004	metals, organochlorine pesticides, SVOCs, conventionals[b]	21	EPA (2005d)
Norfolk CSO sediment remediation project 5-year monitoring program: Annual monitoring report – year 5, April 2004	Norfolk-monit7	2004	metals, PCB Aroclors, SVOCs, conventionals[b]	4	King County (2005d)
Triad approach to characterize PCB in a Washington riverine sediment site (USACE)	Jorgensen August 2004	2004	metals, PCB Aroclors, SVOCs, conventionals[b]	43	Wakeman (2005)
Duwamish/Diagonal perimeter monitoring – pre-dredge	DuwDiag-October2003	2003	metals, PCB Aroclors, organochlorine pesticides, SVOCs, conventionals[b]	12	King County et al. (2005)
Terminal 117 EAA site characterization	T117 Boundary Definition	2003-2004	PCB Aroclors; metals, TBT and SVOCs on selected samples, conventionals[b]	54	Windward et al. (2004a; 2004b)
Boeing Plant 2 transformer investigation – Phase 1	Plant 2-Trans-former Phase1	2003	PCB Aroclors, conventionals[b]	6[c]	Floyd Snider McCarthy (2004)
Norfolk CSO (Duwamish River) sediment cap recontamination – Phase I investigation	Ecology-Norfolk	2002	PCB Aroclors, conventionals[b]	17	Ecology (2003)
Norfolk CSO sediment remediation project 5-year monitoring program: Annual monitoring report – year 3, April 2002	Norfolk-monit5	2002	metals, PCB Aroclors, SVOCs, conventionals[b]	1	King County (2002a)
Norfolk CSO 5-year monitoring program, year 2, April 2001	Norfolk-monit4	2001	metals, PCB Aroclors, SVOCs, conventionals[b]	1	King County (2001b)
Norfolk CSO 5-year monitoring program – 12-month post-construction	Norfolk-monit3	2000	metals, PCB Aroclors, SVOCs, conventionals[b]	1	King County (2000c)
Norfolk CSO 5-year monitoring program – supplemental nearshore sampling	Norfolk-monit2b	2000	metals, PCB Aroclors, SVOCs, conventionals[b]	3	King County (2000b)
Outfall and nearshore sediment sampling report, Duwamish facility	James Hardie Outfall	2000	metals, PCB Aroclors, SVOCs, conventionals[b]	9	Weston (2000)

Sampling Event	Event Code	Year of Sampling	Chemicals	Number of Sampling Locations[a]	Source
Norfolk CSO 5-year monitoring program – 6-month post-construction	Norfolk-monit2a	1999	metals, PCB Aroclors, SVOCs, conventionals[b]	2	King County (2000d)
Norfolk CSO 5-year monitoring program – post-backfill	Norfolk-monit1	1999	metals, PCB Aroclors, SVOCs, conventionals[b]	2	King County (1999f)
EPA site inspection: Lower Duwamish River	EPA SI	1998	metals, PCB Aroclors, selected PCB congeners, and SVOCs; organochlorine pesticides, dioxins and furans, TBT, and VOCs on subset of samples; conventionals[b]	251	Weston (1999a)
King County CSO water quality assessment for the Duwamish River and Elliott Bay	KC WQA	1997	metals, PCB Aroclors, SVOCs, tetrabutyltin, conventionals[b]	14	King County (1999e)
Duwamish Waterway Phase 1 site characterization	Boeing SiteChar	1997	metals, PCB Aroclors, SVOCs, conventionals[b]	79[d]	Exponent (1998)
Duwamish Waterway sediment characterization study	NOAA SiteChar	1997	total PCBs, selected PCB congeners, total polychlorinated terphenyls, conventionals[b]	299	NOAA (1997; 1998)
Seaboard Lumber Phase 2 site investigation	Seaboard-Ph2	1996	metals, PCB Aroclors, SVOCs, conventionals[b]	20	Herrera (1997)
Rhône-Poulenc seep sampling	Rhône-Poulenc RFI-3	1996	metals, phenols	14	Rhône-Poulenc (1996)
RCRA facility investigation Duwamish Waterway sediment investigation, Plant 2 – Phase 2b	Plant 2 RFI-2b	1996	metals, PCB Aroclors, phthalates, conventionals[b]	36	Weston (1998)
Duwamish/Diagonal cleanup study – Phase 2	Duw/Diag-2	1996	metals, PCB Aroclors, SVOCs, conventionals[b]	10	King County (2000a)
Duwamish/Diagonal cleanup study – Phase 1.5	Duw/Diag-1.5	1995	metals, PCB Aroclors, SVOCs, conventionals[b]	9	King County (2000a)
Norfolk CSO sediment cleanup study – Phase 3	Norfolk-cleanup3	1995	PCB Aroclors, SVOCs, conventionals[b]	12	King County (1996)
Norfolk CSO sediment cleanup study – Phase 2	Norfolk-cleanup2	1995	metals, organochlorine pesticides, PCB Aroclors, selected PCB congeners, SVOCs, conventionals[b]	2	King County (1996)
RCRA facility investigation, Duwamish Waterway sediment investigation, Plant 2 – Phase 2a	Plant 2 RFI-2a	1995	metals, PCB Aroclors SVOCs, conventionals[b]	54	Weston (1998)
RCRA facility investigation, Duwamish Waterway sediment investigation, Plant 2 – Phase 1	Plant 2 RFI-1	1995	metals, PCB Aroclors, TPH, SVOCs, VOCs, conventionals[b]	66	Weston (1998)
Duwamish/Diagonal cleanup study – Phase 1	Duw/Diag-1	1994	metals, organochlorine pesticides, PCB Aroclors, SVOCs, conventionals[b]	31	King County (2001a)
Norfolk CSO sediment cleanup study – Phase 1	Norfolk-cleanup1	1994	metals, organochlorine pesticides, SVOCs, PCB Aroclors, conventionals[b]	13	King County (1996)

Sampling Event	Event Code	Year of Sampling	Chemicals	Number of Sampling Locations[a]	Source
Rhône-Poulenc RCRA facility investigation for the Marginal Way facility – Round 2	Rhône-Poulenc RFI-2	1994	SVOCs, conventionals[b]	6	Rhône-Poulenc (1995)
Results of sampling and analysis, sediment monitoring plan, Duwamish Shipyard, Inc.	Duwamish Shipyard	1993	metals, SVOCs, TBT, conventionals[b]	1	Hart Crowser (1993)
Harbor Island remedial investigation	Harbor Island RI	1991	metals, organochlorine pesticides, PCB Aroclors, SVOCs, VOCs, TPH, TBT, conventionals[b]	9	Weston (1993b)

[a] Samples are surface sediment grab samples from 0 to 15 cm unless otherwise noted.

[b] Conventionals may include all or some of the following parameters: grain size, pH, total organic carbon, and total solids.

[c] Five samples were collected from 0 to 5 cm. The top interval of 0 to 15 cm from a subsurface sediment core was also included in the RI baseline surface sediment chemistry dataset.

[d] Sample total does not include three reference samples that were collected upstream of the study area.

CSO – combined sewer overflow
EAA – early action area
EPA – US Environmental Protection Agency
County – King County
LDW – Lower Duwamish Waterway
NOAA – National Oceanic and Atmospheric Administration
PCB – polychlorinated biphenyl
RCRA – Resource Conservation and Recovery Act
RFI – RCRA facility investigation
RI – remedial investigation
SI – site inspection
SVOC – semivolatile organic compound
TBT – tributyltin
TPH – total petroleum hydrocarbon
USACE – US Army Corps of Engineers
VOC – volatile organic compound
WQA – water quality assessment

4.1.2.2 Subsurface sediment

Numerous subsurface sediment sampling events have been conducted since 1990. All subsurface sediment sampling events included in the RI baseline dataset are summarized in Table 4-5, and the locations of subsurface samples collected during site-specific and LDW-wide studies are shown on Maps 4-5 and 4-6, respectively. Sampling locations, including the sample IDs, are presented on Maps 4-7a through 4-7d. Sampling locations with data for specific 2 ft core intervals are shown in Maps 4-8a, 4-8b, and 4-8c.

Subsurface data collected in areas that have since been dredged, as noted in Table 4-5, were not included in the data summary tables in Section 4.2. These data were excluded because they do not represent existing conditions. However, data from areas that have been dredged are presented on subsurface sediment maps in Section 4.2 to show spatial patterns of chemicals at depth from a historical perspective.

Subsurface samples were collected from different intervals within the cores, depending on the objectives of the different sampling events. Subsurface sediment data were summarized in Section 4.2 according to sampling intervals; these intervals

included 1- or 2-ft intervals collected from the uppermost 10 ft of sediment.[20] Subsurface sediment data were also summarized in Section 4.2 from any sample interval. Eighty-eight percent of the subsurface data was included in the 1- and 2-ft data summaries (i.e., 786 of the 895 subsurface samples for which data were available from areas that have not been dredged, including calculated data for some of the 2-ft intervals). The remaining 12% of the samples were not included in the 1- and 2-ft interval data summaries because they were not sectioned into 1- or 2-ft intervals.

[20] For the calculation of summary statistics, actual sampling intervals were rounded to the nearest foot. The data for each interval were then assigned to the 1- or 2-ft sampling category that best approximated the actual sampling interval. If there were data for two 1-ft intervals at a particular location, the concentrations were also averaged to obtain a calculated concentration for the corresponding 2-ft interval; these intervals were included in addition to the 1-ft intervals.

Table 4-5. Summary of subsurface sediment sampling events included in the RI baseline dataset

Sampling Event	Event Code	Year of Sampling	Chemicals	Sample Summary	Sample Location Status	Reference
LDW RI: Subsurface sediment sampling for chemical analyses	LDW Subsurface Sediment 2006	2006	metals, PCB Aroclors, organochlorine pesticides, SVOCs, dioxins and furans, conventionals[a]	213 samples (impact corer or vibracorer) from 56 locations at 1- or 2-ft intervals to depths ranging from 6 to 12 ft below mudline ; 62 samples (impact corer or vibracorer) from 9 locations at 0.5-ft intervals to depths ranging from 2 to 4.5 ft below mudline	Area has not been dredged or remediated.	Windward and RETEC (2007)
Rhône-Poulenc (Rhodia) sediment and porewater investigation	Rhône-Poulenc AugSep 2004	2004	metals, PCB Aroclors, organochlorine pesticides, SVOCs, conventionals[a]	11 subtidal locations (push cores to a depth of 30 cm below mudline) and 13 intertidal locations (clam gun cores to a depth of 1 m below mudline); 2 samples (0 to 10 cm and > 10 cm) from each subtidal and intertidal location	Area has not been dredged or remediated.	EPA (2005d)
Additional vertical characterization, Duwamish Sediment Other Area	DSOAvert char3	2004	PCB Aroclors, conventionals[a]	7 samples (impact corer) from 5 locations; 1 to 3 samples from each location from 60 to 180 cm below mudline	Area has not been dredged or remediated.	MCS (2004b)
Triad approach (immunoassay as a real-time measure) to characterize PCBs in a Washington riverine sediment site	Jorgensen August 2004	2004	metals, PCB Aroclors, conventionals[a]	51 samples (impact corer) from 16 locations; 1 to 5 samples per location from 45 to 180 cm below mudline	Area has not been dredged or remediated.	unpublished data from USACE
Duwamish Sediment Other Area upriver (Area 1) sediment characterization	Jorgensen April 2004	2004	metals, PCB Aroclors, conventionals[a]	75 samples (impact corer) from 22 locations; 2 to 9 samples per location from 0 to 265 cm below mudline	Area has not been dredged or remediated.	MCS (2004c)
Boyer Towing dock replacement	Boyer Towing	2004	metals, PCB Aroclors, SVOCs, conventionals[a]	3 samples (push core) from 3 locations; 30 to 60 cm below mudline	Area has not been dredged.	WR Consulting (2004)
Additional vertical characterization, Duwamish Sediment Other Area	DSOAvert char2	2004	PCB Aroclors, conventionals[a]	28 samples (impact corer) from 15 locations; 1 to 3 samples from each location from 60 to 144 cm below mudline	Area has not been dredged or remediated.	MCS (2004a)
PSDDA characterization at the Lehigh Northwest Duwamish Waterway facility	LehighNW	2004	metals, PCB Aroclors, organochlorine pesticides, SVOCs, conventionals[a]	3 samples (impact corer) from 4 locations; 2 from 0 to 120 cm, 1 from 120 to 150 cm below mudline	Area was dredged in 2004.	MCS (2004d)
Slip 4 EAA site characterization	Slip4-EarlyAction	2004	PCB Aroclors, mercury, conventionals[a]	57 samples (vibracorer) from 11 locations; 4 to 6 samples taken at each location to a depth of 360 cm below mudline	Area has not been dredged or remediated.	Integral (2004)

Sampling Event	Event Code	Year of Sampling	Chemicals	Sample Summary	Sample Location Status	Reference
Terminal 117 (T-117) EAA site characterization	T-117 Boundary Definition	2003-2004	metals, PCB Aroclors; SVOCs (selected samples), conventionals[a]	107 samples (vibracorer) from 25 locations, 3 to 6 samples collected at each core location to a depth of 300 cm	Area has not been dredged or remediated.	Windward (2004a; 2004b)
Preliminary site investigation for the South Park Bridge Project	SouthPark Bridge	2003	metals, TBT, VOCs, SVOCs, pesticides, PCB Aroclors, conventionals[a]	11 samples (mud-rotary drill unit) from 2 locations; 5 to 6 samples from each location at depths from 2.5 to 100 ft below mudline	Area has not been dredged.	Wilbur Consulting (2004)
Boeing Plant 2 transformer investigation – Phase 1	Plant 2-Trans-former Phase1	2003	PCB Aroclors, conventionals[a]	46 samples (impact corer) from 13 locations; 3 to 5 samples at each location from 0 to 240 cm below mudline	Area has not been dredged or remediated.	Floyd Snider McCarthy (2004)
Sediment characterization results for the Duwamish River navigational channel turning basin	Turning-basin	2003	metals, PCB Aroclors, organochlorine pesticides, SVOCs, conventionals[a]	5 samples (vibracorer) from 5 locations; collected to depths of 144 to 390 cm	Area was most recently dredged in 2004.	Anchor (2003)
Glacier NW cement terminal	Glacier NW	2002	metals, PCB Aroclors, organochlorine pesticides, SVOCs, TBT, conventionals[a]	9 samples (vibracorer), each made from 3 separate cores collected to 90 to 150 cm below mudline	Area was dredged in 2005.	Pacific International Engineering (2002)
Data report, DSOA vertical characterization and Outfall 12 data collection. Duwamish sediment other area, Boeing Plant 2	DSOAvert char	2001	PCB Aroclors, conventionals[a]	115 samples (impact corer) from 37 locations; 2 to 6 samples at each location; most locations starting at 60 cm to depths of 150 to 280 cm below mudline	Area has not been dredged or remediated.	Pentec (2001)
Dredge material characterization, Duwamish Yacht Club	Duwam Yacht Club	1999	metals, PCB Aroclors, organochlorine pesticides, SVOCs, VOCs, TBT, conventionals[a]	6 samples (vibracorer) from 6 locations, each made from 2 separate cores collected from 50 to 65 cm below mudline	Area was dredged in 1999.	Hart Crowser (1999)
Sediment sampling and analysis, James Hardie Gypsum, Inc. – Round 1	Hardie Gypsum-1	1999	metals, PCB Aroclors, organochlorine pesticides, SVOCs, VOCs, conventionals[a]	5 samples (vibracorer) made from single cores down to 120 cm below mudline	Area was dredged in 1999.	Spearman (1999)
Sediment sampling and analysis, James Hardie Gypsum, Inc. – Round 2	Hardie Gypsum-2	1999	metals, PCB Aroclors, organochlorine pesticides, SVOCs, VOCs, conventionals[a]	9 samples (vibracorer) made from single cores down to 90 cm below mudline	Area was dredged in 1999.	Spearman (1999)
PSDDA sediment characterization of Duwamish River navigation channel: FY2000 operations and maintenance dredging data report	PSDDA99	1999	metals, PCB Aroclors, organochlorine pesticides, SVOCs, conventionals[a]	20 samples (vibracorer) from 20 locations; 3 borings made at each location; 18 samples from 0 to 120 cm; 2 samples from 120 to 240 cm below mudline	Area was proposed for dredging, but no dredging has been conducted since samples were collected.	SEA (2000a; 2000b)

Sampling Event	Event Code	Year of Sampling	Chemicals	Sample Summary	Sample Location Status	Reference
EPA site inspection: Lower Duwamish River	EPA SI	1998	metals, PCB Aroclors and selected congeners, organochlorine pesticides, SVOCs, VOCs, TBT, conventionals[a]	33 samples (vibracorer) from 17 locations; 1 to 2 samples collected at each location, from 0 to 60 and 60 to 120 cm below mudline	None of the areas were dredged after samples were collected, except at RM 0.5 (DR008), which was dredged in 2003/2004.	Weston (1999a)
Dredge material characterization Hurlen Construction Company & Boyer Alaska Barge Lines berthing areas	Hurlen-Boyer	1998	metals, PCB Aroclors, organochlorine pesticides, SVOCs, TBT, TPH, conventionals[a]	6 samples (vibracorer) from 2 locations at Boyer and 4 locations at Hurlen, each made from 2 separate cores collected from 60 to 120 cm below mudline	Area was dredged in 1998.	Hart Crowser (1998)
PSDDA sediment characterization of Duwamish River navigation channel: FY1999 operations and maintenance dredging data report	PSDDA98	1998	metals, PCB Aroclors, organochlorine pesticides, SVOCs, conventionals[a]	10 samples (vibracorer) from 12 locations; 7 samples from 0 to 60 to 90 cm, each from single location; 3 samples from 2 or 3 locations (0 to 60 cm, 0 to 120 cm, and 120 to 360 cm below mudline)	Area was most recently dredged in 1999.	Striplin (1998)
Duwamish/Diagonal cleanup study – Phase 2	Duw/Diag-2	1996	metals, PCB Aroclors, SVOCs, TPH, conventionals[a]	37 samples (vibracorer) from 14 locations; 1 to 6 samples per core, up to 270 cm below mudline	Area was dredged in 2003/2004.	King County (2000a)
RCRA facility investigation Duwamish Waterway sediment investigation, Plant 2 – Phase 2b	Plant 2 RFI-2b	1996	metals, PCB Aroclors, SVOCs, conventionals[a]	44 samples (vibracorer) from 16 locations; 2 to 4 samples per core, up to 480 cm below mudline	Area has not been dredged or remediated.	Weston (1998)
1996 USACE Duwamish O&M	PSDDA96	1996	metals, PCB Aroclors, organochlorine pesticides, SVOCs, VOCs, conventionals[a]	4 samples (vibracorer) from 4 locations; collected to a depth of 120 cm below mudline	Area was most recently dredged in 1999.	Striplin (1996)
Proposed dredging of Slip No. 4, Duwamish River, Seattle, WA	Slip4-Crowley	1996	metals, PCB Aroclors, organochlorine pesticides, SVOCs, VOCs, TBT, conventionals[a]	4 samples (vibracorer) composited from sediment at 9 locations; collected to a depth of 70 to 130 cm below mudline	Area was dredged in 1996.	PTI (1996)
Lone Star Northwest and James Hardie Gypsum – Kaiser dock upgrade	Lone Star-Hardie Gypsum	1995	metals, PCB Aroclors, organochlorine pesticides, SVOCs, VOCs, conventionals[a]	5 samples (vibracorer) from 4 locations; 4 collected to a depth of 120 to 150 cm, 1 at 120 to 360 cm below mudline	Area was dredged in 1995.	Hartman (1995)
Norfolk CSO sediment cleanup study – Phase 2	Norfolk-cleanup2	1995	metals, PCB Aroclors and selected congeners, organochlorine pesticides, SVOCs, VOCs, TPH, conventionals[a]	27 samples (vibracorer) from 3 locations; collected at 30- or 60-cm intervals up to 180 cm below mudline	Area was dredged and capped in 1999.	King County (1996)

Sampling Event	Event Code	Year of Sampling	Chemicals	Sample Summary	Sample Location Status	Reference
RCRA facility investigation Duwamish Waterway sediment investigation, Plant 2 – Phase 1	Plant 2 RFI-1	1995	metals, PCB Aroclors, TPH, SVOCs, VOCs, conventionals[a]	20 samples (vibracorer) from 12 locations; collected at 15- to 45-cm intervals down to 135 cm below mudline	Area has not been dredged or remediated.	Weston (1998)
Duwamish/Diagonal cleanup study – Phase 1	Duw/Diag-1	1994	metals, PCB Aroclors, organochlorine pesticides, SVOCs, TBT, conventionals[a]	12 samples (vibracorer) from 2 locations; collected at 15- to 30-cm intervals down to 150 cm below mudline	Area was dredged in 2003/2004.	King County (2001a)
Norfolk CSO sediment cleanup study – Phase 1	Norfolk-cleanup1	1994	metals, PCB Aroclors, organochlorine pesticides, SVOCs, VOCs, conventionals[a]	3 samples from 1 location; collected at 15 to 30, 30 to 45, and 45 to 60 cm below mudline	Area was dredged and capped in 1999.	King County (1996)
Lone Star Northwest – West Terminal USACE – Seattle, WA	Lone Star 92	1992	metals, PCB Aroclors, organochlorine pesticides, SVOCs, VOCs, conventionals[a]	1 sample (vibracorer) made from 2 separate cores collected to 120 cm below mudline	Area was dredged in 1995.	Hartman (1992)
Sediment sampling and analysis, South Park Marina, Duwamish Waterway, Seattle, WA	South Park Marina	1991	metals, PCB Aroclors, organochlorine pesticides, SVOCs, conventionals[a]	2 samples (vibracorer), each made from 2 separate cores collected to 120 cm below mudline	Area was dredged in 1992.	Spearman (1991)
Subsurface sediment chemistry and geotechnical analysis related to proposed dredging at Terminal 105[b]	Terminal 105	1985	metals, PCBs, SVOCs, and DDT	35 samples (hollow stem auger) from 12 locations, 2 to 5 samples per core, down to approximately 9 m below mudline	Area has not been dredged.	Port of Seattle unpublished data (1985)

Note: For reference, 1 ft is equal to 30 cm.

[a] Conventionals may include all or some of the following parameters: grain size, pH, total organic carbon, and total solids.

[b] The data from this sampling event were never validated and were collected prior to 1990, which is the earliest year established in the Phase 1 RI for data to be considered acceptable for all uses in the RI/FS. No additional data quality review was conducted for these data; therefore, they were not included in the data summaries or discussions of nature and extent of contamination in Section 4.2.

CSO – combined sewer overflow

DSOA – Duwamish sediment other area (Boeing Plant 2)

EAA – early action area

EPA – Environmental Protection Agency

LDW – Lower Duwamish Waterway

O&M – operation and maintenance

NW – northwest

PCB – polychlorinated biphenyl

PSDDA – Puget Sound Dredged Disposal Analysis

RCRA – Resource Conservation and Recovery Act

RI – remedial investigation

RM – river mile

SVOC – semivolatile organic compound

TBT – tributyltin

TPH – total petroleum hydrocarbons

USACE – US Army Corps of Engineers

VOC – volatile organic compound

4.1.2.3 Fish, shellfish, and benthic invertebrate tissue

Tissue chemistry data for the study area are available for multiple tissue types collected as part of several sampling events conducted since 1990. Data from 295 composite samples of fish and shellfish tissue from both sampling events conducted for the RI and historical sampling events were considered suitable for use (Table 4-6). Approximately 80% of these samples were collected in 2004 and 2005 as part of the RI. The remainder of the samples were collected during six other sampling events from 1992 to 2002. Collection locations for fish tissue samples are shown on Map 4-9, and collection locations for invertebrate and shellfish samples are shown on Map 4-10.

Some additional tissue chemistry data available for the LDW were not included (Table 4-7). King County collected shiner surfperch and English sole data for PCBs in 2006; these data were not included in data summaries in Section 4.2.1 because they were recently received, but were included in the temporal evaluation in Section 4.2.1.4. Juvenile chinook salmon whole-body and stomach-content data from Varanasi et al. (1993) and from the National Marine Fisheries Service (NMFS) (2002) were available but were not used because the quality assurance/quality control (QA/QC) data were not readily available for EPA review. Data for adult salmon were excluded because it was estimated that less than 1% of the PCB body burden in adult salmon migrating through the LDW could have been obtained from prey items consumed in the LDW, based on an analysis presented in Appendix B.

As part of the RI, two individual shiner surfperch were collected in Slip 4, and ten individual shiner surfperch were collected in subarea T2E (RM 2.1 to RM 2.4). These data were not included because only data from composite tissue samples were used. The total PCB concentrations in individual fish samples were generally lower than concentrations in composite samples. In addition, juvenile chinook salmon collected upstream of the LDW during the RI were not used; mean PCB concentrations in these fish were lower than those in fish collected in the LDW.[21]

[21] Data for upstream sampling were presented in the data report for the juvenile chinook salmon sampling event (Windward 2004d).

Table 4-6. Summary of tissue sampling events used in the RI dataset

Study	Year of Sampling	Species	Number of Composite Tissue Samples	Individuals per Sample	Sample Type	Use in Risk Assessments		Chemicals
						HHRA	ERA	
LDW RI: chemical analyses of fish and crab tissue samples collected in 2005 (Windward 2006c)	2005	English sole	10	5	skin-on fillet	X		PCB Aroclors,[c] lipids
			21	5	whole body[a]	X	X	
		Dungeness crab	3	5	edible meat	X	X	
			3	5	hepatopancreas[b]	X	X	
		slender crab	1	5	edible meat	X	X	
			1	10	hepatopancreas[b]	X	X	
		shiner surfperch	22	10	whole body	X	X	
		Pacific staghorn sculpin	4	10	whole body		X	
LDW RI: chemical analyses of fish and crab tissue samples collected in 2004 (Windward 2005b)	2004	English sole	7	5	skin-on fillet	X		metals, SVOCs, organochlorine pesticides, PCB Aroclors (PCB congeners in subset of samples), TBT, lipids
			21	5	whole body	X	X	
		starry flounder	1	5	skin-on fillet	X		
			3	5	whole body	X	X	
		Dungeness crab	7	5	edible meat	X	X	
			3	6 – 15	hepatopancreas[b]	X	X	
		slender crab	12	5	edible meat	X	X	
			4	15 – 18	hepatopancreas[b]	X	X	
		shiner surfperch	24	9 – 10	whole body	X	X	
		striped perch	1	12	skin-on fillet	X		
		pile perch	1	12	skin-on fillet	X		
		Pacific staghorn sculpin	24	7 – 10	whole body		X	
LDW RI: chemical analyses of benthic invertebrate and clam tissue samples and co-located sediment samples (Windward 2005a)	2004	benthic invertebrates	20	nd	whole-body		X	metals, SVOCs, alkylated PAHs, organochlorine pesticides, PCB Aroclors (PCB congeners in subset of samples), TBT, lipids
		Eastern softshell clam	14	19 – 52	whole-body	X	X	
LDW RI: juvenile chinook salmon sampling (Windward 2004d)	2004	chinook salmon	18	7 – 12	whole-body		X	metals, SVOCs, alkylated PAHs, organochlorine pesticides, PCB Aroclors, TBT, lipids
			1	72[d]	stomach contents		X	

Study	Year of Sampling	Species	Number of Composite Tissue Samples	Individuals per Sample	Sample Type	Use in Risk Assessments		Chemicals
						HHRA	ERA	
East Waterway, Harbor Island Superfund site: Tissue chemistry results for juvenile chinook salmon collected from Kellogg Island and East Waterway (Windward 2002)	2002	chinook salmon	6	6-7	whole body		X	mercury, PCB Aroclors, lipids
Waterway Sediment Operable Unit Harbor Island Superfund Site – Assessing human health risks from the consumption of seafood (ESG 1999)	1998	English sole	3	5	skinless fillet	X		mercury, TBT, PCB Aroclors
		red rock crab	2	5	edible meat	X		
		Dungeness crab	1	5	edible meat	X		
		red rock crab/ Dungeness crab	1	5	edible meat	X		
		striped perch	2	5 – 8	skin-on fillet	X		
King County Combined Sewer Overflow Water Quality Assessment for the Duwamish River and Elliott Bay (King County 1999e)[e]	1996 – 1997	Dungeness crab	2	3	edible meat	X	X	metals, TBT, SVOCs, PCB Aroclors, lipids
			1	3	hepatopancreas[b]	X	X	
		English sole	3	20	skinless fillet	X		
		shiner surfperch	3	10	whole body		X	
		wild mussels	22	50 – 100	edible meat	X	X	
		transplanted mussels	32	50 – 100	edible meat			
		amphipods	6	nd	whole body		X	
Puget Sound Ambient Monitoring Program – annual sampling (West et al. 2001)	1992	English sole	3	5 – 20	skinless fillet	X		SVOCs, organochlorine pesticides, PCB Aroclors, arsenic, copper, lead, mercury, lipids
	1995	English sole	3	5 – 20	skinless fillet	X		organochlorine pesticides, PCB Aroclors, arsenic, copper, lead, mercury, lipids
	1997	English sole	3	5 – 20	skinless fillet	X		organochlorine pesticides
Elliott Bay/Duwamish River Fish Tissue Investigation (Battelle 1996; Frontier Geosciences 1996)	1995	English sole	3	6	skinless fillet	X		PCB Aroclors, mercury, methylmercury, TBT, lipids

[a] Whole-body samples include 10 composite samples in which concentrations were calculated using results from separate analyses of fillet tissue and remainder tissue (remainder tissue includes tissue and fluids remaining after fillets were removed from the specimens). The calculated English sole whole-body concentrations were based on the relative weights and total PCB concentrations in skin-on fillet and remainder tissues.

[b] Data from composite hepatopancreas samples were mathematically combined with data from composite samples of edible meat to form composite samples of edible meat plus hepatopancreas. Total PCB concentrations in whole-body (i.e., edible meat plus hepatopancreas) crab were calculated assuming 69% (by weight) edible meat and 31% hepatopancreas, based on the relative weights of these tissues in a 16.6-cm Dungeness crab dissected by Windward in 2004 (unpublished data).

[c] A subset of samples from the 2005 sampling event was also analyzed for PCB congeners. These 2005 PCB congener data are included in the discussion of the nature and extent of PCBs in Section 4.2. However, the 2005 data were not included in the risk assessments because the final data were not available at the time those documents were prepared.

[d] An unknown fraction of these stomachs were empty.

[e] Additional samples of cooked crab and English sole were collected during the King County WQA (King County 1999e), but these data were not used.

ERA – ecological risk assessment
HHRA – human health risk assessment
LDW – Lower Duwamish Waterway
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
SVOC – semivolatile organic compound
RI – remedial investigation
TBT – tributyltin

Table 4-7. Summary of additional tissue sampling events not included in the RI dataset

Study	Year of Sampling	Species	Sample Type	Number of Composite Samples	Chemicals	Reference
King County fish tissue sampling and chemical analysis in the LDW[a]	2006	shiner surfperch	whole body	7[b]	PCB Aroclors, phthalates, lipids	Anchor and King County (2007)
		English sole	whole body	6[b, c]		
LDW RI – juvenile chinook salmon sampling	2003	juvenile chinook salmon	whole body	9[d]	Metals, PCB Aroclors, PAHs, organochlorine pesticides, lipids	Windward (2004d)
Puget Sound Ambient Monitoring Program – annual sampling	1992 – 1998	adult coho and chinook salmon	fillets	153[b]	SVOCs, arsenic, copper, lead, mercury, PCB Aroclors, organochlorine pesticides, lipids	West (2001)

[a] These data were not included in data summaries in Section 4.2.1, but were included in the temporal evaluation in Section 4.2.1.4.

[b] Collected from Area T1 only.

[c] Nine individual English sole whole-body tissue samples were also collected and analyzed for PCB Aroclors, phthalates, and lipids.

[d] Collected from upstream locations in the Green River.

LDW – Lower Duwamish Waterway
PCB – polychlorinated biphenyl
PAH – polycyclic aromatic hydrocarbon
RI – remedial investigation
SVOC – semivolatile organic compound

4.1.2.4 Surface water

LDW surface water data are available from three sampling events conducted for the King County water quality assessment (WQA) in 1996 and 1997 (King County 1999e) and during an additional sampling event in 2005, in which whole water samples were collected and analyzed for PCBs and conventional parameters (Mickelson and Williston 2006) (Table 4-8).

Grab samples were collected along transects at three locations in the LDW in 1996 and 1997: RM 1.1 (in the vicinity of the Brandon CSO outfall), RM 1.9 (in the vicinity of the Southwest Michigan CSO outfall), and RM 4.9 (in the vicinity of the Norfolk CSO outfall) (Map 4-11). Samples were collected weekly from October 1996 to June 1997 at all three locations, except during storm events, when sampling was conducted on three successive days following the storm. Chemicals analyzed in both filtered and unfiltered samples included metals and trace elements and semivolatile organic compounds (SVOCs).

PCBs, pesticides, and polycyclic aromatic hydrocarbons (PAHs) were not analyzed in surface water samples collected in 1996 and 1997 because of the difficulty in obtaining suitably low RLs to allow concentrations of these chemicals to be reliably reported.[22] Instead, King County deployed semipermeable membrane devices (SPMDs) at two locations in the LDW from March 26 to April 8, 1997.[23] Three SPMDs, including one field duplicate SPMD, were deployed just offshore of the Duwamish/Diagonal CSO, and two SPMDs were deployed just offshore of the Brandon Street CSO. The SPMDs were analyzed for PCBs, organochlorine pesticides, and PAHs. Chemical concentrations in water were calculated using the mass of chemicals detected in each SPMD and the chemical-specific partition coefficients between the polyethylene and water, which were derived from laboratory experiments conducted at Battelle's Marine Science Laboratory. These calculated concentrations are summarized in Section 4.2 but were not included in the risk assessments because of uncertainty in using calculated concentrations.

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[22] King County analyzed a limited number of water samples for PCBs, organochlorine pesticides, and PAHs, but all results were reported as not detected. Because obtaining lower detection limits was not possible at that time, analyses for these analytes were discontinued in grab water samples.

[23] The SPMD sampler consisted of layflat, polyethylene tubing, containing a high-molecular-weight neutral lipid. SPMDs mimic the transport across biological membranes and can be used to provide time-averaged chemical concentrations in water, which may include episodic events. Given an adequate deployment time, the concentrations of lipophilic compounds in the SPMD should come close to equilibrium with the concentrations in the water.

Table 4-8. Summary of surface water sampling events included in the RI dataset

Sampling Event	Dates of Sampling	Location	Depths Sampled	Sample Type	Chemicals	Reference
KC WQA	Weekly from October 1996 to June 1997[a]	RM 1.1 (near the Brandon CSO outfall) – east and west banks, mid-channel	1 m below surface and 1 m above river bottom	filtered and unfiltered grab samples	antimony, arsenic, cadmium, cobalt, copper, lead, mercury, nickel, selenium, silver, thallium, vanadium, zinc, and SVOCs[c]	King County (1999e)
		RM 1.9 (near the SW Michigan CSO outfall) – east and west banks, mid-channel				
		RM 4.9 (near the Norfolk CSO outfall) – east and west banks[b]				
KC WQA – SPMD sampling	March 26 to April 8, 1997	Duwamish/Diagonal CSO	1 m and 3 m below surface	SPMDs	PCBs, organochlorine pesticides, and PAHs	King County (1999e)
		Brandon Street CSO				
KC WQA 2005	Four dates in 2005: August 22, September 26, November 28, and December 19	RM 0.0 at the south end of Harbor Island	1 m below surface and 1 m above river bottom	unfiltered grab samples	209 PCB congeners, total PCBs[d]	King County (2005b), Mickelson and Williston (2006)
		RM 3.3 at the South Park Bridge				

[a] After storm events, samples were collected on three successive days following the event.

[b] Only two transects were sampled because of the narrow width and shallow water depth in the LDW at this location.

[c] PCBs, pesticides, and PAHs were not analyzed in these surface water samples because of difficulties with obtaining suitably low RLs.

[d] Total PCB concentrations were calculated as the sum of all detected PCB congener concentrations (data were blank-qualified, and RLs for undetected PCB congeners were not included).

CSO – combined sewer overflow	SPMD – semipermeable membrane devices
KC – King County	SVOC – semivolatile organic compound
PAH – polycyclic aromatic hydrocarbon	SW – southwest
PCB – polychlorinated biphenyl	WQA – water quality assessment
RM – river mile	

In 2005, King County collected water column samples for PCB congener analysis during two dry-weather sampling events in August and September and two wet-weather sampling events in November and December (King County 2005b). Two locations were sampled in the LDW: one location at RM 0.0 at the southern end of Harbor Island, and one at RM 3.3 at the South Park Bridge (Map 4-11). All 209 PCB congeners were analyzed using high-resolution methods, and total PCB concentrations were calculated as the sum of all detected PCB congener concentrations (data were blank-qualified, and RLs for undetected PCB congeners were not included).

4.1.2.5 Seeps

Seep water samples have been collected from shoreline areas along the LDW as part of the RI and other investigations conducted since 1990 (Table 4-9). In addition to the areas sampled for the RI, specific sites that have been sampled include Rhône-Poulenc, Boeing Plant 2, Boeing Isaacson, Terminal 117 (T-117), and GWI (Map 4-11).

Table 4-9. Summary of seep sampling events included in the RI dataset

Site	Location	Number of Seeps Sampled	Year of Sampling	Sampling Method	Chemicals	Data Source
Entire LDW	RM 0.2 to RM 4.3	18	2004	mini-piezometers and funnel with attached tubing to capture direct flow	metals, SVOCs, VOCs, PCB Aroclors, pesticides, and TPH	Windward (2004b)
T-117[a]	RM 3.5 to RM 3.7 E	3	2003	funnel with attached tubing to capture direct flow	metals, SVOCs, and PCB Aroclors	Windward et al. (2004c)
Great Western International[b]	RM 2.3 to RM 2.4 E	5	1999	na	SVOCs and VOCs	Terra Vac and Floyd & Snider (2000)
		9	1998	na	VOCs	Terra Vac and Floyd & Snider (2000)
		10	1998	na	VOCs	Terra Vac and Floyd & Snider (2000)
		4	1997	na	VOCs	Terra Vac and Floyd & Snider (2000)
		5	1996	na	VOCs	Terra Vac and Floyd & Snider (2000)
		7	1995	na	VOCs	Terra Vac and Floyd & Snider (2000)
		7	1995	na	VOCs	Terra Vac and Floyd & Snider (2000)

Site	Location	Number of Seeps Sampled	Year of Sampling	Sampling Method	Chemicals	Data Source
		8	1994	na	VOCs	Terra Vac and Floyd & Snider (2000)
		2	1994	na	VOCs	Terra Vac and Floyd & Snider (2000)
		6	1994	na	VOCs	Terra Vac and Floyd & Snider (2000)
Boeing Plant 2	RM 2.9 to RM 3.6 E	18	1995	capture of direct flow into sampling container	metals, SVOCs, VOCs, and PCB Aroclors, TPH	Weston (1998)
Rhône-Poulenc[c]	RM 4.0 to RM 4.2 E	7	1995	PVC pipe was placed horizontally out from the bank to capture direct flow; or a small hole was dug and contents were transferred to the sample container.	metals, VOCs, and phthalates	Rhône-Poulenc (1996)
Boeing Isaacson	RM 3.7 E	1	2000	A peristaltic pump was used to pump seep water through a 0.45-micron filter into the sample container.	arsenic	ERM and Exponent (2000)

[a] One seep was re-sampled in 2004.

[b] A total of 10 seeps were sampled; some of the seeps were sampled on more than one date.

[c] Data are available only for detected chemicals (metals, bis(2-ethylhexyl) phthalate, and formaldehyde).

LDW – Lower Duwamish Waterway
na – not available
PCB – polychlorinated biphenyl
PVC – polyvinyl chloride
RM – river mile
SVOC – semivolatile organic compound
T-117 – Terminal 117
TPH – total petroleum hydrocarbon
VOC – volatile organic compound

4.1.2.6 Porewater

Table 4-10 summarizes the available porewater datasets collected from the LDW. Samples were collected from Great Western International (GWI) and Boeing Plant 2/ Jorgensen Forge in 2005 and analyzed for volatile organic compounds (VOCs) as part of the RI. At Rhône-Poulenc, porewater samples were collected in 2004 and were analyzed for metals and organic compounds. As part of the EPA site inspection (SI), porewater samples were collected in 1998 throughout the LDW and were analyzed for metals. Porewater samples were collected and analyzed for butyltins during five sampling events: Glacier Northwest (NW), Duwamish Yacht Club, Hurlen-Boyer, PSDDA99, and PSDDA98. Data from the PSDDA99 sampling event are discussed in this RI; samples from the other four events are not discussed because they were collected from areas that have been dredged and thus do not represent current conditions. Porewater locations sampled during the GWI, Boeing Plant 2/Jorgensen

Forge, Rhône-Poulenc, EPA SI, and PSDDA99 sampling events are shown on Map 4-12.

Table 4-10. Summary of porewater sampling events

Site or Sampling Event	Location	Number of Samples	Year of Sampling	Sampling Method	Chemicals	Reference
Great Western International	RM 2.3 to RM 2.4 E	14	2005	mini-piezometer and peeper	VOCs	Windward (2006a)
Boeing Plant 2/ Jorgensen Forge	RM 3.6 E	14	2005	mini-piezometer and peeper	VOCs	Windward (2006a)
Rhône-Poulenc	RM 4.0 to RM 4.2 E	16	2004	mini-peizometer and seepage meter	metals, BTEX, PAHs, phthalates	EPA (2005d)
EPA SI	entire LDW	15	1998	centrifuge from sediment	metals and butyltins	Weston (1999a)
Glacier NW[a]	RM 1.4 to RM 1.5 W	5	2002	na	butyltins	PIE (2002)
Duwamish Yacht Club[a]	RM 4.1 W	2	1999	na	tributyltin	Hart Crowser (1999)
PSDDA99	RM 1.9 to RM 3.4 W	20	1999	na	tributyltin	SEA (2000b)
Hurlen-Boyer[a]	RM 2.4 to RM 2.7 W	2	1998	centrifuge from sediment	tributyltin	Hart Crowser (1998)
PSDDA98[a]	RM 4.4 to RM 4.6 in the navigation channel	2	1998	centrifuge from sediment	tributyltin	SEA (1998)

[a] Data from these events are not summarized in this document because sediment was dredged after porewater samples were collected.

BTEX – benzene, toluene, ethylbenzene, and xylene

EPA – US Environmental Protection Agency

LDW – Lower Duwamish Waterway

na – not available (porewater sampling method was not presented in available documents)

NW – northwest

PAH – polycyclic aromatic hydrocarbon

PIE – Pacific International Engineering, PLLC

PSDDA – Puget Sound Dredged Disposal Analysis

RM – river mile

SEA – Striplin Environmental Associates, Inc.

SI – site inspection

TBT – tributyltin

VOC – volatile organic carbon

4.1.3 Data suitability

Once data were determined to meet met DQOs, the suitability of data for specific purposes in the RI was evaluated according to EPA guidance (EPA 1988, 1992b). Of primary importance was the degree to which the data adequately represented site-related contamination and potential exposure of humans and ecological receptors at the site. Other considerations in determining the suitability of datasets that met DQOs included the availability of data validation results, whether results may have been confounded by analytical interferences, and whether data represented a significant departure from the standard analytical methods used for other samples in the data set.

4.1.3.1 Representativeness

This section provides a discussion of whether the available sediment, tissue, and water data represent the ranges and distributions of chemical concentrations in the LDW.

Surface Sediment

Many historical environmental sampling events have included the collection and analysis of surface sediment from the LDW (Table 4-4). These studies were designed for site-wide or reconnaissance surveys (e.g., Boeing SiteChar, EPA SI, and NOAA SiteChar) or focused investigations of suspected areas of contamination (e.g., Boeing Resource Conservation and Recovery Act [RCRA] facility investigation [RFI], Rhône-Poulenc RFI). Because many of the sediment samples were collected to characterize contaminated areas, the dataset as a whole may have a relatively higher proportion of elevated chemical concentrations than is representative of site-wide conditions within the LDW. Of the 1,365 sampling locations in the RI baseline surface sediment dataset, 880 of these locations were sampled as part of a site-wide sampling event, and 485 locations were sampled during focused investigations, as shown on Maps 4-1 and 4-2, respectively.

Subsurface Sediment

Many historical environmental sampling events have included the collection of subsurface sediment from the LDW (Table 4-5). Of the 895 subsurface samples (from 309 cores that have not been dredged), 265 samples (from 78 cores) were collected as part of site-wide events for a more general characterization of the nature and extent of contamination. Of these 265 samples, 234 samples (from 62 cores) were collected in 2006 as part of the RI and 31 samples (from 16 cores) were collected during the EPA SI in 1998 (Map 4-6). The remaining 630 samples (from 231 cores) were collected from areas of suspected contamination (Map 4-5). Because many of the subsurface sediment samples were collected to characterize contaminated areas, the dataset as a whole likely has a higher proportion of samples with chemical concentrations that are elevated relative to the average chemical concentrations expected throughout the entire LDW.

Tissue

To represent chemical concentrations in tissue within the LDW, data for a variety of biological tissue types are necessary (Table 4-6). In addition, a representative set of tissue data should include spatial coverage for each species according to its home range and appropriate size classes to meet specific objectives of sampling for each organism, and should contain a sufficient number of samples to represent differing exposure regimes within the LDW.

To supplement tissue data for the LDW that were collected prior to the RI, large numbers of fish and crabs were collected as part of the RI data collection efforts for use primarily in the risk assessments (Windward 2004e, f). These data were also used to summarize the nature and extent of chemical contamination in biological matrices of

the LDW. Species were selected to represent exposure of humans and ecological receptors, and thus a variety of fish and crab species were collected. Samples collected for wildlife exposure included the entire specimen, while samples collected for human exposure included only the edible portions of fish or crab species (i.e., fillet and edible meat). The tissue sampling events conducted in 2004 and 2005 provided 28 to 46 composite samples for each of three fish species for whole-body tissue, 18 composite samples of English sole or starry flounder fillet tissue, 23 composite samples of crab edible meat tissue, and 11 composite samples of crab hepatopancreas tissue, all of which exceeded the 6 samples desirable to calculate a 95% upper confidence limit (UCL) concentration for each chemical and tissue type for the LDW (Windward 2004f).

Tissue sampling performed for the RI was designed to generate data representative of the range of concentrations of key chemicals to which an organism may be exposed and to provide spatial coverage of the LDW. For mobile species such as fish and crabs, spatial representation of the LDW was accomplished by collecting samples from four sampling areas distributed throughout the LDW. These areas were selected based on a rolling average concentration of total PCBs in sediment, as discussed in detail in the fish and crab QAPP (Windward 2004f), to ensure that samples were collected within areas representative of low, medium, and high total PCB concentrations in sediment. For benthic invertebrates, 20 sampling locations were chosen to represent areas with a range of concentrations of total PCBs, arsenic, and lead in sediment, as well as to provide spatial coverage of intertidal areas in the LDW. For clams, 10 sampling locations were placed in known clam habitat areas to cover the concentration range of PCBs, cPAHs, and arsenic in sediment (Windward 2004e). Target sizes of fish and crabs collected during the RI were selected to represent the prey size ranges preferred by piscivorous wildlife receptors and reasonable size ranges of seafood consumed by people (Windward 2004f).

Surface Water

Surface water data collected during the King County WQA for many chemicals were based on a large dataset of whole water samples collected weekly from October 1996 to June 1997 along transects at three sites in the LDW (King County 1999e). These data represent a seasonal range of flow conditions at each of the three sites. These data were used in the RI to estimate exposure of wildlife to chemicals through water ingestion in the ERA; these data were also used in describing the nature and extent of contamination in the LDW. These data were considered sufficient for use in the ERA because ingestion of chemicals via water is a very minor component of the total ingestion of chemicals by birds and mammals from prey, sediment, and water combined (Appendix A, Section A.5.1.3.3).

Surface water samples for analysis of PCB congeners were collected at two LDW locations on four different dates in 2005. These data were used to update King County's EFDC model (Mickelson and Williston 2006). The results of the updated EFDC model were used in the RI to refine the predictions of total PCB concentrations in the LDW water column for the food web model (FWM) (Appendix D). In addition,

total PCB concentrations calculated as the sum of congeners from these datasets were used to define the upper and lower limits of total PCB concentrations in water for the FWM. The FWM was used to estimate sediment RBTCs for total PCBs. These data were sufficient for use in the ERA because aqueous PCBs are a very minor component of the total ingested dose, as noted above for other chemicals.

Porewater

Porewater samples were collected as part of the RI to evaluate risks to the benthic invertebrate community [24] in areas assumed to represent worst-case exposure areas based on potential exposure to groundwater contamination. Porewater data were collected from two areas of the LDW (GWI and Boeing Plant 2/Jorgensen Forge) to investigate potential worst-case conditions for discharge of VOCs from the groundwater (Windward 2006a). Other areas in the LDW were assumed to have a lower potential for groundwater discharge of VOCs based on existing source data. Detailed groundwater conceptual site models were developed for both the GWI and Boeing Plant 2/Jorgensen Forge sites to ensure that sampling was conducted in areas with the highest potential for groundwater discharge (Windward 2005f). Porewater samples were collected using peepers deployed 10 cm beneath the sediment surface and left to equilibrate for 2 weeks. Because of their placement within the biologically active zone and the equilibration time allowed, these samples should reasonably represent benthic invertebrate exposure conditions within these areas.

Other historical porewater data provide information about potential sources in the LDW and were therefore included in the characterization of nature and extent of contamination. The Rhône-Poulenc investigation (EPA 2005d) collected porewater data for metals and SVOCs from locations along the shoreline of the facility using seepage meters and piezometers. The locations and methods for sampling porewater were designed to represent groundwater or surface water in the subsurface-surface transition zone from areas most likely to be contaminated from the Rhône-Poulenc site.

The EPA SI collected porewater data for metals from 15 locations throughout the LDW as part of a screening-level investigation of the LDW (Weston 1999a). The objectives for selecting these locations were not presented in the EPA SI documents; it is assumed that these samples were collected to represent a range of conditions throughout the LDW. Although porewater samples were analyzed for butyltins during five sampling events, data from four of these events were collected from areas that were subsequently dredged, so they do not represent current conditions. In the remaining sampling event, 20 samples were collected between RM 1.9 to RM 3.4.

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[24] Because VOCs have a low affinity for sediment, analysis of porewater is the best way to estimate exposure. The primary use of porewater data in the RI was for the evaluation of risk to benthic invertebrates from exposure to VOCs. For other chemicals that have a higher affinity for sediment, risk to benthic invertebrates was better addressed using sediment chemistry data.

Seep Water

As part of the RI, a reconnaissance survey was conducted in May 2004 to identify seeps along the entire LDW shoreline (Windward 2004b). Eighteen seeps from throughout the LDW were selected for collection and chemical analysis of seep water samples. Seeps were selected based on an evaluation of the proximity of the sampling locations to potential sources, surface sediment chemistry, and observations made during the reconnaissance survey. The primary objective of this sampling was to characterize suspected sources of contaminants to the LDW. Although rainfall was lower than usual in the spring of 2004 an analysis of expected groundwater elevations indicated that the lower rainfall would not have affected the ability to identify or sample seeps at that time (Windward 2004g)

In addition to the 18 seep locations sampled as part of the RI, seep data were available for four other sites along the LDW. Available data were used to describe the nature and extent of chemical concentrations in seep water.

4.1.3.2 QA/QC results

All datasets used in the RI (i.e., datasets for sediment, tissue, surface water, porewater, and seeps) were validated by the original authors of the individual studies or by third parties. Summaries of these data validations for historical sampling events have been presented in technical memoranda prepared by Windward (Windward 2005i, j, 2007d). Data validation reports for samples collected as part of the RI were included in the data reports for each sampling event. The only surface sediment chemistry dataset not considered acceptable for all uses in the RI/FS because the DQOs were not met were data collected from the Rhône-Poulenc facility in 1994; these data were not considered acceptable because Form 1s were not available.

Based on data quality, approximately 1,000 analytical results, across all media types, were qualified as unusable out of more than 150,000 analytical results. These data were excluded because of concerns such as extremely low matrix or surrogate spike recoveries, identified by the analytical laboratory or by the data validator. Almost all of the rejected results were for SVOCs (i.e., 1,2,4-trichlorobenzene, 1,2-dichlorobenzene, 2,4-dimethylphenol, 3,3′-dichlorobenzidine, 3-nitroaniline, 4-chloroaniline, 4-nitroaniline, aniline, benzidine, benzoic acid, hexachlorocyclopentadiene, pentachlorophenol, and pyridine). In addition, some sediment results for metals (i.e., antimony, cadmium, silver, and selenium) were rejected, as were several results for organochlorine pesticides (i.e., delta-benzene hexachloride [BHC], endosulfan sulfate, and endrin aldehyde). Data qualified as unusable were not used in the risk assessments or presented in the RI.

Analytical interference with the quantification of organochlorine pesticides from the presence of PCB congeners occurred during the analyses of benthic invertebrate tissue (Windward 2005a) and fish and crab tissue (Windward 2005b). This issue was identified by both the analytical laboratory and the data validator. The organochlorine pesticides were analyzed using EPA Method 8081A (gas chromatography with

electron capture detection [GC/ECD]), which is a standard method used in many environmental investigations for organochlorine pesticides. The detected results for both the benthic invertebrate and clam tissue samples collected in 2004 (Windward 2005a) and the fish and crab tissue samples collected in 2004 (Windward 2005b) were qualified JN, which indicates "the presence of an analyte that has been 'tentatively identified' and the associated numerical value represents its approximate concentration" (EPA 1999c). These data were qualified based on the probable interference in the organochlorine pesticide analysis from PCB congeners.

The JN-qualified results are highly uncertain and biased high. The high bias for DDTs was confirmed by re-analyzing six sediment samples co-located with benthic invertebrate tissue samples and eight fish and crab tissue samples that had high PCB and DDT concentrations using a gas chromatography/mass spectrometry (GC/MS) method, which is not susceptible to analytical interference by PCBs for organochlorine pesticides. Additional details regarding the confirmation analysis results are presented in Section A.2.4.2 of Appendix A. Both the HHRA and the ERA address the uncertainty associated with the use of the JN-qualified organochlorine pesticide data in evaluating risks to human and ecological receptors. This uncertainty is also relevant to the nature and extent characterization of organochlorine pesticides in sediment.

4.1.3.3 Analytical Methods

The sediment surveys from which the RI baseline surface sediment chemistry dataset was compiled used similar or identical analytical methods for most analytes, with one notable exception. PCB analyses for the NOAA site characterization were conducted using high-performance liquid chromatography/photodiode array detection (HPLC/PDA). This method was not used for PCB Aroclor analyses in any of the other events; other analyses were conducted using GC/ECD. NOAA laboratory data for total PCBs are based on a nonstandard analytical method and may not be quantitatively comparable to data generated using standard analytical techniques. Specifically, the NOAA laboratory data for total PCBs reflect the difference between the sum of PCBs and polychlorinated terphenyls (PCTs) and the results of a separate analysis for PCTs alone.

Krahn et al. (1998) reported the results for 30 samples that were analyzed using both HPLC/PDA and GC/ECD methods by two different laboratories.[25] The two laboratories calculated total PCBs for each sample, which were then compared to each other. Total PCB concentrations varied between the two laboratories by as much as a factor of 6 (Krahn et al. 1998). Regression analyses conducted for the two sets of results indicated that the GC/ECD results were lower than the HPLC/PDA results at high PCB concentrations and higher than the HPLC/PDA results at low PCB concentrations (Krahn et al. 1998). The regression coefficient (R^2) between the two sets

[25] HPLC/PDA analyses were conducted by the NOAA laboratory in Seattle; GC/ECD analyses were conducted by Analytical Resources, Inc. in Seattle.

of analyses was 0.92. The differences between the total PCB concentrations calculated by the two laboratories are not surprising given the differences between the two methods, including: 1) different ranges of linear response for the two detectors, 2) differences in methods for calculating total PCBs, 3) differences in methods for quantifying and/or removing analytical interferences, and 4) differences in RLs.

Despite the differences between the two analytical methods for PCBs, data from both methods were used in the RI and risk assessments, although the uncertainty associated with total PCB concentrations may be significant in some areas. Alternative risk calculations are presented in the HHRA (Appendix B, Section B.6) using total PCB data derived solely from summing PCB Aroclor concentrations analyzed using GC/ECD.

Although different laboratories, and in some cases different methods, were used for the various tissue analyses, all tissue data summarized in Table 4-6 met the DQOs established for the project (Section 4.1). The uncertainties associated with combining data from different events that used different analytical methods are described in more detail in the HHRA (Appendix B, Section B.6).

4.1.4 Data reduction

Data reduction refers to methods used to aggregate raw data received from the laboratory for use in the RI/FS. A detailed discussion of data reduction methods is presented in Appendix E, Section E.3, and briefly summarized as follows:

- Chemical concentrations obtained from the analysis of laboratory duplicates or replicates (i.e., two or more analyses of the same sample) were averaged.
- Chemical concentrations of discrete samples collected at a single sampling location that were submitted to the laboratory as individual samples and analyzed separately were averaged and evaluated as a single sample.
- In some instances, the laboratory generated more than one result for a chemical for a given sample if re-analysis was required or if two different analytical methods were used for that chemical. The procedures for selecting the best result are described in Appendix E, Section E.3.
- The precision of each result was stored in the project database by recording the number of significant figures assigned by the laboratory. These significant figures were treated according to methods described in Appendix E, Section E.3.

For several chemicals (PCBs, DDTs, PAHs, and chlordane), total concentrations in individual samples were calculated by summing concentrations of individual components (i.e., seven Aroclor mixtures or individual congeners for total PCBs, six DDT isomers for total DDTs, specific individual PAH compounds for total PAHs, and specific individual chlordane compounds for total chlordane). The treatment of non-detects for these sums were as follows:

- If some of the individual components were detected in a sample and some were not, only the detected concentrations were included in the sum.
- If none of the individual components were detected in a sample, the total concentration was given a value equal to the highest RL of an individual component and assigned a U-qualifier, indicating the lack of detected concentrations.

If none of the components of a mixture were detected (e.g., all Aroclors, DDT isomers, individual PAH compounds, or individual chlordane compounds were non detect), one half the highest RL of an individual component was used for that sample in calculating the mean concentration for a group of samples.

Toxic equivalents (TEQs) of dioxins and furans and PCBs were calculated by summing the products of concentrations and compound-specific toxic equivalency factors (TEFs) for individual congeners of polychlorinated dibenzo-*p*-dioxins and furans or PCBs, as discussed in more detail in Appendix E, Section E.3. Total cPAHs were calculated by summing the products of concentrations and compound-specific potency equivalency factors (PEFs) for individual cPAH compounds, also discussed in more detail in Appendix E, Section E.3. Congeners or individual cPAH compounds that were undetected for a given sample were assigned a value equal to one-half the sample-specific RL for use in the calculations. Only the congeners or compounds that were analyzed were included in the TEQ or cPAH calculations.

4.2 CHEMICAL-SPECIFIC NATURE AND EXTENT INFORMATION

This section presents the nature and extent of contamination in surface sediment, subsurface sediment, tissue, surface water, seeps, and porewater, including maps that show distributions of selected chemicals in the various media. To focus the presentation of data, in sediment, a chemical was included if it was identified as a chemical of concern (COC) in the risk assessments based on a sediment exposure pathway. In tissue, a chemical was included if it was identified as a COC in either the HHRA based on seafood consumption or in the ERA based on a tissue exposure pathway.[26] In addition, data summary tables for tissue included any detected SVOCs and all organochlorine pesticides. In general, data summary tables for surface water, seeps, and porewater included any chemicals with detected concentrations. Data for all chemicals in each media are included in Appendix E.

Detailed discussions in this section are organized as follows: Sections 4.2.1 through 4.2.4 present chemical-specific discussions of four risk driver chemicals (PCBs, arsenic, cPAHs, and dioxins and furans) and Sections 4.2.5 through 4.2.8 present discussions

[26] A chemical was identified as a COC in the ERA if the LOAEL-based HQ was greater than or equal to 1.0 or the SMS was exceeded. A chemical was identified as a COC in the HHRA if the cancer risk estimate was 1×10^{-6} or if the HQ was greater than 1.

for metals and tributyltin (TBT), SVOCs, organochlorine pesticides, and VOCs, respectively.

Summaries presented in this section are based on data of acceptable quality collected from 1990 to the present, as described in Section 4.1. Summary statistics for all chemicals analyzed in each media are presented in Appendix E, Section E.4. Chemical concentrations in surface sediment relative to Washington State Sediment Management Standards (SMS) criteria are discussed in Section 10.2 and in the ERA (as summarized in Section 5.2 and presented in full in Appendix A).

4.2.1 PCBs (Aroclors and congeners)

This section discusses the nature and extent of PCBs in surface sediment, subsurface sediment, tissue, surface water, and seep water. PCBs were not analyzed in porewater samples collected from the LDW.

4.2.1.1 Surface sediment

This section presents a general discussion of total PCB concentrations in surface sediment throughout the LDW based on data in the RI baseline dataset. In addition, surface sediment data are used to elucidate temporal patterns in total PCB concentrations, including: 1) 57 locations with sample pairs representing locations that were resampled at different points in time, 2) data from Slip 4, where multiple sediment investigations have been conducted in a fairly confined area, and 3) 7 locations surrounding the Duwamish/Diagonal EAA which have been monitored on an annual basis since 2003.

This section also discusses spatially weighted average concentrations (SWACs) of total PCBs (as Aroclors) in surface sediment, which were calculated using inverse distance weighting (IDW). IDW methods have evolved throughout the project. As documented in Windward (2006), the IDW interpolation used for the ERA and FWM was based on optimization over broad areas of the LDW, considering a neighborhood of sample concentrations when determining each individual grid cell value. Thiessen polygons, rather than IDW, were used in the HHRA.

The interpolation methods described by Windward (2006) required that the LDW be divided into three separate sections: north, central, and south. Because IDW will be used in the FS to a much greater extent than in the RI, the 2006 interpolation methods were re-evaluated for the FS with two additional objectives: 1) create a streamlined computational method that does not require division of the LDW into separate sections, and 2) optimize the interpolation parameters to enhance prediction accuracy (relative to the 2006 methods) at areas with highest PCB concentrations.[27] The resulting parameterization, which is used in this RI (except for the ERA and FWM,

[27] Prediction accuracy of an IDW interpolation, as determined by a comparison of known to interpolated concentrations, is generally lowest in areas with highest spatial variability. Typically, these areas had the highest PCB concentrations and very steep concentration gradients.

which use the 2006 interpolation) and will be used in the FS, can be implemented across the entire LDW in one step. LDWG is currently preparing a memorandum describing the details of the parameterization.

Percentiles for surface sediment concentrations in the RI baseline dataset were calculated using two separate methods. For the first method, percentiles were calculated as the concentration at or below which the corresponding percent of concentrations in the dataset fall (e.g., the 95th percentile is the concentration at or below which 95% of the concentrations fall). These percentiles are referred to as numerical percentiles. Area-based percentiles were calculated based on IDW interpolations, and thus account for the spatial bias in the sampling designs, which were in many cases focused on areas with higher concentrations. The area-based percentiles were calculated as the surface sediment concentration at which a certain percentage of the LDW area has interpolated values less than or equal to that concentration (e.g., the 95th percentile is the concentration at which 95% of the LDW area has interpolated values at or below that concentration).

LDW-Wide Data

Surface sediment samples collected from 1,030 of the 1,365 locations in the RI baseline dataset were analyzed for total PCBs as Aroclors, and an additional 297 samples were analyzed for total PCBs using NOAA methods.[28] Total PCBs were detected at 94% of the locations where PCB Aroclors were analyzed, with concentrations ranging from 1.6 to 220,000 µg/kg dw (Table 4-11). The highest total PCB concentrations were generally detected in EAAs (Maps 4-13 and 4-14), which are discussed in Section 10. As presented in Table 4-11, the Aroclors detected most frequently in surface sediment were Aroclors 1254 and 1260, at 80% and 81% of the locations, respectively.

Using the updated IDW method discussed above, the SWAC of total PCBs for the LDW from RM 0.0 to 6.0 was 342 µg/kg dw. Five percent of the LDW area from RM 0.0 to RM 6.0 had a spatially weighted total PCB concentration higher than 808 µg/kg dw. The 25th, 50th, and 75th percentile concentrations based on the IDW interpolation were 48, 115, and 212 µg/kg dw, respectively.[29] These area-based percentiles are lower than the numerical-based 25th, 50th, 75th, and 95th percentiles of 58, 140, 390, and 4,300 µg/kg dw, respectively. The area-based percentiles are discussed in this section because the IDW interpolation accounts for spatial bias in the sampling events, which were frequently focused on contaminated areas.

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[28] These additional 297 samples were analyzed for selected PCB congeners, PCTs, and PCBs+PCTs using NOAA's HPLC/PDA and GC/ECD (equivalent to EPA 8082) methods. Total PCBs were calculated as the difference between total PCTs (which were determined using GC/ECD) and PCBs+PCTs, which were determined using HPLC/PDA. This method provides total PCB values but does not quantify the individual Aroclors.

[29] The 25th, 50th, 75th, and 95th percentiles calculated using the intial (2006) IDW interpolation discussed in the ERA and FWM were 51, 130, 220, and 1,117 µg/kg dw, respectively.

Table 4-11. Summary of LDW-wide surface sediment data for detected Aroclors, total PCBs (as Aroclors), and mammalian PCB TEQ

Chemical	Detection Frequency		Unit	Concentration			
	Ratio	%		Minimum Detect	Maximum Detect	Calculated Mean[a]	RL or Range of RLs[b]
Aroclor 1242	108/1,021	11	µg/kg dw	7.8 J	2,700	49	0.87 – 6,100
Aroclor 1248	228/1,030	22	µg/kg dw	6.3	220,000	360	0.87 – 4,300
Aroclor 1254	817/1,022	80	µg/kg dw	2.2	110,000	500	1.3 – 4,300
Aroclor 1254/1260[c]	8/8	100	µg/kg dw	37	800	180	nc
Aroclor 1260	823/1,022	81	µg/kg dw	1.2 J	38,000	600	3.9 – 15,000
Aroclor 1262	2/12	17	µg/kg dw	270	840	97	3.3 – 20
Aroclor 1268	1/11	9	µg/kg dw	460 J	460 J	47	3.3 – 20
Total PCBs (dw)[d]	1,243/1,327	94	µg/kg dw	1.6 J	220,000	1,200	nc
Total PCBs (OC-normalized)[e]	1,145/1203	95	mg/kg OC	0.11 J	10,000	60	nc
PCB TEQ[f]	48/48	100	ng/kg dw	0.0908 J	1,380	45.9	nc

[a] Calculated mean concentration is the average of detected concentrations and one-half the RL for non-detected results.

[b] RLs from non-detect samples only.

[c] Aroclors 1254 and 1260 could not be distinguished from each other in these eight samples, so they were reported together.

[d] Total PCBs (dw) represent the sum of the detected concentrations of the individual Aroclors. If no Aroclors were detected, the RL reported represents the highest RL for an individual Aroclor.

[e] Summary statistics for OC-normalized PCB concentrations were calculated using only samples with TOC contents ≥ 0.5% and ≤ 4.0%.

[f] PCB TEQS were calculated using TEFs for mammals presented in Van den Berg et al. (2006).

dw – dry weight
J – estimated concentration
LDW – Lower Duwamish Waterway
nc – not calculated
OC – organic carbon
PCB – polychlorinated biphenyl
RL – reporting limit
TEQ –toxic equivalent
TOC – total organic carbon

The maximum total PCB concentration was detected in 1995 at location NFK305, on the east side of the LDW between RM 4.8 and RM 4.9 (Map 4-13). Other areas with total PCB concentrations greater than the area-based 95th percentile concentration (808 µg/kg dw) were as follows: near RM 0.5 and RM 0.6 on the east side of the LDW (in the Duwamish/Diagonal EAA), near RM 1.0 in the navigation channel, between RM 2.2 and RM 2.3 on the west side (in EAA 2), in the Slip 4 EAA, between RM 3.2 and RM 3.7 on the east side of the LDW (in the Boeing Plant 2/Jorgensen Forge EAA), between RM 3.5 and RM 3.7 on the west side (in the T-117 EAA), and near RM 4.9 (in the Norfolk EAA) (Map 4-14). Patterns of total PCB concentrations in surface sediment are discussed in greater detail in Section 4.2.1.3 relative to subsurface data.

PCB congeners were analyzed in a subset of the surface sediment samples collected in 2005 for the RI and in samples collected during two sampling events in the late 1990s (the NOAA site characterization in 1997 and the EPA SI in 1998), as shown on Map 4-15. The PCB congener data collected in the late 1990s met DQOs, but RLs were relatively high. As part of the RI, surface sediment samples from 48 locations were analyzed for 26 PCB congeners.[30] The samples collected for PCB congener analyses were selected to provide adequate spatial coverage of the LDW and to ensure that a range of locations with low, moderate, and high total PCB concentrations was represented. The PCB congeners analyzed included the 12 congeners identified by the World Health Organization (WHO) as having dioxin-like properties (i.e., PCB congeners 77, 81, 105, 114, 118, 123, 126, 156, 157, 167, 169, and 189). In addition, six principal PCB congeners (i.e., PCB congeners 66, 101, 110, 138, 153, and 180) were selected for analysis because they were detected in surface sediment at high concentrations relative to the other PCB congeners in the two historical datasets for the LDW. An additional eight PCB congeners that coelute with the PCB congeners listed above were also included in the analysis. In general, the PCB congeners with the highest concentrations at the locations sampled as part of the RI were PCB congeners 90, 110, 118, 129, and 153. Appendix E, Section E.4, presents summary statistics for all PCB congeners in surface sediment.

PCB TEQs were calculated using data for the 12 dioxin-like PCB congeners. Historical PCB congener data were not included in these TEQ calculations because of elevated RLs. Methods for mammalian PCB TEQ calculations are described in Appendix A, Section A.5.1.1. PCB TEQs ranged from 0.0908 to 1,380 ng/kg dw (Table 4-11). Based on a regression analysis, concentrations of total PCBs and PCB TEQs in sediment were significantly correlated (R^2=0.99) for the 48 locations analyzed for PCB congeners in 2005.

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[30] Discrete surface sediment grab samples were collected during Rounds 1 and 2 of the surface sediment sampling event at 32 locations, and composite sediment samples were co-located with benthic invertebrate tissue samples or clam tissue samples at eight locations each.

LDW Resampled Locations

This section presents an evaluation of the change in total PCB concentrations in surface sediment between older samples and newer samples collected within 10 ft of each other.[31] Changes in concentrations could result from the mixing of surface sediment, burial of older sediment by newer sediment with different concentrations, transport of older sediment away from an area, or from resampling slightly different locations in areas with spatial heterogeneity in surface sediment chemical concentrations. Data from the Duwamish/Diagonal area and the Norfolk area were not included in this evaluation; monitoring data for these areas are discussed in detail later in this section for the Duwamish/Diagonal perimeter area, in Section 10.3.1 (Duwamish/Diagonal cap), and in Section 10.3.15 (Norfolk).

Of the 57 locations, total PCB concentrations decreased in surface sediments over time at 30 locations (53%) and increased at 18 locations (32%) (Table 4-12 and Map 4-16). At 26 of the 48 locations, the concentration increase or decrease was ≤ 250 µg/kg dw, as shown on Map 4-16. At the remaining 9 of the 57 locations (16%), concentration differences were within the range of analytical variability (≤ 25% increase or decrease compared to the initial concentration). Three general areas had more than one location with decreases in total PCB concentrations of more than 250 µg/kg dw):

- **RM 0.2 to RM 0.4 –** At two locations in this area, concentrations decreased by 339 and 940 µg/kg dw; at two other resampled locations nearby, concentrations were relatively low and decreased by about 60 µg /kg dw at each location.
- **RM 2.0 to RM 2.3 –** Concentrations at four resampled locations in this area decreased by 680 to 3,470 µg/kg; the only other resampled location in this area (in Slip 3) had a decrease in concentration of 113 µg/kg dw.
- **RM 3.9 to RM 4.0 –** Concentrations at all the resampled locations in this area decreased by 370 to 1,351 µg/kg dw.

The remaining six locations with total PCB concentrations that decreased more than 250 µg/kg dw were at various single locations throughout the LDW.

Table 4-12. Difference in total PCB concentrations at resampled locations sorted by concentration difference

Approximate River Mile	Sampling Location ID		Sampling Date		Concentration (µg/kg dw)		
	Initial	Resampled	Initial	Resampled	Initial	Resampled	Difference
4.7	DR271	LDW-SS148	9/15/1998	3/9/2005	9,400	520	-8,880
3.6	SD-DUW90	SD-343-S	4/4/1996	8/27/2004	7,500	260 J	-7,240

[31] The older data at these resampled locations were not included in the baseline dataset, as described in Section 4.1.2.1. The age difference between the older and newer samples ranged from 1 to 13 years, with an average difference of approximately 8 years.

Approximate River Mile	Sampling Location ID		Sampling Date		Concentration (µg/kg dw)		
	Initial	Resampled	Initial	Resampled	Initial	Resampled	Difference
3.5	WST323	T117-SE-10-G	10/21/1997	12/8/2003	7,900 J	1,200	-6,700
0.8	EST219	LDW-SS27	9/17/1997	1/18/2005	4,400 J	97 J	-4,303
1.4	DR030	LDW-SS50	8/17/1998	1/24/2005	4,800	590	-4,210
2.2	WIT280	B5a-2	10/3/1997	9/24/2004	5,200 J	1,730	-3,470
2.2	DR113	LDW-SS81	8/19/1998	3/8/2005	2,100 J	210	-1,890
3.9	EST144	LDW-SS123	9/25/1997	1/24/2005	1,500 J	149	-1,351
3.9	EIT061	LDW-SS121	9/29/1997	1/25/2005	2,400 J	1,060 J	-1,340
2.1	CH0023	LDW-SS79	10/16/1997	1/24/2005	1,200 J	68	-1,132
0.3	DUD042	LDW-SS17	11/11/1995	1/24/2005	1,060 J	120	-940
2.0	R7	LDW-SS75	10/15/1997	1/21/2005	1,200	520	-680
1.2	DR088	LDW-SS40	8/31/1998	1/18/2005	1,010 J	510 J	-500
3.9	R30	LDW-SS119	10/11/1997	1/19/2005	1,250 J	880 J	-370
0.2	DR035	LDW-SS12	8/11/1998	1/17/2005	510 J	171 J	-339
3.7	R21	LDW-SS113b	10/9/1997	1/20/2005	200	18 J	-182
4.2	R42	LDW-SS129	10/13/1997	1/20/2005	193	19 U	-174
1.5	DR123	LDW-SS57	9/14/1998	1/24/2005	900	750	-150
1.3	DR053	LDW-SS44	8/31/1998	1/21/2005	250 J	103 J	-147
2.1	DR106	LDW-SS76	8/19/1998	1/20/2005	230	117	-113
4.1	A11-05	LDW-SS126	8/18/1994	1/20/2005	109	20 U	-89
4.2	R45	LDW-SS130	10/16/1997	1/20/2005	101	26	-75
1.0	DR020	LDW-SS31	8/17/1998	1/21/2005	169 J	96	-73
4.2	R40	LDW-SS127	10/13/1997	1/20/2005	119	58	-61
0.2	K-05	LDW-SS10	9/27/1991	1/17/2005	91 J	31	-60
0.3	DR079	LDW-SS15	8/24/1998	1/17/2005	187 J	128 J	-59
2.8	DR175	LDW-SS94	8/20/1998	1/21/2005	120	72	-48
4.3	DR286	B10b	8/26/1998	8/19/2004	54	9.8 J	-44.2
1.0	DR019	LDW-SS32	8/17/1998	1/18/2005	162	122 J	**-40**
0	K-11	LDW-SS1	9/30/1991	1/17/2005	200 J	161 J	**-39**
1.7	DR097	LDW-SS63	8/20/1998	1/21/2005	126 J	95	**-31**
3.2	DR202	LDW-SS104	8/27/1998	1/25/2005	98	75	**-23**
4.1	DR238	LDW-SS125	8/27/1998	1/20/2005	40 U	19 U	-21
0	DR076	LDW-SS5	8/24/1998	1/17/2005	40 U	20 UJ	-20
3.8	DR187	LDW-SS115	8/27/1998	1/25/2005	240	220	**-20**
3.1	DR198	LDW-SS102	8/20/1998	1/24/2005	85	74	**-11**
1.9	DR131	LDW-SS70	8/13/1998	1/21/2005	97 J	96	**-1**
3.8	R24	LDW-SS117	10/10/1997	1/20/2005	73	79 J	**6**
1.4	DR065	LDW-SS52	8/17/1998	1/25/2005	190	209	**19**
4.2	EST135	B8b	11/12/1997	8/19/2004	10 J	37 J	27

Approximate River Mile	Sampling Location ID		Sampling Date		Concentration (µg/kg dw)		
	Initial	Resampled	Initial	Resampled	Initial	Resampled	Difference
2.4	WST342	DR141	10/23/1997	8/20/1998	38 J	68 J	30
1.4	DR160	LDW-SS51	8/12/1998	1/18/2005	115	155 J	40
1.0	WST367	DR048	9/19/1997	8/12/1998	29 J	88	59
0.1	K-07	LDW-SS4	9/30/1991	1/17/2005	87 J	153 J	66
4.2	DR242	SB-1	8/24/1998	8/25/2004	93 J	170	77
3.6	EST154	SD-334-S	9/24/1997	8/26/2004	150	290	140
3.6	SD-SWY07	SD-SWY17	6/13/1995	9/9/2003	320	460	140
1.4	DR028	B4b	8/17/1998	8/28/2004	210	400	190
0.9	DR021	LDW-SS319	8/17/1998	10/4/2006	142 J	350	208
2.6	EIT074	LDW-SS88	11/3/1997	1/25/2005	450	660	210
3.6	EST152	SD-309-S	9/24/1997	8/16/2004	290	570	280
0.9	DR085	LDW-SSB2b	8/31/1998	3/11/2005	410 J	790	380
2.8	EST180	LDW-SS92	10/6/1997	1/25/2005	230	970	740
3.7	DR186	LDW-SS111	8/27/1998	1/19/2005	1,180	3,200 J	2,020
3.7	SD-323-S	LDW-SS110	8/17/2004	1/25/2005	9,400 J	13,000	3,600
1.0	DR087	LDW-SS37	8/12/1998	1/18/2005	690	5,100	4,410
3.7	SD-DUW92	SD-320-S	4/2/1996	8/16/2004	1,500	8,900	7,400

Note: Concentrations in bold were within the range of analytical variability (≤25% increase or decrease compared to the initial concentration).

J estimated concentration

RL reporting limit

U not detected at RL shown

UJ not detected at estimated RL shown

Total PCB concentrations increased by more than 250 µg/kg dw at seven locations, most notably on the east side of the river near RM 3.7 (within the Boeing Plant 2/ Jorgensen Forge EAA), where there were three resampled locations with increases ranging from 2,020 to 7,400 µg/kg dw. There was also an area between RM 0.8 and RM 1.1 in the navigation channel with two resampled locations where PCB concentrations increased by 380 and 4,410 µg/kg dw.

The changes in total PCB concentrations over time at the resampled locations could reflect actual differences within an area from changes in sources or as a result of localized disturbances, or could reflect local spatial variability (i.e., hotspots). Therefore, although there is some uncertainty in the interpretation of these data, the more frequent observation of lower concentrations over time indicates that there may be an overall decreasing trend in total PCBs concentrations in the LDW.

Slip 4

Slip 4, located near RM 2.8 on the east side of the LDW (Map 4-17), is one of seven areas within the LDW that have been identified as an EAA[32] (Windward 2003d). Surface sediment data for PCBs have been collected during six sampling events conducted in Slip 4 between 1990 and 2004. Slip 4 is the only area in the LDW sampled during more than two time periods at a relatively high sampling density, providing an opportunity to evaluate the data for temporal trends on an area-specific basis. The western portion of Slip 4 was most recently dredged in 1996 to allow barge and tug access to the Crowley Marine Services pier; this dredged area is shown on Map 4-17.

The six surface sediment investigations conducted in Slip 4 included:

- Environmental site assessment in 1990 (Landau 1990) (9 surface sediment samples)
- NOAA sediment characterization study in 1997 (NOAA 1998) (18 surface sediment samples)
- Phase 1 site characterization study in 1997 (Exponent 1998) (7 surface sediment samples)
- EPA site inspection in 1998 (Weston 1999a) (7 surface sediment samples)
- EAA investigation in 2004 (Integral 2004) (30 surface sediment samples)
- RI benthic invertebrate sampling event in 2004 (Windward 2005a) (3 surface sediment samples collected with co-located clam samples)

To evaluate changes in total PCB concentrations over time, the data were separated into three groups according to the collection dates of 1990, 1997-1998, and 2004. The locations sampled during each of these periods, in addition to the boundary of the currently approved Slip 4 EAA, are shown on Map 4-17. Fewer samples were collected in 1990 relative to the number collected in 1997-1998 and 2004. No locations were resampled at the same coordinates (i.e., within 10 ft of each other). Map 4-17 shows the total PCB concentrations detected at each location during each time period.

Total PCB concentrations in surface sediment were highest at the head of the slip (the eastern 200 ft) in all years but generally decreased with time. In this area, the concentrations ranged from 1,000 to 34,000 µg/kg dw in 1990, from 3,300 to 25,000 µg/kg dw in 1997-1998, and from 490 to 5,100 µg/kg dw in 2004.

From the head of the slip westward to the EAA boundary, total PCB concentrations also generally decreased over time. Total PCB concentrations were high in the only two samples collected in this area in 1990 (5,800 and 11,800 µg/kg dw), were generally between 632 and 7,000 µg/kg dw in 1997-1998 (although one location had a

[32] EAAs are discussed in greater detail in Section 10.

concentration of 16,000 µg/kg dw), and were generally ≤ 710 µg/kg dw in 2004, except for two samples with concentrations of 1,000 and 1,700 µg/kg (Map 4-17).

West of the EAA boundary, differences were not apparent between 1990 and 1997–1998; maximum total PCB concentrations were 730 and 1,670 µg/kg dw for each time period, respectively. Total PCB concentrations were generally lower in 2004, with a maximum concentration of 210 µg/kg dw. In total, these data indicate an overall decrease in total PCB concentrations in Slip 4 surface sediment over the 5-year period between 1998 and 2004. Differences between 1990 and 1997-1998 data were more difficult to discern because of the low sampling density in 1990.

Duwamish/Diagonal Perimeter

Surface sediment sampling has been conducted at the Duwamish/Diagonal EAA on an annual basis since 2003 to monitor sediment quality following remedial actions. This section discusses total PCB data collected at seven locations along the perimeter of the remediated area that have been monitored during five sampling events (Map 4-18). Monitoring data for the cap are discussed in Section 10.3.1.

Remediation at the Duwamish/Diagonal EAA consisted of dredging sediment from the area between RM 0.4 and RM 0.6 from November 2003 to January 2004 (King County et al. 2005). The dredged area was then capped between January and March of 2004 by placing clean cap material over a 5-ac rectangle (Area A) and a 2-ac rectangle (Area B), as shown on Map 4-18. A 9-in. thin-layer placement (enhanced natural recovery) was later placed over an area to the south and west of Area B in February 2005. Monitoring of surface sediment chemistry at the seven perimeter locations began in October 2003 before remediation began and was then conducted on an annual basis beginning in March 2004.

In general, total PCB concentrations in surface sediment at the seven perimeter monitoring locations showed a decreasing trend from 2004 to 2007 (Map 4-18); the 2003 samples are not included in this analysis as they represent conditions just before and just after dredging. At locations 1C, 2C, 8C, and 11C, it appears that lower concentrations in 2004 may have been affected by the higher sand content deposited over some of the perimeter areas during capping operations, as shown on Map 4-18. Total PCB concentrations were lower in 2007 than in 2004 at all locations except one, which had relatively low concentrations. The sand content also remained relatively stable or decreased from 2004 to 2007 at all locations, suggesting that burial by cleaner sediment is contributing to lower sediment concentrations.

4.2.1.2 Subsurface sediment

Subsurface sediment cores were sampled at various intervals during different sampling events. To calculate summary statistics, two categories were created according to specified intervals. The following intervals were selected: 2-ft intervals of 0 to 2 ft, 2 to 4 ft, 4 to 6 ft, 6 to 8 ft, and 8 to 10 ft; and 1-ft intervals (e.g., 0-to-1 ft, 1-to-2 ft) to a depth of 10 ft below mudline. Data were assigned to these intervals by

first rounding (to the nearest foot) the top and bottom depths of each sample (in feet below mudline) and then assigning the data to the appropriate sampling interval. If data were available from two 1-ft intervals at a particular location, the concentrations were also averaged to obtain a calculated concentration for the corresponding 2-ft interval category; these intervals are presented in addition to the 1-ft intervals. For example, if a location had data in both the 0-to-1-ft and 1-to-2-ft intervals, the concentrations were averaged for a calculated concentration in the 0-to-2-ft interval. Total PCB data for the 1- and 2-ft subsurface intervals are shown on Maps 4-19a through 4-19c.

Total PCBs were detected in 74% of the subsurface sediment samples, at concentrations ranging from 0.52 to 890,000 µg/kg dw (Table 4-13). Most of the subsurface samples analyzed for total PCBs were from the top 4 ft of the sediment cores. Data shown on Maps 4-19a through 4-19c and presented in Table 4-13 are discussed in more detail in Section 4.2.1.3.

Table 4-13. Summary of LDW-wide subsurface sediment data for total PCBs (as Aroclors)

Sample Interval (ft)	Detection Frequency		Total PCB Concentration (µg/kg dw)	
	Ratio	%	Minimum Detect	Maximum Detect
All data[a]				
Any interval	609/821	74	0.52 J	890,000
1-ft intervals[b]				
0 to 1	114/118	97	13.5	108,000
1 to 2	83/101	82	19.6	6,500
2 to 3	75/104	72	4 J	32,000
3 to 4	36/49	73	0.52 J	9,100
4 to 5	26/57	46	23	6,400
5 to 6	13/18	72	22	4,700
6 to 7	6/21	29	21	2,400
7 to 8	2/5	40	540	910
8 to 9	3/10	30	25	690
2-ft intervals[b]				
0 to 2	130/137	95	12.9	890,000
2 to 4	106/135	79	3 J	19,000 J
4 to 6	47/60	78	3.9 J	15,000
6 to 8	20/30	67	4.5 J	3,800
8 to 10	12/24	50	14 J	540

[a] Data included in the subsurface dataset are described in Section 4.1.2.2.

[b] For the calculation of summary statistics in this table, actual sampling intervals were rounded to the nearest foot. The data for each interval were then assigned to either the 1- or 2-ft sampling category, whichever best approximated the actual sampling interval. If there were data for two 1-ft intervals at a particular location, the

concentrations were averaged to obtain a calculated concentration for the corresponding 2-ft interval. Some of the subsurface data were collected from intervals other than 1 or 2 ft; therefore, the detection frequency ratio for "any interval" may reflect data for intervals not included in these categories.

dw – dry weight

PCB – polychlorinated biphenyl

J – estimated concentration

Of the subsurface sediment samples collected and analyzed as part of the RI, 27 of the 47 samples analyzed for total PCBs were collected entirely within the lower alluvium unit. The lower alluvium unit is predominantly dense sand that was deposited prior to the industrialization of the area. The depth from the river bottom to the surface of this unit ranged widely from 0.2 to 12 ft, with an average depth of 6.2 ft, and is discussed in greater detail in Section 2.5.4.

Total PCB concentrations in subsurface sediment samples collected from entirely within the lower alluvium unit were ≤ 53 µg/kg dw and less than the SQS in all samples.[33]

4.2.1.3 Surface and subsurface sediment patterns

This section presents the spatial and vertical patterns of total PCB concentrations in surface and subsurface sediment by river mile. These patterns were assessed to identify areas with relatively high concentrations in surface and subsurface sediment, to evaluate the spatial relationships between the two, and to identify areas where maximum total PCB concentrations were detected in uppermost intervals or in deeper subsurface intervals. The integration of this information with the physical CSM (as presented in Section 3) is discussed in Section 4.3.

The LDW was divided into five sections for discussion purposes (RM 0.0 to RM 1.0, RM 1.0 to RM 2.2, RM 2.2 to RM 3.0, RM 3.0 to RM 4.0, RM 4.0 to RM 5.0). Boundaries for some of these sections correspond to the river reaches defined by the STM (i.e., Reach 1 from RM 0.0 to RM 2.2, Reach 2 from RM 2.2 to RM 4.0, and Reach 3 from RM 4.0 to RM 4.8). Maps in this section present data separately for locations within the navigation channel, west of the channel, and east of the channel to facilitate comparisons of river bench vs. navigation channel areas and for consistency with results of sediment transport modeling (Maps 4-20a through 4-20f). According to the STM, bench and channel areas have some distinct differences in sedimentation rates and potential scour depths in some sections of the LDW.

It is important to note that the certainty and scale of the analyses presented in this section (and in Sections 4.2.2 and 4.2.3 for arsenic and cPAHs, respectively), are dictated by the availability of sediment data. For example, in this section, patterns of

[33] Subsurface data are compared to SMS criteria in Appendix J. One sample collected from the 6-to-8.8-ft interval at LDW-SC10 had a total PCB concentration exceeding the SQS of 12 mg/kg OC (35 mg/kg OC; 350 µg/kg dw). This sample interval was identified as lower alluvium in the draft core logs presented in the subsurface sampling data report (Windward and RETEC 2007), but this sample contained silt and sand from the upper alluvium unit.

total PCB concentrations in surface sediment are based on results of IDW interpolation. The uncertainty associated with interpolated concentrations is lower in areas with higher sampling density. The density of subsurface cores is also highly variable and extrapolations for areas with less cores presents some uncertainty. In addition, the interpretation of subsurface data is dictated by the size of the sampling interval within each core, which is variable across the site. There is greater uncertainty in distinguishing vertical patterns in locations with larger sampling intervals. There is also some uncertainty in comparing interpolated surface sediment data to subsurface core data collected at different times because conditions at the surface may have changed.

RM 0.0 to RM 1.0

Total PCB concentrations in surface and subsurface sediment samples collected between RM 0.0 and RM 1.0 were highly variable (Map 4-20a). Within this section of the LDW, three areas had total PCB concentrations in surface sediment greater than the area-based 95th percentile (808 µg/kg dw): 1) RM 0.1 on the eastern shoreline, 2) the vicinity of the Duwamish/Diagonal EAA (between RM 0.4 and RM 0.6), and 3) RM 0.6 on the western shoreline.

At RM 0.1 to RM 0.2 on the eastern shoreline, the peak total PCB concentrations in subsurface cores were detected in the 2-to-4-ft interval (2,900 µg/kg dw at LDW-SC2 and 600 µg/kg dw at LDW-SC4). In a finer-resolution core collected just south of Harbor Island near an SD (LDW-SC1 sampled at 0.5-ft intervals), there was a peak in total PCB concentrations (6,700 µg/kg dw) in the 1-to-2-ft interval, with a lower concentration (85 µg/kg dw) in the uppermost interval (0 to 0.5 ft). Just east of the navigation channel at RM 0.15 (LDW-SC3), interpolated total PCB concentrations were relatively low in the surface sediment, and PCBs were not detected in subsurface sediment samples.

On the west side, a core (LDW-SC5) was collected near a small area at RM 0.15 at which the PCB concentrations in surface sediment estimated by interpolation were relatively high (between the 75th and 95th percentiles). The total PCB concentration in the uppermost 1-ft interval of this core was also within this range (510 µg/kg dw); concentrations decreased with depth. Farther south at RM 0.3 peak concentrations were highest in the 4-to-4.5-ft interval (2,600 µg/kg dw at LDW-SC6). A historical core collected 290 feet to the south (DR068) had the same concentration (2,600 µg/kg dw)in the only interval sampled (0 to 2 ft).

A relatively large number of cores from within and near the Duwamish/Diagonal EAA have been analyzed for total PCBs. Sediments within the EAA were either dredged and capped or covered with a thin layer of sand subsequent to the collection and analysis of these samples. Prior to dredging and capping, total PCB concentrations in this area were high in surface sediment (greater than the 75th percentile) and in subsurface sediment (generally having at least one interval greater than the 95th percentile). A cluster of cores near the center of the dredged area at RM 0.5 (DUD254,

DUD253, DUD020, DUD006, DR008, DUD256, and LDW-SC10) [34] had peak PCB concentrations below 2 ft. In three other cores from the dredged area (DUD252, DUD251, and DUD255) and cores from the thin layer cap area (DUD257, DUD260, DUD262, DUD027, and DUD261)[35] peak total PCB concentrations were in the uppermost intervals.

Relatively few subsurface sediment core samples have been collected from the Kellogg Island area. Interpolated total PCB concentrations in surface sediment in this area were generally low (≤ 115 µg/kg dw), with a few higher concentrations (between the 75th and 95th percentiles) on the west side of the island. Two cores collected at the same location north of the island at RM 0.6 had peak PCB concentrations (~2,000 µg/kg dw) at depth (in the 2-to-2.5-ft interval at LDW-SC12 and in the 2-to-4-ft interval at DR044).

On the western shoreline of the LDW to the north of Kellogg Island, a single core (LDW-SC11) was collected in an area with higher interpolated concentrations of total PCBs in the surface sediment. In this core, total PCB concentrations were high (3,000 µg/kg dw) in the uppermost interval (0 to 0.8 ft), but were not detected below this interval. To the south of the island in an area of low interpolated PCB concentrations in surface sediment, PCBs were highest in the 6-to-7-ft interval (2,400 µg/kg dw) of core LDW-SC19.

Interpolated total PCB concentrations were generally between the 50th and 95th percentile in surface sediments in Slip 1. At the head of the slip, total PCB concentrations in core LDW-SC17 were highest at depth (9,800 µg/kg dw in the 2-to-4-ft interval), although all intervals had high concentrations (greater than the 95th percentile). Near the mouth of the slip, peak total PCB concentrations (ranging from 1,950 to 5,400 µg/kg dw) were detected at depth (in the 2-to-4-ft and 4-to-6-ft intervals) in all three cores collected in this area (LDW-SC15, LDW-SC16, and DR021).

Two cores were collected just downstream of Slip 1 at RM 0.85. In an area of relatively high PCB concentrations in surface sediment in the navigation channel (between the 75th and 95th percentiles), the core from LDW-SC14 had high total PCB concentrations (1,550 to 4,500 µg/kg dw) in the uppermost intervals down to 4.1 ft, with the peak concentration in the 0-to-1.4-ft interval. Near the eastern shoreline of the LDW (LDW-SC13) the highest total PCB concentrations were in the uppermost intervals down to 2 ft, but concentrations were somewhat lower, ranging from 280 to 470 µg/kg dw.

In summary, the LDW between RM 0.0 and RM 1.0 is highly variable. Interpolated concentrations in surface sediment covered all percentile categories, although only a

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[34] Athough not apparent on Map 4-18a, total PCB concentrations were 890 and 1,670 µg/kg dw in the 0-to-3-ft and 3-to-6-ft intervals, respectively, at DUD256.

[35] Athough not apparent on Map 4-18a, total PCB concentrations were 7,700 and 1,260 µg/kg dw in the 0-to-3-ft and 3-to-6-ft intervals, respectively, at DUD255 and 13,200 and 21,000 µg/kg dw in the 0-to-3-ft and 3-to-6-ft intervals, respectively, at DUD261.

few areas had concentrations greater than the 95th percentile. Many of the areas sampled with subsurface cores had peak total PCB concentrations at depth (Map 4-21). Exceptions to this general pattern include a few locations along the western shoreline (LDW-SC5 and LDW-SC11), a location in the navigation channel near RM 0.9, and a number of locations in the Duwamish/Diagonal EAA (Map 4-21), where peak concentrations were detected in the uppermost interval of the cores. At these locations, concentrations were generally lower in interpolated surface sediment than the peak concentrations in subsurface sediment.

RM 1.0 to RM 2.2

Total PCB concentrations in surface and subsurface sediment samples collected between RM 1.0 and RM 2.2 were also variable (Map 4-20b). One area just south of RM 1.0 in the navigation channel had interpolated concentrations in surface sediment as high as 5,100 µg/kg dw. The core collected from this area had the highest concentration in the uppermost interval (3,200 µg/kg dw in LDW-SC20 in the 0-to-2-ft interval). In the core collected on the western shoreline at RM 1.0 (LDW-SC21), the peak total PCB concentration (1,680 µg/kg dw) was detected at depth (4 to 6.2 ft). Cores along the eastern shoreline at RM 1.0 (C2, C3, A1, LDW-SC18, and LDW-SC22) had relatively lower PCB concentrations, but samples from within the historical cores were collected over relatively large intervals (e.g., 4-ft intervals); thus, it was not possible to determine the depth or magnitude of peak concentrations.

Total PCB concentrations in surface sediment areas along the eastern shoreline between RM 1.2 and RM 1.4 were variable. Peak concentrations in cores in this area were detected at depth, with peak concentrations in the 4-to-6-ft, 2-to-4-ft, and 1-to-1.5-ft intervals in cores LDW-SC23, DR025, and LDW-SC27, respectively.[36]

Just downstream of Slip 2 along the eastern shoreline, numerous historical and two cores were collected for the RI and analyzed for PCBs. The historical cores were collected prior to dredging in this area and were generally composited over larger intervals (0 to 3 ft or 0 to 4 ft), so the depth of the peak total PCB concentration is unknown. In the cores collected as part of the RI, peak concentrations (12.9 µg/kg dw at LDW-SC30 and 370 µg/kg dw at LDW-SC31) were in the uppermost intervals.

One core (LDW-SC32) was located within Slip 2. Total PCB concentrations in the uppermost 4 ft of this core were higher than relatively low surface sediment concentrations in the same area (generally > 210 µg/kg dw). The peak total PCB concentration within this core (2,450 µg/kg dw) was detected in the 2-to-4-ft interval.

At the head of the slip near core LDW-SC37, interpolated total PCB concentrations were less than the 75th percentile in surface sediment; a subsurface peak of 950 µg/kg dw was detected in the 1-to-2-ft interval. Near the mouth of Slip 3, in cores DR106 and

[36] Specific total PCB concentrations in samples from LDW-SC27 (not apparent on Map 4-18b) were 110, 260, 83, and 39 µg/kg dw in the 0.5-to-1.0-ft, 1.0-to-1.5-ft, 1.5-to-2.0-ft, and 2.0-to-2.5-ft intervals, respectively.

LDW-SC36, subsurface concentrations of PCBs were lower than those in the interpolated surface sediment (less than the 50th percentile). In the core just upstream from Slip 3 (DR112), the peak concentration was detected in the 2-to-4-ft interval (330 µg/kg dw). Thus, peak concentrations were detected at depth at the head and just downstream of the slip, and were relatively low towards the mouth of the slip.

Multiple historical cores from the navigation channel between RM 1.7 and RM 2.0 were analyzed for total PCBs. These cores, collected over intervals from 0 to 4 ft, had variable total PCB concentrations, which were generally similar to the interpolated concentrations in surface sediment (ranging from below the 25th percentile to the 95th percentile). Vertical patterns were not evident based on these cores because of the large sampling intervals. Core LDW-SC33, just east of the navigation channel at RM 1.9, was analyzed at 0.5-ft intervals down to 3 ft and had a peak total PCB concentration of 4,700 µg/kg dw in the 1-to-1.5-ft interval.[37]

Six locations between RM 1.2 and RM 1.5 on the west side of the navigation channel had sufficient data to determine the depth of peak concentrations within these cores. Two of these cores had the highest concentrations in the uppermost intervals (33 and 280 µg/kg dw in LDW-SC29 and LDW-SC24, respectively), but concentrations in the uppermost core intervals were different than interpolated total PCB concentrations in surface sediment. Cores LDW-SC26 and LDW-SC28 had peak concentrations of 2,300 and 3,200 µg/kg dw in their 6-to-8-ft intervals. Peak concentrations were somewhat lower (750 to 800 µg/kg dw) and shallower (2 to 4 ft or 4 to 6 ft) in the other two cores (LDW-SC25 and DR054). Thus, in this area, peak concentrations were detected at depth in the cores with the highest subsurface concentrations.

Along the western shoreline between RM 1.8 and RM 2.2, four cores were analyzed. Peak concentrations were detected at depth in three of the four cores: LDW-SC34 (280 µg/kg dw in the 1-to-2-ft interval), LDW-SC38a (3,400 µg/kg dw in the 2-to-3-ft interval), and LDW-SC39 (440 µg/kg dw in the 1-to-2-ft interval). In the remaining core (LDW-SC35), the peak concentration was 370 µg/kg dw in the 0-to-2-ft interval, similar to concentrations in surface sediment.

In summary, peak concentrations in this section of the LDW were detected in the uppermost intervals and at depth (Map 4-21). Peak concentrations were at depth for the locations with the highest subsurface concentrations (i.e., greater than the 95th percentile). One notable exception was in the navigation channel near RM 1.0, where the uppermost interval of the core had a peak concentration of 3,200 µg/kg dw. At other locations where the peak concentrations were in the uppermost interval of the core, concentrations were always less than or equal to 370 µg/kg dw.

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[37] Samples were collected from coarser depth intervals in the replicate core collected at this location (LDW-SC201); the peak concentration in this core was 1,450 µg/kg dw in the 0-to-1.5-ft interval.

RM 2.2 to RM 3.0

The highest interpolated total PCB concentrations (i.e., greater than the 95th percentile) in surface sediment between RM 2.2 and RM 3.0 were detected between RM 2.2 and RM 2.3 on the west side of the LDW in EAA 2 (up to 23,000 µg/kg dw), in the Slip 4 EAA (up to 25,000 µg/kg dw), and on the east side of the LDW between RM 2.9 and RM 3.0 in the Boeing Plant 2/Jorgensen Forge EAA (up to 16,000 µg/kg dw) (Map 4-20c). The highest concentrations in subsurface sediment were in Slip 4 (up to 35,000 µg/kg dw).

In the vicinity of EAA 2 between RM 2.2 and RM 2.3 on the western shoreline, two cores were collected. Core LDW-SC40 had lower concentrations of total PCBs in the uppermost interval (160 µg/kg dw in the 0-to-1.3-ft interval) than in the surface sediment; and PCBs were not detected below 1.3 ft. Core DR137, located just downstream from this area, had a peak PCB concentration of 730 µg/kg dw in the 2-to-4-ft interval, higher than the interpolated concentration in surface sediment in this area.

Interpolated total PCB concentrations in surface sediment along the western shoreline between RM 2.3 and RM 3.0 were relatively low (generally ≤ 115 µg/kg dw). Historical cores collected along the western shoreline between RM 2.4 and RM 2.8 had large composite intervals (generally 0 to 3 ft or 0 to 4 ft). The RI core with greater resolution in this area (LDW-SC46), near RM 2.7, had a peak total PCB concentration of 270 µg/kg dw in the 2-to-4-ft interval.

In the navigation channel between RM 2.2 and RM 2.7, total PCB concentrations in surface sediment were generally between the 25th and 75th percentiles. Historical cores in the navigation channel had larger intervals of 2 or 4 ft and had total PCB concentrations similar to interpolated concentrations in surface sediment in that area (with a maximum concentration of 270 µg/kg dw).

Along the eastern shoreline between RM 2.3 and RM 2.8, total PCB concentrations in surface sediment and core intervals were generally similar, except for core LDW-SC43. Interpolated total PCB concentrations in surface sediment near this core were between the 75th and 95th percentiles; PCBs were not detected in the 0-to-2-ft or 2-to-4-ft intervals of this core.

Cores located at the mouth of Slip 4 had total PCB concentrations ranging from 320 to 980 µg/kg dw in the subsurface samples composited over 3 to 4 ft. These concentrations were similar to or higher than concentrations in surface sediment. However, the depth of the peak concentration was not discernable because of the large interval sampled. Seven cores at the head of Slip 4 and halfway through the slip generally had total PCB concentrations greater than the 95th percentile in at least one of the subsurface intervals. The peak total PCB concentration was in the 0-to-2-ft interval in three of the cores and in the 2-to-4-ft or 4-to-6-ft interval in the remaining four cores. High concentrations were detected in the subsurface intervals in these seven cores. For example, core SC01 had a peak concentration of 35,000 µg/kg dw in the 0-to-2-ft

interval, and core SC03 had a peak concentration of 15,000 µg/kg dw in the 4-to-6-ft interval.

In cores collected from RM 2.9 to RM 3.0 on the eastern shoreline, data were not available from the uppermost 2 ft, so the location of the peak concentrations within those cores is not known. Concentrations in intervals between 2 and 5 ft were highly variable ranging from non-detect in cores DUW106, DUW138, and DUW109 to 9,200 µg/kg dw in core DUW 37.

In summary, total PCB concentrations were variable in this section of the LDW (Map 4-21). Outside of Slip 4, in areas with sufficient data, peak concentrations were either low (≤ 160 µg/kg dw) or were detected at depth within the cores. Within Slip 4, concentrations were greater than the 95th percentile both at depth and in uppermost core intervals. Peak concentrations in subsurface cores from Slip 4 ranged from 990 to 35,000 µg/kg dw. The depths of peak concentrations in Slip 4 were highly variable.

RM 3.0 to RM 4.0

The highest total PCB concentrations (less than the 95th percentile) in surface sediment between RM 3.0 and RM 4.0 were detected between RM 3.5 and RM 3.9 on the east side of the LDW (up to 38,000 µg/kg dw in the Boeing Plant 2/Jorgensen Forge EAA), and on the west side of the LDW between RM 3.5 and RM 3.7 (up to 110,000 µg/kg dw) in the T-117 EAA (Maps 4-20d and 4-20e).

Surface sediment concentrations were generally low (between the 25th and 50th percentiles) along the western shoreline from RM 3.0 to RM 3.5. Subsurface samples in this area had relatively low total PCB concentrations, except for cores LDW-SC47 at RM 3.05 and SB-5 at RM 3.35; peak concentrations were detected in the 1-to-2-ft interval (2,000 µg/kg dw) and 2.5-to-5-ft interval (1,720 µg/kg dw), respectively, in these cores. Thus, total PCB concentrations were either low or the peak concentrations were detected at depth in this area.

In cores collected from the T-117 EAA between RM 3.5 and RM 3.7 on the western shoreline, the highest total PCB concentrations in subsurface samples (SE-70, SE-20, SE-21, SE-25, and SE-31) were in the same locations as the highest concentrations in surface sediment. The seven highest concentrations in subsurface sediment samples (ranging from 2,800 to 51,000 µg/kg dw) were all detected within the uppermost subsurface intervals.

Peak total PCB concentrations in subsurface cores LDW-SC49a, SD-321-C, and SD-322-C, in the navigation channel between RM 3.5 and RM 3.7, were detected at depth (> 1 ft), with concentrations ranging from 750 to 2,770 µg/kg dw. All other cores in the navigation channel between RM 3.0 and RM 4.0 had low total PCB concentrations in subsurface intervals (≤ 115 µg/kg dw).

In cores collected in the Boeing Plant 2/Jorgensen Forge EAA between RM 3.0 and RM 3.7 along the eastern shoreline, data in the uppermost 1- or 2-ft intervals were generally only available for cores collected between RM 3.5 and RM 3.7. Interpolated

concentrations in surface sediment for most of this area were greater than the 95th percentile. In general, most locations north of RM 3.6 with available data had peak concentrations in the uppermost intervals; these peak concentrations were greater than the 95th percentile. South of RM 3.6, there were a number of cores with peak concentrations in deeper intervals; these peak concentrations were generally less than the 95th percentile.

Surface sediment along the eastern shoreline between RM 3.7 and RM 3.9 generally had relatively high interpolated total PCB concentrations (between the 75th and 95th percentiles). Two cores in this area (LDW-SC50a and DR220) had similar concentrations in the uppermost intervals (780 and 830 µg/kg dw, respectively), and the other two cores (LDW-SC51 and LDW-SC52) had higher concentrations (1,290 and 3,000 µg/kg dw, respectively) in the uppermost intervals.

In summary, two EAAs with high total PCB concentrations are located within this reach. Vertical gradients were generally difficult to discern in the Boeing Plant 2/Jorgensen Forge EAA, but in general the highest concentrations appeared to be in the uppermost intervals, particularly north of RM 3.6. In the T-117 EAA, peak concentrations were generally found in the uppermost intervals of the cores. Along the eastern shoreline south of RM 3.7, peak concentrations were generally in the uppermost intervals. In cores between RM 3.0 and RM 3.5, in the navigation channel and along the western shoreline, total PCB concentrations were generally low or peaks were detected at depth (Map 4-21).

RM 4.0 to RM 5.0

Total PCB concentrations in surface and subsurface sediment between RM 4.0 and RM 5.0 were generally relatively low (less than the 50th percentile) (Map 4-20f), except for the eastern shoreline at RM 4.1 and the area between RM 4.8 and RM 5.0 (including the Norfolk EAA), where concentrations were greater than the 95th percentile in surface and subsurface sediment.

At RM 4.1 on the eastern shoreline, one small area had interpolated concentrations in surface sediment greater than the 95th percentile; the only subsurface concentration greater than the 95th percentile near RM 4.1 on the eastern shoreline was also near this small area (2,500 µg/kg dw in core SH-04 in the only interval analyzed at 0.33 to 0.82 ft). Total PCB concentrations in both surface and subsurface sediment around this small area were generally much lower although they appear to be higher in the uppermost interval of the cores (between the 50th and 75th percentiles) than in the interpolated surface sediment.

Between RM 4.8 and RM 5.0, total PCB concentrations were variable. They were highly elevated in surface sediments at RM 4.85 but were low at depth in nearby core SC55 (13.5 µg/kg dw in the 0-to-1-ft interval). Within the Norfolk EAA, peak total PCB concentrations were 81,700 µg/kg dw in the 1-to-2-ft interval at NFK008 and 247 µg/kg dw in the 0-to-1-ft interval at nearby location NFK009. The area within the Norfolk EAA was dredged and backfilled with clean material after these cores were

collected. The interpolated surface sediment data shown on Map 4-20f reflect post-dredging conditions.

A few other locations had total PCB concentrations in surface sediment that were between the 75th and 95th percentiles. Concentrations in subsurface sediment at these locations were similar (LDW-SC56) or lower (e.g., LDW-SC54) than interpolated concentrations in surface sediments in those areas.

In summary, total PCB concentrations in both surface and subsurface sediments were generally lower in this section of the LDW than in downstream areas, except at one location near RM 4.1 and locations near the Norfolk EAA. Concentrations in all cores from the navigation channel were low.

4.2.1.4 Tissue

The RI tissue dataset contained data from 308 composite samples of fish and shellfish collected in the LDW between 1992 and 2005.[38] Over 80% of these samples were collected in 2004 and 2005 as part of the RI, as described in Section 4.1.1.3. These data were used in the risk assessments.

In addition, total PCB data are available for seven composite shiner surfperch samples, six composite English sole samples, and nine individual English sole samples that were collected by King County from Area T1 in 2006 (Anchor and King County 2007). The 2006 data were not used in the ERA, HHRA, or FWM, but are useful in understanding temporal patterns of total PCBs in tissue.

Table 4-14 summarizes the total PCB (as Aroclors) data in the RI tissue dataset. PCBs were detected in most of the tissue samples collected between 1992 and 2005, with a few exceptions.[39] In whole-body composite samples of fish, the lowest mean total PCB concentration within an area was 71 µg/kg ww in juvenile chinook salmon collected between RM 0.5 and RM 0.9. The highest mean concentration in fish within a sampling area was 4,300 µg/kg ww in shiner surfperch in Area T2 (Table 4-14; Map 4-22). In general, mean concentrations in whole-body fish were highest in English sole and shiner surfperch (ranging from 1,000 to 3,900 µg/kg ww in English sole and 457 to 4,300 µg/kg in shiner surfperch) and lowest in juvenile chinook salmon (ranging from 71 to 260 µg/kg ww).

One shiner surfperch composite sample collected from Subarea T2E in 2004 had a total PCB concentration (18,400 µg/kg ww) that was much higher than any other samples in the LDW.[40] The total PCB concentration in this sample based on the sum of PCB

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[38] An additional 35 whole-body concentrations calculated from concentrations in fillet and remainder samples were also included in the dataset (see Section 4.1.2).

[39] PCBs were not detected in 4 of the 22 wild mussel samples, 19 of the 32 transplanted mussel samples, 1 of the 20 benthic invertebrate samples, and 3 of the 13 Dungeness crab edible-meat samples.

[40] Using Rosner's test for outliers from a log-normal distribution, this value was considered a statistical outlier ($p < 0.005$).

congeners was also very high (12,230 µg/kg ww), providing laboratory confirmation of the initial result. To better understand the variability of total PCB concentrations in shiner surfperch collected from Subarea T2E in 2004, 10 archived fish from this subarea were analyzed individually.[41] Total PCB concentrations in these individual fish ranged from 172 to 1,140 µg/kg ww, with a mean concentration of 640 µg/kg ww. Based on these data, it is likely that one or more of the 10 fish included in the composite sample with 18,400 µg/kg ww had a very high concentration of total PCBs.

[41] A total of 20 shiner surfperch (> 80 mm) were collected in subarea T2E in 2004. Ten of these fish were included in the initial composite sample for this area, and the 10 remaining fish were archived frozen as individual fish.

Table 4-14. Summary of tissue data for total PCBs (as Aroclors)

Tissue Type	Area[a]	1992 – 2003				2004				2005				2006			
		N	Total PCBs (µg/kg ww)			N	Total PCBs (µg/kg ww)			N	Total PCBs (µg/kg ww)			N	Total PCBs (µg/kg ww)		
			Min	Max	Mean[b]		Min	Max	Mean[b]		Min	Max	Mean[b]		Min	Max	Mean[b]
Fish, whole body																	
Juvenile chinook salmon	RM 0.5 – RM 0.9[c]	15	9.3 J	320 J	71	nd				nd				nd			
	RM 2.8 – RM 2.9[d]	9	6.9 J	1,200	260	nd				nd				nd			
English sole[e]	T1	nd				6	2,700	4,700	3,700	6[f]	1,120	2,200	1,600	6	630	1,470	920
	T2	nd				6	3,300	4,200	3,900	6[f]	1,600	2,400	2,000	nd			
	T3	nd				6	1,320	4,300	2,600	6[f]	610	2,200	1,400	nd			
	T4	nd				3	1,640	1,800	1,700	3[f]	910	1,180	1,000	nd			
Pacific staghorn sculpin	T1	nd				6	580	860	730	1	720 J	720 J	720	nd			
	T2	nd				6	620	1,260	770	1	620	620	620	nd			
	T3	nd				6	810	2,800	1,500	1	590	590	590	nd			
	T4	nd				6	510	1,300	780	1	430	430	430	nd			
Shiner surfperch	T1	3	350	620	500	6	970	1,830	1,400	6	530 J	960 J	780	7	250	730	457
	T2	nd				6	1,260	18,400 J	4,300	6	660	2,000	1,300	nd			
	T3	nd				6	1,280	8,800	3,800	6	700	2,400	1,500	nd			
	T4	nd				6	640	960	800	4	540	600	580	nd			
Starry flounder	T4	nd				3	450	660	570	nd				nd			

Table 4-14, cont. Summary of tissue data for total PCBs (as Aroclors)

Tissue Type	Area[a]	1992 – 2003				2004				2005				2006			
		N	Total PCBs (µg/kg ww)			N	Total PCBs (µg/kg ww)			N	Total PCBs (µg/kg ww)			N	Total PCBs (µg/kg ww)		
			Min	Max	Mean[b]		Min	Max	Mean[b]		Min	Max	Mean[b]		Min	Max	Mean[b]
Fish, fillet																	
English sole (with skin)	T1	nd				2	1,330	1,600	1,500	3	920	1,450	1,200	nd			
	T2	nd				2	1,840	2,010	1,930	3	850	1,400	1,000	nd			
	T3	nd				2	850	1,640	1,200	3	450	860	680	nd			
	T4	nd				1	710	710	710	1	530	530	530	nd			
English sole (without skin)	T1 and RM 1.1 – RM 1.4[g]	12	79	360	190	nd				nd				nd			
	RM 2.1 – RM 2.9	1	300 J	300 J	300	nd				nd				nd			
	RM 2.9 – RM 3.6	1	280 J	280 J	280	nd				nd				nd			
	RM 3.6 – RM 4.4	1	530 J	530 J	530	nd				nd				nd			
Starry flounder (with skin)	T4	nd				1	450	450	450	nd				nd			
Pile perch (with skin)	T3	nd				1	300	300	300	nd				nd			
Striped perch (with skin)	RM 0.0 – RM 0.2	2	164 J	171 J	168	nd				nd				nd			
	RM 4.0 – RM 4.1	nd				1	630	630	630	nd				nd			
Crab, edible meat																	
Dungeness crab	T1	3	80 J	180	132	3	206 J	290	230	1	20 U	20 U	20 U	nd			
	T2	nd				nd				nd				nd			
	T3	nd				3	212	300	250	1	20 U	20 U	20 U	nd			
	T4	nd				1	240 J	240 J	240	1	20 U	20 U	20 U	nd			

Table 4-14, cont. Summary of tissue data for total PCBs (as Aroclors)

Tissue Type	Area[a]	1992 – 2003				2004				2005				2006			
		N	Total PCBs (µg/kg ww)			N	Total PCBs (µg/kg ww)			N	Total PCBs (µg/kg ww)			N	Total PCBs (µg/kg ww)		
			Min	Max	Mean[b]		Min	Max	Mean[b]		Min	Max	Mean[b]		Min	Max	Mean[b]
Red rock crab	T1	2	85	164	125	nd				nd				nd			
Red rock/Dungeness crab	T1	1	60	60	60	nd				nd				nd			
Slender crab	T1	nd				3	210 J	390 J	270	nd				nd			
	T2	nd				6	108	260 J	190	1	64	64	64	nd			
	T3	nd				3	146	220	180	nd				nd			
	T4	nd				nd				nd				nd			
Crab, hepatopancreas																	
Dungeness crab	T1	1	1,700	1,700	1,700	1	4,000	4,000	4,000	1	1,420	1,420	1,420	nd			
	T2	nd				nd				nd				nd			
	T3	nd				1	4,500	4,500	4,500	1	1,310	1,310	1,310	nd			
	T4	nd				1	5,500	5,500	5,500	1	1,320	1,320	1,320	nd			
Slender crab	T1	nd				1	1,490 J	1,490 J	1,490	nd				nd			
	T2	nd				2	1,950 J	2,190 J	2,070	1	660	660	660	nd			
	T3	nd				1	1,640	1,640	1,640	nd				nd			
	T4	nd				nd				nd				nd			
Invertebrates, whole body																	
Amphipods	RM 0.5 – RM 0.7	4	106	410	230	nd				nd				nd			
Benthic invertebrates	LDW-wide[h]	nd				20	60 J	1,400	270	nd				nd			
Shellfish																	

Table 4-14, cont. Summary of tissue data for total PCBs (as Aroclors)

Tissue Type	Area[a]	1992 – 2003				2004				2005				2006			
		N	Total PCBs (µg/kg ww)			N	Total PCBs (µg/kg ww)			N	Total PCBs (µg/kg ww)			N	Total PCBs (µg/kg ww)		
			Min	Max	Mean[b]		Min	Max	Mean[b]		Min	Max	Mean[b]		Min	Max	Mean[b]
Clams	LDW-wide[h]					14	24	580 J	140					nd			
Mussels, wild	RM 0.3 – RM 1.2[i]	22	16	60	34	nd				nd				nd			
Mussels, transplanted	RM 0.5 – RM 1.2	32	35.9	73.1	27					nd				nd			

[a] Tissue sampling areas are shown on Maps 4-9 and 4-10.

[c] Includes nine samples collected between RM 0.5 and RM 0.9 during the RI sampling event in 2003 and six samples collected between RM 0.8 and RM 0.9 during the East Waterway sampling event (both locations are within Area T1).

[d] These nine samples were collected between RM 2.6 and RM 2.9 (slightly downstream from Area T3) during the RI sampling event.

[e] In addition to the composite samples presented in this table, nine individual whole-body English sole samples were collected in 2006 by King County in Area T1. Total PCB concentrations in these samples ranged from 240 to 1,210 µg/kg ww, with a mean concentration of 546 µg/kg ww (Anchor and King County 2007)

[f] Includes calculated "whole-body" samples (three each from Areas T1, T2, and T3 and one from T4). Concentrations in "whole-body" samples were calculated using results from separate analyses of fillet and remainder composite samples (i.e., all remaining tissue and fluids after fillets were removed from the specimens). The calculated English sole "whole-body" concentrations were based on the relative weights of and total PCB concentrations in skin-on fillet and remainder tissues collected in 2005.

[g] Includes data from three samples in the EVS 95 event collected between RM 1.1 and RM 1.4 (slightly upstream from Area T1), and nine samples collected between RM 0.5 and RM 1.0 (within Area T1) during the PSAMP and KC WQA sampling efforts.

[h] Benthic invertebrate and clam samples were collected from 10 locations throughout the LDW (Map 4-10).

[i] There was one additional wild mussel sampling location at the mouth of Slip 4 near RM 2.9.

J – estimated concentration

n – number of samples

nd – no data

U – not detected at given RL

Fillet samples from four different fish species were also analyzed. The largest fillet dataset for PCBs was available for English sole, with 15 samples (without skin) collected between 1992 and 1998 and 17 samples (with skin) collected in 2004 and 2005. Other species with available fillet data were pile perch (one sample), starry flounder one sample), and striped perch (three samples); all of these fillet samples were analyzed with skin. In all but 2 of 15 fillet samples collected in the 1990s without skin, total PCB tissue concentrations were lower (79 to 530 µg/kg ww) than those in samples collected in 2004 and 2005 with skin (450 to 2,010 µg/kg ww). This difference may be related to the year of sampling (1992 to 1998 vs. 2004 and 2005) or to the absence of skin.[42]

Mean total PCB concentrations within sampling areas for crab edible meat, benthic invertebrates, and clams were below 270 µg/kg dw; in general, these concentrations were lower than those in fish (Table 4-14). In crabs, mean total PCB concentrations (non-detected to 270 µg/kg ww) in edible-meat samples from the four tissue sampling areas were much lower than those in hepatopancreas samples (660 to 5,500 µg/kg ww) (Table 4-14). Mean concentrations in invertebrate samples (amphipods collected from RM 0.5 to RM 0.7 and benthic invertebrates collected from throughout the LDW) were low (230 and 270 µg/kg ww, respectively) relative to concentrations in fish tissue, as were soft-shell clam tissues collected from throughout the LDW (140 µg/kg ww). The lowest mean total PCB concentrations in any of the tissue types were detected in wild and transplanted mussels collected primarily from RM 0.3 to RM 1.2 (34 and 27 µg/kg, respectively). Mean total PCB concentrations in Dungeness crab edible meat samples collected in 2005 (20 U µg/kg ww) were lower than in mussels, but concentrations in 2005 (230 to 250 µg/kg ww) were higher than in mussels.

Temporal Evaluation

Mean total PCB concentrations in whole-body English sole and shiner surfperch were higher in 2004 than in 2005 in all sampling areas (Figure 4-1) and statistical tests were conducted to evaluate whether the difference between years was significant. Based on the results of a two-way analysis of variance (ANOVA), the annual mean of LDW-wide, $\log_{10}$-transformed, total PCB concentrations in whole-body English sole and shiner surfperch was significantly higher in 2004 than in 2005 (Figure 4-1, year effect in two-way ANOVA significant for both species[43]). Within each area, the

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[42] The potential influence of skin on the total PCB concentrations in the 2004 English sole fillet samples was investigated to facilitate comparison of 2004 samples with historical data. One fillet with skin and one fillet without skin were created from each of 10 individual fish archived from Area T1, for a total of 20 samples. Total PCB concentrations in the fillets with skin (420 to 1,440 µg/kg ww) and without skin (230 to 1,570 µg/kg ww) were not significantly different.

[43] A two-way ANOVA was conducted with factors year (2004, 2005), area (T1 to T4), and year-by-area interaction. The interaction term is included to assess whether the difference between years was consistent in all areas (or likewise, if the differences among areas were consistent in both years). The interaction term was not significant for either species ($p = 0.81$ for English sole; $p = 0.73$ for shiner

difference between years was significant in all areas for English sole[44] and in area T3 for shiner surfperch.[45]

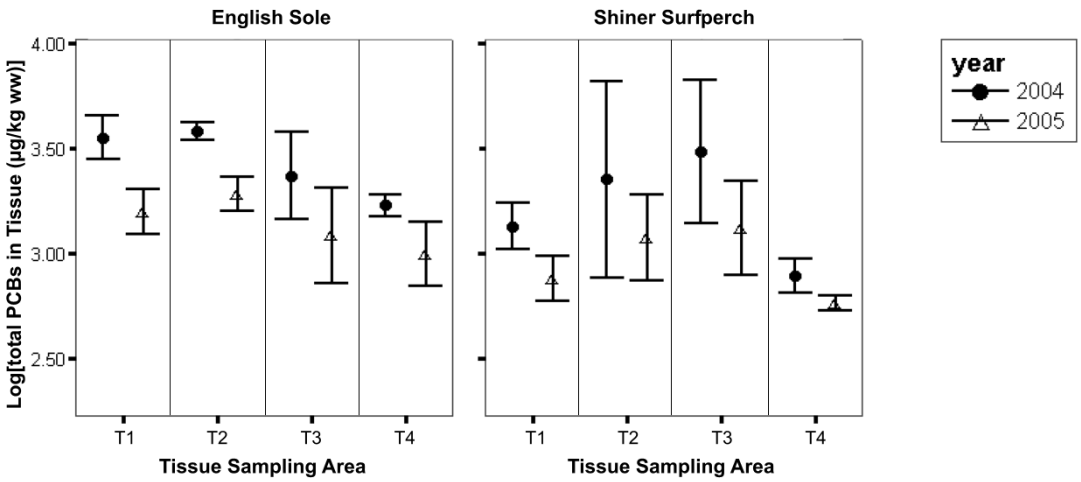

Figure 4-1. Mean and 95% confidence intervals for total PCBs in English sole and shiner surfperch tissues in 2004 and 2005

Four years of data were available for whole-body English sole and shiner surfperch samples from Area T1, the only area for which within-species data were available for three years or more (Figures 4-2 and 4-3). For English sole, $\log_{10}$-transformed total wet-weight PCB concentrations in tissues were significantly greater in 2004 than in 2005 and 2006 and in 2005 than 2006[46]. For shiner surfperch, $\log_{10}$-transformed total wet-weight PCB concentrations in tissue were significantly greater in 2004 than 2005,

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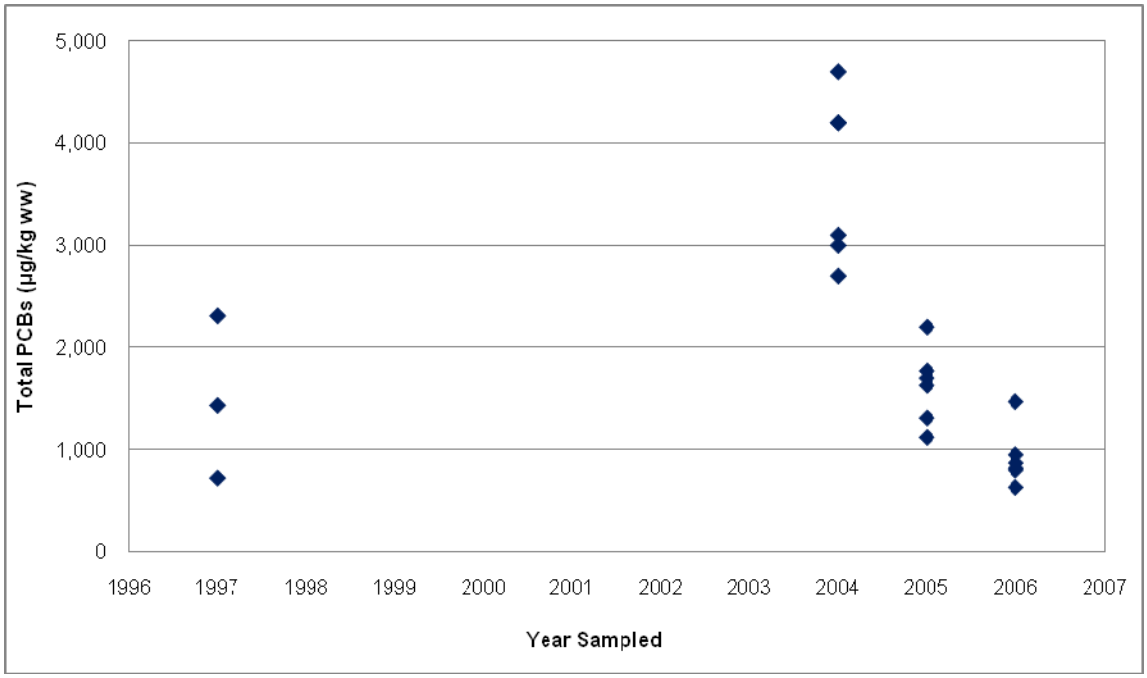
surfperch), indicating that the difference between years was consistent across tissue sampling areas and that each main effect (year, area) may then be interpreted as averaging over the other effect. The differences between years (averaging over areas) was significant ($p < 0.0005$ for English sole; $p = 0.001$ for shiner surfperch). Although these p-values may not be exact because variances were not equal across all groups (Levene's test, English sole $p < 0.0005$, shiner surfperch $p = 0.014$) and because sample size was not equal in all groups (English sole T4 $n = 3$, shiner surfperch T4 $n = 4$ in 2005), the high significance levels of the year effect indicates that the differences are significant.

[44] Based on individual one-way ANOVA testing for year effect within area (T1 $p < 0.0005$, T2 $p < 0.0005$, T3 $p = 0.001$, T4 $p = 0.039$) with Sidak's correction for multiple comparisons.

[45] Based on individual one-way ANOVA testing for year effect within area (T1 $p = 0.074$, T2 $p = 0.047$, T3 $p = 0.011$, T4 $p = 0.390$) with Sidak's correction for multiple comparisons.

[46] Based on Tukey post hoc multiple pairwise comparisons run after finding a significant effect of year in a one-way ANOVA testing for effects of year ($p < 0.0005$ for English sole; variances equal using Levene's test for homogeneity of variance $p = 0.982$). Results of Tukey post hoc multiple pairwise comparisons indicated 2004 > 2005 ($p < 0.0005$), 2004 > 2006 ($p<0.0005$), and 2005 > 2006 ($p = 0.003$). Data from 1997 are shown for informational purposes on Figure 4-2 but were not included in the statistical testing because of data quality concerns.

2006, and 1997.[47] Concentrations in tissue were also significantly greater in 2005 than 2006 but neither 2005 or 2006 concentrations were significantly different than concentrations in 1997.[48]



Note: Samples from 1997 consisted of tissue remaining after subsamples of fillets were removed. In addition, livers were removed from some fish in these composite samples. These data are presented for informational purposes only.

Figure 4-2. Total PCB concentrations in 1997, 2004, 2005, and 2006 English sole whole-body samples collected in Area T1.

[47] Based on Tukey post hoc multiple pairwise comparisons run after finding a significant effect of year in a one-way ANOVA testing for effects of year ($p < 0.0005$ for shiner perch; variances equal using Levene's test for homogeneity of variance $p = 0.632$). Tukey post hoc results indicated 2004 > 2005 ($p = 0.019$), 2004 > 2006 ($p < 0.0005$), and 2004 > 1997 ($p = 0.001$).

[48] Results of Tukey post hoc multiple pairwise comparisons indicated 2005 > 2006 ($p = 0.017$), 2005 = 1997 ($p = 0.177$), and 2006 = 1997 ($p = 0.959$).

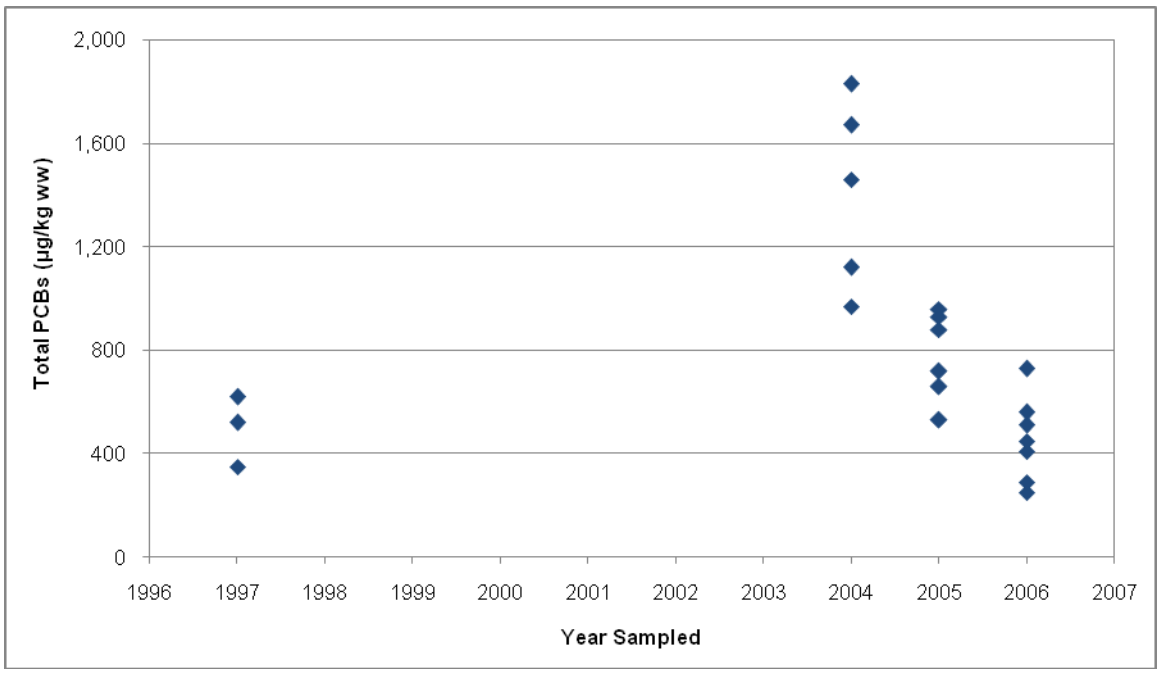


Figure 4-3. Total PCB concentrations in 1997, 2004, 2005, and 2006 shiner surfperch whole-body samples collected in Area T1.

When the data were lipid normalized, the only result that changed was significance of the differences between mean concentrations in 2005 and 2006 for both English sole and shiner perch. Differences were significant using the non-normalized data and became insignificant using the normalized data ($p = 0.101$ for English sole and $p = 0.194$ for shiner surfperch based on Tukey post hoc multiple pairwise comparison)[49].

Spatial Trends

Statistical tests were conducted to evaluate whether there were differences among the four sampling areas in either 2004 or in 2005. For both English sole and shiner surfperch, the relative magnitudes (rank ordering) of mean $\log_{10}$-transformed total PCB concentrations in all four sampling areas were consistent in 2005 and 2004 (Figure 4-1; interaction effect in two-way ANOVA not significant[50]). Both species had their lowest mean tissue concentrations in Area T4 in both years. In 2004, the mean of $\log_{10}$-transformed concentrations in Area T4 was significantly lower than mean of $\log_{10}$-transformed concentrations from the two areas with the highest mean concentrations (Areas T1 and T2 for English sole; Areas T2 and T3 for shiner

[49] Results of Tukey post hoc multiple pairwise comparisons for both species indicated 2004 > 2005 ($p = 0.001$) and 2004 > 2006 ($p<0.0005$). For shiner surfperch, the mean of $\log_{10}$-transformed lipid normalized concentrations was significantly greater in 2004 than in 1997 (Tukey post hoc multiple pairwise comparison $p = 0.035$) and was not different in 2005 and 2006 than in 1997 (Tukey post hoc multiple pairwise comparison $p = 0.740$ and $p = 0.060$ respectively).

[50] See methods and results of two-way ANOVA discussed in Section 4.2.1.4 (Footnote 25).

surfperch)[51]. Also in that year, the mean of log_{10}-transformed concentrations in tissues from the two areas with the highest mean concentrations did not differ significantly[52] and the mean of log_{10}-transformed concentrations in tissue samples from the two areas with the lowest concentrations did not differ significantly[53] (Areas T3 and T4 for English sole; Areas T1 and T4 for shiner surfperch). Statistical differences between concentrations in areas with intermediate concentrations were marginally significant.[54]

In 2005, fewer statistically significant differences existed between the modeling areas. For English sole, concentrations in Area T4 were lower than concentrations in T2,[55] but for shiner surfperch, there were no significant differences among areas.[56]

Map 4-22 presents total PCB concentrations in benthic invertebrate and clam tissue samples[57] at each location, along with total PCB concentrations in co-located surface sediment samples. The sample locations were selected to cover the range of total PCB concentrations detected in the LDW so that relationships between the concentrations of those chemicals in sediment and in benthic invertebrate and clam tissue could be evaluated throughout the entire range (Windward 2004e). This relationship was evaluated for sediment and benthic invertebrates using regression analyses, as described below. The relationship between sediment and clams was evaluated using the FWM, as described in Appendix D.

The regression relationship for total PCBs in sediment and benthic invertebrates was used in the ERA to estimate concentrations in tissue throughout the LDW using the RI baseline dataset; actual tissue data were available from 20 locations (Figure 4-4). When both sediment and tissue total PCB concentrations were log_{10}-transformed, the regression was significant ($p < 0.0005$) and explained 69% of the variance ($R^2 = 0.69$) in the concentrations in benthic invertebrate tissues (Figure 4-5).

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[51] Based on post hoc multiple pairwise ANOVA comparisons run after finding a significant effect of area in a one-way ANOVA testing for effects of year ($p < 0.0005$ for both species; see methods and results of two-way ANOVA discussed in Section 4.2.1.4. (Footnote 25). For log-transformed tissue concentrations in English sole, T4 < T1 ($p = 0.003$); and T4 < T2 ($p = 0.008$). For log-transformed tissue concentrations in shiner surfperch, T4 < T3 ($p = 0.001$); and T4 < T2 ($p = 0.009$).

[52] For both species, $p > 0.92$.

[53] For both species, $p > 0.43$.

[54] For English sole mean log-transformed tissue concentrations were lower in T3 than T2 ($p = 0.049$); and for shiner surfperch they were lower in T1 than T3 ($p = 0.075$).

[55] For English sole, mean log-transformed tissue concentrations were lower in T4 than T2 ($p = 0.025$) and in T3 than T2 ($p = 0.080$). The lowest p-value for all other pairwise comparisons was 0.20.

[56] For shiner surfperch, the lowest pairwise comparison p-value was 0.13.

[57] Dry weight concentrations were used for benthic invertebrate tissue to be consistent with the ERA.

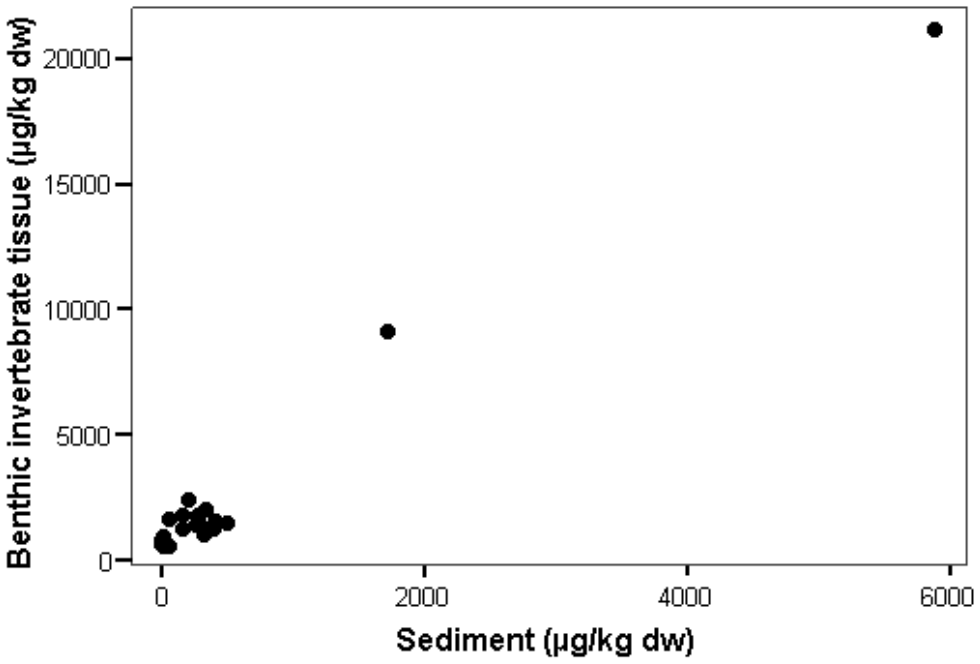


Figure 4-4. Total PCB concentrations in co-located sediment and benthic invertebrate tissue samples

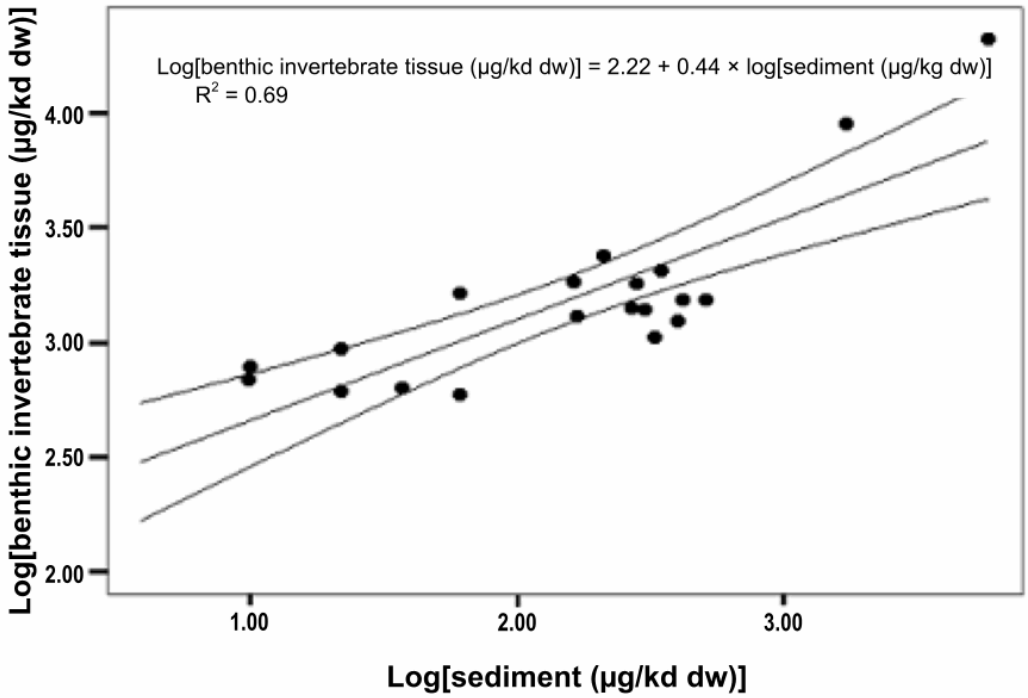


Figure 4-5. Regression for log_{10}-transformed total PCB concentrations in co-located sediment and benthic invertebrate tissue samples, including 95% confidence intervals

Although the residuals were normally distributed (R^2 of regression residuals vs. expected normal scores = 0.96), when plotted against log sediment concentrations (or estimated tissue values), the variance in the residuals does not appear homogeneous and the two largest residuals are associated with the two highest sediment values.

The highest total PCB concentration in sediment (5,900 µg/kg dw at location B3b in Slip 1) was co-located with the highest concentration in benthic invertebrates (21,000 µg/kg dw) (Figure 4-4). Although this sediment-tissue pair could be considered an outlier in terms of its sediment concentration (Leverage value = 0.256) and its tissue concentration (Studentized deleted residual = 2.95, p = 0.009), it did not have a significant effect on the estimated tissue concentration at its sediment concentration (DFFit = 0.197), the estimate of the intercept (DFbeta0 = 0.196), the estimate of the slope (DFbeta1 = 0.104), or the regression relationship in general (Cook's Distance = 1.34, p = 0.261). This indicates that the relationship between tissue and sediment is consistent across the range of sediment concentrations that were sampled and so the regression was considered a reliable means for estimating total PCB concentrations in benthic invertebrate tissue throughout the LDW.

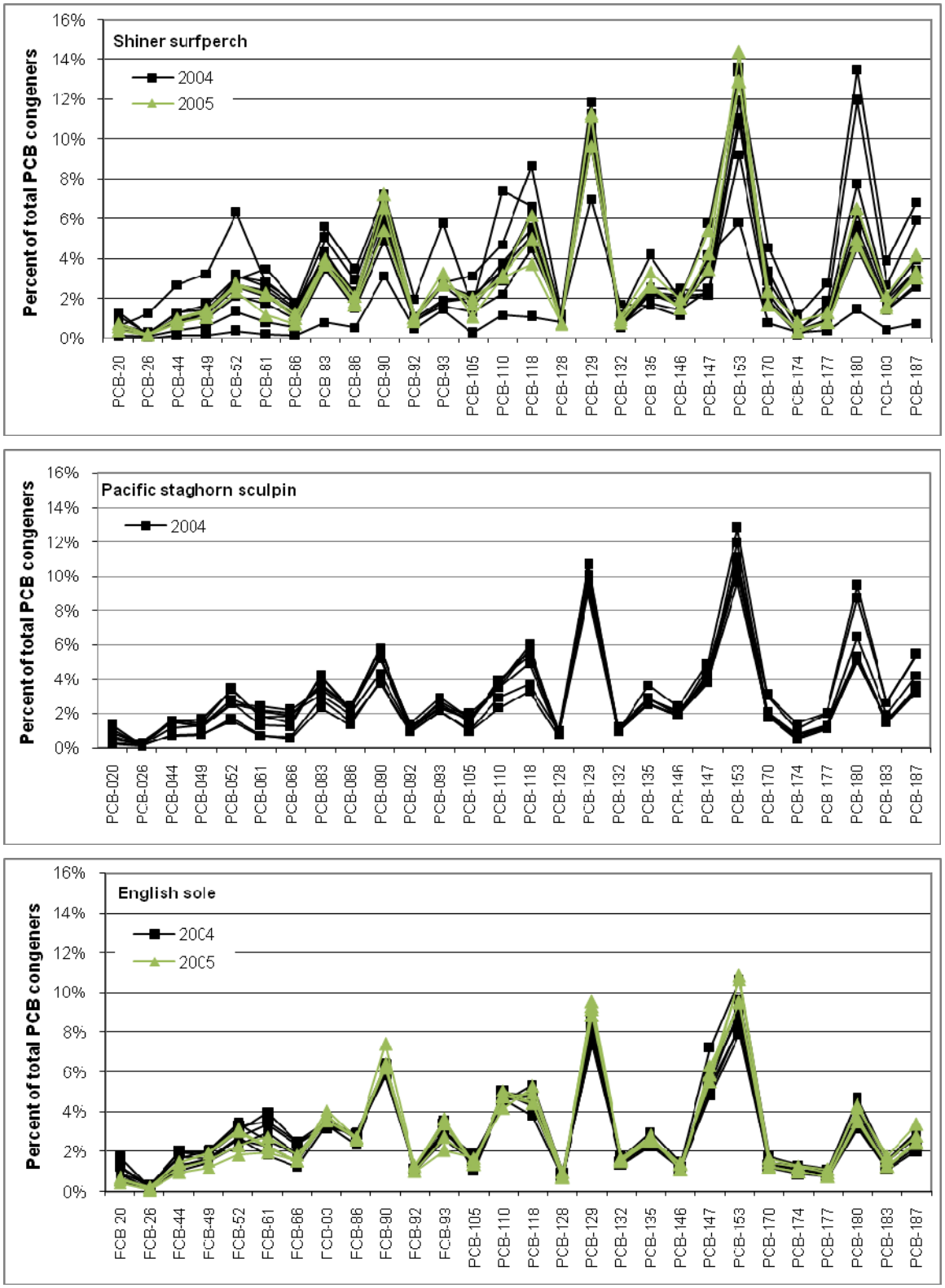
PCB Congener Pattern

PCB congener data were available for all tissue types except amphipods, mussels, red rock crab, and juvenile chinook salmon. PCB congeners were analyzed in a subset of all tissue types sampled in 2004 and 2005 as part of the RI. PCB congener data are also available for three English sole fillet samples from the 1995 Elliott Bay/Duwamish River fish tissue investigation (EVS unpublished).

Differences in the relative abundances of individual PCB congeners within a tissue type and between years (2004 and 2005) were evaluated by calculating the percentage of the total PCB concentration represented by each PCB congener in order to identify the PCB congeners that contributed the most to the total PCB concentrations for each sample independent of the total PCB concentrations. This analysis was conducted to determine if patterns were different in different species with different exposure regimes. Data from 2004 and 2005 were compared because different Aroclors were quantitated in tissue samples collected during these two years.

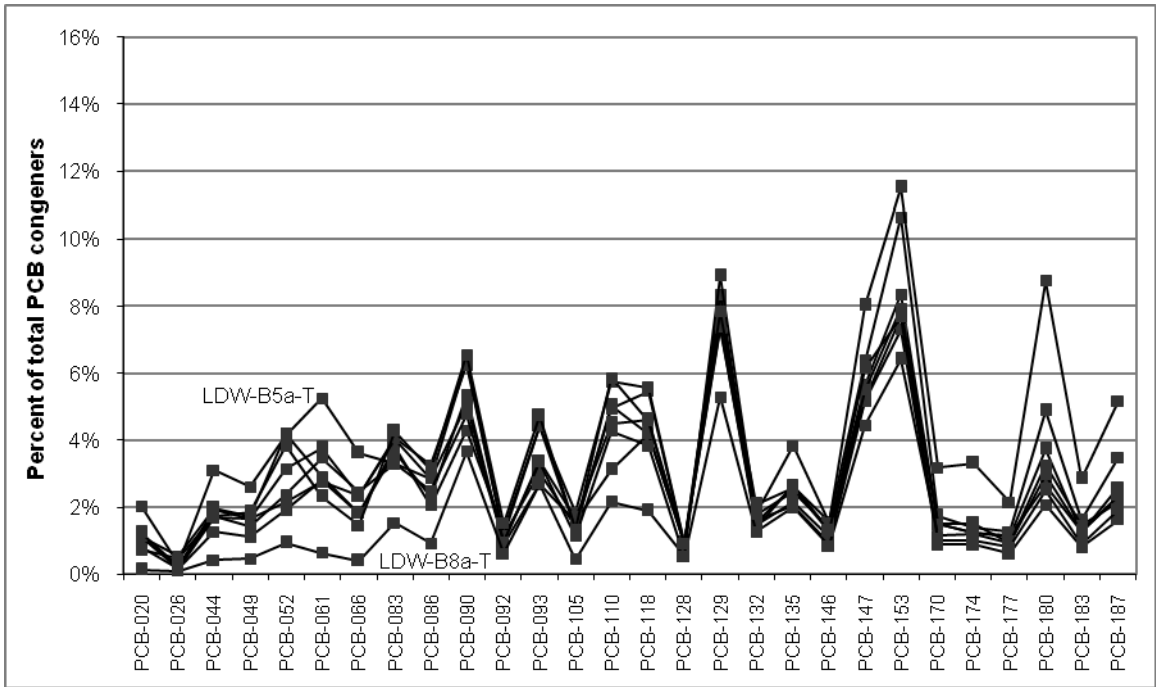
A PCB congener was selected for inclusion in plots showing relative abundances of individual PCB congeners within a sample if the PCB congener was detected in every sample and if the concentration comprised at least 2% of the total PCB concentration in any single sample. Twenty-eight PCB congeners were identified using these criteria. The mean PCB congener patterns among species with available data collected in 2004 are presented in Figures 4-6, 4-7, 4-8, and 4-9 for fish, benthic invertebrates, clams, and crabs, respectively.[58] The sum of the concentrations of these 28 PCB congeners represented 76 to 84% of the total PCB concentrations in tissue samples.

[58] The percentage of total PCB concentrations for each PCB congener and tissue type was calculated as the mean PCB congener concentration of a particular tissue type divided by the mean total PCB concentration (sum of all PCB congeners) of that tissue type.



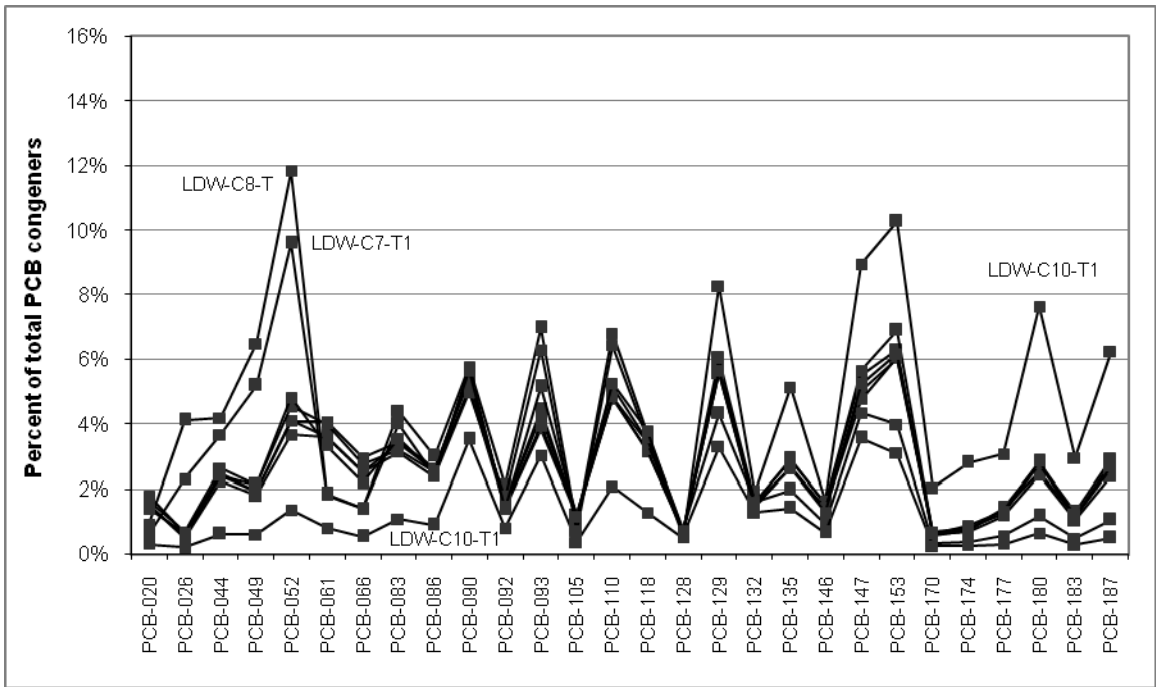
Note: Each line represents an individual composite sample.

Figure 4-6. PCB congener patterns in each whole-body composite sample of shiner surfperch and English sole collected in 2004 and 2005 and Pacific staghorn sculpin collected in 2004



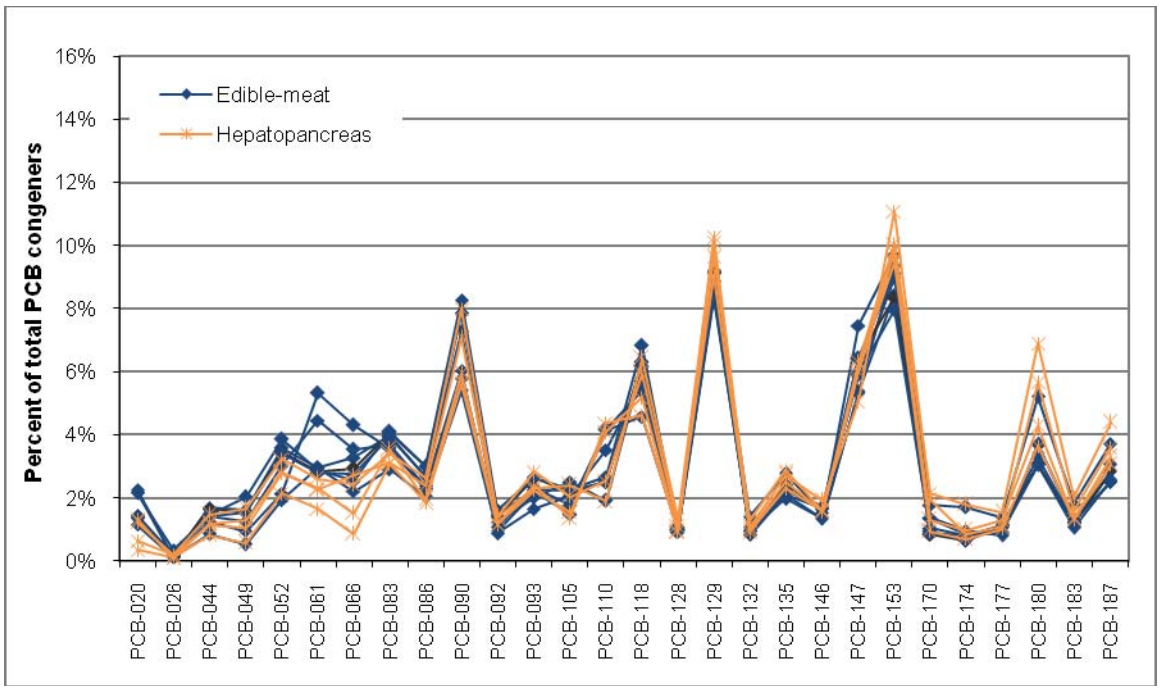
Note: Each line represents an individual composite sample.

Figure 4-7. PCB congener patterns in each whole-body composite benthic invertebrate sample collected in 2004



Note: Each line represents an individual composite sample.

Figure 4-8. PCB congener patterns in each whole-body composite soft-shell clam sample collected in 2004



Note: Each line represents an individual composite sample.

Figure 4-9. PCB congener patterns in each edible-meat and hepatopancreas crab composite sample collected in 2004

PCB congener patterns were similar among individual samples and between years in the whole-body composite tissue samples of shiner surfperch, Pacific staghorn sculpin, and English sole (Figure 4-6). The shiner surfperch samples had the greatest variability in PCB congener patterns among individual composite samples. Two of the nine shiner surfperch samples (collected from Subareas T3-E and T3-F) had higher contributions from the more highly chlorinated PCB congeners (PCB-180, PCB-183, and PCB-187). One shiner surfperch sample (collected from subarea T2-E) had higher relative contributions from the less chlorinated PCB congeners (PCB-044, PCB-049 and PCB-052).

PCB congener patterns in benthic invertebrate, clam, and crab tissue samples were generally similar to one another and to those in fish tissue (Figure 4-10). One benthic invertebrate sample (LDW-B8a from RM 3.5 on the east side of the LDW) had higher relative concentrations of several of the more highly chlorinated PCB congeners. Two of the eight clam samples (LDW-C8-T and LDW-C7-T1 from Slip 4) had higher relative concentrations of several less chlorinated PCB congeners, and one clam sample (LDW-C10-T from near RM 3.6 on the west side of the LDW) had relatively high concentrations of more highly chlorinated PCB congeners. The influence of the two clam samples from Slip 4 with high contributions of less chlorinated PCB congeners resulted in a mean pattern in clams that differed slightly from the patterns in other tissue types (Figure 4-10). The three clam samples with distinct congener patterns also had the highest total PCB concentrations in clam tissue samples.

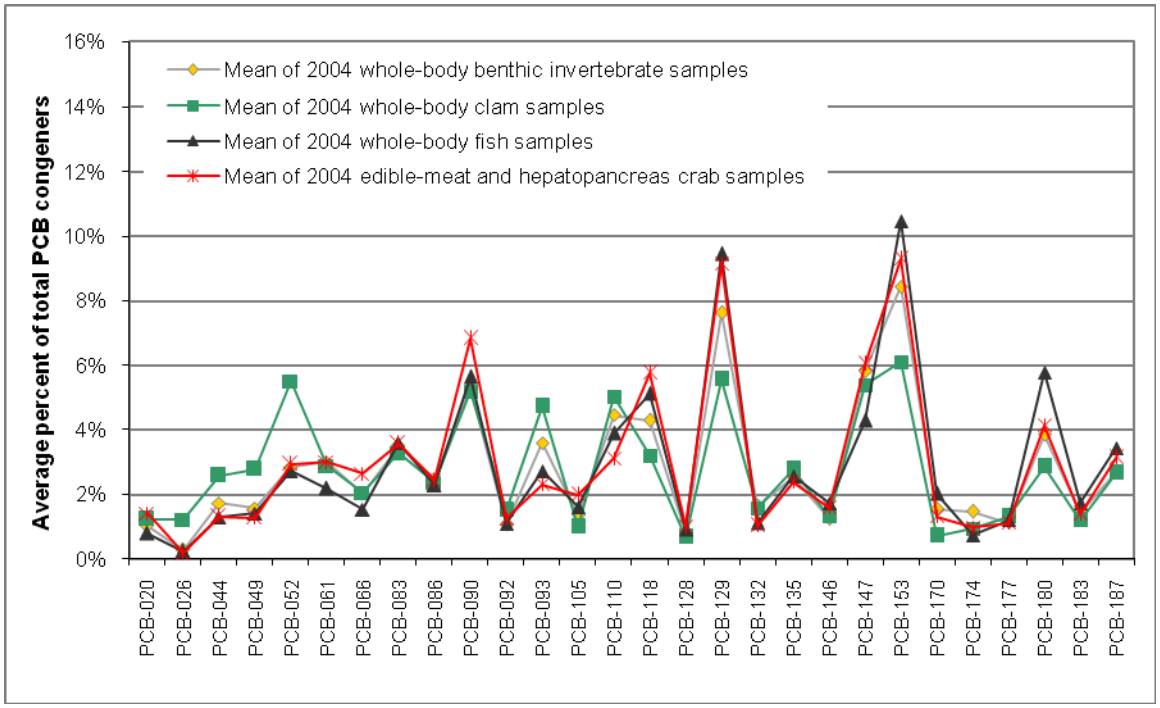


Figure 4-10. Mean PCB congener patterns for benthic invertebrate, clam, fish, and crab samples from the LDW

The differences in PCB congener patterns among the shiner surfperch samples likely reflect differences in exposures of individual fish. This effect is particularly evident in Subareas T2-E, T3-E, and T3-F. The three composites collected from these subareas with distinctly different congener patterns had the three highest total PCB concentrations in fish tissue samples. For English sole, Pacific staghorn sculpin, and crabs, the PCB congener patterns were similar among samples collected from different areas and among these different species.

PCB congener patterns in the 2004 and 2005 samples were similar in shiner surfperch and English sole composite samples. However, subtle differences in the PCB congener patterns in the 2004 samples compared to the 2005 samples resulted in differences in the Aroclor characterization of these samples. The 2004 samples were quantitated as a mixture of Aroclors 1248, 1254 and 1260. In 2005, the samples were quantitated as a mixture of Aroclors 1254 and 1260, except for one shiner surfperch sample that also had Aroclor 1248 quantitated (Area T2-E). In 2006 (Area T1 only), Aroclors 1254 and 1260 were the only Aroclors quantitated.

The toxic potency of a mixture of PCB congeners can be represented by the ratio of TEQ to total PCBs (sum of PCB congeners) in a sample. Mean potencies calculated for each tissue type collected in 2004 were similar (Table 4-15 and Figure 4-11). Individual composite samples with the lowest potencies were those with the highest total PCB concentrations. For example, the two shiner surfperch composite samples (collected from Subareas T2-E and T3-E) with the highest total PCB (sum of congeners) concentrations (12,230 and 8,010 µg/kg ww) had the lowest potencies (6.0 and

4.5 µg TEQ/g total PCBs, respectively) because the concentrations of dioxin-like PCB congeners relative to the overall concentration were low for the mixture present. The potencies were similar in the English sole and shiner surfperch tissues collected in 2004 and 2005 (Table 4-15). PCB TEQs for the various tissue types are summarized in the HHRA (Appendix B).

Table 4-15. Potency of dioxin-like PCB congeners in various tissue types

Tissue type	2004 Samples		2005 Samples	
	n	Potency Range (and Mean) (µg TEQ/g PCBs)	n	Potency Range (and Mean) (µg TEQ/g PCBs)
English sole – whole body	7	5.6 – 11.3 (8.9)	3	7.2 – 9.1 (8.2)
English sole – fillet	7	7.8 – 11.5 (9.8)	0	na
Pacific staghorn sculpin – whole body	8	5.3 – 13.2 (10.2)	0	na
Shiner surfperch – whole body	9	4.5 – 17.9 (11.8)	3	7.0 – 12.8 (10.8)
Dungeness crab – edible meat	3	9.2 – 12.6 (11.0)	0	na
Dungeness crab – hepatopancreas	2	7.5 – 9.3 (8.4)	0	na

n – number of samples

na – not available

PCB – polychlorinated biphenyl

TEQ – toxic equivalent

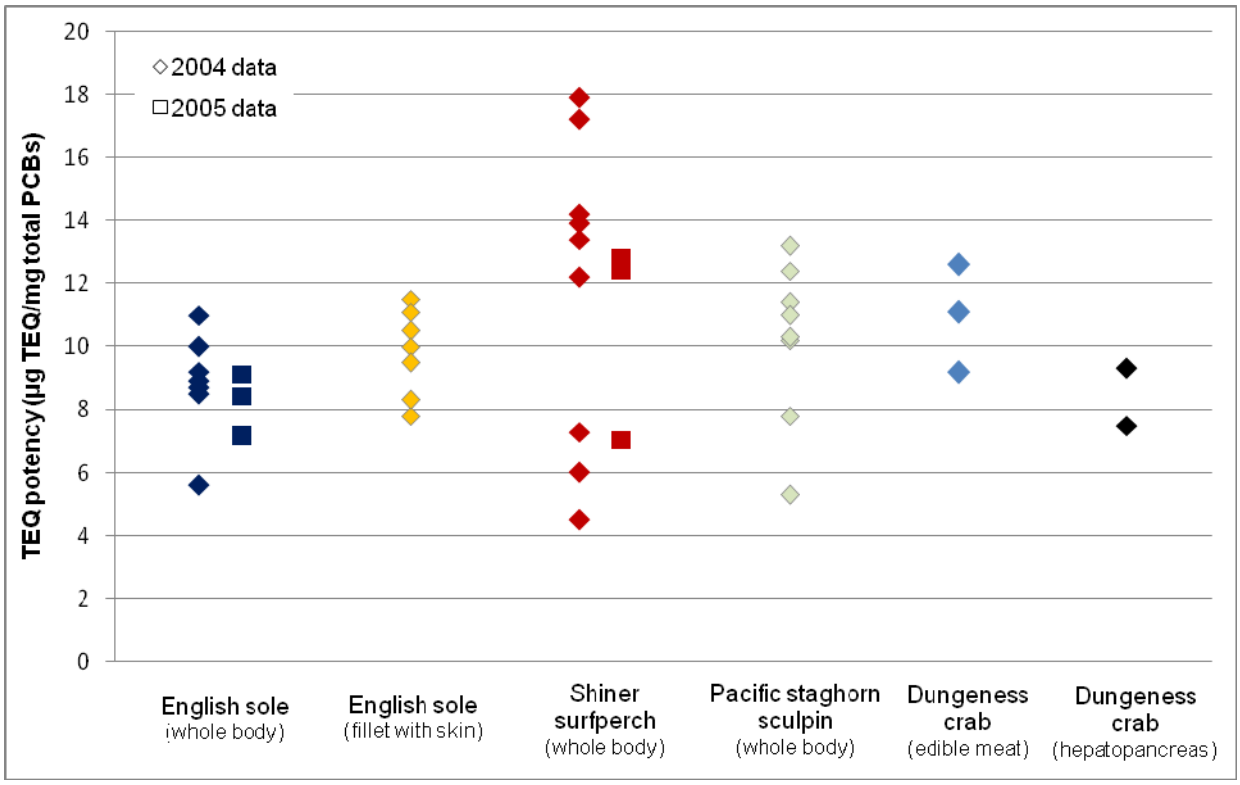


Figure 4-11. Calculated potencies by tissue type

4.2.1.5 Surface water

Concentrations of PCB congeners in surface waters were estimated on the basis of concentrations in semi-permeable membrane devices (SPMD) deployed in the LDW as part of the King County WQA in 1997, and were more recently analyzed in whole-water samples collected by King County in 2005 (Mickelson and Williston 2006). SPMD data are not summarized in the RI because of the uncertainty associated with the analysis; in addition, the PCB congener analysis included only 17 PCB congeners.[59] The whole-water samples were collected from two locations in the LDW: RM 0.0 at the southern end of Harbor Island (LTKE03) and RM 3.3 at the South Park Bridge (LTUM03). Samples were also collected from Elliott Bay and upstream of the LDW in the Green River. Each of these locations was sampled during two dry-weather periods (August and September) and two wet-weather periods (November and December). All 209 PCB congeners were analyzed in unfiltered water samples using high-resolution methods. Total PCB concentrations were calculated as the sum of detected PCB congeners.

Total PCB concentrations (sum of PCB congener concentrations) in water at both depths in both locations were lowest in November (Table 4-16). Flow, represented as the daily mean discharge from USGS Gauge 12113000 at Auburn, was also highest in November (1,060 cfs) compared to August, September, and December (277, 378, and 550 cfs, respectively).

The highest flow rate was associated with the lowest total PCB concentration at each LDW location and depth, and the lowest flow rate was associated with the highest concentration, except at RM 0.0 at 1 m below the surface.

Table 4-16. Summary of surface water data for total PCBs (as sum of PCB congeners)

Location ID	Collection Depth	Total PCB Concentration (pg/L)[a]			
		August (277 cfs)[b]	September (378 cfs)[b]	November (1,060 cfs)[b]	December (550 cfs)[b]
LTKE03 (RM 0.0)	1 m below the surface (brackish water)	1,796	1,024	591	1,947 J
	1 m above the bottom (saltwater)	1,814	nc	250	599
LTUM03 (RM 3.3)	1 m below the surface (brackish water)	1,592 J	1,452 J	398	1,122
	1 m above the bottom (saltwater)	3,211	1,883 J	132	1,341
LTED04 (Elliott Bay)	15 m below the surface	56.3	62.6	98.2	98.9
TGS/1 (Green River)	surface	222	784	751 J	40.8

[59] Total PCB concentrations calculated from SPMD data (ranging from 92 to 193 pg/L based on congener sums and 161 to 320 pg/L based on Aroclor sums and average partitioning coefficients) (Windward 2003a), were generally lower than concentrations in 2005 in whole-water samples.

[a] Total PCB concentration represents the sum of detected PCB congener concentrations. RLs for non-detects were not included in the calculation. Laboratory duplicate and field replicate samples were averaged using the data management rules presented in Appendix E, Section E.3. Detected results for select PCB congeners that were within 5 times the detected concentration in the associated method blank sample were qualified as non-detected at elevated RLs. Alternate data management procedures and data validation criteria were used to calculate total PCB concentrations in the King County technical memorandum that presented these data (Mickelson and Williston 2006).

[b] Daily mean discharge flow rate in the Green River at USGS Gauge 12113000 at Auburn.

cfs – cubic feet per second

ID – identification

nc – not calculated (a number of PCB congener results were rejected because method performance criteria were not met during analysis; therefore, total PCB concentrations were not calculated)

PCB – polychlorinated biphenyl

RL – reporting limit

In August and September, total PCB concentrations in samples collected from 1 m above the bottom at RM 3.3 were higher than those in samples collected from 1 m below the surface at the same location; concentrations were similar in surface and bottom water in August samples from RM 0.0. In contrast, in November and December, total PCB concentrations in bottom water were lower than those in the surface water, except for the sample from RM 3.3 in December.

Total PCB concentrations were lower in Elliott Bay samples than in LDW samples (Table 4-16). Concentrations in the Green River in August, September, and December were also lower than those in the LDW. In November, total PCB concentrations in Green River samples were highly variable; relative to the LDW samples, total PCB concentrations were higher in the original sample (819 pg/L) but were lower in the field duplicate sample (65 pg/L).

Additional water samples were collected from the Green River location (TGS/1) five times in 2007 (from March to June) for PCB congener analyses; total PCB concentrations in these samples were generally lower than in 2005, ranging from 38.45 pg/L to 132.94 pg/L (Williston 2007). Four samples were also collected from a location at RM 6.3 (downstream of TGS/1) during the 2007 sampling event. The total PCB concentrations in these samples ranged from 99.48 pg/L to 514.4 pg/L (Williston 2007).

4.2.1.6 Seeps

PCB Aroclors were analyzed in unfiltered seep water samples collected at 16 locations throughout the LDW during the seep sampling event conducted in 2004 for the RI. In addition, PCB Aroclors were analyzed in seep water samples collected at three T-117 locations in 2003 and in unfiltered seep water samples collected at 17 Boeing Plant 2 locations in 1995 (Map 4-11).

PCBs were detected in unfiltered seep water samples collected from three of the locations sampled for the RI (SP-54 in the inlet at RM 2.2, SP-64 south of Kellogg Island, and SP-71 at RM 0.015 on the west side), from one location at T-117 (Seep 3), and from four locations at Boeing Plant 2 (SE-SWY01, SW-SWY01, SE-SWY03, and SE-

84102) (Table 4-17). The maximum detected PCB concentration was 8.9 µg/L in an unfiltered sample collected at SP-54 near RM 2.2 in EAA 2. Unfiltered seep samples may contain sediment that is suspended during sampling. PCBs were also detected in one filtered seep sample collected from SP-54 in the inlet at RM 2.2 (EAA 2).

Table 4-17. Summary of detected concentrations of PCBs in seep water

Sampling Event	Chemical	Detection Frequency		Concentration (µg/L)		
		Ratio	%	Minimum Detect	Maximum Detect	RL or Range of RLs
RI – LDW-wide	**Total (not filtered or centrifuged)**					
	Aroclor 1248	2/14	14	0.092	4.7	0.017 – 0.02
	Aroclor 1254	3/14	21	0.020 J	2.3 J	0.017 – 0.02
	Aroclor 1260	2/14	14	0.16	1.9 J	0.017 – 0.02
	Total PCBs	3/14	21	0.020 J	8.9 J	nc
	Filtered[a]					
	Aroclor 1248	1/16	6	0.21	0.21	0.017
	Aroclor 1260	1/16	6	0.047	0.047	0.017
	Total PCBs	1/16	6	0.26	0.26	nc
T-117[b]	**Total (not filtered or centrifuged)**					
	Aroclor 1260	1/3	33	0.94 J	0.94 J	1.0
	Total PCBs	1/3	33	0.94 J	0.94 J	nc
Boeing Plant 2	**Total (not filtered or centrifuged)**					
	Aroclor 1254	1/17	6	0.93 J	0.93 J	1 – 1.5
	Aroclor 1260	3/17	18	1.7 J	4.6	1
	Total PCBs	4/17	24	0.93 J	4.6	nc

[a] A 1-µm filter was used during the RI sampling event.

[b] Seep 3 was resampled because it was suspected that the detected PCB concentration of 0.94 J µg/L may have been associated with suspended solids in the water sample. The sample was then centrifuged and analyzed, and PCBs were not detected at a reporting limit of 0.033 µg/L.

.LDW – Lower Duwamish Waterway
J – estimated concentration
nc – not calculated
PCB – polychlorinated biphenyl
RL – reporting limit
T-117 – Terminal 117

4.2.1.7 Summary of PCB data

A large sediment dataset is available to describe the nature and extent of PCB contamination in the LDW, including a total of 1,327 surface sediment samples from which lateral and vertical distributions of PCBs in LDW sediments can be described. Total PCB concentrations in surface sediment ranged from 1.6 to 220,000 µg/kg dw; the area-based 95th and 50th percentile concentrations calculated from the RI baseline dataset were 808 and 115 µg/kg dw, respectively. The highest total PCB concentrations in surface sediment were detected between RM 0.4 and RM 0.6 on the east side of the LDW (Duwamish/Diagonal EAA), in the Slip 4 EAA, between RM 3.3 and RM 3.7 on the east side (the Boeing Plant 2/Jorgensen Forge EAA), and between RM 4.8 and RM 4.9 (including the Norfolk EAA). Many samples in these areas had

concentrations greater than the area-based 95th percentile of 808 µg/kg dw. The SWAC for the area between RM 0.0 and RM 6.0 was 342 µg/kg dw.

In general, the highest total PCB concentrations in subsurface samples were detected in the same areas with higher concentrations in surface sediment. At many locations in the LDW, sediment at deeper intervals had higher PCB concentrations than did sediment at uppermost intervals. There were several exceptions to this pattern, in which relatively high total PCB concentrations were detected in the uppermost interval: 1) in the navigation channel near RM 0.9 and RM 1.0, 2) near RM 2.2 on the western shoreline, 3) towards the head of Slip 4, 4) near RM 3.6 east of the navigation channel, and 5) between RM 3.5 and RM 3.7 west of the navigation channel. Discussions of how these patterns compare to the physical CSM are presented in Section 4.3.

The fish and invertebrate tissue dataset provides good spatial coverage of the waterway, represents various exposure regimes of biota, and provides representation of species of interest to the ecological risk assessment (as either receptors or prey for receptors) and to the human health risk assessment (as species and size classes used by people). In whole-body composite samples of fish, mean total PCB concentrations ranged from 71 µg/kg ww in juvenile chinook salmon in T1 to 4,320 µg/kg ww in shiner surfperch in Area T2. In crabs, sampling area mean total PCB concentrations in edible-meat samples (non-detected to 270 µg/kg ww) were much lower than those in hepatopancreas samples (660 to 5,500 µg/kg ww). Benthic invertebrate samples had relatively low mean concentrations (230 and 270 µg/kg ww, respectively), as did soft-shell clams collected from throughout the LDW (143 µg/kg ww). Thus, total PCB concentrations in invertebrate (except crab hepatopancreas samples) tended to be lower than in fish, and concentrations in juvenile chinook salmon were lower than concentrations in English sole and shiner surfperch.

The LDW-wide mean total PCB concentrations in whole-body English sole and shiner surfperch were higher in 2004 than in 2005. When the differences between years were evaluated separately for each area, the results showed that for English sole the differences were significant in all areas, and for shiner surfperch, the differences were significant in area T3 only. A second analysis was conducted using whole-body English sole and shiner surfperch data from Area T1, the area with the most temporal data. For both species, total PCB concentrations from Area T1 were significantly greater in 2004 than in other years with available data (2005 and 2006 for English sole; and 2004, 2005, and 1997 for shiner surfperch). Total PCB concentrations in English sole tissues were similar in 1997, 2005, and 2006.

Total PCB concentrations in tissues of both English sole and shiner surfperch were significantly different among areas (sampling area effect significant, $p < 0.001$). The lowest mean tissue concentrations for both species were detected in Area T4 in both 2004 and 2005. In 2004, mean concentrations in Area T4 were significantly lower than concentrations in the two areas with the highest means (Areas T1 and T2 for English sole and Areas T2 and T3 for shiner surfperch). In 2005, concentrations in Area T4

were significantly lower than in Areas T2 for English sole, and there were no significant differences among areas for shiner surfperch.

Total PCB concentrations in sediment were significantly and positively correlated with concentrations in benthic invertebrate tissue.

In general, the PCB congeners patterns for each tissue type were similar, except for two clam samples from Slip 4 with higher contributions of less chlorinated PCB congeners. Of the three fish species analyzed for PCB congeners, shiner surfperch had the greatest variability in congener patterns among the individual samples. The TEQ potencies were similar in the English sole and shiner surfperch tissues collected in 2004 and 2005 although different Aroclors were quantitated in the 2 years.

The available dataset for water samples is small relative to the tissue and sediment datasets. Surface water grab samples were collected from the LDW on four sampling dates between August and December 2005 from two locations (near Harbor Island at RM 0.0 and near the South Park Bridge at RM 3.3) at two different depths. Total PCB concentrations ranged from 132 to 3,211 pg/L with an overall mean concentration of 1,277 pg/L. Concentrations were lowest at all locations in November 2005, when flow rates were the highest. Concentrations in the LDW were higher than those in Elliot Bay and in most cases were higher than in the Green River.

PCBs were detected in unfiltered seep water samples collected from three of the 16 locations sampled as part of the RI, one of the T-117 locations, and four of the Boeing Plant 2 locations. Because these samples were not filtered, these detections may represent entrained sediment at locations with elevated total PCB concentrations in sediment. Total PCBs were detected in one filtered sample collected from the inlet at RM 2.2 (EAA 2). The maximum detected PCB concentration was 8.9 µg/L in an unfiltered sample collected at a seep in EEA 2. PCBs were not analyzed in any porewater samples collected from the LDW.

4.2.2 Arsenic

This section summarizes the nature and extent of arsenic contamination of surface sediment, subsurface sediment, tissue, surface water, seeps, and porewater. The extent of arsenic in site media is described in detail in this section because arsenic is a risk driver chemical.

4.2.2.1 Surface sediment

Surface sediment samples collected from 852 of the 1,365 locations in the RI baseline dataset were analyzed for total arsenic. Arsenic was detected in 93% of surface sediment samples analyzed for arsenic, with concentrations ranging from 1.2 mg/kg dw to 1,100 mg/kg dw and a mean concentration of 17 mg/kg dw (Table 4-18). Arsenic concentrations in surface sediment were much less variable than total PCB concentrations; the numerical 95th percentile based on arsenic concentrations from the surface sediment dataset was 30 mg/kg dw (Map 4-23). The 75th, 50th, and 25th percentiles were 15, 11, and 7.9 mg/kg dw, respectively. Percentiles calculated on an

area-basis were similar; these percentiles were calculated as described in Section 4.2.1.1 using Thiessen polygons. Area-based 95th, 75th, 50th, and 25th percentiles were 24, 13, 11, and 7.9 mg/kg dw, respectively (Map 4-24).

Table 4-18. Summary of surface sediment data for total arsenic

Chemical	Detection Frequency		Total Arsenic Concentration (mg/kg dw)			
	Ratio	%	Minimum Detection	Maximum Detection	Calculated Mean[a]	RL or Range of RLs[b]
Arsenic	794/852	93	1.2	1,100	17	3.1 – 31

[a] Calculated mean concentration is the average of detected concentrations and one-half the RL for non-detected results.

[b] RLs based only on non-detect samples.

dw – dry weight

RL – reporting limit

The maximum arsenic concentration in surface sediment was detected at sampling location LDW-SS114, on the east side of the LDW between RM 3.7 and RM 3.8 (Map 4-23). Other areas with arsenic concentrations in surface sediment greater than the 95th percentile (30 mg/kg dw) were: 1) near RM 0.1 on the east side of the navigation channel, 2) between RM 0.4 and RM 0.6 in the Duwamish/Diagonal EAA, 3) at the head of Slip 1, 4) between RM 1.3 and RM 1.5 on the west side of the navigation channel, 5) at the mouth and head of Slip 3, 6) between RM 3.5 and RM 3.8 on the east side of the navigation channel in and just upstream of the Boeing Plant 2/ Jorgensen Forge EAA, and 7) near RM 5.0 (Map 4-24).

Forty-seven of the locations where surface sediment samples were collected and analyzed for arsenic were within 10 ft of where samples had previously been collected (Table 4-19). The older data were excluded from the RI baseline dataset, as described in Section 4.1.2.1 and Appendix E; however, older data for stations that were re-occupied in the RI were retained for the evaluation of changes in concentrations between older samples and newer samples only.

Of the 47 locations, arsenic concentrations decreased in surface sediments over time at 14 locations (30%) and increased at 9 locations (19%) (Table 4-12 and Map 4-25). At the remaining 24 locations (51%), concentration differences were within the range of analytical variability (≤ 25% increase or decrease compared to the initial concentration) (Table 4-19 and Map 4-25). The area with both the highest negative (-323 and -122.8 mg/kg dw) and highest positive (+105 mg/kg dw) changes in arsenic concentrations was located between RM 1.3 and RM 1.4 on the west side of the LDW, perhaps because of high spatial variability in this area. At all other locations, changes in arsenic concentrations were relatively small (between -16.6 and +14.5 mg/kg dw).

Table 4-19. Difference in arsenic concentrations at resampled locations, sorted by concentration difference

Approximate River Mile	Sampling Location ID		Sampling Date		Arsenic Concentration (mg/kg dw)		
	Initial	Resampled	Initial	Resampled	Initial	Resampled	Difference
1.3	SS-2	LDW-SS48	8/17/1993	1/18/2005	1,130	807	-323
1.4	SS-4	LDW-SS55	8/17/1993	1/24/2005	140	17.2	-122.8
1.5	DR123	LDW-SS57	9/14/1998	1/24/2005	52	35.4	-16.6
4.2	R42	LDW-SS129	10/13/1997	1/20/2005	21.1	10.6	-10.5
3.7	SD-323-S	LDW-SS110	8/17/2004	1/25/2005	32	24.7	**-7.3**
0	K-11	LDW-SS1	9/30/1991	1/17/2005	13	6.2	-6.8
4.3	DR286	B10b	8/26/1998	8/19/2004	11	5.05 J	-5.95
1.0	DR019	LDW-SS32	8/17/1998	1/18/2005	21	15.7	**-5.3**
4.2	R40	LDW-SS127	10/13/1997	1/20/2005	18.4	13.2	-5.2
1.7	DR097	LDW-SS63	8/20/1998	1/21/2005	15	10.2	-4.8
0	DR076	LDW-SS5	8/24/1998	1/17/2005	11	6.5	-4.5
0.2	DR035	LDW-SS12	8/11/1998	1/17/2005	17	13.0	**-4**
3.8	DR187	LDW-SS115	8/27/1998	1/25/2005	48	44.4	**-3.6**
0.3	DR079	LDW-SS15	8/24/1998	1/17/2005	15	11.5	**-3.5**
1.0	DR087	LDW-SS37	8/12/1998	1/18/2005	17 J	13.6	**-3.4**
3.7	R21	LDW-SS113b	10/9/1997	1/20/2005	10.8	8.3	**-2.5**
3.9	R30	LDW-SS119	10/11/1997	1/19/2005	12.4	10.9	**-1.5**
0.9	DR085	LDW-SSB2b	8/31/1998	3/11/2005	17	16.5	**-0.5**
4.1	DR238	LDW-SS125	8/27/1998	1/20/2005	8.9	8.6	**-0.3**
0.9	DR021	LDW-SS319	8/17/1998	10/4/2006	15	14.8	**-0.2**
0.3	DUD042	LDW-SS17	11/11/1995	1/24/2005	15 J	14.9	**-0.1**
3.1	DR198	LDW-SS102	8/20/1998	1/24/2005	6.7	6.6	**-0.1**
1.4	DR028	B4b	8/17/1998	8/28/2004	9.9	10.3 J	**0.4**
2.0	R7	LDW-SS75	10/15/1997	1/21/2005	7.9	8.3	**0.4**
4.1	A11-05	LDW-SS126	8/18/1994	1/20/2005	6.5	7.3	**0.8**
4.1	06-intsed-2	SH-04	7/1/1996	8/24/2004	8.0	8.8	**0.8**
4.2	R45	LDW-SS130	10/16/1997	1/20/2005	13.9	15.0	**1.1**
2.1	DR106	LDW-SS76	8/19/1998	1/20/2005	13	14.5	**1.5**
1.2	DR088	LDW-SS40	8/31/1998	1/18/2005	15	16.7	**1.7**
4.2	DR242	SB-1	8/24/1998	8/25/2004	20	22	**2**
1.4	DR030	LDW-SS50	8/17/1998	1/24/2005	14	16.3	**2.3**
0.2	K-05	LDW-SS10	9/27/1991	1/17/2005	10	12.4	**2.4**
3.2	DR202	LDW-SS104	8/27/1998	1/25/2005	8.1	11.5	3.4
4.0	07-intsed-1	SH-02	7/1/1996	8/24/2004	7.0	11	4
3.8	R24	LDW-SS117	10/10/1997	1/20/2005	10.2	14.4	4.2
2.2	DR113	LDW-SS81	8/19/1998	3/8/2005	13	18.1	5.1

Approximate River Mile	Sampling Location ID		Sampling Date		Arsenic Concentration (mg/kg dw)		
	Initial	Resampled	Initial	Resampled	Initial	Resampled	Difference
1.4	DR065	LDW-SS52	8/17/1998	1/25/2005	10	15.5	5.5
1.9	DR131	LDW-SS70	8/13/1998	1/21/2005	8.1	14.8	6.7
3.7	DR186	LDW-SS111	8/27/1998	1/19/2005	25	31.7	6.7
1.4	DR160	LDW-SS51	8/12/1998	1/18/2005	9.6	16.9	7.3
3.7	SD-DUW92	SD-320-S	4/2/1996	8/16/2004	12	20	8
0.1	K-07	LDW-SS4	9/30/1991	1/17/2005	12	21.2	9.2
4.7	DR271	LDW-SS148	9/15/1998	3/9/2005	6.4	15.6	9.2
1.3	DR053	LDW-SS44	8/31/1998	1/21/2005	35 J	46.8	11.8
2.8	DR175	LDW-SS94	8/20/1998	1/21/2005	12	26.5	14.5
1.0	DR020	LDW-SS31	8/17/1998	1/21/2005	99	122	**23**
1.4	SS-3	LDW-SS49	8/17/1993	1/26/2005	66.0	171	105

Note: Concentrations in bold were within the range of analytical variability (≤25% increase or decrease compared to the initial concentration).

dw – dry weight

ID – identification

J – estimated concentration

4.2.2.2 Subsurface sediment

Subsurface sediment cores were sampled at various intervals during the different sampling events. To calculate summary statistics, data were categorized according to specified sampling intervals, as described for total PCBs in Section 4.2.1.2.

Arsenic was detected in 82% of the subsurface sediment samples that were analyzed for arsenic, at concentrations ranging from 3.9 to 2,000 mg/kg dw (Table 4-20). Most of the subsurface arsenic data were collected in the top 4 ft of sediment cores. Arsenic data in 1- and 2-ft subsurface sampling intervals for the top 6 ft of sediment are shown graphically on Maps 4-26a through 4-26c. Data on these maps and in Table 4-20 are discussed in detail for individual cores in the following section.

Table 4-20. Summary of subsurface sediment data for total arsenic

Sampling Interval (ft)	Detection Frequency		Total Arsenic Concentration (mg/kg dw)		
	Ratio	%	Minimum Detection	Maximum Detection	RL or Range of RLs[a]
All data[b]					
Any interval	267/325	82	3.9	2,000	3.0 – 10.0
1-ft intervals[c]					
0 to 1	60/63	95	4.7	707	6.0 – 6.2
1 to 2	46/59	78	5.2	281	4.5 – 8.00
2 to 3	7/10	70	5.9	161	6.0 – 6.1
3 to 4	4/9	44	9	21	6.0 – 7

Sampling Interval (ft)	Detection Frequency		Total Arsenic Concentration (mg/kg dw)		
	Ratio	%	Minimum Detection	Maximum Detection	RL or Range of RLs[a]
8 to 9	1/1	100	8	8	na
2-ft intervals[c]					
0 to 2	71/76	93	5.3	494	5.0 – 7
2 to 4	60/73	82	5.0	2,000	6 – 8.00
4 to 6	5/6	83	14	270	7.00
6 to 8	4/4	100	20	1,890	na
8 to 10	2/3	67	14	21	7.00

[a] RLs are presented only if there were no detected concentrations.

[b] Data included in the subsurface dataset are described in Section 4.1.2.2.

[c] For the calculation of summary statistics in this table, actual sampling intervals were rounded to the nearest foot. The data for each interval were then assigned to the 1- or 2-ft sampling category that best approximated the actual sampling interval. If there were data for two 1-ft intervals at a particular location, the concentrations were averaged to obtain a calculated concentration for the corresponding 2-ft interval. Some of the subsurface data were collected from intervals other than 1 or 2 ft; therefore, the detection frequency ratio for "any interval" may reflect data for intervals not included in these categories.

dw – dry weight

na – not applicable

RL – reporting limit

4.2.2.3 Surface and subsurface sediment patterns

This section evaluates arsenic concentrations in surface and subsurface sediment to determine whether peak concentrations are found at the surface or at depth. The specific depths of peak concentrations within sediment cores were also evaluated. The percentiles for arsenic concentrations in sediment discussed in this section were calculated on an area-basis, using the method described in Section 4.2.1.1 and Thiessen polygon data.

RM 0.0 to RM 1.0

Between RM 0.0 and RM 1.0, areas with the highest arsenic concentrations in surface sediment were located in Slip 1 (up to 725 mg/kg dw) and near RM 0.1 on the east side (up to 123 mg/kg dw) (Map 4-27a). These two areas also had the highest concentrations in subsurface cores collected in this section of the LDW. In the Duwamish/Diagonal EAA, between RM 0.4 and RM 0.7, arsenic concentrations in surface sediment were also higher than the area-based 95th percentile of 24 mg/kg dw prior to dredging (Map 4-27a); the maximum concentration in this area was 55 mg/kg dw.

Two subsurface cores were collected along the eastern shoreline of the LDW near RM 0.1 and RM 0.2. Peak arsenic concentrations in both cores were at depth. At LDW-SC2, the area with the high arsenic concentrations in surface sediment, the maximum concentration (270 mg/kg dw) was in the 4-to-6-ft interval although the arsenic concentration was also relatively high (190 mg/kg dw) in the 0-to-2-ft interval.

At LDW-SC4, the peak concentration (63 mg/kg dw) was detected in the 1-to-2-ft interval (Maps 4-27a).

Maximum arsenic concentrations were low (non-detected to 22 mg/kg dw) in the three cores collected between RM 0.0 and RM 0.2 near the navigation channel and on the west side of the navigation channel. Peak arsenic concentrations were slightly higher (41 to 44 mg/kg dw) in two cores collected farther south along the western shoreline near RM 0.3 (LDW-SC6 and DR068). The peak concentration was at depth (2-to-4-ft interval) in LDW-SC6 and unknown in the other core (DR068) because only one interval was sampled.

Multiple cores were collected and analyzed for arsenic between RM 0.4 to RM 0.6 in the Duwamish/Diagonal EAA (including several cores just outside of the area to the north and to the east). Many of these locations had arsenic concentrations in surface sediment greater than the area-based 95th percentile of 24 mg/kg dw. The highest arsenic concentrations in surface and subsurface sediment in this area were 55 and 36 mg/kg dw, respectively. Some peak concentrations were located in the uppermost intervals and some were at depth. Most of the arsenic concentrations in subsurface sediment were relatively low; only four cores (LDW-SC9, DUD254, DUD020, and DUD261) had concentrations greater than the area-based 95th percentile. These peak concentrations were located at depth in these four cores.

Two cores were collected within the navigation channel just outside of the Duwamish/Diagonal EAA. Peak arsenic concentrations in these cores (62 mg/kg dw near RM 0.35 and 30 mg/kg dw near RM 0.5) were at depth (4-to-6-ft interval and 1-to-3-ft interval, respectively). The peak concentration in a third core in the navigation channel (24 mg/kg dw near RM 0.9) was in the uppermost interval.

Relatively few cores were collected in the vicinity of Kellogg Island. In cores collected at three locations north and south of the island (two locations near RM 0.6 and one location near RM 1.0), arsenic concentrations were less than or equal to the 95th percentile and varied little with depth. In a fourth core, which was collected in the vicinity of the island along the western shoreline of the LDW between RM 0.5 and RM 0.6 (LDW-SC11), a peak concentration of 28 mg/kg dw was detected in the uppermost interval.

In a core collected at the head of Slip 1, the peak arsenic concentration was at depth (170 mg/kg dw in the 1-to-2-ft interval). In three cores collected toward the mouth of Slip 1 and one core collected just downstream of Slip 1, arsenic concentrations were much lower (≤ 30 mg/kg dw) and did not vary much with depth. Three of these four cores had peak concentrations in the uppermost interval, and one core had a peak concentration at depth (24 mg/kg dw in the 2-to-4-ft interval of DR021).

In summary, two areas had high arsenic concentrations (> 100 mg/kg dw) in surface sediment; cores in these areas had peak concentrations at depth (Map 4-28). Arsenic concentrations in surface and subsurface sediment in the Duwamish/Diagonal EAA were also somewhat elevated (> 30 mg/kg dw) prior to dredging. In other areas, there

were generally low arsenic concentration gradients within cores, with some peak concentrations at depth and some in the uppermost interval. In all of the cores with concentrations greater than 30 mg/kg dw (and with data for more than one interval), the peak concentrations were at depth.

RM 1.0 to RM 2.2

Between RM 1.0 and RM 2.2, the highest arsenic concentrations were detected at two locations: 1) between RM 1.3 and RM 1.5 on the west side of the LDW (up to 807 and 1,890 mg/kg dw in surface and subsurface sediment, respectively), and 2) at the head of Slip 3 (up to 81 and 2,000 mg/kg dw in surface and subsurface sediment, respectively) (Map 4-27b). In the other areas, surface and subsurface sediment concentrations were generally less than the 95th percentile.

West of the navigation channel, four cores were collected between RM 1.3 and RM 1.4 (LDW-SC25, DR054, LDW-SC26, and LDW-SC28). High arsenic concentrations were detected in subsurface sediment in these cores, with peak concentrations ranging from 250 to 1,890 mg/kg dw (Maps 4-27b). The peak concentrations were at depth in each of these four cores; the two cores with the highest concentrations (760 and 1,890 mg/kg dw) each had peak concentrations at an interval of approximately 6 to 8 ft. Between RM 1.4 and RM 1.5, four cores were sampled at only one interval each, with concentrations ranging from 11 to 181 mg/kg dw; this area was dredged after the cores had been collected. In the only other core collected between RM 1.3 and RM 1.5 (LDW-SC29), arsenic concentrations were relatively low (≤ 14 mg/kg dw) and did not vary much with depth.

One core was collected from the head of Slip 3 (LDW-SC37). The peak concentration in this core (2,000 mg/kg dw) was detected at depth (2-to-4-ft interval). Two cores collected toward the mouth of Slip 3 had low arsenic concentrations (less than the 75th percentile) that varied only slightly with depth.

Other cores collected between RM 1.0 and RM 2.2 had relatively low arsenic concentrations (≤ 20 mg/kg dw) that varied little with depth, with four exceptions. In these four cores, arsenic concentrations were higher (ranging from 30 to 56 mg/kg dw). Peak concentrations were in the uppermost intervals at two of these locations (30 mg/kg dw in the 0-to-1-ft interval at LDW-SC24 and 56 mg/kg dw in the 0-to-2-ft interval at LDW-SC33) and in deeper intervals at the other two locations (40 mg/kg dw in the 1-to-2-ft interval at LDW-SC32 and 34 mg/kg dw in the 2-to-4-ft interval at LDW-SC21).

In summary, the two areas with relatively high arsenic concentrations in surface and subsurface sediment had peak concentrations at depth within cores (Map 4-28). Most of the other subsurface sediment cores had relatively low arsenic concentrations that varied only slightly with depth.

RM 2.2 to RM 3.0

Arsenic concentrations were generally low in both surface and subsurface sediment samples collected between RM 2.2 and RM 3.0 (Map 4-27c). Only one area had concentrations that exceeded the area-based 95th percentile in either surface or subsurface sediment. A core collected in this area (LDW-SC45), located just downstream from Slip 4 on the east side of the LDW, had a peak arsenic concentration of 25 mg/kg dw at depth (2-to-4-ft interval). Four other subsurface cores were collected between RM 2.2 and RM 3.0 on the east side. Arsenic was not detected in the core collected at LDW-SC43 and was detected at relatively low concentrations (≤ 20 mg/kg dw) in the other three cores (LDW-SC41, LDW-SC42, and LDW-SC44). Arsenic concentrations in these three cores varied only slightly with depth (Map 4-27c).

Four cores were collected within Slip 4 from an area that was subsequently dredged. Each of these cores was sampled at only one relatively large sampling interval (generally 4-ft intervals). Arsenic concentrations ranged from 4 to 16 mg/kg dw in these cores, similar to surface sediment concentrations in the same area.

Arsenic concentrations were all less than 10 mg/kg dw in the five cores collected in the navigation channel. Only one core (DR171) was sampled at more than one interval, and arsenic concentrations were similar in both intervals in that core.

All of the 12 cores collected on the west side of the navigation channel had arsenic concentrations ≤ 20 mg/kg dw. Three of these twelve cores were sampled at more than one interval. There were minor arsenic concentration gradients with depth in these three cores.

In summary, arsenic concentrations were ≤ 25 mg/kg dw in this section of the LDW and did not vary much with depth. In the core with the highest concentration (25 mg/kg dw), the peak concentration was in the 2-to-4-ft interval.

RM 3.0 to RM 4.0

Arsenic concentrations in surface and subsurface sediment were generally less than or equal to the 75th percentile (13 mg/kg dw) between RM 3.0 and RM 4.0, except for a few areas between RM 3.5 and RM 3.8 along the eastern shoreline (Map 4-27d). The highest surface sediment concentrations were detected in the area between RM 3.7 and RM 3.8 (up to 1,100 mg/kg dw), just south of the Boeing Plant 2/Jorgensen Forge EAA. One of the cores collected in this area (LDW-SC50) had a peak concentration in the uppermost interval (707 mg/kg dw), with decreasing concentrations at depth. The other core in this area (LDW-SC51) had a peak concentration at depth (55 mg/kg dw in the 2-to-3.8-ft interval).

In surface sediment between RM 3.4 and RM 3.7, within the Boeing Plant 2/Jorgensen Forge EAA, arsenic concentrations were generally less than the 95th percentile, with a few locations with higher concentrations, up to 57 mg/kg dw. In subsurface sediment, concentrations were generally ≤ 21 mg/kg dw and many cores were sampled at only

one interval. Three cores (SD-DUW53, SD-312-C, and SD-323-C) had peak concentrations higher than 21 mg/kg dw (31 mg/kg dw in the 8-to-12-ft interval, 26.7 mg/kg dw in the 1-to-2-ft interval, and 22.8 mg/kg dw in the 1-to-2-ft interval, respectively).

On the west side of the navigation channel, all subsurface core samples had low arsenic concentrations (≤ 11 mg/kg dw), with the exception of one sample (SB-5) collected near RM 3.3. The core at this location had a peak concentration of 22 mg/kg dw at an interval of 5 to 7.5 ft, compared to somewhat lower concentrations (8 and 8.5 mg/kg dw) in shallower intervals.

Arsenic concentrations in the navigation channel were all less than the 75th percentile, and most of the cores were sampled at only one interval. In the two cores sampled at more than one interval, concentrations varied only slightly with depth.

In summary, this section of the LDW generally had low arsenic concentrations in surface and subsurface sediment, and concentrations did not vary much with depth. The highest subsurface concentrations were in three cores on the east side of the waterway; two of these occurred in areas where surface sediments were also greater than the area-based 95th percentile concentration, and two further south were at concentrations between the area-based 75th and 95th percentile concentrations. In the core with highest arsenic concentration (707 mg/kg dw) near RM 3.8, the peak concentration was in the uppermost interval collected from 0 to 1 ft.

RM 4.0 to RM 5.0

Arsenic concentrations were all ≤ 20 mg/kg dw in both surface and subsurface sediment between RM 4.0 and RM 5.0, with the exception of concentrations at two surface sediment locations, one in Slip 6 (26.2 mg/kg dw) and the other near RM 5.0 (51 mg/kg dw) (Map 4-27e).

Surface and subsurface sediment concentrations in Slip 6 were between the 75th and 95th percentile concentrations. Of the 11 cores collected in Slip 6, only two had more than one interval sampled. Subsurface sediment concentrations in cores collected downstream from Slip 6 on the eastern shoreline were less than or equal to the 50th percentile concentration.

Near RM 5.0, all four subsurface cores had arsenic concentrations less than the 50th percentile. No core was collected near the area with a high concentration of 51 mg/kg dw in surface sediment; therefore, concentration gradients with depth could not be evaluated at this location.

At all other locations between RM 4.0 and RM 5.0 (i.e., in the navigation channel and to the west of the navigation channel), subsurface concentrations were less than the 75th percentile, and, in general, only one interval was sampled. Also, many of the intervals were greater than 4 ft deep, so concentration gradients with depth could not be evaluated.

In summary, most of the surface and subsurface sediment arsenic concentrations in this section of the LDW were less than the 95th percentile of 24 mg/kg dw. In the few cores that had more than one sampling interval, concentrations were ≤ 20 mg/kg dw and varied little with depth.

4.2.2.4 Tissue

Total arsenic was detected in all tissue samples in which it was analyzed. In whole-body fish, the mean total arsenic concentration was higher in English sole (3.274 mg/kg ww) than in other species (0.738 to 0.99 mg/kg ww) (Table 4-21 and Figure 4-12). Likewise, in fillet samples, the mean concentration was higher in English sole (5.255 mg/kg ww with skin and 11 mg/kg ww without skin) compared to other species (0.274 to 0.899 mg/kg ww with skin), although there were only three fillet samples for species other than English sole.

Table 4-21. Summary of LDW-wide tissue data for total arsenic

Tissue Type	Detection Frequency		Total Arsenic Concentration (mg/kg ww)		
	Ratio	%	Minimum Detect	Maximum Detect	Mean
Fish, whole body					
English sole	21/21	100	2.230	4.330	3.274
Pacific staghorn sculpin	24/24	100	0.364	1.430	0.738
Shiner surfperch	27/27	100	0.715	1.4	0.99
Starry flounder	3/3	100	0.793	0.973	0.887
Fish, fillet					
English sole (with skin)	7/7	100	3.965	6.890	5.255
English sole (without skin)	9/9	100	9.0	15	11
Pile perch (with skin)	1/1	100	0.563	0.563	0.563
Starry flounder (with skin)	1/1	100	0.899	0.899	0.899
Striped perch (with skin)	1/1	100	0.274	0.274	0.274
Crab, edible meat					
Dungeness crab	9/9	100	2.540	13	5.5
Slender crab	12/12	100	1.670	3.570	2.612
Crab, hepatopancreas					
Dungeness crab	4/4	100	3.080	7.0	4.6
Slender crab	4/4	100	2.230	3.310	2.645
Crab, whole body (calc'd)[a]					
Dungeness crab	9/9	100	2.707 M	11.0 M	5.2
Slender crab	12/12	100	1.844 M	3.489 M	2.622
Invertebrates, whole body					
Amphipod	4/4	100	0.96	1.5	1.2
Benthic invertebrates	20/20	100	0.573	17.40	2.34

Tissue Type	Detection Frequency		Total Arsenic Concentration (mg/kg ww)		
	Ratio	%	Minimum Detect	Maximum Detect	Mean
Shellfish					
Mussels, wild (edible meat)	22/22	100	0.34	1.1 J	0.81
Mussels, transplanted (edible meat)	35/35	100	0.573	1.42	0.791
Soft-shell clam (whole body)	14/14	100	1.300 J	5.870 J	3.073

[a] Data from composite hepatopancreas samples were mathematically combined with data from composite samples of edible meat to form composite samples of edible meat plus hepatopancreas. Arsenic concentrations in whole-body (i.e., edible meat plus hepatopancreas) crab were calculated assuming 69% (by weight) edible meat and 31% hepatopancreas, based on the relative weights of these tissues in a 16.6-cm Dungeness crab dissected by Windward in 2004 (unpublished data).

J – estimated concentration

LDW –Lower Duwamish Waterway

M – calculated whole-body concentration

na – not applicable

ww –wet weight

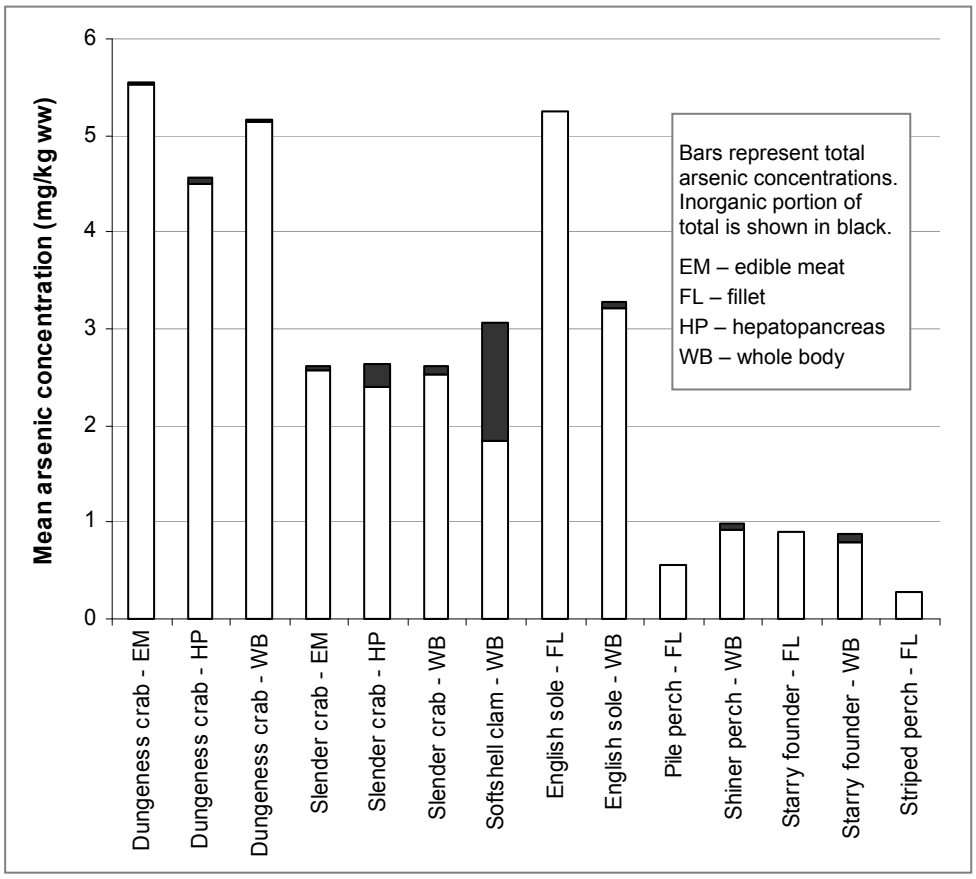


Figure 4-12. Inorganic arsenic concentrations as a percentage of total arsenic concentrations in tissue

Mean total arsenic concentrations were higher in Dungeness crabs than in slender crabs (5.2 and 2.645 mg/kg ww, respectively, in calculated whole-body samples) (Table 4-21). Mean total arsenic concentrations were lower in mussels (0.81 to 0.791 mg/kg dw) than in clams (3.073 mg/kg ww).

The highest sample-specific total arsenic concentration in any tissue sample (17.4 mg/kg ww) was in a benthic invertebrate composite sample (B3b) collected from Slip 1 (Map 4-29). The co-located sediment sample from this location had one of the highest arsenic concentrations (725 mg/kg dw) in surface sediment samples collected from the LDW. Slip 1 was also the location of a sediment core (LDW SC17) with relatively high arsenic concentrations at all intervals, to a depth of 8.5 feet.

Inorganic arsenic is the most toxic form of arsenic to humans and is a known human carcinogen, so a subset of tissue types consumed by humans (i.e., fish, crabs, and clams) was also analyzed for inorganic arsenic as part of the RI (Table 4-22). The concentration patterns of inorganic arsenic were different than those of total arsenic. The highest mean inorganic arsenic concentration (1.24 mg/kg ww) was detected in soft-shell clams. Mean inorganic arsenic concentrations in all other species were much lower, ranging from 0.002 to 0.24 mg/kg ww.

Table 4-22. Summary of LDW-wide tissue data for inorganic arsenic

Tissue Type	Detection Frequency		Inorganic Arsenic Concentration (mg/kg ww)			
	Ratio	%	Minimum Detect	Maximum Detect	Calculated Mean[a]	RL or Range of RLs[b]
Fish, whole body						
English sole	7/7	100	0.020	0.090	0.051	na
Shiner surfperch	8/8	100	0.020	0.160	0.070	na
Starry flounder	1/1	100	0.090	0.090	0.090	na
Fish, fillet						
English sole (with skin)	6/7	86	0.003	0.006 J	0.004	0.003
Pile perch (with skin)	0/1	0	nd	nd	0.0050	0.010
Starry flounder (with skin)	0/1	0	nd	nd	0.002	0.003
Striped perch (with skin)	0/1	0	nd	nd	0.0050	0.010
Crab, edible meat						
Dungeness crab	2/2	100	0.010	0.010	0.010	na
Slender crab	4/4	100	0.030	0.030	0.030	na
Crab, hepatopancreas						
Dungeness crab	2/2	100	0.050	0.090	0.070	na
Slender crab	4/4	100	0.080	0.330	0.24	na
Crab, whole body (calc'd)[c]						
Dungeness crab	nc	nc	0.022 JM	0.035 M	0.029	na
Slender crab	nc	nc	0.046 M	0.123 M	0.098	na
Shellfish, whole body						

Tissue Type	Detection Frequency		Inorganic Arsenic Concentration (mg/kg ww)			
	Ratio	%	Minimum Detect	Maximum Detect	Calculated Mean[a]	RL or Range of RLs[b]
Soft-shell clam	8/8	100	0.132	3.27	1.24	na

[a] Calculated mean concentration is the average of detected concentrations and one-half the RL for non-detected results.

[b] RLs are based only on non-detect samples.

[c] Data from composite hepatopancreas samples were mathematically combined with data from composite samples of edible meat to form composite samples of edible meat plus hepatopancreas. Inorganic arsenic concentrations in whole-body (i.e., edible meat plus hepatopancreas) crab were calculated assuming 69% (by weight) edible meat and 31% hepatopancreas, based on the relative weights of these tissues in a 16.6-cm Dungeness crab dissected by Windward in 2004 (unpublished data). Detection frequencies were not calculated for these samples because they do not represent individually analyzed samples.

J – estimated concentration

M – calculated whole-body concentration

na – not applicable

nc – not calculated

nd – not detected

RL – reporting limit

ww –wet weight

The amount of inorganic arsenic in fish and shellfish tissue as a percentage of total arsenic is presented in Table 4-23 and shown in Figure 4-12. Clam tissue had the highest percentage of inorganic arsenic, with a mean of 40% in the 8 samples analyzed for inorganic arsenic and a range of 10% to 68% in individual samples. The ratio of inorganic to total arsenic was much lower in crab and fish tissues, typically less than 5% and always less than 15%.

Table 4-23. Inorganic arsenic as a percentage of total arsenic in tissue samples

Tissue Type	Number of Samples	Inorganic Arsenic as a Percentage of Total Arsenic		
		Mean (%)	Minimum (%)	Maximum (%)
Fish, whole body				
English sole	7	2	1	3
Shiner surfperch	8	7	2	13
Starry flounder	1	9	9	9
Fish, fillet				
English sole	7	0	0	0
Pile perch	1	1	1	1
Starry flounder	1	0	0	0
Striped perch	1	2	2	2
Crab, edible meat				
Dungeness crab	2	0	0	0
Slender crab	4	1	1	1
Crab, hepatopancreas				

Tissue Type	Number of Samples	Inorganic Arsenic as a Percentage of Total Arsenic		
		Mean (%)	Minimum (%)	Maximum (%)
Dungeness crab	2	2	1	3
Slender crab	4	10	2	15
Crab, whole body (calc'd)[a]				
Dungeness crab	2	1	1	1
Slender crab	4	4	1	5
Shellfish, whole body				
Soft-shell clam	8	40	10	68

[a] Data from composite hepatopancreas samples were mathematically combined with data from composite samples of edible meat to form composite samples of edible meat plus hepatopancreas.

For comparison, the fractions of inorganic arsenic in tissue collected from Puget Sound in other studies were well below 10%. The mean inorganic arsenic fraction for 16 fish and crab samples collected throughout Puget Sound was 0.29% (Ecology 2002a). The inorganic arsenic fractions calculated from the same study from 13 clam samples was 0.77%, much lower than the percentages found in the RI samples.

Mean total arsenic concentrations in English sole and Pacific staghorn sculpin were similar among areas but were slightly higher in Area T1 (Map 4-29). Total arsenic concentrations in shiner surfperch tissue were similar among areas. In crab tissue, the highest total arsenic concentration was in Area T4, but only one Dungeness crab sample was collected in this area. Excluding this sample, the highest mean total arsenic concentrations in crab were in samples from Area T1.

Map 4-29 shows arsenic concentrations in benthic invertebrate and clam tissue samples at each location, along with concentrations in co-located sediment samples. The sample locations were selected to represent areas that covered the range of arsenic concentrations in sediment throughout the LDW so that a relationship between chemical concentrations in sediment and tissue could be evaluated (Windward 2004e). Regression analyses were conducted to evaluate this relationship for arsenic concentrations in sediment and tissue (both benthic invertebrates and clams). [60]

The regression relationship for sediment and benthic invertebrates was used in the ERA to estimate total arsenic concentrations in tissue throughout the LDW using the RI baseline dataset; actual tissue data were available from 20 locations. Log_{10}-transformed total arsenic concentrations in benthic invertebrate tissues were regressed against log_{10}-transformed total arsenic concentrations in sediments. Although the variance in tissue increased with sediment concentration (i.e., all of the largest residuals were found at sediment concentrations greater than 8 µg/kg dw), residuals

[60] Dry weight concentrations were used for benthic invertebrate tissue to be consistent with the ERA. Wet weight concentrations were used for clam tissue to be consistent with the HHRA; clam consumption was not a primary exposure pathway in the ERA.

from the regression were approximately normally distributed (R^2 of residuals against expected normal scores = 0.94), the regression was significant ($p < 0.0005$), and 78% of the variance in tissue was explained by the regression ($R^2 = 0.78$). (Figure 4-13).

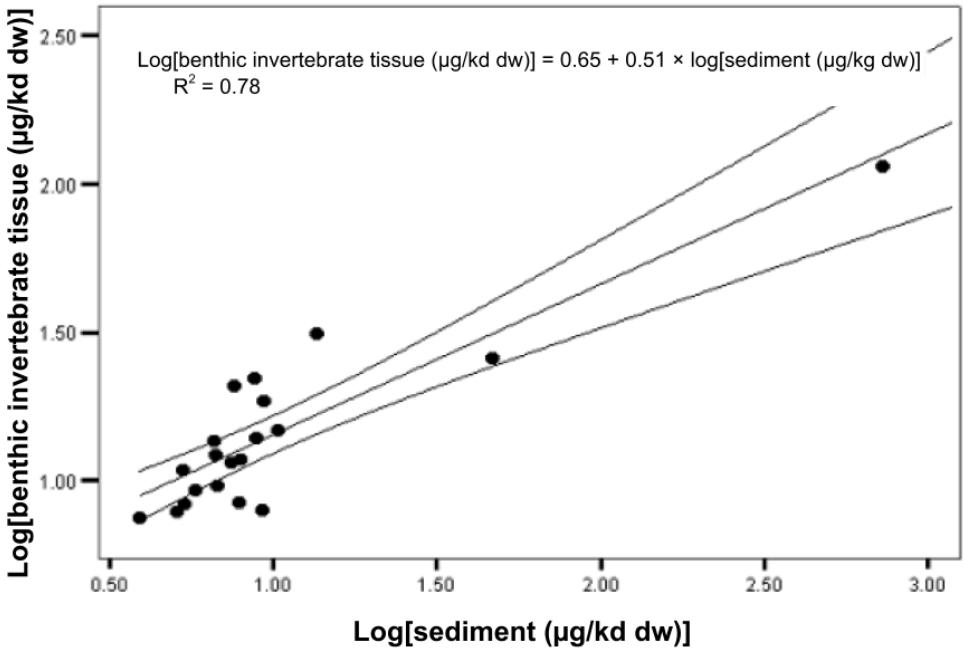


Figure 4-13. Regression equation for log_{10}-transformed total arsenic concentrations in co-located sediment and benthic invertebrate tissue samples, including 95% confidence intervals

The highest arsenic sediment concentration (725 mg/kg dw at location B3b in Slip 1) was co-located with the highest benthic invertebrate tissue concentration (17.4 mg/kg ww). This sediment-tissue pair could be considered an outlier in terms of its sediment concentration (Leverage value = 0.76), but it was not an outlier with respect to its tissue concentration (Studentized deleted residual = 0.58, p = 0.57). It did not have a significant effect on the estimated tissue concentration at its sediment concentration (DFFit = 0.151), the estimate of the intercept (DFbeta0 = 0.067), the estimate of the slope (DFbeta1 = 0.076), or the regression relationship in general (Cook's Distance = 0.757, p = 0.395). These results indicate that the relationship between tissue and sediment is consistent across the range of sediment concentrations that was sampled and the regression was considered a reliable means for estimating total arsenic concentrations in benthic invertebrate tissue throughout the LDW.

For clams, inorganic arsenic in tissue rather than total arsenic was more relevant for the regression because arsenic is a risk driver for human health based on the toxicity of inorganic arsenic in seafood. When total arsenic in sediment was log_{10}-transformed, the regression with inorganic arsenic in clam tissues was significant ($p = 0.005$) and explained 76% of the variance in tissue concentrations of inorganic arsenic ($R^2 = 0.76$).

However, the relationship was weak because one sediment-tissue pair had a large effect on the relationship. The relationship between total arsenic concentrations in sediment and total arsenic concentrations in clam tissue was also evaluated to determine if the relationship was better described using total arsenic rather than inorganic arsenic in tissue. When both sediment and tissue concentrations were $\log_{10}$-transformed, the regression was significant ($p < 0.006$) but explained only 48% of the variance ($R^2 = 0.48$) and was also strongly affected by a single data pair. The regression relationship for total arsenic in sediment and inorganic arsenic in clams is discussed further in Section 8.3.2.

4.2.2.5 Surface water

Dissolved and total arsenic were analyzed in hundreds of water samples collected from October 1996 to June 1997 at three LDW locations as part of the King County WQA (King County 1999e). Samples at these three locations were collected at two depths: 1 m below the surface and 1 m above the sediment. Mean dissolved and total arsenic concentrations in individual samples ranged from 0.175 to 1.57 µg/L (Table 4-24). Mean dissolved and total arsenic concentrations were slightly higher at the downstream end of the LDW (at RM 1.1 and RM 1.9) than at the upstream end at RM 4.9. All concentrations were well below the Washington State acute and chronic marine water quality standards of 69 and 36 µg/L, respectively.

Table 4-24. Summary of surface water data for arsenic

Location ID	Detection Frequency		Concentration (µg/L)		
	Ratio	%	Minimum Detect	Maximum Detect	Mean
Dissolved arsenic					
BRN (RM 1.1)	72/72	100	0.237	1.46	0.855
SWM (RM 1.9)	72/72	100	0.224	1.42	0.806
NFK (RM 4.9)	24/24	100	0.175	0.456	0.319
Total arsenic					
BRN (RM 1.1)	167/167	100	0.292	1.53	0.931
SWM (RM 1.9)	167/167	100	0.282	1.57	0.880
NFK (RM 4.9)	56/56	100	0.183	0.868	0.493

BRN – LDW near Brandon CSO
LDW – Lower Duwamish Waterway
NFK – LDW near Norfolk CSO
RM – river mile
SWM – LDW near SW Michigan CSO

4.2.2.6 Seeps

Arsenic was analyzed in 17 seep water samples collected from the Boeing Plant 2 site in 1995, in 7 seep water samples collected from the Rhône-Poulenc site in 1995, in 1 seep water sample from the Boeing Isaacson site in 2000, in 3 seep water samples from T-117 in 2003, and in 16 seep water samples from locations throughout the LDW

during the RI sampling event in 2004 (Table 4-25). The RI locations were selected based on their proximity to potential sources in consultation with EPA and Ecology.

Table 4-25. Concentrations of arsenic in LDW-wide seep water samples

Chemical	Detection Frequency		Concentration (µg/L)			
	Ratio	%	Minimum Detect	Maximum Detect	Calculated Mean[a]	RL or Range of RLs[b]
Boeing Plant 2						
Dissolved arsenic	10/10	100	6	10	9	na
Total arsenic	16/17	94	2	20	9	5
Rhône-Poulenc						
Total arsenic	1/7	14	31.0	31.0	nc	nr
RI – LDW-wide						
Dissolved arsenic	16/16	100	0.054	253	22	na
Total arsenic	13/13	100	0.058	287	28	na
T-117						
Dissolved arsenic	0/3	0	nd	nd	25	50
Total arsenic	0/1	0	nd	nd	25	50
Boeing Isaacson						
Dissolved arsenic	1/1	100	7	7	7	na

[a] Calculated mean concentration is the average of detected concentrations and one-half the RL for non-detected results.

[b] RLs are based only on non-detect samples.

LDW – Lower Duwamish Waterway

na – not applicable (detected in all samples)

nc – not calculated (RLs for non-detects not reported for this sampling event)

nd – not detected

nr – not reported (RLs for non-detects not reported for this sampling event)

RL – reporting limit

At Boeing Plant 2, total arsenic concentrations in seep water were generally low, ranging from 2 to 20 µg/L; dissolved arsenic concentrations ranged from 6 to 10 µg/L. At Boeing Isaacson, the single seep sample had a dissolved arsenic concentration of 7 µg/L. These concentrations were less than the Washington State acute and chronic marine water quality standards of 69 and 36 µg/L, respectively.[61]

At Rhône-Poulenc, total arsenic was detected in one of the seven seep water samples at a concentration of 31 µg/L; RLs were not available for the samples with non-detects. At T-117, arsenic was not detected in seep water samples at an RL of 50 µg/L.

[61] Seep data were compared to water quality standards for information purposes only because these standards are not intended for the type of exposure conditions generally present in intertidal seeps. The comparison to chronic water quality standards is conservative because the data were collected at only one point in time and may not accurately reflect chronic conditions.

During the RI sampling event in 2004, dissolved arsenic concentrations were greater than the Washington State acute marine water quality standard of 69 µg/L in two seeps (72.4 µg/L at location SP-61 [RM 1.5 W] and 253 µg/L at location SP-76 [Slip 1 at RM 1.0 E]) (Map 4-11). The sampling locations with the highest arsenic concentrations in seep water were near locations with high arsenic concentrations in surface sediment (725 mg/kg dw at B3b in Slip 1 and 161 mg/kg dw at LDW-SS56 near RM 1.4 W). It is not known if these water quality standards would have been exceeded after the dilution of seep water in receiving water of the LDW. At the other 14 seep locations sampled for the RI, arsenic concentrations (total and dissolved) were less than 7 µg/L.

4.2.2.7 Porewater

Total arsenic was analyzed in 16 porewater samples collected from 14 locations at the Rhône-Poulenc site in 2004 and in 15 porewater samples collected throughout the LDW during the EPA SI (Weston 1999a). Arsenic was not detected in any of the Rhône-Poulenc samples, but the RL (450 µg/L) was higher than the Washington State acute marine water quality standard of 69 µg/L. Total arsenic was detected in 12 of the 15 porewater samples collected during the EPA SI at concentrations ranging from 26 to 114 µg/L, with a mean concentration of 50 µg/L. Some of these concentrations exceeded the Washington State acute marine water quality standard of 69 µg/L; this comparison is conservative because these samples were unfiltered.

4.2.2.8 Summary of arsenic data

Arsenic concentrations in surface sediment ranged from 1.2 mg/kg dw to 1,100 mg/kg dw; the 95th and 50th percentile concentrations calculated from the RI baseline dataset were 30 and 15 mg/kg dw, respectively. The highest concentration (1,100 mg/kg dw) was detected between RM 3.7 and RM 3.8 on the east side of the navigation channel. Surface sediment samples collected from several other areas, including from RM 0.4 to RM 0.7 on the east side and from RM 1.3 to RM 1.5 on the west side, had high arsenic concentrations (i.e., greater than the 95th percentile).

Areas with relatively high arsenic concentrations in surface sediment generally also had high concentrations in subsurface sediment. Specific areas with high subsurface concentrations were: 1) Slip 3 (2,000 mg/kg dw), 2) between RM 1.3 and RM 1.4 (1,890 mg/kg dw), 3) between RM 3.7 and RM 3.8 on the eastern shoreline (707 mg/kg dw), 4) Slip 1 (170 mg/kg dw), and 5) near RM 0.1 along the eastern shoreline (270 mg/kg dw). In all of these areas, peak arsenic concentrations were detected at depth within the cores, except for the core collected between RM 3.7 and RM 3.8, which had a peak concentration in the uppermost interval of the core.

In tissue, mean total arsenic concentrations were highest in Dungeness crab tissue (5.5 mg/kg ww in edible meat and 5.2 mg/kg ww in whole-body samples) and in English sole fillet tissue (5.255 mg/kg ww with skin and 11 mg/kg ww without skin). Soft-shell clams had the highest mean inorganic arsenic concentration (1.24 mg/kg ww); mean inorganic arsenic concentrations in other tissue types were ≤ 0.24 mg/kg ww.

Regression analyses showed a significant and strong positive relationship between log-transformed total arsenic concentrations in benthic invertebrate tissue and co-located samples ($R^2 = 0.78$, $p<0.0005$). The relationship between the inorganic arsenic concentrations in clam tissue and the co-located log-transformed total arsenic concentrations in sediment was also significant ($R^2 = 0.76$, $p = 0.059$), but was strongly influenced by a single sediment-tissue pair, as discussed further in Section 8.3.2.

In surface water, mean dissolved and total arsenic concentrations were higher at the two downstream locations (RM 1.1 and RM 1.9), with concentrations ranging from 0.806 to 0.931 µg/L, than at the upstream location at RM 4.9 (with concentrations ranging from 0.319 to 0.493 µg/L). All concentrations were well below Washington State marine water quality criteria.

The highest dissolved arsenic concentrations in seep water (72.4 and 253 µg/L) were detected near RM 1.5 on the west side and in Slip 1, respectively. These areas also had high arsenic concentrations in surface sediment. Total arsenic concentrations in porewater samples collected from 15 locations throughout the LDW ranged from 26 to 114 µg/L.

4.2.3 cPAHs

This section summarizes the nature and extent of cPAH contamination in surface sediment, subsurface sediment, and tissue. Total cPAH concentrations in sediment and tissue were calculated based on concentrations of seven individual PAHs (benzo(a)pyrene, benzo(b)fluoranthene, benzo(a)anthracene, benzo(k)fluoranthene, indeno(1,2,3-cd)pyrene, dibenz(a,h)anthracene, and chrysene) and their toxicity relative to benzo(a)pyrene, using PEFs from California EPA (California EPA 1994) as described in Section 4.1.4 and in Appendix E, Section E3. cPAH concentrations in surface water, seep water, and porewater were not calculated because these media types are not primary exposure pathways for humans, and cPAHs were identified as a primary risk driver chemical only for human health seafood consumption and direct sediment contact pathways. Data for individual and total PAHs are summarized in Section 4.2.6.

4.2.3.1 Surface sediment

Surface sediment samples from 828 of the 1,365 locations in the RI baseline dataset were analyzed for at least one individual cPAH compound. Most of the 828 samples were analyzed for all cPAH compounds; samples from 17 locations were analyzed for only a subset of the individual cPAH compounds.[62] At least one cPAH compound was detected in 94% of sediment samples analyzed for PAHs, with cPAH concentrations ranging from 9.7 to 11,000 µg/kg dw (Table 4-26). The numerical 95th percentile based on concentrations in the surface sediment dataset was 1,500 µg/kg dw (Map 4-30). The 75th, 50th, and 25th percentiles were 540, 260, and 102 µg/kg dw, respectively. Area-

[62] The cPAHs in the total cPAH calculation always included at least five cPAH compounds.

based 95th, 75th, 50th, and 25th percentiles were 1,100, 440, 230, and 97 µg/kg dw, respectively (Map 4-31). The highest RLs (ranging from 65 to 130 µg/kg dw) were from samples collected in 1995. More sensitive analytical methods (e.g., EPA 8270-SIM) were used for recent data collection efforts, achieving RLs of 45 µg/kg dw or lower. The wide range of RLs in the surface sediment dataset for cPAHs is attributed to the use of different analytical methods (EPA 8270 full-scan or EPA 8270-SIM) and required analytical dilutions for some samples.

Table 4-26. Summary of surface sediment data for cPAHs

Chemical	Detection Frequency		Concentration (µg/kg dw)[a]			
	Ratio	%	Minimum Detect	Maximum Detect	Calculated Mean[b]	RL or Range of RLs[c]
cPAHs	780/828	94	9.7 J	11,000	500	9.0 – 130

[a] Total cPAHs were calculated by summing the products of concentrations and compound-specific PEFs for individual cPAH compounds, as discussed in detail in Appendix E, Section E.3.

[b] Calculated mean concentration is the average of detected concentrations and one-half the RL for non-detected results.

[c] RLs based only on non-detect samples.

cPAH – carcinogenic polycyclic aromatic hydrocarbon

dw – dry weight

J – estimated concentration

RL – reporting limit

The maximum cPAH concentration was detected at a location near RM 3.6 in the T-117 EAA (T117-SE-37-G). Other areas with the highest cPAH concentrations (greater than both the numerical and area-based 95th percentiles) were: 1) in the Slip 4 EAA (7,100 and 5,100 µg/kg dw at C8 and DR178, respectively), 2) in the RM 3.8 EAA on the eastern shoreline (6,600 and 3,500 µg/kg dw at R23 and R22, respectively), 3) near RM 0.6 on the western shoreline (4,200 µg/kg dw at LDW-SS312), 4) at RM 0.0 near Harbor Island (2,200 µg/kg dw at LDW-SS301), and 5) in the Duwamish/Diagonal EAA (2,900 µg/kg dw at DUD005). Other sample locations or areas with cPAH concentrations greater than the numerical or area-based 95th percentiles are shown on Maps 4-30 and 4-31.

Forty-four locations were resampled and analyzed for cPAHs between 2004 and 2006 at locations within 10 ft of the original samples;[63] most of the original samples were collected in 1997 or 1998. Of the 44 locations, cPAH concentrations decreased in surface sediments during this 6-to-9-yr time period at 34 locations (77%) and increased at 4 locations (9%) (Table 4-27 and Map 4-32). At the remaining 6 locations (14%), concentration differences were within the range of analytical variability (≤ 25% increase or decrease compared to the initial concentration) (Table 4-27; Map 4-32). The highest increase in cPAH concentration (1,830 µg/kg dw) was located in Slip 6.

[63] The older data within 10 ft of the newer sample were excluded from the baseline dataset, as described in Section 4.1.2.1.

Table 4-27. Difference in chemical concentrations of cPAHs at resampled locations sorted by concentration difference

River Mile	Sampling Location ID		Sampling Date		Concentration (µg/kg dw)[a]		
	Initial	Resampled	Initial	Resampled	Initial	Resampled	Difference
4.2	R40	LDW-SS127	10/13/1997	1/20/2005	31,000	640	-30,360
4.2	R42	LDW-SS129	10/13/1997	1/20/2005	8,600	860	-7,740
4.2	R45	LDW-SS130	10/16/1997	1/20/2005	4,800	370	-4,430
3.8	DR187	LDW-SS115	8/27/1998	1/25/2005	5,600	2,400	-3,200
2.8	DR175	LDW-SS94	8/20/1998	1/21/2005	2,000	100	-1,900
1.0	DR019	LDW-SS32	8/17/1998	1/18/2005	2,100	340	-1,760
1.0	DR020	LDW-SS31	8/17/1998	1/21/2005	1,900	600	-1,300
1.3	DR053	LDW-SS44	8/31/1998	1/21/2005	1,700 J	670	-1,030
1.0	DR087	LDW-SS37	8/12/1998	1/18/2005	1,200	210	-990
1.7	DR097	LDW-SS63	8/20/1998	1/21/2005	1,000	190	-810
1.3	SS-2	LDW-SS48	8/17/1993	1/18/2005	2,160 J	1,400	-760
0.1	K-07	LDW-SS4	9/30/1991	1/17/2005	1,000 J	270	-730
1.4	SS-3	LDW-SS49	8/17/1993	1/26/2005	1,080	400	-680
0.2	DR035	LDW-SS12	8/11/1998	1/17/2005	840	200	-640
0.3	DUD042	LDW-SS17	11/11/1995	1/24/2005	1,080 J	440	-640
1.4	DR065	LDW-SS52	8/17/1998	1/25/2005	700	160	-540
1.5	DR123	LDW-SS57	9/14/1998	1/24/2005	770	350 J	-420
0	K-11	LDW-SS1	9/30/1991	1/17/2005	530 J	130	-400
2.1	DR106	LDW-SS76	8/19/1998	1/20/2005	510	110	-400
1.4	DR160	LDW-SS51	8/12/1998	1/18/2005	540	170	-370
1.4	SS-4	LDW-SS55	8/17/1993	1/24/2005	559	190	-369
3.7	SD-323-S	LDW-SS110	8/17/2004	1/25/2005	600	250	-340
0.2	K-05	LDW-SS10	10/14/1991	1/17/2005	800 J	480	-320
0.3	DR079	LDW-SS15	8/24/1998	1/17/2005	460	140	-320
1.4	DR028	B4b	8/17/1998	8/28/2004	600	300	-300
0.9	DR021	LDW-SS319	8/17/1998	10/4/2006	830	560 J	-270
4.7	DR271	LDW-SS148	9/15/1998	3/9/2005	430	230	-200
3.9	R30	LDW-SS119	10/11/1997	1/19/2005	420	260	-160
1.2	DR088	LDW-SS40	8/31/1998	1/18/2005	230	95	-135
0.9	DR085	LDW-SSB2b	8/31/1998	3/11/2005	390	260	-130
1.9	DR131	LDW-SS70	8/13/1998	1/21/2005	500	410	**-90**
3.1	DR198	LDW-SS102	8/20/1998	1/24/2005	150	61	-89
3.2	DR202	LDW-SS104	8/27/1998	1/25/2005	130	52 J	-78
4.3	DR286	B10b	8/26/1998	8/19/2004	100	24 J	-76
2.0	R7	LDW-SS75	10/15/1997	1/21/2005	170	130	**-40**
3.8	R24	LDW-SS117	10/10/1997	1/20/2005	100	78	-22

River Mile	Sampling Location ID		Sampling Date		Concentration (µg/kg dw)[a]		
	Initial	Resampled	Initial	Resampled	Initial	Resampled	Difference
3.7	R21	LDW-SS113b	10/9/1997	1/20/2005	190	190	**0**
1.4	DR030	LDW-SS50	8/17/1998	1/24/2005	400	410	**10**
4.1	DR238	LDW-SS125	8/27/1998	1/20/2005	160	170	**10**
4.1	A11-05	LDW-SS126	8/18/1994	1/20/2005	130	180	**50**
0	DR076	LDW-SS5	8/24/1998	1/17/2005	18	89	71
2.2	DR113	LDW-SS81	8/19/1998	3/8/2005	140	270	130
3.7	DR186	LDW-SS111	8/27/1998	1/19/2005	1,200	1,900	700
4.2	DR242	SB-1	8/24/1998	8/25/2004	470 J	2,300 J	1,830

Notes: Concentrations in bold were within the range of analytical variability (≤25% increase or decrease compared to the initial concentration). All seven cPAH compounds were analyzed in all samples presented in this table.

[a] Total cPAHs were calculated by summing the products of concentrations and compound-specific PEFs for individual cPAH compounds, as discussed in detail in Appendix E, Section E.3.

cPAH – carcinogenic polycyclic aromatic hydrocarbons

J estimated concentration

4.2.3.2 Subsurface sediment

Subsurface sediment cores were sampled at various intervals during the different sampling events. To calculate summary statistics, data were categorized according to specified intervals, as described for total PCBs in Section 4.2.1.2.

cPAHs were detected in 83% of the subsurface samples in which they were analyzed, at concentrations ranging from 18 to 7,000 µg/kg dw (Table 4-28). Most of the subsurface cPAH data were collected in the top 4 ft of the sediment cores. cPAH data in the 1- and 2-ft intervals for the top 6 ft of sediment are shown on Maps 4-33a through 4-33c. Data on these maps and in Table 4-28 are discussed in detail for individual cores in the following section (Section 4.2.3.3).

Table 4-28. Summary of subsurface sediment data for cPAHs

Sampling Interval (ft)	Detection Frequency		Concentration (µg/kg dw)[a]		
	Ratio	%	Minimum Detect	Maximum Detect	RL or Range of RLs[b]
All data[c]					
Any interval	252/304	83	18	7,000	3.9 – 400
1-ft intervals[d]					
0 to 1	60/64	94	19 J	4,400	18 – 400
1 to 2	34/45	76	19 J	7,000	17 – 35
2 to 3	6/10	60	18	530	18 – 90
3 to 4	2/9	22	19 J	52 J	17 – 36
4 to 5	1/1	100	75 J	75 J	na
2-ft intervals[d]					
0 to 2	71/74	96	41.6 J	4,900	18

Sampling Interval (ft)	Detection Frequency		Concentration (µg/kg dw)[a]		
	Ratio	%	Minimum Detect	Maximum Detect	RL or Range of RLs[b]
2 to 4	56/73	77	18 J	5,600	17 – 35
4 to 6	9/10	90	54.1	1,300	48
6 to 8	5/5	100	48	4,000 J	na
8 to 10	4/5	80	93 J	290	4.89

[a] Total cPAHs were calculated by summing the products of concentrations and compound-specific PEFs for individual cPAH compounds, as discussed in detail in Appendix E, Section E.3.

[b] RLs were based only on non-detect samples.

[c] All data in the subsurface dataset described in Section 4.1.2.2 were included in this summary.

[d] For the calculation of summary statistics in this table, actual sampling intervals were rounded to the nearest foot. The data for each interval were then assigned to the 1- or 2-ft sampling category that best approximated the actual sampling interval. If there were data for two 1-ft intervals at a particular location, the concentrations were averaged to obtain a calculated concentration for the corresponding 2-ft interval. Some of the subsurface data were collected from intervals other than 1 or 2 ft; therefore, the detection frequency ratio for "any interval" may reflect data for intervals not included in these categories.

cPAH – carcinogenic polycyclic aromatic hydrocarbon
dw – dry weight
J – estimated concentration
na – not applicable
nc – not calculated
RL – reporting limit

4.2.3.3 Surface and subsurface sediment patterns for cPAHs

cPAH concentrations in surface and subsurface sediment were evaluated to determine whether peak concentrations were detected in the uppermost interval of sediment cores or at depth. The specific depths of peak concentrations within sediment cores were also evaluated to assess patterns (Maps 4-34a through 4-34e). The percentiles of cPAH concentrations in sediment discussed in this section were calculated on an area basis.[64]

RM 0.0 to RM 1.0

In this reach, cPAH concentrations in surface and subsurface sediment were variable; many surface and subsurface sediment samples from RM 0.0 to RM 1.0 had cPAH concentrations higher than the area-based 75th percentile of 440 µg/kg dw. Areas with the highest cPAH concentrations (i.e., greater than the area-based 95th percentile of 1,100 µg/kg dw) in both surface and subsurface sediment included RM 0.4 to RM 0.6 (the Duwamish/Diagonal EAA), near RM 0.1 on the west side of the navigation channel, near RM 0.5 on the western shoreline, and in Slip 1 (Map 4-34a).

Six cores were collected north of RM 0.3. Three of these cores, along the eastern shoreline and near the navigation channel, had relatively low cPAH concentrations in subsurface sediment. Two of the cores, one near Harbor Island and one near RM 0.3 on the west side, had higher cPAH concentrations, with peak values detected in the

64 Area-based percentiles were calculated using Thiessen polygons and thus account for the spatial bias in the sampling designs. The area-based percentiles were calculated as the surface sediment concentration at which a particular percentage of the LDW area had interpolated values less than or equal to that concentration (e.g., the 95th percentile is the concentration at which 95% of the LDW area has interpolated values ≤ 1,100 µg/kg dw).

uppermost interval (620 µg/kg dw at LDW-SC1 and 560 µg/kg dw at LDW-SC6). The remaining core (LDW-SC5), near RM 0.1, had a peak cPAH concentration of 1,900 µg/kg dw at depth (in the 1-to-2.2-ft interval).

In the vicinity of the Duwamish/Diagonal EAA, the highest surface and subsurface cPAH concentrations were generally located near RM 0.45. Three of the four cores with high subsurface cPAH concentrations (i.e., greater than the area-based 95th percentile) in the vicinity of the EAA were collected near RM 0.45; two of the cores in this area had peak concentrations in the uppermost interval (1,700 µg/kg dw at DR008 and 1,420 µg/kg dw at DUD256), and one core had a peak cPAH concentration at depth (1,630 µg/kg dw at DUD254). The depth of the peak in the fourth core (1,560 µg/kg dw at DUD258 near RM 0.5) is not known because only one interval (0 to 3 ft) was analyzed. The other cores collected within and close to the EAA had peak cPAH concentrations ranging from 35 to 940 µg/kg dw. Some of these peak cPAH concentrations were in the uppermost intervals, and some were at depth, with no clear spatial pattern.

In Slip 1, the core collected at the head of the slip (LDW-SC17) had a relatively high cPAH concentration (2,400 µg/kg dw) at an interval of 6 to 8.6 ft; concentrations were also high (1,800 µg/kg dw) in the uppermost interval. In a core collected near the mouth of the slip (LDW-SC16), the peak cPAH concentration (1,300 µg/kg dw) was detected at depth (in the 4-to-6-ft interval). Peak cPAH concentrations were also at depth (in the 2-to-4-ft interval) in two other cores collected at the mouth of Slip 1 (550 µg/kg dw at LDW-SC15 and 760 µg/kg dw at DR021); concentrations in the uppermost intervals of these cores were only slightly lower. The core collected just downstream from Slip 1 (LDW-SC13) had a peak concentration of 1,200 µg/kg dw in the 2-to-4-ft interval.

At the remaining locations between RM 0.0 and RM 1.0,[65] some of the peak cPAH concentrations were at depth, and some were in the uppermost interval; there were no notable concentration gradients within these cores. Peak cPAH concentrations in these cores ranged from 140 to 590 µg/kg dw, except for one core, located between RM 0.5 and RM 0.6 on the western shoreline, which had a peak cPAH concentration of 4,400 µg/kg dw in the uppermost interval (0 to 0.8 ft). The deeper intervals of this core did not have detectable cPAH concentrations.

In summary, cPAH concentrations in surface and subsurface sediment in this section of the LDW were variable; many subsurface cores had cPAH concentrations greater than the area-based 75th percentile (Map 4-35). In Slip 1, high cPAH concentrations were detected primarily at depth within the cores. At other locations with high cPAH

[65] The remaining cores were located at RM 0.35 on the western shoreline (one core), near RM 0.5 on the western shoreline (one core), in the channel (three cores), north of Kellogg Island (two cores), and south of Kellogg Island (one core).

concentrations (i.e., greater than the area-based 95th percentile), peak concentrations were in the uppermost intervals at four locations and at depth at two locations.

RM 1.0 to RM 2.2

Concentrations of cPAHs were variable in both surface and subsurface sediment in this section of the LDW. Along the eastern shoreline between RM 1.0 and RM 1.5, subsurface cPAH concentrations were relatively low (≤ 610 µg/kg dw) in cores that were sectioned into more than one subsurface interval, with the exception of location LDW-SC23, which had a high cPAH concentration (4,600 µg/kg dw) in the 3-to-3.5-ft interval (Map 4-34b).

Along the western shoreline between RM 1.0 and RM 1.3, two cores were collected; peak cPAH concentrations (500 µg/kg dw in both cores) were detected at depth (2 to 4 ft) in the core near RM 1.0 and in the uppermost interval in the core near RM 1.2.

Four cores collected between RM 1.3 and RM 1.4 on the west side of the navigation channel had high cPAH concentrations at depth (Map 4-34b). In two of these cores (LDW-SC25 and DR054), the peak cPAH concentrations (980 and 2,000 mg/kg dw, respectively) were in the 2-to-4-ft interval. In the other two cores (LDW-SC26 and LDW-SC28), the peak cPAH concentrations (4,000 and 1,400 µg/kg dw, respectively) were within the approximate interval of 6 to 8 ft. Between RM 1.4 and RM 1.5, four cores were sampled at only one interval each, with concentrations ranging from 280 to 620 µg/kg dw; this area was dredged after the cores were collected. In the only other core collected between RM 1.3 and RM 1.5 (LDW-SC29), cPAH concentrations were relatively low (≤ 150 µg/kg dw).

Between RM 1.5 and RM 1.7 on the east side of the navigation channel in an area that has been dredged, cPAH concentrations in subsurface sediment were generally less than the 75th percentile, and most locations were sampled at only one relatively large interval (generally greater than 3 ft). Therefore, peak cPAH concentrations could not be determined at these locations. Similarly, in the navigation channel, subsurface cPAH concentrations were less than the 75th percentile, and samples were collected at only one relatively large interval of 4 ft.

The subsurface sample with the highest cPAH concentration in the entire LDW (7,000 µg/kg dw) was collected from the head of Slip 3, in the 1-to-2-ft interval. The concentration in the uppermost interval (0 to 1 ft) was lower (2,800 µg/kg dw), as were surface sediment concentrations based on Thiessen polygons (≤ 1,100 µg/kg dw).

cPAH concentrations in the remaining nine core locations on the east and west sides of the navigation channel between RM 1.7 and RM 2.2 were ≤ 750 µg/kg dw. Peak concentrations were at depth, except at locations LDW-SC39 and LDW-SC36, which had relatively low peak concentrations (260 and 110 µg/kg dw, respectively) in the uppermost interval.

In summary, in this section of the LDW, peak cPAH concentrations were detected at depth at all of the locations that had high subsurface cPAH concentrations (i.e., greater

than the 95th percentile), and at most of the locations that had cPAH concentrations greater than the 75th percentile (Map 4-35).

RM 2.2 to RM 3.0

In the section of the LDW between RM 2.2 and RM 3.0, cPAH concentrations were generally variable, with concentrations between the 25th and 95th percentile in both surface and subsurface sediment (Map 4-34c). The highest concentrations in both surface and subsurface sediment were located in Slip 4 and near RM 2.7 on the west shoreline.

In Slip 4, subsurface cPAH concentrations ranged from 280 to 3,400 µg/kg dw in four cores sampled over relatively large intervals (ranging from the surface to approximately 3 to 4 ft); thus, the depth of peak cPAH concentrations could not be determined. This area was subsequently dredged. Just downstream from Slip 4, a peak cPAH concentration of 1,000 µg/kg dw was detected at depth (in the 2-to-4-ft interval of core LDW-SC45).

Four cores were collected on the east side of the navigation channel between RM 2.2 and RM 2.7. cPAH concentrations were very low in one core (≤ 20 µg/kg dw at LDW-SC43). In the other three cores, peak cPAH concentrations ranged from 120 to 550 µg/kg dw; peak cPAH concentrations were detected at depth within each core.

In five cores collected within the navigation channel, cPAH concentrations were ≤ 290 µg/kg dw. Samples were collected from one interval in four of these cores. In the remaining core (DR171), cPAH concentrations were similar (250 to 290 µg/kg dw) in the uppermost intervals and at depth.

On the west side of the navigation channel, the highest subsurface cPAH concentrations were detected between RM 2.6 and RM 2.8. In four of the five cores collected in this area, only one interval (approximately 0 to 3 ft) was sampled in each core; cPAH concentrations in these cores ranged from 340 to 2,800 µg/kg dw. The area was dredged after these cores were collected. The remaining core (LDW-SC46) was collected after dredging. The peak cPAH concentration (1,200 µg/kg dw) at this location was detected at depth (in the 1-to-2-ft interval). The cPAH concentration was also relatively high in the uppermost interval (840 µg/kg dw in the 0-to-1-ft interval).

The remaining five cores on the west side of the navigation channel between RM 2.2 and RM 2.6 had subsurface cPAH concentrations of ≤ 570 µg/kg dw. Only two of these cores were sampled at more than one interval. The peak cPAH concentration was at depth in core DR137 (290 µg/kg dw in the 2-to-4-ft interval) and in the uppermost interval in core LDW-SC40 (51 µg/kg dw in the 0-to-1.3-ft interval).

In summary, all of the cores sampled at more than one interval and with cPAH concentrations ≥ 290 µg/kg dw had peak cPAH concentrations at depth (Map 4-35). The location of the peak cPAH concentration is unknown in many cores collected between RM 2.2 and RM 3.0 because they were sampled at only one interval.

RM 3.0 to RM 4.0

In general, cPAH concentrations in surface and subsurface sediment between RM 3.0 and RM 4.0 were less than or equal to the area-based 50th percentile (227 µg/kg dw) (Map 4-34d). Subsurface cPAH concentrations were ≤ 460 µg/kg dw at all locations except two. On the eastern shoreline, peak concentrations were in the uppermost interval near RM 3.3 (1,500 µg/kg dw at SD-04107) and near RM 3.8 (2,200 µg/kg dw at LDW-SC51). At the other six locations on the east side of the navigation channel, subsurface cPAH concentrations ranged from 18 to 460 µg/kg dw. Some of the peak cPAH concentrations were at depth, and some were in the uppermost interval at these locations.

Eight cores were collected within the navigation channel between RM 3.0 and RM 4.0. Subsurface cPAH concentrations in these cores ranged from 33 to 270 µg/kg dw. The locations of peak cPAH concentrations in six of these cores are unknown because samples were collected from only one interval (either 0 to 2 ft or 0 to 4 ft). At the remaining two locations, the peak cPAH concentration of 40 µg/kg dw was detected within the uppermost interval (0 to 1 ft) of core LDW-SC48 and the peak concentration of 270 µg/kg dw was detected at depth (1-to-2-ft interval) in core LDW-SC49.

On the west side of the navigation channel, cPAH concentrations ranged from 18 to 340 µg/kg dw in cores collected from 12 locations. At two of these locations near RM 3.4, cPAHs were not detected in samples collected from the 0-to-4-ft interval. At five locations between RM 3.5 and RM 3.7 (in the T-117 EAA), there were no data for the uppermost intervals; cPAH concentrations in the samples collected at depth were ≤ 97.5 µg/kg dw. At the remaining five locations, where samples were collected at more than one interval, peak cPAH concentrations were ≤ 340 µg/kg dw and were detected at depth in four cores and in the uppermost interval in one core (Map 4-34d).

In summary, cPAH concentrations in surface and subsurface sediment were generally less than the 50th percentile in this section of the LDW. At the two locations with the highest cPAH concentrations in subsurface sediment (2,200 and 1,500 µg/kg dw), peak concentrations were in the uppermost interval within the cores. At other locations where more than one interval was sampled, some peak concentrations were at depth, and some were in the uppermost interval.

RM 4.0 to RM 5.0

Most of the cPAH concentrations in both surface and subsurface sediment between RM 4.0 and RM 5.0 were less than the 50th percentile (230 µg/kg dw), except in and around Slip 6 and upstream of RM 4.6 (Map 4-34e).

In Slip 6, surface and subsurface cPAH concentrations were generally highest toward the head of the slip. Seventeen of the 19 subsurface sediment cores within the slip and downstream from the slip on the east side of the navigation channel were sampled at relatively shallow single intervals (either 0.33 to 0.69 ft or 0.33 to 0.82 ft); therefore, insufficient data were available to evaluate the depth of peak cPAH concentrations at

these locations. cPAH concentrations in the eight shallow samples from within the slip ranged from 630 µg/kg dw at the mouth of the slip to 2,100 µg/kg dw at the head of the slip. cPAH concentrations were generally lower downstream of the slip, ranging from 310 to 880 µg/kg dw. In one of the cores with available data from the head of the slip, the peak cPAH concentration (1,200 µg/kg dw) was detected in the uppermost interval (0 to 2 ft). In the other core with available data, just outside the slip, concentrations were similar in the two intervals (230 and 210 µg/kg dw in the 0-to-2-ft and 2-to-4-ft intervals, respectively).

All cores collected in the navigation channel between RM 4.0 and RM 5.0 were sampled at one interval and had relatively low cPAH concentrations, ranging from 14 to 170 µg/kg dw, except for one. This core (from location "Average of 10-12") had a peak cPAH concentration (1,100 µg/kg dw) in the uppermost interval (0 to 4 ft) and a lower cPAH concentration (57 µg/kg dw) at depth (4-to-11-ft interval).

Ten cores were collected along the west side of the LDW between RM 4.0 and RM 5.0. cPAH concentrations in these cores were less than or equal to the 50th percentile (230 µg/kg dw). Six of these cores were sampled at one interval (approximately 0 to 2 ft) in an area that was subsequently dredged near RM 4.1 (Map 4-34e). In the remaining four cores with available data, three cores had peak cPAH concentrations in the uppermost interval, and one had a peak cPAH concentration at depth; all cPAH concentrations were less than the 50th percentile.

Four cores were collected between RM 4.8 and RM 5.0, three of which were within the Norfolk EAA. The core collected outside the EAA had no detected cPAH concentrations. In the two cores within the EAA (NFK009 and NFK008) and one core just outside the EAA (NFK207), peak concentrations (220, 270, and 1,460 µg/kg dw, respectively) were all in the uppermost interval of the core, within the 0-to-1-ft interval. The area within the Norfolk EAA was dredged and backfilled with clean material after the two cores at NFK009 and NFK008 were collected. The interpolated surface sediment data shown on Map 4-34f reflect post-dredging conditions.

In summary, most of the subsurface cPAH concentrations between RM 4.0 and RM 5.0 were less than the 50th percentile, except in Slip 6 and in the Norfolk EAA, which had concentrations up to 2,100 and 1,460 µg/kg dw, respectively. Of the 11 cores sampled at more than one interval in this section of the LDW, all but one had the peak concentration in the uppermost interval (Map 4-35).

4.2.3.4 Tissue

The tissue dataset for the RI contained cPAH data for 214 composite samples of fish, shellfish, and benthic invertebrates; all of these samples were analyzed for each of the seven individual cPAH compounds. Table 4-29 presents a summary of cPAH concentrations for each tissue type. At least one cPAH compound was detected in 65% of tissue samples. The highest cPAH concentrations were detected in invertebrates (clams, mussels, and benthic invertebrates), with mean concentrations ranging from 15 to 44 µg/kg ww. cPAH concentrations in individual composite samples of all other

species were less than 3 µg/kg ww, with the exception one Pacific staghorn sculpin sample with a concentration of 36 µg/kg ww from subarea T3. cPAHs were not detected in any of the other 23 Pacific staghorn sculpin samples, although the RLs for these samples (36 to 360 µg/kg ww) were elevated because analytical dilutions were used to minimize matrix interferences. The Pacific staghorn sculpin samples were analyzed by the same low-level analytical method, EPA 8270-SIM, as the other tissue samples collected in 2004. RLs for historical samples were generally higher than for samples collected in 2004; this uncertainty was discussed in Section B.6.1.1.5 of the HHRA.

Table 4-29. Summary of cPAH data in tissue

Tissue Type	Detection Frequency		Concentration (µg/kg ww)[a]			
	Ratio	%	Minimum Detect	Maximum Detect	Calculated Mean[b]	RL or Range of RLs[c]
Fish, whole body						
English sole	18/21	86	0.45 J	2.8	1.6	0.45
Pacific staghorn sculpin	1/24	4	36 J	36 J	130	36 – 360
Shiner surfperch	24/27	89	0.37 J	2.2	3.1	43
Starry flounder	3/3	100	0.47 J	0.66 J	0.59	na
Fish, fillet						
English sole (with skin)	4/7	57	0.37 J	0.53	0.35	0.45
English sole (without skin)	0/6	0	nd	nd	8.9	6.4 – 29
Pile perch (with skin)	1/1	100	0.43 J	0.43 J	0.43	na
Starry flounder (with skin)	1/1	100	0.64 J	0.64 J	0.64	na
Striped perch (with skin)	1/1	100	0.43 J	0.43 J	0.43	na
Crab, edible meat						
Dungeness crab	6/9	67	0.54 J	0.84 J	3.7	0.65 – 29
Slender crab	2/12	17	0.33	0.63 J	0.35	0.65
Crab, hepatopancreas						
Dungeness crab	2/4	50	0.73 J	0.73 J	6.5	6.2 – 43
Slender crab	4/4	100	0.68 J	0.71	0.70	na
Crab, whole body (calc'd)[d]						
Dungeness crab	nc	nc	0.60 JM	2.4 JM	2.6	17
Slender crab	nc	nc	0.45 M	0.67 M	0.65	na
Invertebrates, whole body						
Amphipod	0/4	0	nd	nd	22	43
Benthic invertebrates	20/20	100	4.2 J	290 J	44	na
Shellfish						
Mussels, wild (edible meat)	11/22	50	30	33	23	29
Mussels, transplanted (edible meat)	16/34	47	29	35	27	29 – 50

Tissue Type	Detection Frequency		Concentration (µg/kg ww)[a]			
	Ratio	%	Minimum Detect	Maximum Detect	Calculated Mean[b]	RL or Range of RLs[c]
Soft-shell clam (whole body)	14/14	100	6.8	44	15	na

[a] Total cPAHs were calculated by summing the products of concentrations and compound-specific PEFs for individual cPAH compounds, as discussed in detail in Appendix E, Section E.3.

[b] Calculated mean concentration is the average of detected concentrations and one-half the RL for non-detected results.

[c] RLs are based only on non-detect samples.

[d] Data from composite hepatopancreas samples were mathematically combined with data from composite samples of edible meat to form composite samples of edible meat plus hepatopancreas. cPAH concentrations in whole-body (i.e., edible meat plus hepatopancreas) crab were calculated assuming 69% (by weight) edible meat and 31% hepatopancreas, based on the relative weights of these tissues in a 16.6-cm Dungeness crab dissected by Windward in 2004 (unpublished data). Detection frequencies were not calculated for these samples because they do not represent individually analyzed samples.

cPAH – carcinogenic polycyclic aromatic hydrocarbon
J – estimated concentration
M – calculated whole-body concentration
na – not applicable
nc – not calculated
nd – not detected
RL – reporting limit
ww – wet weight

Mean cPAH concentrations by sampling area for the various tissue types are presented on Map 4-36. cPAH concentrations in whole-body English sole samples were significantly lower in Area T4 than in Areas T1 and T2. For shiner surfperch, mean concentrations were significantly lower in Area T4 than in Area T3. In Dungeness crab samples, cPAH concentrations were similar among all areas, except the single sample collected from Area T4 had a higher concentration.

All 20 benthic invertebrate composite tissue samples and all 14 clam composite tissue samples had detected concentrations of cPAHs. The maximum cPAH concentration in benthic invertebrate tissue (290 µg/kg ww) was detected at location B3b in Slip 1; this location also had the highest co-located sediment cPAH concentration (Map 4-36). The maximum cPAH concentration in clams (44 µg/kg ww) was detected at location C8 in Slip 4, which had the highest cPAH concentration in sediment co-located with clam tissue samples (Map 4-36).

The regression relationship for cPAHs in sediment and clams was evaluated in Section 8.3.3 to determine whether sediment RBTCs could be developed. The relationship was considered uncertain and therefore was not used to calculated RBTCs.

4.2.3.5 Summary of cPAH data

cPAH concentrations in surface sediment ranged from 9.7 to 11,000 µg/kg dw; the area-based 95th and 50th percentiles calculated from the RI baseline dataset were 1,100 and 230 µg/kg dw, respectively. The highest cPAH concentrations in surface sediment were in the T-117 EAA, in the Slip 4 EAA, in the RM 3.8 EAA on the eastern shoreline, near RM 0.6 on the western shoreline, near RM 0.0 close to Harbor Island, and in the Duwamish/Diagonal EAA.

Areas with the highest cPAH concentrations in subsurface sediment were Slip 3, near RM 1.2 on the east side, near RM 0.5 on the western shoreline, and near RM 1.4 west of the navigation channel. Peak cPAH concentrations were generally detected at depth within cores at locations with more than one interval sampled and with concentrations greater than the area-based 95th percentile (1,100 µg/kg dw). Cores with peak cPAH concentrations detected in uppermost intervals were located at the head of Slip 6, in the Norfolk EAA, near RM 0.5 on the western shoreline, and in some of the cores from the Duwamish/Diagonal EAA.

The highest cPAH concentrations in tissue were detected in clams, mussels, and benthic invertebrates, with mean concentrations ranging from 15 to 44 µg/kg ww. cPAH concentrations detected in individual composite samples of all other species were less than 3 µg/kg ww, with the exception of one Pacific staghorn sculpin sample with a concentration of 36 µg/kg ww. Although low-level analytical methods were used for all tissue samples collected in 2004, RLs were high for Pacific staghorn sculpin samples because analytical dilutions were used to minimize matrix interferences, resulting in reduced sensitivity of the analysis.

Mean cPAH concentrations in English sole (whole-body tissue) were significantly lower in Area T4 than in Areas T1 and T2. For shiner surfperch, mean concentrations were significantly lower in Area T4 than in Area T3. The relationship between cPAH concentrations in clam tissue and sediment was relatively uncertain (Section 8.3.3).

4.2.4 Dioxins and furans

This section summarizes the nature and extent of dioxin and furan contamination in surface and subsurface sediment samples. Dioxins and furans have not been analyzed in tissue, surface water, seep water, or porewater samples collected from the LDW.

4.2.4.1 Surface sediment

Surface sediment samples from within the LDW area were analyzed for dioxins and furans from 23 locations as part of the RI (Windward 2005d) and from 24 locations as part of the EPA SI (Weston 1999a). At least one dioxin or furan congener was detected in every sample, with individual congener concentrations ranging from 0.0730 to 241,000 ng/kg dw (Table 4-30).

Table 4-30. Summary of dioxin and furan data in LDW surface sediment

Chemical	Detection Frequency		Concentration (ng/kg dw)			
	Ratio	%	Minimum Detect	Maximum Detect	Calculated Mean[a]	RL or Range of RLs[b]
2,3,7,8-TCDD	22/47	47	0.0890 J	30.6	1.6	0.27 – 1.1
1,2,3,7,8-PeCDD	24/47	51	0.284 J	57.1	4.7	0.53 – 4.1
1,2,3,4,7,8-HxCDD	24/47	51	0.382 J	124	9.2	0.72 – 5.4
1,2,3,6,7,8-HxCDD	39/47	83	1.73 J	3,400	120	0.74 – 4.1
1,2,3,7,8,9-HxCDD	35/47	74	1.19 J	315	27	0.84 – 4.8

Chemical	Detection Frequency		Concentration (ng/kg dw)			
	Ratio	%	Minimum Detect	Maximum Detect	Calculated Mean[a]	RL or Range of RLs[b]
1,2,3,4,6,7,8-HpCDD	45/47	96	41.4 J	73,700	2,800	0.99 – 1.1
OCDD	47/47	100	7.8 J	241,000	18,000	na
2,3,7,8-TCDF	38/47	81	0.426 J	397	14	0.18 – 1.4
1,2,3,7,8-PeCDF	23/47	49	0.214 J	69.3	5.02	0.28 – 5.0
2,3,4,7,8-PeCDF	24/47	51	0.392 J	230	17	0.44 – 5.4
1,2,3,4,7,8-HxCDF	34/47	72	0.694 J	2,530	130	0.29 – 4.2
1,2,3,6,7,8-HxCDF	24/47	51	0.335 J	365	23	0.22 – 4.3
1,2,3,7,8,9-HxCDF	22/47	47	0.0730 J	33.8 J	1.97	0.12 – 2.4
2,3,4,6,7,8-HxCDF	24/47	51	0.307 J	302 J	14	0.29 – 2.5
1,2,3,4,6,7,8-HpCDF	44/47	94	6.71	40,300	1,200	0.62 – 7.7
1,2,3,4,7,8,9-HpCDF	31/47	66	0.421 J	3,720	120	0.77 – 4.2
OCDF	46/47	98	12.5	93,700	3,000	0.74 – 0.74
Dioxin and furan mammalian TEQ (Van den Berg et al. 2006)	47/47	100	1.1 J	2,100 J	na	na

[a] Calculated mean concentration is the average of detected concentrations and one-half the RL for non-detected results.

[b] RLs are based only on non-detect samples.

dw – dry weight
HpCDD – heptachlorodibenzo-*p*-dioxin
HpCDF – heptachlorodibenzofuran
HxCDD – hexachlorodibenzo-*p*-dioxin
HxCDF – hexachlorodibenzofuran
J – estimated concentration
LDW – Lower Duwamish Waterway
na – not applicable
OCDD – octachlorodibenzo-*p*-dioxin
OCDF – octachlorodibenzofuran
PeCDD – pentachlorodibenzo-*p*-dioxin
PeCDF – pentachlorodibenzofuran
RL – reporting limit
TCDD – tetrachlorodibenzo-*p*-dioxin
TCDF – tetrachlorodibenzofuran
TEQ – toxic equivalent
WHO – World Health Organization

TEQs were calculated using mammalian TEFs for 17 dioxin or furan congeners.[66] Methods for these calculations are described in Appendix B, Section B.2.2.4. Dioxin and furan TEQ concentrations in surface sediment ranged from 1.1 to 2,100 ng/kg dw (Table 4-30). Thirty-seven of the 47 dioxin and furan TEQs were less than 20 ng/kg dw. The three highest dioxin and furan TEQs (2,100, 565, and 463 ng/kg dw) were detected in surface sediment samples collected from the embayment between RM 1.4 and RM 1.5 on the west side of the LDW (Map 4-37). Other areas with notable dioxin and furan TEQ concentrations were: 1) the inlet at RM 2.2 in EAA 2 on the west side of

[66] For the purposes of describing the distribution of dioxin contamination in the LDW, TEQ concentrations were calculated using TEFs for mammals as presented in Van den Berg et al. (2006). If a dioxin or furan congener concentration was reported as undetected, then the TEF was multiplied by half the RL.

the LDW (412 ng/kg dw), 2) near RM 0.5 E in the Duwamish/Diagonal EAA (180 ng/kg dw), 3) near RM 1.0 in the navigation channel (124 ng/kg dw), and 4) near RM 3.6 E in the Boeing Plant 2/Jorgensen Forge EAA (101 ng/kg dw).

4.2.4.2 Subsurface sediment

Subsurface sediment samples were analyzed for dioxins and furans from eight locations as part of the RI. These locations were selected because of relatively high dioxin and furan TEQs in nearby surface sediment. Dioxin and furan TEQs at these eight subsurface sediment locations ranged from 0.147 to 194 ng/kg dw (Table 4-31). The highest concentration (194 ng/kg dw) was detected in the 4-to-6-ft interval in the core from location LDW-SC20 in the navigation channel at RM 1.0.

Table 4-31. Summary of dioxin and furan mammalian TEQ data in LDW subsurface sediment

Subsurface Sediment			Surface Sediment	
Subsurface Core Sampling Location	Sampling Interval (ft)	TEQ (ng/kg dw)[a]	Sampling Location Closest to Subsurface Core	TEQ (ng/kg dw)[a]
LDW-SC19 (RM 1.0 south of Kellogg Island)	0 – 1	22.8 J	LDW-SS36	27.1 J
	1 – 2	20.1 J		
	2 – 4	20.5 J		
LDW-SC20 (RM 1.0 in the navigation channel)	0 – 2	38.7 J	LDW-SS37	124 J
	2 – 4	27.1		
	4 – 6	194 J		
	8 – 10	5.60 J		
LDW-SC26 (RM 1.4 on the west side of the navigation channel)	0 – 1	15.9 J	none[b]	no data
	1 – 2	13.1 J		
	2 – 4	22.4 J		
	6 – 8	136 J		
LDW-SC28 (RM 1.4 on the west side of the navigation channel)	0 – 1	19.9 J	none[b]	no data
	1 – 2	14.8		
	2 – 4	18.5 J		
LDW-SC29 (RM 1.4 west of the Glacier NW dock)	0 – 1	54.1 J	LDW-SS56	2,100 J
	1 – 2	1.03 J		
	2 – 4	0.147 J		
LDW-SC39 (between RM 2.1 and RM 2.2 on the western shoreline)	0 – 1	7.91 J	none[b]	no data
	1 – 2	12.4 J		
	2 – 4	13.1 J		
LDW-SC40 (near RM 2.3 on the western shoreline)	0 – 1	6.71 J	none[b]	no data
	1 – 2	0.485 J		
	2 – 4	0.355 J		
LDW-SC41 (near RM 2.4 on	0 – 1	13.8	LDW-SS83	33.3 J
	1 – 2	12.5 J		

Subsurface Sediment			Surface Sediment	
Subsurface Core Sampling Location	Sampling Interval (ft)	TEQ (ng/kg dw)[a]	Sampling Location Closest to Subsurface Core	TEQ (ng/kg dw)[a]
the east side in the Myrtle Street Embayment)	2 – 4	14.0 J		

[a] TEQs were calculated using TEFs for mammals presented in Van den Berg et al. (2006)

[b] No surface samples were collected near subsurface cores.

dw – dry weight

J – estimated concentration

LDW – Lower Duwamish Waterway

RM – river mile

TEQ – toxic equivalent

Core LDW-SC29, collected near the surface sediment location with the highest TEQ (2,100 ng/kg dw) had low TEQs in comparison (Table 4-31). The highest TEQ in this subsurface core was 54.1 ng/kg dw in the 0-to-1-ft interval. In core LDW-SC26 collected just outside the embayment, the highest TEQ (136 ng/kg dw) was detected at an interval of 6 to 8 ft. A third subsurface core collected in that general area (LDW-SC28) had relatively low TEQs (14.8 to 19.9 ng/kg dw) in all three subsurface intervals.

In core LDW-SC20 near RM 1.0 in the navigation channel, the highest TEQ (194 ng/kg dw in the 4-to-6-ft interval) was higher than the TEQ in the surface sediment sample from that area (124 ng/kg dw).

In the other two cores collected from locations where surface sediment data were available (LDW-SC19 and LDW-SC41), TEQs in subsurface samples were slightly lower than TEQs in associated surface sediment samples (up to 22.8 ng/kg dw in subsurface samples and up to 33.3 ng/kg dw in surface samples) (Table 4-31). There was little variability in TEQs among the different subsurface intervals in each core (Table 4-31).

4.2.5 Metals

This section summarizes the nature and extent of metals contamination in surface sediment, subsurface sediment, tissue, surface water, seeps, and porewater.

4.2.5.1 Surface sediment

Surface sediment samples collected from 884 locations were analyzed for at least one metal or TBT. Table 4-32 presents detection frequencies and a summary of concentrations in surface sediment for nine metals (cadmium, chromium, copper, lead, mercury, nickel, silver, vanadium, and zinc) and TBT.[67]

[67] A metal was included if it was identified as a COC in the risk assessments.

Table 4-32. Summary of surface sediment data for metals and TBT

Detection Frequency			Concentration (mg/kg dw)			
Chemical	Ratio	%	Minimum Detect	Maximum Detect	Calculated Mean[a]	RL or Range of RLs[b]
Cadmium	584/838	70	0.030 J	120	1	0.04 – 2.5
Chromium	850/850	100	4.8	1,100 J	40	na
Copper	852/852	100	5	12,000 J	100	na
Lead	852/852	100	2	23,000	100	na
Mercury	747/869	86	0.021	4.6 J	0.2	0.02 – 0.10
Nickel	812/812	100	5	910	30	na
Silver	499/823	61	0.020	270	1	0.046 – 5
Vanadium	597/597	100	15	150	58	na
Zinc	849/849	100	16	9,700	190	na
Tributyltin as ion	142/158	90	0.00028 J	3.0	0.09	0.001 – 0.0053

[a] Calculated mean concentration is the average of detected concentrations and one-half the RL for non-detected results.

[b] RLs are based only on non-detect samples.

J – estimated concentration

na – not applicable

RL – reporting limit

Maps 4-38 through 4-47 summarize the concentrations of these metals and TBT in surface sediment samples collected from locations throughout the LDW. Concentrations of metals are shown as individual locations. These data are most relevant on a point-by-point basis, consistent with SMS, because metals other than arsenic are risk drivers for the benthic invertebrate community only. Therefore, only numerical percentiles are discussed. Most of the samples with concentrations that exceeded the 95th percentile[68] for each of the nine metals discussed in this section were located either downstream of RM 2.0 or between RM 3.5 and RM 3.7 on the east side of the navigation channel (within the Boeing Plant 2/Jorgensen Forge EAA). There were several areas of the LDW where the highest concentrations (i.e., greater than the 95th percentile) of various metals were co-located. Most notable of these areas was the Boeing Plant 2/Jorgensen Forge EAA. The maximum concentrations for each of the nine metals (as presented in Table 4-32) were detected in samples collected between RM 3.5 and RM 3.6 within the Boeing Plant 2/Jorgensen Forge EAA.

The Duwamish/Diagonal EAA and its vicinity also had multiple locations with high concentrations of all metals (but most notably cadmium, mercury, and silver). Other metals also had high concentrations at a few locations in this area.

[68] The SQS for each metal and the SL for nickel were greater than or equal to the corresponding 95th percentile concentrations.

Other locations with concentrations greater than the 95th percentile for multiple metals included:

- One area near RM 0.1 on the eastern shoreline (cadmium, copper, lead, silver, and zinc)
- One sampling location near RM 1.0 in the navigation channel (cadmium, chromium, mercury, and silver)
- Slip 1 (cadmium, copper, lead, mercury, nickel, zinc, and vanadium)
- RM 1.3 to RM 1.5 on the west side of the navigation channel (copper, zinc, and vanadium at multiple locations and one location that also had concentrations of cadmium, chromium, lead, mercury, and nickel greater than the 95th percentile)
- RM 2.2 on the west side of the navigation channel (cadmium, chromium, lead, mercury, silver, and zinc)
- Slip 4 (cadmium, lead, mercury, silver, and vanadium)

Spatial patterns for TBT were different than those for the nine metals. The maximum concentration of 3.0 mg/kg dw was detected at sampling location LDW-SS46, located at RM 1.3 on the west side of the LDW (Map 4-47). In general, this area and the area downstream of RM 1.0 had the highest TBT concentrations.

4.2.5.2 Subsurface sediment

Subsurface sediment samples collected from 205 cores (471 samples) prior to dredging; were analyzed for at least one metal or TBT. Subsurface sediment cores were sampled at various intervals during the different sampling events. To calculate summary statistics, data were categorized according to specified subsurface intervals, as described for total PCBs in Section 4.2.1.2. Table 4-33 presents a summary of subsurface metals and TBT data for samples collected from any interval, and Table 4-34 presents a summary of data for samples collected from specific 1- and 2-ft intervals within the top 10 ft of sediment.

Table 4-33. Summary of data for metals and TBT from any subsurface sediment core interval

Chemical	Detection Frequency[a]		Concentration (mg/kg dw)		
	Ratio	%	Minimum Detect	Maximum Detect	RL or Range of RLs[b]
Cadmium	257/388	66	0.088	20.4	0.18 – 0.9
Chromium	397/397	100	6.67	386	na
Copper	397/397	100	7.1	2,940	na
Lead	361/394	92	2.8	3,520 J	2 – 3.6
Mercury	341/436	78	0.020	10	0.020 – 0.100
Nickel	317/317	100	4.8	226	na
Silver	166/372	45	0.050	7.5	0.30 – 1

Chemical	Detection Frequency[a]		Concentration (mg/kg dw)		
	Ratio	%	Minimum Detect	Maximum Detect	RL or Range of RLs[b]
Vanadium	251/251	100	18	223	na
Zinc	396/396	100	16.2 J	4,720	na
Tributyltin as ion	58/95	61	0.00055 J	6.2	0.0010 – 0.0054

Note: All data in the subsurface dataset described in Section 4.1.2.2 were included in this summary.

[a] Subsurface sediment samples collected from areas that were subsequently dredged are not included in subsurface sediment data summarized in this table.

[b] RLs are based only on non-detect samples.

J – estimate concentration

na – not applicable

RL – reporting limit

Table 4-34. Summary of subsurface sediment data for metals and TBT in samples collected from 1- and 2-ft intervals within the top 10 ft of sediment

Chemical	Sample Interval (ft)[a]	Detection Frequency		Concentration (mg/kg dw)		
		Ratio	%	Minimum Detect	Maximum Detect	RL or Range of RL[b]
Cadmium – 1-ft intervals	0 to 1	52/78	67	0.3	4.5	0.2 – 0.9
	1 to 2	51/80	64	0.3	7.6	0.2 – 0.5
	2 to 3	11/19	58	0.3	18.7	0.20 – 0.38
	3 to 4	1/9	11	0.8	0.8	0.2 – 0.36
	4 to 5	3/9	33	0.50	3.8 J	0.20 – 0.30
	6 to 7	0/5	0	nd	nd	0.20 – 0.30
	8 to 9	1/4	25	0.70	0.70	0.20
Cadmium – 2-ft intervals	0 to 2	72/97	74	0.22	18	0.20 – 0.5
	2 to 4	47/73	64	0.3	15	0.2 – 0.8
	4 to 6	6/6	100	0.7	4.0	na
	6 to 8	3/4	75	1.4	20.4	0.3
	8 to 10	3/3	100	0.12	1.9	0.30
Chromium – 1-ft intervals	0 to 1	86/86	100	6.67	143 J	na
	1 to 2	80/80	100	9.96	135	na
	2 to 3	20/20	100	10.9	300	na
	3 to 4	9/9	100	8.9	24.2	na
	4 to 5	9/9	100	9.9	140 J	na
	6 to 7	5/5	100	10	19.1	na
	8 to 9	4/4	100	8.3	26.5	na
Chromium – 2-ft intervals	0 to 2	97/97	100	11.4 J	210 J	na
	2 to 4	73/73	100	8.1	386	na
	4 to 6	6/6	100	22	92	na

Chemical	Sample Interval (ft)[a]	Detection Frequency		Concentration (mg/kg dw)		
		Ratio	%	Minimum Detect	Maximum Detect	RL or Range of RL[b]
	6 to 8	4/4	100	13.0	160	na
	8 to 10	3/3	100	11.20	54.7	na
Copper – 1-ft intervals	0 to 1	86/86	100	10.9	327	na
	1 to 2	80/80	100	7.1	339	na
	2 to 3	20/20	100	9.20	599	na
	3 to 4	9/9	100	9.00	37.9	na
	4 to 5	9/9	100	7.4	67.3	na
	6 to 7	5/5	100	9.0	26.8	na
	8 to 9	4/4	100	7.5	31.9	na
Copper – 2-ft intervals	0 to 2	97/97	100	10.7	800	na
	2 to 4	73/73	100	7.6	2,940	na
	4 to 6	6/6	100	29.7	663	na
	6 to 8	4/4	100	23.2	1,950	na
	8 to 10	3/3	100	15.00	89.4	na
Lead – 1-ft intervals	0 to 1	84/84	100	4.9	639	na
	1 to 2	73/80	91	2.8	514	2.0 – 3
	2 to 3	18/19	95	3	356	2.0
	3 to 4	5/9	56	3	37 J	2 – 3.6
	4 to 5	8/9	89	3.0	64	2.0
	6 to 7	2/5	40	3.0	11	2.0
	8 to 9	1/4	25	43	43	2.0
Lead – 2-ft intervals	0 to 2	95/97	98	3	772	2.5 – 3
	2 to 4	65/73	89	3	3,520 J	2 – 3
	4 to 6	6/6	100	18.0	1,210	na
	6 to 8	4/4	100	42	1,350	na
	8 to 10	3/3	100	3.00	89	na
Mercury – 1-ft intervals	0 to 1	81/84	96	0.05	0.71	0.05 – 0.100
	1 to 2	54/66	82	0.06	0.6	0.040 – 0.0600
	2 to 3	13/19	68	0.070	0.64	0.030 – 0.060
	3 to 4	2/9	22	0.07	0.17	0.020 – 0.06
	4 to 5	3/10	30	0.060	0.24 J	0.040 – 0.060
	6 to 7	2/5	40	0.050	0.16	0.040 – 0.050
	8 to 9	1/3	33	0.19	0.19	0.050
Mercury – 2-ft intervals	0 to 2	102/107	95	0.05	10	0.04 – 0.07
	2 to 4	68/84	81	0.07	1.4	0.020 – 0.06
	4 to 6	9/15	60	0.080	0.98	0.050 – 0.060
	6 to 8	6/15	40	0.075	4.34	0.040 – 0.060
	8 to 10	4/12	25	0.25	0.89	0.050 – 0.080

Chemical	Sample Interval (ft)[a]	Detection Frequency		Concentration (mg/kg dw)		
		Ratio	%	Minimum Detect	Maximum Detect	RL or Range of RL[b]
Nickel – 1-ft intervals	0 to 1	65/65	100	4.8	36	na
	1 to 2	53/53	100	6.92	51.1	na
	2 to 3	10/10	100	9	32	na
	3 to 4	9/9	100	6 J	25	na
	8 to 9	1/1	100	5	5	na
Nickel – 2-ft intervals	0 to 2	76/76	100	7 J	37	na
	2 to 4	70/70	100	6	226	na
	4 to 6	5/5	100	13	34	na
	6 to 8	4/4	100	10	69	na
	8 to 10	3/3	100	12.00	24	na
Silver – 1-ft intervals	0 to 1	15/62	24	0.500	3.0	0.4 – 1
	1 to 2	23/80	29	0.5	2.6	0.3 – 1
	2 to 3	8/19	42	0.060	7.3	0.40 – 0.5
	3 to 4	1/9	11	0.61	0.61	0.3 – 0.5
	4 to 5	1/9	11	1.2	1.2	0.30 – 0.40
	6 to 7	0/5	0	nd	nd	0.30 – 0.40
	8 to 9	0/4	0	nd	nd	0.30 – 0.40
Silver – 2-ft intervals	0 to 2	52/97	54	0.050	3.3	0.4 – 1
	2 to 4	34/73	47	0.080	5	0.3 – 1
	4 to 6	4/6	67	1.4	4.3	0.400 – 1
	6 to 8	3/4	75	2	3	0.4
	8 to 10	2/3	67	0.5	2.3	0.40
Vanadium – 1-ft intervals	0 to 1	64/64	100	18	85	na
	1 to 2	37/37	100	39.6	84.3	na
	2 to 3	9/9	100	47.0	67	na
	3 to 4	8/8	100	36.1	71.5	na
	8 to 9	1/1	100	37.5	37.5	na
Vanadium – 2-ft intervals	0 to 2	73/73	100	37	87	na
	2 to 4	68/68	100	26	223	na
	4 to 6	5/5	100	39.8	95.1	na
	6 to 8	4/4	100	41.0	112	na
	8 to 10	2/2	100	63.9	75.4	na
Zinc – 1-ft intervals	0 to 1	86/86	100	18.6	1,260	na
	1 to 2	80/80	100	20.2	2,050	na
	2 to 3	19/19	100	22.3	1,770	na
	3 to 4	9/9	100	18.5	69	na
	4 to 5	9/9	100	20.8	324 J	na
	6 to 7	5/5	100	22.0	47.0	na

Chemical	Sample Interval (ft)[a]	Detection Frequency		Concentration (mg/kg dw)		
		Ratio	%	Minimum Detect	Maximum Detect	RL or Range of RL[b]
	8 to 9	4/4	100	17.6	84.7	na
Zinc –2-ft intervals	0 to 2	97/97	100	21	1,660	na
	2 to 4	73/73	100	16.2 J	4,720	na
	4 to 6	6/6	100	69.0	1,430	na
	6 to 8	4/4	100	88.4	4,550	na
	8 to 10	3/3	100	21.00	186	na
Tributyltin as ion – 1-ft intervals	0 to 1	11/11	100	0.0055	0.22	na
	1 to 2	6/14	43	0.021	0.35	0.0036 – 0.0054
	2 to 3	0/1	0	nd	nd	0.0010
	3 to 4	0/1	0	nd	nd	0.0036
	8 to 9	0/1	0	nd	nd	0.0036
Tributyltin as ion – 2-ft intervals	0 to 2	22/27	81	0.0030	2.5	0.0010 – 0.0049
	2 to 4	16/26	62	0.010	2.1	0.0010 – 0.0038
	4 to 6	3/3	100	0.014	1.0	na
	6 to 8	2/2	100	3.4	6.2	na
	8 to 10	0/1	0	nd	nd	0.0052569

[a] For the calculation of summary statistics in this table, actual sampling intervals were rounded to the nearest foot. The data for each interval were then assigned to the 1- or 2-ft sampling category that best approximated the actual sampling interval. If there were data for two 1-ft intervals at a particular location, the concentrations were averaged to obtain a calculated concentration for the corresponding 2-ft interval. Some of the subsurface data were collected from intervals other than 1 or 2 ft; therefore, the detection frequency ratio for "any interval" in Table 4-33 may reflect data for intervals not included in these categories.

[b] RLs based only on non-detect samples.

J – estimated concentration

na – not applicable

nd – not detected

RL – reporting limit

Over 90% of the samples presented in Table 4-34 were collected from the top 4 ft of sediment. There were relatively few samples analyzed from depths greater than 4 ft below mudline; most of those samples were collected and analyzed as part of the RI at locations with SMS exceedances in upper intervals or during other investigations targeting specific locations.

The highest copper and zinc concentrations were detected in a subsurface sediment sample collected from the head of Slip 3 and the highest cadmium concentration was in a subsurface sediment sample collected from the head of Slip 1. The highest concentrations of lead, chromium, and silver in any subsurface sediment samples (Table 4-33) were detected in cores collected from the 0-to-3-ft or 0-to-1-ft intervals within or just outside of the Duwamish/Diagonal EAA. In this area, high metal concentrations were detected in samples collected from both uppermost core intervals as well as deeper intervals within the cores. High metal concentrations were also

detected in subsurface samples from both uppermost intervals and at depth in the Boeing Plant 2/Jorgensen Forge EAA.

Other notable locations with subsurface metal concentrations greater than the 95th percentile for multiple metals include:

- RM 0.1 on the east side (LDW-SC2) (cadmium and silver, with peak concentrations in the 2-to-4-ft interval and lead and zinc, with peak concentrations in the 4-to-6-ft interval)
- RM 0.5 on the western shoreline (LDW-SC11) (cadmium, copper, lead, mercury, and zinc, with peak concentrations in the 0-to-1-ft interval)
- Slip 1 (DR021, LDW-SC16 and LDW-SC17) (up to nine metals in each core, with peak concentrations at intervals of 2 to 4 ft, 4 to 6 ft, or 6 to 8 ft)
- RM 1.3 to RM 1.5 on the west side of the navigation channel (DR054, LDW-SC25, LDW-SC26, LDW-SC27, and LDW-SC28) (up to eight metals in each core, with peak concentrations primarily at intervals of 2 to 4 ft, 4 to 6 ft, or 6 to 8 ft, but some high concentrations were also in the 0-to-2-ft interval)
- Slip 3 (LDW-SC37) (nine metals, with peak concentrations primarily in the 2-to-4-ft interval)
- RM 3.9 on the east side (LDW-SC52) (lead and mercury, with peak concentrations in the 0-to-1-ft interval; chromium, nickel, and silver, with peak concentrations in the 1-to-2-ft interval; and cadmium with the same concentration in both the 0-to-1-ft and 1-to-2-ft intervals)

TBT concentrations in subsurface sediment greater than the 95th percentile were located primarily near RM 1.3 and RM 1.4 on the west side. The two highest concentrations (6.2 and 3.4 mg/kg dw) were in the 6-to-8-ft interval in cores LDW-SC26 and LDW-SC28, respectively, collected in this area.

4.2.5.3 Tissue

Table 4-35 presents the detection frequencies and ranges of concentrations of seven metals and TBT that were identified as COCs in specific tissue types.[69] Appendix E, Section E.4.3, presents a summary of data for other metals and inorganic substances that were analyzed in tissue but are not discussed in this section.

The tissue types with the highest mean concentrations of each metal (including TBT) were invertebrates (i.e., clams, crabs, benthic invertebrates, and amphipods). In general, fish fillet samples had the lowest mean concentrations of TBT and metals, except for mercury. Mean concentrations in whole-body fish samples were higher than those in fish

[69] A metal is discussed in this section if it was detected in any tissue sample and was identified as a COC in either the HHRA based on seafood consumption or in the ERA based on a tissue exposure pathway.

fillets for TBT and all metals, except mercury. In crab hepatopancreas, mean concentrations of cadmium, copper, lead, and TBT were higher than in crab edible meat.

TBT concentrations were higher in soft-shell clams relative to other tissue types. TBT concentrations in the 14 clam samples ranged from 0.15 to 0.66 mg/kg ww; whereas the mean concentrations in other tissue types ranged from 0.0033 to 0.029 mg/kg ww.

As part of the ERA, regression analyses were conducted using co-located sediment and benthic invertebrate tissue data for metals and TBT (see Appendix A, Attachment 11). TBT was the only analyte with a significant relationship ($R^2 = 0.59$, $p < 0.05$). However, there is uncertainty in this relationship because of one sediment-tissue pair that was highly influential (Appendix A, Attachment 11).

Table 4-35. Summary of metals and TBT data in tissue composite samples

Tissue Type[a]	Unit	Detection Frequency		Concentration			
		Ratio	%	Minimum Detect	Maximum Detect	Calculated Mean[b]	RL or Range of RLs[c]
Cadmium							
Crab, edible meat	mg/kg ww	21/21	100	0.0055	0.0444	0.023	na
Crab, hepatopancreas	mg/kg ww	8/8	100	0.11	0.8530	0.48	na
Crab, whole body (calc'd)[d]	mg/kg ww	nc	nc	0.04 M	0.2951 M	0.16	na
Fish, fillet (with or without skin)	mg/kg ww	1/13	8	0.0013 J	0.0013 J	nc	0.0042 – 0.0079
Fish, whole body	mg/kg ww	75/75	100	0.002 J	0.0240	0.01	na
Amphipod, whole body	mg/kg ww	4/4	100	0.017	0.15	0.058	na
Benthic invertebrates, whole body	mg/kg ww	20/20	100	0.0175 J	0.202 J	0.060	na
Mussels, wild (edible meat)	mg/kg ww	22/22	100	0.19	0.84	0.49	na
Mussels, transplanted (edible meat)	mg/kg ww	35/35	100	0.231	0.781	0.379	na
Soft-shell clam, whole body	mg/kg ww	14/14	100	0.064	0.148	0.10	na
Chromium							
Crab, edible meat	mg/kg ww	2/21	10	0.13	0.16	0.05	0.07 – 0.11
Crab, hepatopancreas	mg/kg ww	3/8	38	0.04 J	0.16	0.05	0.01 – 0.08
Crab, whole body (calc'd)[d]	mg/kg ww	nc	nc	0.07 JM	0.140 M	0.04	0.03 – 0.05
Fish, fillet (with or without skin)	mg/kg ww	2/13	15	0.054	0.062	nc	0.049 – 0.12
Fish, whole body	mg/kg ww	53/75	71	0.06 J	3.74	0.2	0.10 – 0.14
Amphipod, whole body	mg/kg ww	4/4	100	0.45 J	0.56 J	0.52	na
Benthic invertebrates, whole body	mg/kg ww	20/20	100	0.08	3.90	1.0	na
Mussels, wild (edible meat)	mg/kg ww	21/22	95	0.10	0.35	0.16	0.050
Mussels, transplanted (edible meat)	mg/kg ww	33/35	94	0.059	0.305	0.1	0.05
Soft-shell clam, whole body	mg/kg ww	14/14	100	0.36	1.32	0.67	na

Table 4-35, cont. Summary of metals and TBT data in tissue composite samples

Tissue Type[a]	Unit	Detection Frequency		Concentration			
		Ratio	%	Minimum Detect	Maximum Detect	Calculated Mean[b]	RL or Range of RLs[c]
Copper							
Crab, edible meat	mg/kg ww	21/21	100	4.430	16	7.5	na
Crab, hepatopancreas	mg/kg ww	8/8	100	12.4	49.9	28	na
Crab, whole body (calc'd)[d]	mg/kg ww	nc	nc	6.9 M	24 M	14	na
Fish, fillet (with or without skin)	mg/kg ww	19/19	100	0.18	1.390	0.67	na
Fish, whole body	mg/kg ww	75/75	100	0.494	3.470	1.4	na
Amphipod, whole body	mg/kg ww	4/4	100	9.8	30	19	na
Benthic invertebrates, whole body	mg/kg ww	20/20	100	1.940 J	21.8	8.52	na
Mussels, wild (edible meat)	mg/kg ww	22/22	100	0.58	1.7 J	1.2	na
Mussels, transplanted (edible meat)	mg/kg ww	35/35	100	0.513	2.08	1.13	na
Soft-shell clam, whole body	mg/kg ww	14/14	100	3.50	7.30	5.34	na
Lead							
Crab, edible meat	mg/kg ww	21/21	100	0.012	0.24 J	0.045	na
Crab, hepatopancreas	mg/kg ww	8/8	100	0.037	0.2690	0.11	na
Crab, whole body (calc'd)[d]	mg/kg ww	nc	nc	0.020 M	0.22 JM	0.067	na
Fish, fillet (with or without skin)	mg/kg ww	10/19	53	0.006	0.137	0.04	0.020 – 0.030
Fish, whole body	mg/kg ww	75/75	100	0.012	0.946	0.17	na
Amphipod, whole body	mg/kg ww	4/4	100	0.95	7.4	3.7	na
Benthic invertebrates, whole body	mg/kg ww	20/20	100	0.1430	14.60	1.77	na
Mussels, wild (edible meat)	mg/kg ww	22/22	100	0.13	0.72	0.41	na
Mussels, transplanted (edible meat)	mg/kg ww	35/35	100	0.043	0.288	0.13	na
Soft-shell clam, whole body	mg/kg ww	14/14	100	0.368	6.370	1.96	na
Mercury							
Crab, edible meat	mg/kg ww	25/25	100	0.023	0.11	0.057	na

Table 4-35, cont. Summary of metals and TBT data in tissue composite samples

Tissue Type[a]	Unit	Detection Frequency		Concentration			
		Ratio	%	Minimum Detect	Maximum Detect	Calculated Mean[b]	RL or Range of RLs[c]
Crab, hepatopancreas	mg/kg ww	8/8	100	0.020	0.067	0.031	na
Crab, whole body (calc'd)[d]	mg/kg ww	nc	nc	0.022 M	0.097 M	0.046	na
Fish, fillet (with or without skin)	mg/kg ww	27/27	100	0.013	0.083	0.040	na
Fish, whole body	mg/kg ww	81/81	100	0.005	0.088	0.03	na
Amphipod, whole body	mg/kg ww	4/4	100	0.0067	0.017	0.011	na
Benthic invertebrates, whole body	mg/kg ww	18/20	90	0.002 J	0.044	0.01	0.004 – 0.009
Mussels, wild (edible meat)	mg/kg ww	21/21	100	0.0088	0.023	0.013	na
Mussels, transplanted (edible meat)	mg/kg ww	35/35	100	0.0051	0.0088	0.0066	na
Soft-shell clam, whole body	mg/kg ww	14/14	100	0.009	0.022	0.02	na
Vanadium							
Crab, edible meat	mg/kg ww	0/19	0	nd	nd	nc	0.14 – 0.21
Crab, hepatopancreas	mg/kg ww	4/7	57	0.11 J	0.2 J	0.1	0.13 – 0.21
Crab, whole body (calc'd)[d]	mg/kg ww	nc	nc	0.13 M	0.2 JM	0.1	0.08 – 0.11
Fish, fillet (with or without skin)	mg/kg ww	0/10	0	nd	nd	nc	0.2 – 0.25
Fish, whole body	mg/kg ww	56/72	78	0.1 J	1.23	0.3	0.2 – 0.25
Benthic Invertebrates,	mg/kg ww	20/20	100	0.26	3.04	1.2	na
Mussels, wild (edible meat)	mg/kg ww	8/8	100	0.058	0.26	0.15	na
Mussels, transplanted (edible meat)	mg/kg ww	10/10	100	0.119	0.281	0.195	na
Soft-shell clam, whole body	mg/kg ww	14/14	100	0.68	2.65	1.3	na
Zinc							
Crab, edible meat	mg/kg ww	21/21	100	26.1	39.3	34	na
Crab, hepatopancreas	mg/kg ww	8/8	100	14.8	33.6	23	na
Crab, whole body (calc'd)[d]	mg/kg ww	nc	nc	24.6 M	37.3 M	31	na
Fish, fillet (with or without skin)	mg/kg ww	13/13	100	3.8	9.33	7.0	na

Table 4-35, cont. Summary of metals and TBT data in tissue composite samples

Tissue Type[a]	Unit	Detection Frequency		Concentration			
		Ratio	%	Minimum Detect	Maximum Detect	Calculated Mean[b]	RL or Range of RLs[c]
Fish, whole body	mg/kg ww	75/75	100	9.95	28.0	16	na
Amphipod, whole body	mg/kg ww	4/4	100	7.9	26	14	na
Benthic invertebrates, whole body	mg/kg ww	20/20	100	8.22	43.6	20.9	na
Mussels, wild (edible meat)	mg/kg ww	22/22	100	17	44	30	na
Mussels, transplanted (edible meat)	mg/kg ww	35/35	100	4.78	20.6	9.71	na
Soft-shell clam, whole body	mg/kg ww	14/14	100	16.1	32.3	23.7	na
Tributyltin as ion							
Crab, edible meat	mg/kg ww	9/25	36	0.00081 J	0.082	0.0062	0.0015 – 0.0020
Crab, hepatopancreas	mg/kg ww	6/8	75	0.00050 J	0.059	0.019	0.0015
Crab, whole body (calc'd)[d]	mg/kg ww	nc	nc	0.00120 JM	0.075 M	0.0099	0.0008
Fish, fillet (with or without skin)	mg/kg ww	14/21	67	0.0012 J	0.016 J	0.0033	0.00074 – 0.0020
Fish, whole body	mg/kg ww	78/92	85	0.0018 J	0.18	0.027	0.0015 – 0.0021
Amphipod, whole body	mg/kg ww	4/4	100	0.018	0.036	0.029	na
Benthic invertebrates, whole body	mg/kg ww	19/20	95	0.0038 J	0.092	0.028	0.0050
Mussels, wild (edible meat)	mg/kg ww	22/22	100	0.012	0.037	0.023	na
Mussels, transplanted (edible meat)	mg/kg ww	32/32	100	0.00935	0.0369	0.0196	na
Soft-shell clam, whole body	mg/kg ww	14/14	100	0.15	0.66	0.32	na

Note: A metal is included in this table if it was detected in any tissue sample and was identified as a COC in either the HHRA based on seafood consumption or in the ERA based on a tissue exposure pathway.

[a] Crab tissue includes a combination of Dungeness, slender, and red rock crabs. Fish tissue includes English sole, Pacific staghorn sculpin, pile perch, shiner surfperch, starry flounder, and striped perch, as available.

[b] Calculated mean concentration is the average of detected concentrations and one-half the RL for non-detected results. Mean concentrations were not calculated if the detection frequency was ≤ 25%.

[c] RLs are based only on non-detect samples.

[d] Data from composite hepatopancreas samples were mathematically combined with data from composite samples of edible meat to form composite samples of edible meat plus hepatopancreas. Concentrations in whole-body (i.e., edible meat plus hepatopancreas) crab were calculated assuming 69% (by weight) edible meat and 31% hepatopancreas, based on the relative weights of these tissues in a 16.6-cm Dungeness crab dissected by Windward in 2004 (unpublished data). Detection frequencies were not calculated for these samples because they do not represent individually analyzed samples.

Table 4-35, cont. Summary of metals and TBT data in tissue composite samples

J – estimated concentration
M – calculated whole-body concentration
na – not applicable
nc – not calculated
nd – not detected
RL – reporting limit
ww – wet weight

4.2.5.4 Surface water

Table 4-36 presents a summary of metal concentrations in LDW surface water samples collected as part of the King County WQA (Map 4-11). Chromium, lead, vanadium, and zinc were detected in every surface water sample. Silver was not detected in any sample. TBT was not analyzed in surface water samples. Mean concentrations of chromium, copper, lead, nickel, vanadium, and zinc in unfiltered samples and copper, lead, and zinc in filtered samples were higher in samples from RM 4.9 than in samples from RM 1.1 and RM 1.9. Concentrations in filtered samples did not exceed Washington State acute or chronic marine water quality standards for dissolved metals.[70]

Table 4-36. Summary of surface water data for metals from three locations in the LDW collected from 1- and 2-ft intervals within the top 10 ft of sediment

Location ID	Detection Frequency		Concentration (µg/L)		
	Ratio	%	Minimum Detect	Maximum Detect	RL or Range of RLs[a]
Cadmium (chronic WQS = 9.3 µg/L; acute WQS = 42 µg/L)					
Filtered					
BRN (RM 1.1)	72/72	100	0.011	0.0755	na
SWM (RM 1.9)	72/72	100	0.0079	0.0795	na
NFK (RM 4.9)	12/22	55	0.0072	0.021	0.0067 – 0.0073
Unfiltered					
BRN (RM 1.1)	173/173	100	0.0088	0.0780	na
SWM (RM 1.9)	171/173	99	0.0076	0.391	0.0067 – 0.0068
NFK (RM 4.9)	49/56	88	0.0071	0.032	0.0068 – 0.0073
Chromium (chronic WQS = 50 µg/L; acute WQS = 1,100 µg/L)					
Filtered					
BRN (RM 1.1)	68/68	100	0.11 J	0.576 J	na
SWM (RM 1.9)	56/56	100	0.10 J	0.453 J	na
NFK (RM 4.9)	24/24	100	0.14 J	0.423 J	na
Unfiltered					
BRN (RM 1.1)	165/165	100	0.240 J	1.74 J	na
SWM (RM 1.9)	157/157	100	0.277 J	2.32 J	na
NFK (RM 4.9)	58/58	100	0.256 J	2.37 J	na

[70] The upstream sampling location near the Norfolk CSO could have been freshwater (i.e., low salinity) during certain sampling events, in which case it would be more appropriate to use freshwater water quality standards. Freshwater standards for metals are dependent on hardness and would need to be calculated based on conditions at the time each sample was collected.

Location ID	Detection Frequency		Concentration (µg/L)		
	Ratio	%	Minimum Detect	Maximum Detect	RL or Range of RLs[a]
Copper (chronic WQS = 3.1 µg/L; acute WQS = 4.8 µg/L)					
Filtered					
BRN (RM 1.1)	65/65	100	0.384 J	1.54 J	na
SWM (RM 1.9)	64/66	97	0.370 J	1.89 J	0.354 – 0.368
NFK (RM 4.9)	23/24	96	0.496 J	1.34 J	0.628
Unfiltered					
BRN (RM 1.1)	166/166	100	0.536 J	5.83 J	na
SWM (RM 1.9)	166/167	99	0.474 J	4.03 J	0.643
NFK (RM 4.9)	58/58	100	0.728 J	4.24 J	na
Lead (chronic WQS = 8.1 µg/L; acute WQS = 210 µg/L)					
Filtered					
BRN (RM 1.1)	70/70	100	0.010 J	0.553 J	na
SWM (RM 1.9)	66/66	100	0.0077 J	0.198 J	na
NFK (RM 4.9)	24/24	100	0.0611 J	0.343 J	na
Unfiltered					
BRN (RM 1.1)	171/171	100	0.0459 J	1.45 J	na
SWM (RM 1.9)	167/167	100	0.0570 J	1.57 J	na
NFK (RM 4.9)	58/58	100	0.143 J	2.81 J	na
Mercury (chronic WQS = 0.025 µg/L; acute WQS = 1.8 µg/L)					
Filtered[c]					
BRN (RM 1.1)	8/9	89	0.00013	0.00051	0.0001
NFK (RM 4.9)	6/6	100	0.00046	0.00071	na
Unfiltered					
BRN (RM 1.1)	9/15	60	0.00056	0.00336	0.20
SWM (RM 1.9)	0/6	0	nd	nd	0.20
NFK (RM 4.9)	6/8	75	0.00104	0.00689	0.20
Nickel (chronic WQS = 8.2 µg/L; acute WQS = 74 µg/L)					
Filtered					
BRN (RM 1.1)	61/65	94	0.274 J	0.910 J	0.154 – 0.387
SWM (RM 1.9)	54/60	90	0.259 J	1.50 J	0.118 – 0.306
NFK (RM 4.9)	21/21	100	0.190 J	0.472 J	na
Unfiltered					
BRN (RM 1.1)	145/147	99	0.394 J	2.11 J	0.348 – 0.386
SWM (RM 1.9)	137/143	96	0.328 J	2.88 J	0.291 – 0.575
NFK (RM 4.9)	55/55	100	0.434 J	2.91 J	na
Silver (no chronic WQS; acute WQS = 1.9 µg/L)					
Filtered					

Location ID	Detection Frequency		Concentration (µg/L)		
	Ratio	%	Minimum Detect	Maximum Detect	RL or Range of RLs[a]
BRN (RM 1.1)	0/72	0	nd	nd	0.11 – 0.13
SWM (RM 1.9)	0/72	0	nd	nd	0.097 – 0.13
NFK (RM 4.9)	0/24	0	nd	nd	0.10 – 0.13
Unfiltered					
BRN (RM 1.1)	0/173	0	nd	nd	0.10 – 0.13
SWM (RM 1.9)	0/173	0	nd	nd	0.10 – 0.13
NFK (RM 4.9)	0/58	0	nd	nd	0.11 – 0.13
Vanadium (no chronic or acute WQS)					
Filtered					
BRN (RM 1.1)	50/50	100	0.193	1.57	na
SWM (RM 1.9)	42/42	100	0.370	1.56	na
NFK (RM 4.9)	18/18	100	0.141	0.400	na
Unfiltered					
BRN (RM 1.1)	133/133	100	0.267	2.96	na
SWM (RM 1.9)	125/125	100	0.220	3.99	na
NFK (RM 4.9)	46/46	100	0.315	3.57	na
Zinc (chronic WQS = 81 µg/L; acute WQS = 90 µg/L)					
Filtered					
BRN (RM 1.1)	72/72	100	0.750 J	5.39 J	na
SWM (RM 1.9)	72/72	100	0.825 J	4.09 J	na
NFK (RM 4.9)	22/22	100	0.881 J	5.24 J	na
Unfiltered					
BRN (RM 1.1)	173/173	100	0.70 J	8.34 J	na
SWM (RM 1.9)	173/173	100	1.08 J	6.62 J	na
NFK (RM 4.9)	56/56	100	0.979 J	9.04 J	na

[a] RLs are based only on non-detect samples.

[b] Mercury data are not available for filtered samples from the SWM (RM 1.9) location.

BRN – LDW near Brandon CSO
J – estimated concentration
na – not applicable
nc – not calculated
nd – not detected
NFK – LDW near Norfolk CSO
RM – river mile
SWM – LDW near SW Michigan CSO
WQS – marine water quality standard

4.2.5.5 Seeps

Seep water samples were collected and analyzed for metals during four sampling events (Table 4-37; Map 4-11). At Boeing Plant 2, 10 filtered seep water samples and 17 unfiltered samples were analyzed for metals. During the Rhône-Poulenc sampling event, unfiltered seep water samples from seven locations were analyzed for metals,

with only detected concentrations reported. At T-117, three filtered and three unfiltered seep water samples were analyzed for metals. As part of the LDW-wide RI sampling event, 16 filtered and 13 unfiltered seep water samples were analyzed for metals. No seep water samples were analyzed for TBT during any of the four sampling events.

Table 4-37. Summary or seep water data for metals

Chemical	Detection Frequency		Concentration (µg/L)			Location of Maximum Detect
	Ratio	%	Minimum Detect	Maximum Detect	RL or Range of RLs[a]	
Cadmium (chronic WQS = 9.3 µg/L; acute WQS = 42 µg/L)						
Boeing Plant 2						
Cadmium (filtered)	0/10	0	nd	nd	2 – 10	na
Cadmium (unfiltered)	0/17	0	nd	nd	2 – 10	na
Rhône-Poulenc						
Cadmium (unfiltered)	2/7	29	2.1	11.6	nr	02-SP
T-117						
Cadmium (filtered)	0/3	0	nd	nd	2.00	na
Cadmium (unfiltered)	0/3	0	nd	nd	2.0	na
LDW RI						
Cadmium (filtered)	16/16	100	0.009	0.508	na	SP-82
Cadmium (unfiltered)	13/13	100	0.022	0.710	na	SP-54
Chromium (chronic WQS = 50 µg/L; acute WQS = 1,100 µg/L)						
Boeing Plant 2						
Chromium (filtered)	0/10	0	nd	nd	5 – 20	na
Chromium (unfiltered)	6/17	35	6	49	5 – 20	SE-SWY03
Rhône-Poulenc						
Chromium (unfiltered)	1/7	14	202	202	nr	02-SP
T-117						
Chromium (filtered)	2/3	67	8.00 J	9.00 J	5.00	SEEP_2
Chromium (unfiltered)	2/3	67	6.00	7.00	5.00	SEEP_3
LDW RI						
Chromium (filtered)	0/16	0	nd	nd	1.51 – 9.74	na
Chromium (unfiltered)	1/13	8	74.9	74.9	0.07 – 11.4	SP-54
Copper (chronic WQS = 3.1 µg/L; acute WQS = 4.8 µg/L)						
Boeing Plant 2						
Copper (filtered)	1/10	10	8	8	2 – 10	SE-94105
Copper (unfiltered)	11/17	65	2	60	2 – 10	SE-SWY03
Rhône-Poulenc						
Copper (unfiltered)	1/7	14	203	203	nr	02-SP
T-117						
Copper (filtered)	3/3	100	4 J	5.00 J	na	SEEP_2, SEEP_1

Chemical	Detection Frequency		Concentration (µg/L)			Location of Maximum Detect
	Ratio	%	Minimum Detect	Maximum Detect	RL or Range of RLs[a]	
Copper (unfiltered)	3/3	100	2.50	4.00	na	SEEP_3
LDW RI						
Copper (filtered)	7/16	44	8.16 J	22.8	3.28 – 7.77	SP-80
Copper (unfiltered)	10/13	77	8.06 J	50.9	4.75 – 6.47	SP-76
Lead (chronic WQS = 8.1 µg/L; acute WQS = 210 µg/L)						
Boeing Plant 2						
Lead (filtered)	0/10	0	nd	nd	1 – 20	na
Lead (unfiltered)	14/17	82	1	104	20	SE-SWY02
Rhône-Poulenc						
Lead (unfiltered)	1/7	14	44.1	44.1	nr	02-SP
T-117						
Lead (filtered)	0/3	0	nd	nd	20	na
Lead (unfiltered)	0/3	0	nd	nd	20	na
LDW RI						
Lead (filtered)	16/16	100	0.036	3	na	SP-76
Lead (unfiltered)	13/13	100	0.080	296	na	SP-54
Mercury (chronic WQS = 0.025 µg/L; acute WQS = 1.8 µg/L)						
Boeing Plant 2						
Mercury (filtered)	0/10	0	nd	nd	0.1	na
Mercury (unfiltered)	3/17	18	0.2	0.2	0.1	SE-31003, SE-SWY01, SE-SWY03
Rhône-Poulenc						
Mercury (unfiltered)	1/7	14	0.65	0.65	nr	05-SP
T-117						
Mercury (unfiltered)	0/3	0	nd	nd	0.10	na
LDW RI						
Mercury (filtered)	16/16	100	0.00062	0.0153	na	SP-76
Mercury (unfiltered)	13/13	100	0.00061	0.582	na	SP-54
Nickel (chronic WQS = 8.2 µg/L; acute WQS = 74 µg/L)						
Boeing Plant 2						
Nickel (filtered)	0/10	0	nd	nd	10 – 50	na
Nickel (unfiltered)	4/17	24	10	50	10 – 50	SE-SWY04
Rhône-Poulenc						
Nickel (unfiltered)	1/7	14	70.0	70.0	nr	02-SP
LDW RI						
Nickel (filtered)	12/16	75	0.84	5.25	0.040	SP-20
Nickel (unfiltered)	12/13	92	2.80	8.83	0.040	SP-20
Silver (chronic WQS value; acute WQS = 1.9 µg/L)						
Boeing Plant 2						

Chemical	Detection Frequency		Concentration (µg/L)			Location of Maximum Detect
	Ratio	%	Minimum Detect	Maximum Detect	RL or Range of RLs[a]	
Silver (filtered)	0/10	0	nd	nd	3 – 20	na
Silver (unfiltered)	0/17	0	nd	nd	3 – 20	na
T-117						
Silver (filtered)	0/3	0	nd	nd	3.00	na
Silver (unfiltered)	0/3	0	nd	nd	3.00	na
LDW RI						
Silver (filtered)	14/16	88	0.012	0.112	0.015	SP-20
Silver (unfiltered)	11/13	85	0.025	0.11	0.015	SP-82
Vanadium (no chronic or acute WQS values)						
Boeing Plant 2						
Vanadium (filtered)	2/10	20	2	4	2 – 10	SE-11001
Vanadium (unfiltered)	15/17	88	2	50	2 – 10	SE-31003
Rhône-Poulenc						
Vanadium (unfiltered)	7/7	100	2.8	400	na	02-SP
Zinc (chronic WQS = 81 µg/L; acute WQS = 90 µg/L)						
Boeing Plant 2						
Zinc (filtered)	3/10	30	30	90	4 – 20	SE-SWY04
Zinc (unfiltered)	14/17	82	6	200	20	SE-SWY04
Rhône-Poulenc						
Zinc (unfiltered)	1/7	14	223	223	nr	02-SP
T-117						
Zinc (filtered)	0/3	0	nd	nd	6.00	na
Zinc (unfiltered)	1/3	33	7.00 J	7.00 J	6.00	SEEP_1
LDW RI						
Zinc (filtered)	16/16	100	3.29	161	na	SP-82
Zinc (unfiltered)	13/13	100	3.49	322	na	SP-54

[a] RLs are based only on non-detect samples.

J – estimated concentration

na – not applicable

nc – not calculated because no RLs reported for non-detects

nd – not detected

nr – not reported; RLs for non-detects not reported for this sampling event

RI – remedial investigation

RL – reporting limit

WQS – marine water quality standard

The unfiltered seep water sample collected from location 02-SP near Slip 6 (Map 4-11) had the highest concentrations of cadmium, chromium, copper, nickel, and vanadium of any samples collected during the four sampling events. Metals concentrations in filtered samples were compared to the Washington State acute marine water quality standards, as shown in Table 4-37. Comparisons were not made to chronic water quality standards because chronic standards are based on 4-day average concentrations not to be exceeded more than once every 3 yrs. Because of the tidal

cycle, comparison of intertidal seep samples to acute standards is more appropriate. Acute standards represent 1-hr average concentrations not to be exceeded more than once every 3 yrs on average.

Copper and zinc were the only metals with concentrations in filtered seep samples greater than acute marine water quality standards. Ten seep water samples (one from Boeing Plant 2, two from T-117, and seven from the LDW RI) had copper concentrations in filtered samples that were greater than the acute marine water quality standard. Two filtered seep water samples collected during the LDW RI had zinc concentrations greater than the acute marine water quality standard. It is not known if these exceedances would have been present after dilution in receiving waters of the LDW.

4.2.5.6 Porewater

Metals were analyzed in 15 unfiltered porewater samples collected during the 2004 Rhône-Poulenc sampling event and in 15 unfiltered porewater samples collected throughout the LDW during the 1998 EPA SI. In addition, TBT was analyzed in 20 unfiltered samples collected by USACE between RM 1.8 and RM 3.2 in 1999 (Table 4-38). No comparison to water quality standards was made because all of the porewater samples were unfiltered. TBT was detected in 95% of the porewater samples at concentrations ranging from 0.010 to 0.15 µg/L, with the exception of one sample collected near RM 2.5 that had a concentration of 0.55 µg/L, which was greater than the DMMP screening level of 0.15 µg/L.

Table 4-38. Summary of porewater data for detected metals

Chemical	Detection Frequency		Concentration (µg/L)		
	Ratio	%	Minimum Detect	Maximum Detect	RL or Range of RLs[a]
Rhône-Poulenc					
Total copper	4/15	27	5.4	264	5
Total mercury	10/15	67	0.0016	0.408	0.2
Total zinc	3/15	20	263	1,560	50
EPA SI					
Total cadmium	4 /15	27	4 J	4 J	5
Total copper	4/15	27	1	5	4
Total lead	13/15	87	0.6 J	4	1
Total silver	6/15	40	0.3 J	0.5 J	1
Total vanadium	15/15	100	3 J	22	na
Total zinc	3/15	20	4 J	6 J	10
PSDDA99					
TBT	19/20	95	0.010 J	0.55	0.020

[a] RLs are based only on non-detect samples.

J – estimated concentration

na – not applicable
RL – reporting limit

4.2.5.7 Summary of metals and TBT data

Most samples with concentrations greater than the numerical 95th percentile for each metal were located either downstream of RM 2.2 or between RM 3.5 and RM 3.7 (within the Boeing Plant 2/Jorgensen Forge EAA). The Duwamish/Diagonal EAA and the Boeing Plant 2/Jorgensen Forge EAA had many of the highest metals concentrations in surface sediment. Other notable areas with high concentrations of more than one metal in surface sediment include Slip 1, Slip 4, RM 1.3 to RM 1.5 on the west side, and RM 2.2 on the west side. TBT concentrations in surface sediment were highest near RM 1.3 on the west side of the LDW and downstream of RM 1.0.

In tissue, the highest mean concentration of metals (including TBT) were detected in invertebrate species (i.e., clams, crabs, benthic invertebrates, and amphipods). Mean concentrations of metals (except mercury) and TBT in whole-body fish samples were higher than those in fish fillets. In crab hepatopancreas, mean concentrations of cadmium, copper, lead, and TBT were higher than those in crab edible meat.

In surface water, higher mean concentrations of chromium, copper, lead, nickel, vanadium, and zinc in unfiltered samples and copper, lead, and zinc in filtered samples were detected in samples from RM 4.9 than in samples from RM 1.1 and RM 1.9. Concentrations in filtered samples did not exceed Washington State acute or chronic marine water quality standards for dissolved metals.

An unfiltered seep water sample collected near Slip 6 had the highest concentrations of cadmium, chromium, copper, nickel, and vanadium. In filtered seep water samples, copper and zinc were the only metals with concentrations that exceeded their acute marine water quality standards. It is not known if these exceedances would have been present after dilution in receiving waters of the LDW.

Thirty unfiltered porewater samples collected from throughout the LDW were analyzed for metals; seven metals were detected in these samples. Twenty porewater samples were collected between RM 1.8 and RM 3.2 and analyzed for TBT. The highest TBT concentration (0.55 µg/L) was detected in a sample collected near RM 2.5. This was the only sample with a TBT concentration that exceeded the DMMP SL of 0.15 µg/L.

4.2.6 SVOCs

This section summarizes the nature and extent of SVOC contamination in surface sediment, subsurface sediment, tissue, surface water, seeps, and porewater.

4.2.6.1 Surface sediment

Table 4-39 presents detection frequencies and a summary of concentrations in surface sediment for selected SVOCs.[71] Surface sediment samples collected from 828 of the 1,365 locations in the RI baseline dataset were analyzed for one or more PAHs, samples from 832 locations were analyzed for one or more phthalates, and samples from 831 locations were analyzed for at least one other SVOC included in Table 4-39.

Table 4-39. Summary of surface sediment data for selected SVOCs

Chemical	Detection Frequency		Concentration (µg/kg dw)			
	Ratio	%	Minimum Detect	Maximum Detect	Calculated Mean[a]	RL or Range of RLs[b]
PAHs						
2-Methylnaphthalene	139/818	17	1.0 J	3,300	nc	1.0 – 2,000
Acenaphthene	304/828	37	1.0 J	5,200	70	1.8 – 2,000
Anthracene	576/828	70	2.0	10,000	100	13 – 2,000
Benzo(a)anthracene	748/828	90	7.3 J	8,400	320	6.4 – 200
Benzo(a)pyrene	747/822	91	6.5	7,900	310	6.4 – 350
Benzo(g,h,i)perylene	682/823	83	6.1	3,800	200	13 – 2,000
Total benzofluoranthenes	757/822	92	6.6 J	17,000	740	nc
Chrysene	773/828	93	12	7,700	500	18 – 170
Dibenzo(a,h)anthracene	436/828	53	1.6 J	1,500	60	1.0 – 2,000
Dibenzofuran	248/827	30	1.0 J	4,200	50	1.7 – 2,000
Fluoranthene	797/828	96	18	24,000	900	19 – 340
Fluorene	382/828	46	1.4 J	6,800	80	1.8 – 2,000
Indeno(1,2,3-cd)pyrene	726/823	88	6.5	4,300	200	6.4 – 1,600
Naphthalene	152/818	19	3.0 J	5,300	nc	1.0 – 2,000
Phenanthrene	759/828	92	7.1	28,000	400	18 – 200
Pyrene	788/828	95	19	16,000	700	18 – 170
Total HPAH	803/828	97	20	85,000	4,000	nc
Total LPAH	763/828	92	9.1	44,000	700	nc
Total PAH	805/828	97	20	128,000	4,000	nc
Phthalates						
BEHP	674/832	81	5.4	14,000	600	15 – 1,500
BBP	425/822	52	2.0	7,100	80	1.8 – 2,000
Dimethyl phthalate	156/822	19	2.0 J	200	nc	1.8 – 2,000
Chlorobenzenes						
1,2,4-Trichlorobenzene	5/816	1	1.6 J	72 J	nc	0.40 – 2,000

[71] An SVOC was included if it was identified as a COC in the risk assessments based on a sediment exposure pathway.

Chemical	Detection Frequency		Concentration (µg/kg dw)			
	Ratio	%	Minimum Detect	Maximum Detect	Calculated Mean[a]	RL or Range of RLs[b]
1,2-Dichlorobenzene	17/816	2	1.3 J	520 J	nc	0.40 – 2,000
1,4-Dichlorobenzene	39/816	5	1.7 J	1,600 J	nc	0.20 – 2,000
Hexachlorobenzene	46/819	6	0.4 J	95 J	nc	0.11 – 2,000
Other SVOCs						
2,4-Dimethylphenol	5/813	1	6.1	290 J	nc	6.0 – 2,000
4-Methylphenol	82/831	10	4.8 J	4,600 J	nc	8.6 – 2,000
Benzoic acid	70/822	9	54 J	4,500	nc	13 – 3,000
Benzyl alcohol	15/812	2	8.2 J	670	nc	9.2 – 690
n-Nitrosodiphenylamine	23/818	3	6.5	230	nc	1.8 – 2,000
Pentachlorophenol	12/785	2	14 J	410	nc	7.6 – 4,900
Phenol	257/831	31	10 J	2,800	90	7.3 – 790

Note: SVOCs were included in this table if they were identified as COCs in the ERA or HHRA based on a sediment exposure pathway.

[a] Calculated mean concentration is the average of detected concentrations and one-half the RL for non-detected results. Mean concentrations were not calculated if the number of samples was ≤ 20 or the detection frequency was ≤ 25% because it is uncertain if these means would represent LDW-wide conditions.

[b] RLs are based only on non-detect samples.

BBP -- butyl benzyl phthalate
BEHP -- bis(2-ethylhexyl) phthalate
dw – dry weight
na – not applicable
nc – not calculated
nd – not detected
J – estimated concentration
HPAH – high-molecular-weight polycyclic aromatic hydrocarbon
LPAH – low-molecular-weight polycyclic aromatic hydrocarbon
PAH – polycyclic aromatic hydrocarbon
RL – reporting limit
SVOC – semivolatile organic compound

At least one individual PAH listed in Table 4-39 was detected in samples from 97% of the locations. The individual PAHs with the highest mean concentrations were fluoranthene (900 µg/kg dw), pyrene (700 µg/kg dw), chrysene (500 µg/kg dw), and phenanthrene (400 µg/kg dw). Concentrations of total high-molecular-weight polycyclic aromatic hydrocarbons (HPAHs) were generally higher than those of total LPAHs (means of 4,000 and 700 µg/kg, respectively).

The maximum total HPAH concentration (85,000 µg/kg dw) was detected at a location near RM 3.6 in the T-117 EAA (T117-SE-37-G). Other areas with high total HPAH concentrations were: 1) in the navigation channel west of the Duwamish/Diagonal EAA (51,000 µg/kg dw at DR044), 2) at RM 3.8 in the Boeing Plant 2/Jorgensen Forge EAA on the eastern shoreline (51,000 µg/kg dw at R23), 3) near RM 2.7 on the western shoreline (48,000 µg/kg dw at LDW-SS95), and 4) near RM 1.0 on the eastern shoreline (42,000 µg/kg dw at LDW-SS35).

The maximum total LPAH concentration (44,000 µg/kg dw) was detected at a location near RM 2.7 on the western shoreline (LDW-SS95). Other areas with high total LPAH concentrations were: 1) in the T-117 EAA (43,000 µg/kg dw at T117-SE-37-G), 2) near

RM 1.0 on the eastern shoreline (34,000 µg/kg dw at LDW-SS35), and 3) in the Boeing Plant 2/Jorgensen Forge EAA at RM 3.6 (10,900 µg/kg dw at SD-307-S). Several other areas also had total HPAH and LPAH concentrations greater than the numerical 95th percentile, as shown on Maps 4-48 and 4-49.

Of the three phthalate compounds presented in Table 4-39, bis(2-ethylhexyl) phthalate (BEHP) was detected most frequently (in 81% of surface sediment samples) and had the highest mean concentration (600 µg/kg dw). Butyl benzyl phthalate (BBP) and dimethyl phthalate were detected less frequently (in 52 and 19% of samples, respectively). The majority of locations with high BEHP concentrations in surface sediment (i.e., greater than the numerical 95th percentile, or 2,407 µg/kg dw) were between RM 0.4 and RM 0.6 in and near the Duwamish/Diagonal EAA. Other areas with more than one sample with high BEHP concentrations were near RM 2.2 on the west side of the LDW (EAA 2), Slip 4, and near RM 3.5 on the east side (in the Boeing Plant 2/Jorgensen Forge EAA) (Map 4-50). Most of locations with high BBP concentrations in surface sediment were located in the vicinity of the Duwamish/Diagonal EAA (Map 4-51), as were the highest dimethyl phthalate concentrations.

Each of the four chlorobenzenes listed in Table 4-39 was detected in 6% or fewer of the surface sediment samples. Samples with detected concentrations of 1,2-dichlorobenzene, 1,4-dichlorobenzene, and 1,2,4-trichlorobenzene were collected primarily between RM 0.4 and RM 0.6 on the east side of the LDW in the vicinity of the Duwamish/Diagonal EAA. The highest hexachlorobenzene concentrations (95 and 63 µg/kg dw) were detected near RM 1.8 west of the navigation channel (LDW-SS68) and between RM 3.0 and RM 3.1 on the western shoreline (B7a), respectively. All other detected hexachlorobenzene concentrations were less than 30 µg/kg dw.

Among the seven chemicals in the "other SVOC" category in Table 4-39, phenol was detected most frequently (in 31% of the samples). The highest concentrations of phenol were primarily located between RM 3.5 and RM 3.7 in the Boeing Plant 2/Jorgensen Forge EAA, near RM 3.6 in the T117 EAA, and in Slip 6 (Map 4-52). The highest concentrations of benzoic acid were located primarily between RM 4.0 and RM 4.2 and in Slip 6 (Map 4-53).

An analysis was conducted to assess changes in BEHP concentrations at 44 locations sampled in the 1990s that were resampled during sampling events in 2004 through 2006; these locations were resampled within 10 ft of the original sample[72] (Table 4-40). This analysis was conducted for BEHP because it is a risk driver chemical that exceeds the SMS criteria in surface sediment more frequently than other SVOCs.

Of the 44 locations, BEHP concentrations decreased in surface sediments over time at 35 locations (79%) and increased at 3 locations (7%) (Table 4-12 and Map 4-54). At the

[72] The older data within 10 ft of the newer sample were excluded from the baseline dataset as described in Section 4.1.2.1.

remaining 6 locations (14%), concentration differences were within the range of analytical variability (≤ 25% increase or decrease compared to the initial concentration) (Table 4-40; Map 4-54). At the three locations with higher concentrations in more recent samples, differences ranged from 190 to 980 µg/kg dw, with the greatest increase located at the head of Slip 6. Concentration differences ranged from 44 µg/kg dw to 6,080 µg/kg dw at the 35 locations with lower concentrations in more recent samples, with the greatest decrease located near RM 0.0. Historically, phthalates have been common contaminants in laboratory hardware, and therefore elevated sample concentrations in older samples may have reflected laboratory contamination in addition to environmental contamination. This issue increases the uncertainty associated with evaluating trends in phthalate concentrations because decreasing phthalate concentrations over time may be an artifact of overestimation of phthalate concentration in older samples as a result of laboratory contamination.

Table 4-40. Difference in BEHP concentrations at resampled locations sorted by concentration difference

Chemical/ River Mile	Location Name		Date		Concentration (µg/kg dw)		
	Initial	Resampled	Initial	Resampled	Initial	Resampled	Difference
0	DR076	LDW-SS5	8/24/1998	1/17/2005	6,100	20 U	-6,080
1.3	DR053	LDW-SS44	8/31/1998	1/21/2005	3,800 J	120	-3,680
1.4	SS-4	LDW-SS55	8/17/1993	1/24/2005	2,200	98	-2,102
1.4	DR160	LDW-SS51	8/12/1998	1/18/2005	1,900	120 U	-1,780
4.2	R40	LDW-SS127	10/13/1997	1/20/2005	1,400	140	-1,260
3.8	DR187	LDW-SS115	8/27/1998	1/25/2005	1,500	330	-1,170
4.2	R45	LDW-SS130	10/16/1997	1/20/2005	1,200	72	-1,128
1.7	DR097	LDW-SS63	8/20/1998	1/21/2005	1,200	150	-1,050
0.3	DR079	LDW-SS15	8/24/1998	1/17/2005	1,100	64 U	-1,036
1.4	DR030	LDW-SS50	8/17/1998	1/24/2005	1,500	560	-940
0.3	DUD042	LDW-SS17	11/11/1995	1/24/2005	2,000	1,100	-900
1.3	SS-2	LDW-SS48	8/17/1993	1/18/2005	1,600	770	-830
1.4	SS-3	LDW-SS49	8/17/1993	1/26/2005	960	160	-800
3.8	R24	LDW-SS117	10/10/1997	1/20/2005	940	140	-800
4.2	R42	LDW-SS129	10/13/1997	1/20/2005	930	170	-760
2.2	DR113	LDW-SS81	8/19/1998	3/8/2005	910	190 U	-720
0.1	K-07	LDW-SS4	9/30/1991	1/17/2005	740 J	83 U	-657
0.2	K-05	LDW-SS10	10/14/1991	1/17/2005	710 UJ	82 U	-628
1.0	DR019	LDW-SS32	8/17/1998	1/18/2005	710	93 U	-617
0.2	DR035	LDW-SS12	8/11/1998	1/17/2005	720	180	-540
2.1	DR106	LDW-SS76	8/19/1998	1/20/2005	460	59	-401
1.0	DR020	LDW-SS31	8/17/1998	1/21/2005	550	160	-390
1.4	DR065	LDW-SS52	8/17/1998	1/25/2005	410	95	-315

Chemical/ River Mile	Location Name		Date		Concentration (µg/kg dw)		
	Initial	Resampled	Initial	Resampled	Initial	Resampled	Difference
1.5	DR123	LDW-SS57	9/14/1998	1/24/2005	560	290	-270
1.4	DR028	B4b	8/17/1998	8/28/2004	390	140 J	-250
3.7	SD-323-S	LDW-SS110	8/17/2004	1/25/2005	410	170	-240
2.8	DR175	LDW-SS94	8/20/1998	1/21/2005	270	46	-224
0	K-11	LDW-SS1	9/30/1991	1/17/2005	290 UJ	67 U	-223
0.9	DR021	LDW-SS319	8/17/1998	10/4/2006	710	520	-190
3.9	R30	LDW-SS119	10/11/1997	1/19/2005	460	280	-180
1.2	DR088	LDW-SS40	8/31/1998	1/18/2005	410	270	-140
4.3	DR286	B10b	8/26/1998	8/19/2004	150 UJ	35 J	-115
2.0	R7	LDW-SS75	10/15/1997	1/21/2005	180	74	-106
4.7	DR271	LDW-SS148	9/15/1998	3/9/2005	260	160 U	-100
3.2	DR202	LDW-SS104	8/27/1998	1/25/2005	80	36 J	-44
4.1	DR238	LDW-SS125	8/27/1998	1/20/2005	130	97 J	**-33**
3.1	DR198	LDW-SS102	8/20/1998	1/24/2005	150 UJ	130	**-20**
3.7	R21	LDW-SS113b	10/9/1997	1/20/2005	220	200	**-20**
0.9	DR085	LDW-SSB2b	8/31/1998	3/11/2005	340	350	**10**
4.1	A11-05	LDW-SS126	8/18/1994	1/20/2005	81	92 J	**11**
1.0	DR087	LDW-SS37	8/12/1998	1/18/2005	570	760	190
1.9	DR131	LDW-SS70	8/13/1998	1/21/2005	1,500	1,700	**200**
3.7	DR186	LDW-SS111	8/27/1998	1/19/2005	210	580	370
4.2	DR242	SB-1	8/24/1998	8/25/2004	620	1,600	980

Note: Concentrations in bold were within the range of analytical variability (≤ 25% increase or decrease compared to the initial concentration).

J estimated concentration

U not detected at reporting limit shown

UJ not detected at estimated reporting limit shown

4.2.6.2 Subsurface sediment

Detection frequencies and ranges of concentrations of SVOCs in all subsurface sediment samples collected from any interval are presented in Table 4-41. The most frequently detected SVOCs in subsurface sediment were PAH compounds; individual PAHs were detected in 15 to 88% of all samples. Other SVOCs that were most frequently detected were BEHP and BBP in 71 and 52% of all samples, respectively.

Table 4-41. Summary of subsurface sediment data from any interval for selected SVOCs

Chemical	Detection Frequency[a]		Concentration (µg/kg dw)		
	Ratio	%	Minimum Detect	Maximum Detect	RL or Range of RLs[b]
PAHs					
2-Methylnaphthalene	43/281	15	2.9 J	4,500	1.6 – 400
Acenaphthene	94/304	31	1.3 J	4,600	4.30 – 400
Anthracene	200/304	66	4.3 J	1,900	4.30 – 400
Benzo(a)anthracene	243/304	80	12 J	4,500	4.30 – 180
Benzo(a)pyrene	237/304	78	11 J	5,300	4.30 – 400
Benzo(g,h,i)perylene	207/298	69	11 J	1,000	4.30 – 400
Total benzofluoranthenes	248/304	82	21 J	10,200	nc
Chrysene	247/304	81	12 J	7,200	4.30 – 180
Dibenzo(a,h)anthracene	119/304	39	3.2 J	660 J	4.30 – 910
Dibenzofuran	58/304	19	2.9 J	1,700	1.7 – 400
Fluoranthene	262/304	86	12 J	13,000	4.30 – 340
Fluorene	104/304	34	4.9 J	4,300	2.3 – 400
Indeno(1,2,3-cd)pyrene	212/298	71	10 J	1,500	4.30 – 400
Naphthalene	70/293	24	1.9 J	3,400	1.7 – 400
Phenanthrene	244/304	80	10 J	13,000	4.30 – 400
Pyrene	266/304	88	9.9 J	10,000	4.30 – 170
Total HPAH	268/304	88	9.9 J	47,000	nc
Total LPAH	244/304	80	12.00	27,000 J	nc
Total PAH	269/304	88	9.9 J	57,000 J	nc
Phthalates					
BEHP	216/306	71	12 J	5,100	19 – 2,400
BBP	147/283	52	5.6 J	610	5.8 – 400
Dimethyl phthalate	20/283	7	4.2 J	8,800	2.4 – 400
Chlorobenzenes					
1,2,4-Trichlorobenzene	27/293	9	3.6 J	110 J	0.87 – 400
1,2-Dichlorobenzene	31/292	11	0.98 J	160	0.78 – 400
1,4-Dichlorobenzene	56/292	19	2.1 J	750 J	0.86 – 400
Hexachlorobenzene	3/280	1	4.6 J	10	0.78 – 400
Other SVOCs					
2,4-Dimethylphenol	21/281	7	3.7 J	46	5.8 – 790
4-Methylphenol	15/304	5	8.6 J	110 J	3.8 – 400
Benzoic acid	122/304	40	35 J	3,000 J	58 – 4,000
Benzyl alcohol	32/281	11	7.7 J	210	4.9 – 2,000

Chemical	Detection Frequency[a]		Concentration (µg/kg dw)		
	Ratio	%	Minimum Detect	Maximum Detect	RL or Range of RLs[b]
n-Nitrosodiphenylamine	1/281	< 1	33	33	2.9 – 7,300
Pentachlorophenol	54/304	18	16 J	930 J	12 – 3,100
Phenol	57/304	19	12 J	3,100	4.4 – 790

Note: All data in the subsurface dataset described in Section 4.1.2.2 were included in this summary. SVOCs were included in this table if they were identified as COCs in the ERA or HHRA based on a sediment exposure pathway.

[a] Subsurface sediment samples collected from areas that were subsequently dredged are not included in this table.

[b] RLs are based only on non-detect samples.

BBP – butyl benzyl phthalate

BEHP – bis(2-ethylhexyl) phthalate

dw – dry weight

HPAH – high-molecular-weight polycyclic aromatic hydrocarbon

LPAH – low-molecular-weight polycyclic aromatic hydrocarbon

J – estimated concentration

nc – not calculated

PAH – polycyclic aromatic hydrocarbon

RL – reporting limit

SVOC – semivolatile organic compound

U – not detected at reporting limit shown (check all for consistency)

Detection frequencies and ranges of concentrations of SVOCs in subsurface sediment collected from specific 1- and 2-ft intervals[73] within the top 10 ft of sediment are presented in Table 4-42. Over 90% of the samples represented in these tables were collected from the top 4 ft of sediment. Relatively few samples were analyzed from depths greater than 4 ft below mudline; most of these samples were collected and analyzed as part of the RI at locations with SMS exceedances of SVOCs in upper intervals.

Table 4-42. Summary of subsurface sediment data for SVOCs in samples collected from 1- and 2-ft intervals within the top 10 ft of sediment

Chemical	Sample Interval (ft)[a]	Detection Frequency		Concentration (µg/kg dw)		
		Ratio	%	Minimum Detect	Maximum Detect	RL or Range of RLs[b]
PAHs						
Total HPAH – 1-ft intervals	0 to 1	60/64	94	122 J	34,700	nc
	1 to 2	38/45	84	13 J	40,000	nc
	2 to 3	7/10	70	115 J	4,400	nc
	3 to 4	4/9	44	21 J	323 J	nc
	4 to 5	1/1	100	580 J	580 J	nc

[73] Subsurface sediment cores were sampled at various depth intervals during different sampling events. To calculate summary statistics, data were categorized according to specified depth intervals, as described for total PCBs in Section 4.2.1.2.

Chemical	Sample Interval (ft)[a]	Detection Frequency		Concentration (µg/kg dw)		
		Ratio	%	Minimum Detect	Maximum Detect	RL or Range of RLs[b]
Total HPAH – 2-ft intervals	0 to 2	72/74	97	80	27,900	nc
	2 to 4	62/73	85	9.9 J	47,000	nc
	4 to 6	10/10	100	140	22,000	nc
	6 to 8	5/5	100	60 J	38,000 J	nc
	8 to 10	5/5	100	58.0	2,040	nc
Total LPAH – 1-ft intervals	0 to 1	54/64	84	12 J	2,100 J	nc
	1 to 2	34/45	76	24 J	7,500	nc
	2 to 3	4/10	40	20	2,630 J	nc
	3 to 4	2/9	22	43 J	220 J	nc
	4 to 5	1/1	100	47 J	47 J	nc
Total LPAH – 2-ft intervals	0 to 2	72/74	97	15 J	3,800 J	nc
	2 to 4	56/73	77	20	27,000 J	nc
	4 to 6	9/10	90	141 J	3,900	nc
	6 to 8	4/5	80	770 J	9,800	nc
	8 to 10	5/5	100	12.00	275 J	nc
Total PAH – 1-ft intervals	0 to 1	61/64	95	24	36,200	nc
	1 to 2	38/45	84	13 J	42,600 J	nc
	2 to 3	7/10	70	135 J	5,000 J	nc
	3 to 4	4/9	44	21 J	366 J	nc
	4 to 5	1/1	100	630 J	630 J	nc
Total PAH – 2-ft intervals	0 to 2	73/74	99	24	29,500 J	nc
	2 to 4	62/73	85	9.9 J	57,000 J	nc
	4 to 6	10/10	100	281 J	23,000 J	nc
	6 to 8	5/5	100	60 J	46,000 J	nc
	8 to 10	5/5	100	70.0	2,210	nc
Phthalates						
BEHP – 1-ft intervals	0 to 1	44/64	69	22	2,100	27 – 1,400
	1 to 2	24/45	53	13 J	3,900	19 – 400
	2 to 3	7/10	70	20 J	820	20 – 59
	3 to 4	2/9	22	17 J	24	19 – 20
	4 to 5	1/1	100	290	290	na
BEHP– 2-ft intervals	0 to 2	65/74	88	12 J	2,400	20 – 530
	2 to 4	51/73	70	13 J	3,900	19 – 280
	4 to 6	8/10	80	56 J	2,200	20.0 – 65
	6 to 8	4/5	80	1,000	3,800	65
	8 to 10	2/5	40	72.00	260	61 – 66
BBP– 1-ft intervals	0 to 1	33/41	80	5.9	610	5.9 – 180
	1 to 2	22/45	49	14	400	5.8 – 36
	2 to 3	3/10	30	6.6	10	5.9 – 400
	3 to 4	2/9	22	6.4	7.7	5.8 – 20
	4 to 5	1/1	100	18	18	na

Chemical	Sample Interval (ft)[a]	Detection Frequency		Concentration (µg/kg dw)		
		Ratio	%	Minimum Detect	Maximum Detect	RL or Range of RLs[b]
BBP – 2-ft intervals	0 to 2	51/74	69	5.9	610	5.8 – 110
	2 to 4	34/73	47	5.8 J	180	5.8 – 42
	4 to 6	5/10	50	12 J	48	6.5 – 36
	6 to 8	4/5	80	15 J	35 J	6.5
	8 to 10	0/5	0	nd	nd	6.1 – 20.0
Dimethyl phthalate – 1-ft intervals	0 to 1	3/41	7	13 J	1,700	13 – 180
	1 to 2	1/45	2	16 J	16 J	12 – 130
	2 to 3	0/10	0	nd	nd	14 – 400
	3 to 4	0/9	0	nd	nd	14 – 20
	4 to 5	0/1	0	nd	nd	6.6
Dimethyl phthalate – 2-ft intervals	0 to 2	5/74	7	9.9 J	84	12 – 110
	2 to 4	4/73	5	30	8,800	14 – 140
	4 to 6	2/10	20	20	210	6.5 – 44
	6 to 8	3/5	60	16	69 J	6.5
	8 to 10	0/5	0	nd	nd	6.6 – 62
Chlorobenzenes						
1,2,4-Tri-chlorobenzene – 1-ft intervals	0 to 1	5/42	12	3.6 J	18 J	5.8 – 180
	1 to 2	6/46	13	4.1 J	17 J	5.8 – 20.0
	2 to 3	0/11	0	nd	nd	5.9 – 400
	3 to 4	0/10	0	nd	nd	5.8 – 20
	4 to 5	0/2	0	nd	nd	6.6 – 8.3
	5 to 6	0/1	0	nd	nd	8.3
	6 to 7	0/1	0	nd	nd	8.2
	7 to 8	0/1	0	nd	nd	9.5
	8 to 9	0/1	0	nd	nd	9.5
	9 to 10	0/1	0	nd	nd	8.4
1,2,4-Tri-chlorobenzene – 2-ft intervals	0 to 2	11/76	14	3.9 J	13 J	5.8 – 110
	2 to 4	4/74	5	4.1 J	110 J	5.8 – 25.0
	4 to 6	2/11	18	11	18	6.5 – 20.0
	6 to 8	3/6	50	9.8	14	6.5 – 8.2
	8 to 10	0/5	0	nd	nd	5.40 – 6.6
1,2-Dichlorobenzene – 1-ft intervals	0 to 1	4/42	10	1.7 J	17	2.0 – 180
	1 to 2	4/46	9	2.9 J	9.6	0.78 – 20.0
	2 to 3	0/11	0	nd	nd	0.86 – 400
	3 to 4	0/10	0	nd	nd	0.86 – 6.0
	4 to 5	1/2	50	10	10	1.7
	5 to 6	0/1	0	nd	nd	1.7
	6 to 7	0/1	0	nd	nd	1.6
	7 to 8	0/1	0	nd	nd	1.9
	8 to 9	0/1	0	nd	nd	1.9

Chemical	Sample Interval (ft)[a]	Detection Frequency		Concentration (µg/kg dw)		
		Ratio	%	Minimum Detect	Maximum Detect	RL or Range of RLs[b]
	9 to 10	0/1	0	nd	nd	1.7
1,2-Dichlorobenzene – 2-ft intervals	0 to 2	8/76	11	1.7 J	20	1.7 – 110
	2 to 4	8/74	11	3.6 J	150	0.86 – 42
	4 to 6	2/11	18	10	12	1.7 – 20.0
	6 to 8	3/6	50	12	160	1.6 – 6.6
	8 to 10	0/5	0	nd	nd	1.10 – 6.6
1,4-Dichlorobenzene – 1-ft intervals	0 to 1	15/42	36	3.0 J	750 J	2.0 – 180
	1 to 2	9/46	20	3.5 J	17 J	1.7 – 20
	2 to 3	2/11	18	2.1 J	3.0 J	1.8 – 400
	3 to 4	0/10	0	nd	nd	0.86 – 6.0
	4 to 5	1/2	50	7.9	7.9	1.7
	5 to 6	0/1	0	nd	nd	1.7
	6 to 7	0/1	0	nd	nd	1.6
	7 to 8	0/1	0	nd	nd	1.9
	8 to 9	0/1	0	nd	nd	1.9
	9 to 10	0/1	0	nd	nd	1.7
1,4-Dichlorobenzene – 2-ft intervals	0 to 2	16/76	21	3.0 J	380 J	1.7 – 110
	2 to 4	13/74	18	2.1 J	38	1.6 – 42
	4 to 6	7/11	64	3.9 J	31	1.7 – 20.0
	6 to 8	4/6	67	4.0 J	24	1.6 – 6.5
	8 to 10	0/5	0	nd	nd	1.10 – 6.6
Hexachlorobenzene – 1-ft intervals	0 to 1	1/41	2	5.9	5.9	0.83 – 180
	1 to 2	1/45	2	10	10	0.78 – 20.0
	2 to 3	0/10	0	nd	nd	0.86 – 400
	3 to 4	0/9	0	nd	nd	0.86 – 6.0
	4 to 5	0/1	0	nd	nd	6.6
Hexachlorobenzene – 2-ft intervals	0 to 2	2/74	3	5.9	10	0.78 – 110
	2 to 4	0/73	0	nd	nd	0.86 – 42
	4 to 6	0/10	0	nd	nd	6.5 – 20.0
	6 to 8	1/5	20	4.6 J	4.6 J	6.5 – 6.6
	8 to 10	0/5	0	nd	nd	6.1 – 20.0
Other SVOCs						
2,4-Dimethylphenol – 1-ft intervals	0 to 1	2/41	5	25 J	27 J	5.8 – 350
	1 to 2	3/45	7	9.5 J	14 J	5.8 – 30
	2 to 3	0/10	0	nd	nd	5.9 – 790
	3 to 4	0/9	0	nd	nd	5.8 – 34
	4 to 5	0/1	0	nd	nd	6.6

Chemical	Sample Interval (ft)[a]	Detection Frequency		Concentration (µg/kg dw)		
		Ratio	%	Minimum Detect	Maximum Detect	RL or Range of RLs[b]
2,4-Dimethylphenol – 2-ft intervals	0 to 2	5/74	7	9.5 J	27 J	5.8 – 330
	2 to 4	5/73	7	6.3 J	46	5.8 – 42
	4 to 6	3/10	30	6.5 J	9.2	6.5 – 20.0
	6 to 8	4/5	80	8.5 J	24 J	6.5
	8 to 10	3/5	60	3.7 J	10	6.6 – 20.0
4-Methylphenol – 1-ft intervals	0 to 1	2/64	3	17 J	42 J	20 – 310
	1 to 2	1/45	2	24	24	19 – 130
	2 to 3	0/10	0	nd	nd	20 – 400
	3 to 4	0/9	0	nd	nd	19 – 34
	4 to 5	0/1	0	nd	nd	66
4-Methylphenol – 2-ft intervals	0 to 2	3/74	4	13 J	42 J	20 – 110
	2 to 4	3/73	4	23	110 J	19 – 140
	4 to 6	1/10	10	42 J	42 J	20.0 – 66
	6 to 8	2/5	40	37 J	48 J	65 – 66
	8 to 10	0/5	0	nd	nd	20.0 – 66
Benzoic acid – 1-ft intervals	0 to 1	40/64	62	52 J	2,000 J	59 – 2,700
	1 to 2	23/45	51	48 J	450 J	59 – 200
	2 to 3	5/10	50	66	110 J	100 – 4,000
	3 to 4	6/9	67	35 J	130	58 – 140
	4 to 5	0/1	0	nd	nd	590
Benzoic acid – 2-ft intervals	0 to 2	41/74	55	50 J	490	58 – 1,100
	2 to 4	32/73	44	54 J	3,000 J	58 – 350
	4 to 6	0/10	0	nd	nd	200 – 590
	6 to 8	1/5	20	320 J	320 J	590
	8 to 10	0/5	0	nd	nd	200 – 620
Benzyl alcohol – 1-ft intervals	0 to 1	10/41	24	18 J	200	29 – 890
	1 to 2	7/45	16	19 J	210	19 – 99
	2 to 3	0/10	0	nd	nd	29 – 2,000
	3 to 4	0/9	0	nd	nd	29 – 34
	4 to 5	0/1	0	nd	nd	33
Benzyl alcohol – 2-ft intervals	0 to 2	11/74	15	18 J	200	19.0 – 550
	2 to 4	7/73	10	20 J	34 J	20.0 – 210
	4 to 6	1/10	10	52	52	20.0 – 33
	6 to 8	0/5	0	nd	nd	33
	8 to 10	0/5	0	nd	nd	20.0 – 38
n-Nitroso-diphenylamine – 1-ft intervals	0 to 1	1/41	2	33	33	8.2 – 320
	1 to 2	0/45	0	nd	nd	5.8 – 620
	2 to 3	0/10	0	nd	nd	5.9 – 400
	3 to 4	0/9	0	nd	nd	5.8 – 34
	4 to 5	0/1	0	nd	nd	32

Chemical	Sample Interval (ft)[a]	Detection Frequency		Concentration (µg/kg dw)		
		Ratio	%	Minimum Detect	Maximum Detect	RL or Range of RLs[b]
n-Nitroso-diphenylamine – 2-ft intervals	0 to 2	1/74	1	33	33	6 – 870
	2 to 4	0/73	0	nd	nd	5.8 – 7,300
	4 to 6	0/10	0	nd	nd	20.0 – 510
	6 to 8	0/5	0	nd	nd	6.5 – 2,600
	8 to 10	0/5	0	nd	nd	20.0 – 65
Pentachlorophenol – 1-ft intervals	0 to 1	15/64	23	16 J	930 J	29 – 3,100
	1 to 2	8/45	18	17 J	120 J	29 – 100
	2 to 3	1/10	10	26 J	26 J	29 – 2,000
	3 to 4	0/9	0	nd	nd	29 – 34
	4 to 5	0/1	0	nd	nd	33
Pentachlorophenol – 2-ft intervals	0 to 2	17/74	23	17 J	730	29 – 550
	2 to 4	10/73	14	18 J	190	29 – 180
	4 to 6	9/10	90	36	63	100
	6 to 8	4/5	80	45	800	33
	8 to 10	1/5	20	19 J	19 J	30 – 98.00
Phenol – 1-ft intervals	0 to 1	17/64	27	14 J	3,100	20 – 350
	1 to 2	9/45	20	15 J	150	19 – 130
	2 to 3	1/10	10	13 J	13 J	20 – 790
	3 to 4	2/9	22	13 J	13 J	19 – 140
	4 to 5	0/1	0	nd	nd	66
Phenol – 2-ft intervals	0 to 2	20/74	27	14 J	180	19 – 220
	2 to 4	17/73	23	13 J	110	19 – 140
	4 to 6	0/10	0	nd	nd	20.0 – 73
	6 to 8	0/5	0	nd	nd	65 – 66
	8 to 10	0/5	0	nd	nd	20.0 – 66

Note: All data in the subsurface dataset described in Section 4.1.2.2 were included in this summary. SVOCs were included in this table if they were identified as COCs in the ERA or HHRA based on a sediment exposure pathway.

[a] For the calculation of summary statistics in this table, actual sampling intervals were rounded to the nearest foot. The data for each interval were then assigned to the 1- or 2-ft sampling category that best approximated the actual sampling interval. If there were data for two 1-ft intervals at a particular location, the concentrations were averaged to obtain a calculated concentration for the corresponding 2-ft interval. Some of the subsurface data were collected from intervals other than 1 or 2 ft; therefore, the detection frequency ratio for "any interval" in Table 4-41 may reflect data for intervals not included in these categories.

[b] RLs are based only on non-detect samples.

BBP – butyl benzyl phthalate
BEHP – bis(2-ethylhexyl) phthalate
dw – dry weight
HPAH – high-molecular-weight polycyclic aromatic hydrocarbon
LPAH – low-molecular-weight polycyclic aromatic hydrocarbon
J – estimated concentration
na – not applicable
nc – not calculated
nd – not detected
PAH – polycyclic aromatic hydrocarbon
RL – reporting limit
SVOC – semivolatile organic compound

PAH data in 1- and 2-ft subsurface intervals for the top 6 ft of sediment are shown on Maps 4-55a through 4-55c (total HPAHs) and Maps 4-56a through 4-56c (total LPAHs). The maximum total HPAH concentration (64,000 µg/kg dw) was detected at a location

near RM 1.2 on the eastern shoreline (LDW-SC23) at an interval of 3 to 3.5 ft. Other areas with high total HPAH concentrations include:

- Slip 3 (LDW-SC37) (47,000 µg/kg dw in the 2-to-4-ft interval)
- Near RM 2.7 on the western shoreline (C3; within an area that was subsequently dredged) (41,000 µg/kg dw in the 0-to-3.3-ft interval)
- Near RM 1.4 on the west side of the navigation channel (LDW-SC26) (38,000 µg/kg dw in the 6-to-8-ft interval)
- Near RM 0.5 on the western shoreline (LDW-SC11) (34,700 µg/kg dw in the 0-to-1-ft interval)

The maximum total LPAH concentration (27,000 µg/kg dw) was detected in Slip 1 (LDW-SC17) at an interval of 2 to 4 ft. High total HPAH concentrations were also reported in the following areas:

- Near RM 1.2 on the east side of the navigation channel (LDW-SC23) (25,000 µg/kg dw in the 3-to-3.5-ft interval)
- Slip 3 (LDW-SC37) (10,500 µg/kg dw in the 2-to-4-ft interval)
- Near RM 2.7 on the western shoreline (C3; within an area that was subsequently dredged) (10,000 µg/kg dw in the 0-to-3.3-ft interval)
- Near RM 1.4 on the west side of the navigation channel (LDW-SC26) (8,500 µg/kg dw in the 6-to-8-ft interval)

BEHP, BBP, and dimethyl phthalate were detected in 71, 52, and 7%, respectively, of all subsurface sediment samples in which they were analyzed from any interval (Table 4-42). BEHP data in 1- and 2-ft subsurface intervals for the top 6 ft of sediment are shown on Maps 4-57a through 4-57c. Most of the locations with high BEHP concentrations in any subsurface interval were located between RM 0.4 and RM 0.6, in the vicinity of the Duwamish/Diagonal EAA. The maximum concentration in this area was 18,000 µg/kg dw in the 3-to-6-ft interval at DUD261. The highest BBP concentration in subsurface sediment was also detected in a sample from the Duwamish/Diagonal EAA (1,300 µg/kg in the 2-to-3-ft interval at DUD006).

Chlorobenzenes were detected in 1 to 19% of all subsurface samples in which they were analyzed from any interval (Table 4-42). Of the other SVOCs listed in Table 4-42, benzoic acid was the most frequently detected (in 40% of all subsurface samples) and had the highest mean concentration (200 µg/kg dw). Other SVOCs were detected in 17% or less of all subsurface samples in which they were analyzed from any interval.

4.2.6.3 Tissue

Table 4-43 presents the detection frequency and range of SVOC concentrations in tissue.[74] Of the SVOCs listed in Table 4-43, PAHs, benzoic acid, and phenol were the only compounds detected in 17% or more of the composite tissue samples.

Table 4-43. Summary of data for selected SVOCs in tissue samples

Chemical	Detection Frequency		Concentration (µg/kg ww)		
	Ratio	%	Minimum Detect	Maximum Detect	Range of RLs[a]
PAHs					
Total HPAH	180/235	77	0.21 J	3,200 J	nc
Total LPAH	150/235	64	0.094 J	450 J	nc
Total PAH	187/235	80	0.43 J	3,630 J	nc
Phthalates[b]					
BEHP	30/235	13	28	2,200 J	3.6 – 5,000
BBP	25/230	11	190 J	2,800	9 – 4,000
Dimethyl phthalate	6/235	3	7.6 J	420 JM	3.6 – 2,900
Chlorobenzenes					
Hexachlorobenzene	23/235	10	0.38 JN	6.6 JN	0.8 – 26
Other SVOCs					
2,4-Dimethylphenol	2/235	1	71 J	210 J	3.6 – 1,200
4-Methylphenol	20/235	9	15 J	44,000	3.6 – 1,200
Benzoic acid	154/235	66	340 J	54,000	36 – 12,000
Benzyl alcohol	44/235	19	12 J	3,450	3.6 – 1,200
Carbazole	2/235	1	6,000	14,000	3.6 – 2,900
n-Nitrosodiphenylamine	1/235	<1	170 J	170 J	3.6 – 2,900
Pentachlorophenol	16/235	7	1.1 J	4,700	1.7 – 5,800
Phenol	40/235	17	17 J	2,200	3.6 – 1,500

Note: An SVOC is included in this table if it was detected in any tissue sample and was identified as a chemical of potential concern in either the HHRA based on seafood consumption or in the ERA based on a tissue exposure pathway.

[a] RLs are based only on non-detect samples.

[b] In addition to data presented in this table, six English sole whole-body composite samples, nine English sole individual whole-body samples, and seven shiner surfperch composite samples collected by King County in 2006 were analyzed for phthalates. In these samples, one English sole sample had a detected concentration of BEHP (170 µg/kg ww). Phthalates were not detected in any other samples, at RLs ranging from 50 to 200 µg/kg ww.

BBP – butyl benzyl phthalate

BEHP – bis(2-ethylhexyl) phthalate

[74] An SVOC is discussed in this section if it was detected in any tissue sample and was identified as a chemical of potential concern in either the HHRA based on seafood consumption or in the ERA based on a tissue exposure pathway.

J – estimated concentration

M – calculated whole-body concentration

N – tentative identification

nc – not calculated

In general, individual PAHs were detected frequently in all tissue types except Pacific staghorn sculpin (in only 3 of the 24 samples), English sole fillets without skin (in none of the 6 samples), and amphipods (in only 1 of the 4 samples) (Table 4-44). Although the same analytical method was used to analyze the Pacific staghorn sculpin samples as the other tissue samples collected in 2004, the RLs for PAHs and phthalates were high for Pacific staghorn sculpin samples because analytical dilutions were used to minimize matrix interferences. cPAH concentrations in tissue are discussed in Section 4.2.3.4. Benzoic acid, which, after PAHs, was the next most frequently detected SVOC in tissue, was detected frequently in fish, benthic invertebrates, mussels, and clams but was not detected in any crab samples (Table 4-44).

Table 4-44. Summary of detection frequencies for SVOCs in tissue samples

Chemical	Detection Frequency (%)																	
	Fish – Whole Body				Fish – Fillet					Crab – EM		Crab – HP		Invertebrates and Shellfish				
	English Sole (n = 21)	Pacific Staghorn Sculpin (n = 24)	Shiner Surfperch (n = 27)	Starry Flounder (n = 3)	English Sole – with Skin (n = 7)	English Sole – without Skin (n = 6)	Pile Perch – with Skin (n = 1)	Starry Flounder with Skin (n = 1)	Striped Perch – with Skin (n = 1)	Dungeness Crab (n = 9)	Slender Crab (n = 12)	Dungeness Crab (n = 4)	Slender Crab (n = 4)	Amphipod – WB (n = 4)	Benthic Invert – WB (n = 20)	Mussels, Wild – EM (n = 22)	Mussels, Transplanted – EM (n = 34)	Soft-shell Clam – WB (n = 14)
PAHs																		
Total HPAH (calc'd)	100	4	89	100	57	0	100	100	100	78	100	50	100	25	100	95	71	100
Total LPAH (calc'd)	100	8	93	100	100	0	100	100	100	78	100	75	100	0	100	0	29	100
Total PAH (calc'd)	100	12	93	100	100	0	100	100	100	78	100	75	100	25	100	95	71	100
Phthalates																		
BEHP	0	0	19	0	29	0	0	0	0	0	0	0	25	50	25	9	0	71
BBP	14	0	29	0	0	0	0	0	0	0	0	25	75	0	5	0	0	0
Dimethyl phthalate	0	0	0	0	0	0	0	0	0	0	8	0	25	0	0	0	0	0
Chlorobenzenes																		
Hexachlorobenzene	19	8	4	0	14	0	0	0	0	11	0	25	0	0	0	0	0	64
Other SVOCs																		
2,4-Dimethylphenol	0	4	0	0	0	0	0	0	0	0	0	0	0	0	5	0	0	0
4-Methylphenol	0	17	4	0	0	0	0	0	0	0	0	0	0	0	30	0	6	50
Benzoic acid	67	96	63	100	86	0	100	100	0	0	0	0	0	0	95	100	100	100
Benzyl alcohol	52	29	22	0	0	0	100	0	0	0	8	25	0	0	40	5	15	0
Carbazole	0	0	7	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
n-Nitrosodiphenylamine	0	0	0	0	0	0	0	0	0	0	0	0	0	0	5	0	0	0
Pentachlorophenol	24	12	7	33	0	0	0	0	0	0	0	0	0	0	25	0	0	0
Phenol	0	8	48	0	0	0	0	0	0	0	8	0	0	50	45	0	0	86

BBP -- butyl benzyl phthalate
BEHP -- bis(2-ethylhexyl) phthalate
EM – edible meat
HP – hepatopancreas
HPAH – high-molecular-weight polycyclic aromatic hydrocarbon
LPAH – low-molecular-weight polycyclic aromatic hydrocarbon
PAH – polycyclic aromatic hydrocarbon
SVOC – semivolatile organic compound

BEHP was detected most frequently in soft-shell clams (in 10 of 14 samples); only 20 other samples of various tissue types had detectable concentrations of BEHP (Table 4-45). The highest concentrations of BEHP were detected in a benthic invertebrate composite tissue sample from LDW B1b near RM 0.1 (2,200 µg/kg ww) and a shiner surfperch sample collected from Area T2E (2,100 µg/kg ww).

Table 4-45. Summary of bis(2-ethylhexyl) phthalate data in specific tissue types

Tissue Type	Detection Frequency		Concentration (µg/kg ww)		
	Ratio	%	Minimum Detect	Maximum Detect	RL or Range of RLs[a]
Fish, whole body					
English sole	0/21	0	nd	nd	66 – 3,600
Pacific staghorn sculpin	0/24	0	nd	nd	490 – 5,000
Shiner surfperch	5/27	19	280 J	2,100 J	24 – 3,600
Starry flounder	0/3	0	nd	nd	66 – 67
Fish, fillet					
English sole (with skin)	2/7	29	1,100	1,300 J	67 – 130
English sole (without skin)	0/6	0	nd	nd	3.6 – 16
Pile perch (with skin)	0/1	0	nd	nd	67
Starry flounder (with skin)	0/1	0	nd	nd	67
Striped perch (with skin)	0/1	0	nd	nd	67
Crab, edible meat					
Dungeness crab	0/9	0	nd	nd	16 – 130
Slender crab	0/12	0	nd	nd	66 – 260
Crab, hepatopancreas					
Dungeness crab	0/4	0	nd	nd	24 – 130
Slender crab	1/4	25	100 J	100 J	66 – 230
Crab, whole body (calc'd)[b]					
Dungeness crab	nc	nc	nd	nd	9 – 55
Slender crab	nc	nc	100 JM	100 JM	33 – 100
Invertebrates, whole body					
Amphipod	2/4	50	170	530	24
Benthic invertebrates	5/20	25	1,100 J	2,200 J	2,500
Shellfish					
Mussels, wild (edible meat)	2/22	9	28	190	16
Mussels, transplanted (edible meat)	0/34	0	nd	nd	16 – 26
Soft-shell clam (whole body)	10/14	71	56 J	220 J	490 – 500

Note: In addition to data presented in this table, six English sole whole-body composite samples, nine English sole individual whole-body samples, and seven shiner surfperch composite samples collected by King County in 2006 were analyzed for BEHP. In these samples, one English sole sample had a detected concentration of BEHP (170 µg/kg ww). BEHP was not detected in any other samples, at RLs ranging from 50 to 200 µg/kg ww.

[a] RLs are based only on non-detect samples.

[b] Data from composite hepatopancreas samples were mathematically combined with data from composite samples of edible meat to form composite samples of edible meat plus hepatopancreas. Concentrations in whole-body (i.e., edible meat plus hepatopancreas) crab were calculated assuming 69% (by weight) edible meat and 31% hepatopancreas, based on the relative weights of these tissues in a 16.6-cm Dungeness crab dissected by Windward in 2004 (unpublished data). Detection frequencies were not calculated for these samples because they do not represent individually analyzed samples.

J – estimated concentration
M – calculated whole-body concentration
na – not applicable
nc – not calculated
nd – not detected
RL – reporting limit
ww – wet weight

BEHP RLs were elevated (up to 5,000 µg/kg ww) for some of the English sole, Pacific staghorn sculpin, shiner surfperch, and benthic invertebrate samples because of analytical dilutions. Original RLs in 49 tissue samples of various types were even higher (up to 7,200 µg/kg ww), resulting in the need for re-analysis by a different laboratory using additional sample extract clean-up steps to achieve lower RLs. BEHP was detected in only one of the re-analyzed tissue samples at a much lower concentration of 100 µg/kg ww; RLs for the re-analyses of the remaining 48 samples ranged from 66 to 230 µg/kg ww. BEHP was detected only once in the 2006 King County sampling event in Area T1 (170 µg/kg ww in an English sole composite sample) (Anchor and King County 2007). RLs for the 2006 King County analyses with English sole and shiner surfperch from Area T1 ranged from 50 to 200 µg/kg ww. The reanalysis of the subset of 2004 tissues indicates that 2004 RLs would have been much lower if samples had not been diluted, and the more recent results from 2006 indicate that BEHP is rarely detected in fish.

RLs for pentachlorophenol (PCP) were also elevated (up to 5,800 µg/kg ww) for some of the English sole, Pacific staghorn sculpin, shiner surfperch, and benthic invertebrate samples because of analytical dilutions. Forty-nine samples of various tissue types with RLs of 5,700 to 5,800 µg/kg ww were re-analyzed using a different analytical method to achieve lower RLs. PCP was detected in six of the re-analyzed tissue samples at low concentrations ranging from 1.1 to 2.8 µg/kg ww; RLs for the re-analyses ranged from 3.3 to 11 µg/kg ww. In the original analyses, PCP was detected in two shiner surfperch whole-body samples at 2,200 µg/kg ww. Neither of these detected results were confirmed by the re-analysis; both samples had non-detected concentrations with RLs of 4.5 and 4.6 µg/kg ww. The results from the re-analysis were selected as the final result because of the enhanced sensitivity of the analytical method.

4.2.6.4 Surface water

On multiple occasions, surface water samples were collected in the LDW near three CSO outfalls and analyzed for SVOCs as part of the King County WQA (King County 1999e). Four SVOCs were detected in surface water samples: benzoic acid, BEHP, di-n-butyl phthalate, and phenol (Table 4-46). Detection frequencies of these four chemicals were generally low, ranging from 2 to 38% at the three locations. BEHP and di-n-butyl phthalate were detected at all three locations, with concentrations ranging from 0.14 to 23.8 µg/L and from 0.25 to 1.2 µg/L, respectively. However, it should be noted that phthalates were often detected in the method blank samples, and therefore, these results could be influenced by laboratory contamination. During data validation, detected sample concentrations were compared to the detected method blank concentrations, and sample concentrations less than 10 times the concentration in the associated method blank samples were qualified as nondetected at elevated RLs (151 results for BEHP and 9 results for di-n-butyl phthalate). Benzoic acid was detected at the two downstream locations, with concentrations ranging from 0.98 to 1.5 µg/L. Phenol was detected once, at RM 1.9, at a concentration of 2.01 µg/L. There are no Washington State water quality standards for the SVOC compounds that were detected in LDW surface water samples.

Table 4-46. Summary of surface water data for detected SVOCs at three locations in the LDW

Chemical by Location ID	Detection Frequency		Concentration (µg/L)		
	Ratio	%	Minimum Detect	Maximum Detect	RL or Range of RLs[a]
BRN (RM 1.1)					
Benzoic acid	5/39	13	0.98	1.4	0.94 – 0.97
BEHP	7/39	18	0.21	3.51	0.14 – 1.31
Di-n-butyl phthalate	2/39	5	0.25	0.26	0.24
SWM (RM 1.9)					
Benzoic acid	3/42	7	1.1	1.5	0.94 – 0.97
BEHP	8/42	19	0.302	23.8	0.14 – 0.667
Di-n-butyl phthalate	2/42	5	0.27	0.483	0.24
Phenol	1/42	2	2.01	2.01	0.94 – 0.97
NFK (RM 4.9)					
BEHP	4/13	31	0.14	0.253	0.15 – 0.859
Di-n-butyl phthalate	5/13	38	0.39	1.2	0.24 – 0.46
Phenanthrene					

[a] RLs are based only on non-detect samples.

BEHP -- bis(2-ethylhexyl) phthalate
BRN – near Brandon CSO
RL – reporting limit
RM – river mile

ID – identification
NFK –near Norfolk CSO
SVOC – semivolatile organic compound
SWM – near SW Michigan CSO

PAHs were not detected in any of the surface water samples collected by King County during the WQA (King County 1999e).

Because many organic compounds are not typically detected in ambient grab samples, particularly in the dissolved phase, King County also deployed SPMDs to collect data for selected organic compounds, including PAHs, at 1 and 3 m below the water surface near the Duwamish/Diagonal (RM 0.5) and Brandon (RM 1.1) CSO outfalls, as described in Section 4.1.2.4. Ambient PAH concentrations in water were calculated using the mass of chemicals detected in each SPMD and the chemical-specific partition coefficients between the polyethylene and water, which were derived from laboratory experiments conducted at Battelle's Marine Science Laboratory.

Calculated concentrations of individual PAHs ranged from 0.0027 to 30 ng/L (Table 4-47). None of the calculated water concentrations for PAHs were higher than the lowest RLs achieved for PAHs during the analysis of the grab water samples (100 ng/L). The individual PAHs with the highest calculated concentrations were acenaphthene (30 ng/L), phenanthrene (25 ng/L), fluorene (20 ng/L), and fluoranthene (20 ng/L).

Table 4-47. Calculated PAH concentrations of in LDW surface water collected near the Duwamish/Diagonal and Brandon CSOs

Chemical	Calculated Concentration (ng/L)			
	Duwamish/Diagonal CSO		Brandon CSO	
	1-m Depth	3-m Depth	1-m Depth	3-m Depth
Acenaphthene	30	16	19	18
Acenaphthylene	1.6	0.91	0.80	0.71
Anthracene	2.3	1.6	1.6	1.4
Benzo(a)anthracene	0.36	0.21	0.22	0.17
Benzo(a)pyrene	0.033	0.030	0.024	0.020
Benzo(b)fluoranthene	0.15	0.11	0.095	0.081
Benzo(g,h,i)perylene	0.048	0.061	0.030	0.027
Benzo(k)fluoranthene	0.12	0.027	0.078	0.066
Chrysene	0.40	0.24	0.21	0.17
Dibenzo(a,h)anthracene	0.0044	0.0052	0.0034	0.0027
Fluoranthene	20	7.0	7.5	6.3
Fluorene	20	11	11	9.9
Indeno(1,2,3-c,d)pyrene	0.022	0.019	0.015	0.0093
Naphthalene	16[a]	nd	6.6[a]	nd
Phenanthrene	25	13	13	11

Chemical	Calculated Concentration (ng/L)			
	Duwamish/Diagonal CSO		Brandon CSO	
	1-m Depth	3-m Depth	1-m Depth	3-m Depth
Pyrene	7.0	3.8	3.8	3.1

[a] Results for naphthalene are highly uncertain because high concentrations were found in the trip blanks (Mickelson 2007).

CSO – combined sewer overflow

nd –non-detect

4.2.6.5 Seeps

SVOCs were analyzed in 3 seep water samples collected at T-117; 7 seep water samples collected at Rhône-Poulenc in 1995; 9 and 5 seep water samples collected at GWI in 1998 and 1999, respectively; 17 seep water samples collected at Boeing Plant 2; and 16 seep water samples collected throughout the LDW as part of the RI in 2004 (Map 4-11).

SVOCs were detected in one unfiltered seep water sample from both the T-117 and Boeing Plant 2 sampling events, in two unfiltered seep water samples from the Rhône-Poulenc sampling event, and in unfiltered and filtered seep water samples collected at one location (SP-54 in EEA 2) during the RI event (Table 4-48). SVOCs were also detected in seep water samples collected at GWI; it is not known if these samples were filtered. There are no Washington State water quality standards for the SVOC compounds that were detected in LDW seep water samples.

Table 4-48. Summary of seep water data for detected SVOCs

Sampling Event/Year	Chemical[a]	Detection Frequency		Detected Concentration (µg/L)	RL (µg/L)[b]
		Ratio	%		
Boeing Plant 2 (1995)	fluoranthene	1/17	6	1.3	1.0
	phenanthrene	1/17	6	1.0	1.0
Rhône-Poulenc RFI-3 (1995)	BEHP	2/7	29	13.8 – 27.0	nr
GWI (1998)	1,2-dichlorobenzene	1/9	11	1.3	1.0
GWI (1999)	acenaphthene	1/5	20	1.3	0.5
	1,2-dichlorobenzene	1/5	20	1.0	0.5
T-117 (2003)	BEHP	1/3	33	8.9 J	1.0

Sampling Event/Year	Chemical[a]	Detection Frequency		Detected Concentration (µg/L)	RL (µg/L)[b]
		Ratio	%		
LDW RI (2004)	**Unfiltered**				
	1,2-dichlorobenzene	1/16	6	2.9	1.0
	1,3-dichlorobenzene	1/16	6	58.3	1.0
	1,4-dichlorobenzene	1/16	6	40.2	1.0
	Filtered				
	1,3-dichlorobenzene	1/15	7	3.6	1.0
	1,4-dichlorobenzene	1/15	7	3.9	1.0

[a] Samples collected at Boeing Plant 2, Rhône-Poulenc, and T-117 were unfiltered, as were some of the samples collected for the RI; it is not known if samples collected from GWI were filtered.

[b] RLs are based only on non-detect samples.

BEHP -- bis(2-ethylhexyl) phthalate

GWI – Great Western International

J – estimated concentration

nr – not reported; RLs for non-detected chemicals were not reported for this sampling event.

RFI – RCRA facility investigation

RI – remedial investigation

SVOC – semivolatile organic compound

T-117 – Terminal 117

4.2.6.6 Porewater

The only porewater samples from the LDW that have been analyzed for SVOCs were those collected during the 2004 Rhône-Poulenc sampling event. Table 4-49 summarizes concentrations of SVOCs that were detected in at least one porewater sample from this event. Individual PAHs were detected in the sample from location SH-02 at concentrations up to 0.32 µg/L, and BEHP was detected at locations SH-02, SH-03, SH-04, SH-05, and SH-08 at concentrations up to 390 µg/L (Map 4-12). There are no Washington State water quality standards for SVOCs detected in porewater samples collected at the Rhône-Poulenc site.

Table 4-49. Summary of porewater data from Rhône-Poulenc for detected SVOCs

Chemical	Detection Frequency		Concentration (µg/L)		
	Ratio	%	Minimum Detect	Maximum Detect	RL or Range of RLs[a]
PAHs					
Benzo(a)pyrene	1/9	11	0.0320 J	0.0320 J	0.37
Benzo(b)fluoranthene	1/9	11	0.0390 J	0.0390 J	0.37
Benzo(k)fluoranthene	1/9	11	0.0390 J	0.0390 J	0.37
Fluoranthene	1/9	11	0.26 J	0.26 J	0.37
Pyrene	1/9	11	0.32 J	0.32 J	0.37
Phthalates					

Chemical	Detection Frequency		Concentration (µg/L)		
	Ratio	%	Minimum Detect	Maximum Detect	RL or Range of RLs[a]
BEHP[b]	5/9	56	2.0	390	1.9

[a] RLs are based only on non-detect samples.

[b] BEHP was not detected in two associated method blanks at RLs of 2 µg/L.

BEHP -- bis(2-ethylhexyl) phthalate

J – estimated concentration

PAH – polycyclic aromatic hydrocarbon

RL – reporting limit

SVOC – semivolatile organic compound

4.2.6.7 Summary of SVOC data

PAHs and BEHP were the most frequently detected SVOCs in surface sediment. The mean concentrations of total HPAHs, total LPAHs, and BEHP were 4,000, 700, and 600 µg/kg dw, respectively. Areas with the highest concentrations of PAHs (i.e., greater than the numerical 95th percentile) were generally located in the following areas: the Duwamish/Diagonal EAA, Slip 1, between RM 1.3 and RM 1.5 on the west side, Slip 4, the Boeing Plant 2/Jorgensen Forge EAA, T-117, RM 3.8 EAA, and Slip 6. The majority of locations with the highest BEHP concentrations in surface sediment were between RM 0.4 and RM 0.6 in and near the Duwamish/Diagonal EAA. At locations that were resampled throughout the LDW over time, BEHP concentrations were lower in the more recent samples at 86% of the locations, but interpreting temporal trends from available phthalate data may be confounded by laboratory contamination with phthalates reflected in historical results.

At least one PAH compound was detected in nearly all surface sediment samples, and PAHs were also detected frequently in tissue. In subsurface sediment, areas with the highest total LPAH and HPAH concentrations included RM 0.5 on the west side; RM 1.2 on the east side; RM 1.4 on the west side; RM 2.7 on the west side; Slips 1, 3, 4, and 6; the Duwamish/Diagonal EAA; and the Boeing Plant 2/Jorgensen Forge EAA. Most of the locations with the highest BEHP concentrations in subsurface samples were located between RM 0.4 and RM 0.6, in the vicinity of the Duwamish/Diagonal EAA.

In general, individual PAHs were detected frequently in most tissue types. Although low-level analytical methods were used for all tissue samples collected in 2004, RLs were high for Pacific staghorn sculpin samples because analytical dilutions were used to minimize matrix interferences, resulting in reduced sensitivity of the analysis. PAHs were infrequently detected in Pacific staghorn sculpin at these elevated RLs. Benzoic acid, which was the second most frequently detected SVOC in tissue, was detected frequently in fish, benthic invertebrates, mussels, and clams, but was not detected in any crab samples. Other SVOCs, most notably the phthalates that were commonly detected in sediments, were generally infrequently detected in tissue, even when

highly sensitive analytical methods were applied. The reanalysis of the subset of 2004 tissues for BEHP indicates that 2004 RLs would have been much lower if samples had not been diluted because of matrix interferences, and the more recent results from 2006 indicate that BEHP is rarely detected in fish.

SVOCs were detected infrequently in surface water, seep, and porewater samples collected from the LDW. There are no Washington State water quality standards available for comparison to the SVOCs detected in the LDW.

4.2.7 Organochlorine pesticides

This section summarizes the nature and extent of organochlorine pesticide contamination in surface sediment, subsurface sediment, tissue, surface water, and seeps. None of the porewater samples collected from the LDW were analyzed for organochlorine pesticides.

Some of the pesticide data are uncertain because of analytical interference from the presence of PCB congeners, as identified by both the analytical laboratory and the data validators during the pesticide analyses of benthic invertebrate, fish, and crab tissue as part of the RI. Samples with this probable interference were assigned a JN-qualifier, indicating the results are highly uncertain and biased high. A more detailed discussion of this data quality issue is provided in Section 4.1.3.2.

4.2.7.1 Surface sediment

Table 4-50 presents a summary of organochlorine pesticides detected in surface sediment. Most organochlorine pesticides were detected in 6% or fewer of the surface sediment samples in which they were analyzed. 4,4'-DDD, 2,4'-DDT, 4,4'-DDT, and gamma-chlordane were the organochloride pesticides most frequently detected in surface sediment, in 34%, 31%, 21%, and 17% of the samples in which they were analyzed, respectively. Other DDT compounds were detected at frequencies of 5 to 31%, and other chlordane compounds were detected at frequencies of 9 to 11%.

Table 4-50. Summary of surface sediment data for detected organochlorine pesticides

Pesticide	Detection Frequency		Concentration (µg/kg dw)		
	Ratio	%	Minimum Detect	Maximum Detect	RL or Range of RLs[a]
DDT and metabolites					
2,4'-DDD	5/93	5	1.6 JN	10 JN	0.97 – 34
2,4'-DDE	2/93	2	2.8 JN	11 JN	0.97 – 34
2,4'-DDT	29/93	31	0.24 JN	11 JN	1.9 – 460
4,4'-DDD	66/193	34	0.29 JN	840	0.81 – 540
4,4'-DDE	29/193	15	0.28 JN	370 J	0.81 – 800
4,4'-DDT	40/193	21	0.48 JN	1,700	0.81 – 56

Pesticide	Detection Frequency		Concentration (µg/kg dw)		
	Ratio	%	Minimum Detect	Maximum Detect	RL or Range of RLs[a]
Total DDTs	76/193	39	0.72 JN	2,900 J	nc
Chlordane compounds					
alpha-Chlordane	13/149	9	0.10 JN	36	0.72 – 30
gamma-Chlordane	25/149	17	0.20 JN	200	0.72 – 96
Chlordane	5/44	11	25	62	7.7 – 330
Total chlordanes	28/149	19	0.20 JN	230	nc
Aldrin/dieldrin					
Aldrin	4/193	2	0.014 J	1.6 JN	0.40 – 56
Dieldrin	8/193	4	0.099 J	280	0.81 – 91
Other pesticides					
alpha-BHC	3/193	2	0.14 J	1.8 JN	0.40 – 56
beta-BHC	4/193	2	0.087 J	13	0.40 – 56
gamma-BHC	11/193	6	0.050 J	6.7 JN	0.40 – 56
delta-BHC	3/155	2	0.081 J	11 JN	0.40 – 56
alpha-Endosulfan	9/147	6	0.18 JN	71 JN	0.40 – 100
beta-Endosulfan	3/149	2	0.47 J	10 JN	0.81 – 200
Endosulfan	1/46	2	0.11 J	0.11 J	0.81 – 56
Endosulfan sulfate	3/191	2	0.63 JN	25 JN	0.81 – 200
Endrin	4/193	2	0.99 JN	9.1	0.81 – 200
Endrin aldehyde	6/182	3	0.28 JN	130	0.81 – 250
Endrin ketone	7/137	5	0.83 JN	110 JN	0.81 – 200
Heptachlor	6/193	3	0.12 J	5.2	0.43 – 70
Heptachlor epoxide	5/193	3	0.47 JN	4.9 JN	0.40 – 510
Methoxychlor	11/193	6	0.34 JN	99	0.97 – 330
Mirex	3/93	3	0.29 JN	1.0 JN	0.97 – 34
Toxaphene	2/191	1	340 JN	6,300 JN	1.0 – 4,300

[a] RLs based only on non-detect samples.

BHC – benzene hexachloride
dw – dry weight
J – estimated concentration
LDW – Lower Duwamish Waterway
N – tentative identification
nc – not calculated
RL – reporting limit

The highest concentration of total DDTs (2,900 µg/kg ww) in surface sediment was detected in sample DR178 collected in Slip 4 (Map 4-58). This location also had a high PCB concentration in surface sediment (7,000 µg/kg dw), indicating a potential for analytical interference. Other areas with relatively high total DDT concentrations in surface sediment were located between RM 0.4 and RM 0.6 on the east side of the

LDW (Duwamish/Diagonal EAA), the inlet at RM 2.2 on the west side of the LDW (EAA 2), and at the mouth of Slip 3.

4.2.7.2 Subsurface sediment

Organochlorine pesticides were analyzed in a subset of subsurface sediment samples collected in the LDW. Of the samples analyzed for organochlorine pesticides, 17% had one or more detected DDT compounds (Table 4-51). Other organochlorine pesticides were detected in 17% or fewer of the samples in which they were analyzed.

Table 4-51. Summary of subsurface sediment data for organochlorine pesticides from any interval

Pesticide	Detection Frequency		Concentration (µg/kg dw)		
	Ratio	%	Minimum Detect	Maximum Detect	RL or Range of RLs[a]
DDT and metabolites					
2,4'-DDD	0/40	0	nd	nd	1.9 – 29
2,4'-DDE	0/40	0	nd	nd	1.9 – 100
2,4'-DDT	0/40	0	nd	nd	1.9 – 29
4,4'-DDD	13/95	14	0.27 J	10	1.5 – 39
4,4'-DDE	14/95	15	0.45 J	18 J	0.62 – 90
4,4'-DDT	5/95	5	0.64 J	2.8	0.96 – 160
Total DDTs	16/95	17	0.27 J	20 J	nc
Chlordane compounds					
alpha-Chlordane	1/89	1	0.44 J	0.44 J	0.052 – 14
gamma-Chlordane	4/87	5	0.22 J	0.71 J	0.92 – 82
Chlordane	1/6	17	29.0	29.0	7.5 – 35
Total chlordanes	4/89	4	0.22 J	1.59 J	nc
Aldrin/dieldrin					
Aldrin	1/95	1	1.5 J	1.5 J	0.33 – 16.00
Dieldrin	3/95	3	0.80 J	1.8	0.14 – 95.00
Other pesticides					
alpha-BHC	3/71	4	0.20 J	0.47 J	0.086 – 14
beta-BHC	0/66	0	nd	nd	0.96 – 24
gamma-BHC	0/95	0	nd	nd	0.099 – 14
delta-BHC	7/48	15	7.0	1,100	0.96 – 14
alpha-Endosulfan	0/62	0	nd	nd	0.96 – 14
beta-Endosulfan	0/62	0	nd	nd	1.90 – 29
Endosulfan	0/4	0	nd	nd	1.5 – 1.6
Endosulfan sulfate	0/62	0	nd	nd	1.90 – 69
Endrin	0/66	0	nd	nd	1.5 – 150

Pesticide	Detection Frequency		Concentration (µg/kg dw)		
	Ratio	%	Minimum Detect	Maximum Detect	RL or Range of RLs[a]
Endrin aldehyde	1/66	2	10	10	1.5 – 29
Endrin ketone	0/62	0	nd	nd	1.90 – 29
Heptachlor	2/95	2	0.25 J	1.3	0.097 – 14
Heptachlor epoxide	0/66	0	nd	nd	0.96 – 120
Methoxychlor	0/66	0	nd	nd	1.0 – 140
Mirex	0/40	0	nd	nd	1.9 – 29
Toxaphene	0/66	0	nd	nd	10 – 1,400

[a] RLs are based only on non-detect samples.

BHC – benzene hexachloride
dw – dry weight
J – estimated concentration
LDW – Lower Duwamish Waterway
N – tentative identification
nc – not calculated
nd – not detected at any location
RL – reporting limit

Total DDT concentrations in subsurface sediment ranged from 0.27 to 20 µg/kg dw, with the three highest concentrations (17, 18, and 20 µg/kg dw) detected during the EPA SI sampling conducted in 1998. These concentrations were detected in the 2-to-4-ft interval of core DR025 at RM 1.2, in the 0-to-2-ft interval of core DR137 at RM 2.2 near EAA 2, and in the 2-to-4-ft interval of core DR021 at RM 0.9 at the mouth of Slip 1, respectively.

4.2.7.3 Tissue

Table 4-52 presents the detection frequency and range of organochlorine pesticide concentrations in LDW tissue samples. 2-4′-DDT and gamma chlordane were the most frequently detected pesticides (91 and 85%, respectively, of all tissue samples in which they were analyzed). Other DDT compounds were detected at a frequency of 8 to 78%, and other chlordane compounds were detected at a frequency of 0 to 33% in all tissue samples in which they were analyzed. Heptachlor epoxide, alpha-endosulfan, and beta-BHC were detected in 33%, 26%, and 26%, respectively, of all tissue samples in which they were analyzed. The remaining organochlorine pesticides were detected in 16% or fewer of all tissue samples in which they were analyzed. The maximum tissue concentrations of most organochlorine pesticides have a JN-qualifier, as shown in Table 4-52. This qualifier indicates probable interference in the analysis from PCB congeners, resulting in data that are highly uncertain and biased high.

Table 4-52. Summary of organochlorine pesticide data in all tissue samples

Chemical	Detection Frequency		Concentration (µg/kg ww)		
	Ratio	%	Minimum Detect	Maximum Detect	RL or Range of RLs[a]
DDT and metabolites					

Chemical	Detection Frequency		Concentration (µg/kg ww)		
	Ratio	%	Minimum Detect	Maximum Detect	RL or Range of RLs[a]
2,4'-DDD	46/179	26	0.49 J	57 JN	0.18 – 88
2,4'-DDE	14/179	8	0.72 J	110 JN	0.26 – 45
2,4'-DDT	163/179	91	1.8 J	440 JN	0.22 – 34
4,4'-DDD	126/212	59	0.23 JN	20 JN	0.13 – 18
4,4'-DDE	155/212	73	0.70 JN	46 JN	1.0 – 14
4,4'-DDT	165/212	78	0.56 J	470 JN	0.38 – 82
Total DDTs	186/212	88	1.1	1,020 JN	nc
Chlordane compounds					
alpha-Chlordane	44/188	23	0.60 JN	12 JN	0.36 – 18
gamma-Chlordane	160/188	85	0.52 J	330 JN	0.14 – 56
Oxychlordane	6/18	33	0.23 J	1.5 J	0.12 – 1.0
cis-Nonachlor	0/18	0	nd	nd	0.12 – 2.3
trans-Nonachlor	6/18	33	0.70 J	3.2	0.14 – 1.5
Chlordane	0/24	0	nd	nd	6.7
Total chlordane	165/188	88	0.24 J	330 JN	nc
Aldrin/dieldrin					
Aldrin	6/212	3	0.61 J	6.2 JN	0.20 – 18
Dieldrin	12/212	6	0.76 J	5.7	0.11 – 34
Other pesticides					
alpha-BHC	13/212	6	0.20 JN	21 JN	0.16 – 18
alpha-Endosulfan	55/212	26	0.21 J	30 JN	0.13 – 15
beta-BHC	55/212	26	0.27 JN	15 JN	0.38 – 18
beta-Endosulfan	33/212	16	0.94 J	44 JN	0.35 – 18
delta-BHC	3/212	1	0.51 JN	2.2 JN	0.34 – 18
Endosulfan sulfate	4/212	2	0.73 J	2.3	0.27 – 190
Endrin	34/212	16	0.10 JN	40 JN	0.099 – 72
Endrin aldehyde	31/212	15	0.42 JN	78 JN	0.17 – 18
Endrin ketone	4/179	2	0.60 JN	8.4 JN	0.29 – 18
gamma-BHC	17/212	8	0.51 JN	7 JNM	0.28 – 18
Heptachlor	10/212	5	0.96 J	9.7 JN	0.45 – 18
Heptachlor epoxide	69/212	33	0.72 J	45 JN	0.15 – 18
Methoxychlor	11/212	5	0.49	130 JN	0.27 – 23
Mirex	0/179	0	nd	nd	0.27 – 18
Toxaphene	0/212	0	nd	nd	10 – 4,800

[a] RLs are based only on non-detect samples.

BHC – benzene hexachloride
J – estimated concentration
M – calculated whole-body concentration
N – tentative identification
nc – not calculated
nd – not detected
RL – reporting limit
ww – wet weight

4.2.7.4 Surface water

Surface water grab samples collected as part of the King County WQA (King County 1999e) were not analyzed for organochlorine pesticides. Instead, organochlorine pesticides were analyzed in the SPMDs deployed at 1 and 3 m below the water surface just offshore of the Duwamish/Diagonal and Brandon CSOs (at RM 0.5 and RM 1.1, respectively), as described in Section 4.1.2.4. Organochlorine pesticide concentrations in the SPMDs were used to calculate concentrations in surface water. Calculated concentrations were very low, ranging from 0.000034 to 0.033 ng/L (Table 4-53).

Table 4-53. Calculated organochlorine pesticide concentrations in surface water collected near Duwamish/Diagonal and Brandon CSOs

Pesticide	Calculated Concentration (ng/L)			
	Duwamish/Diagonal CSO		Brandon CSO	
	1-m Depth	3-m Depth	1-m Depth	3-m Depth
2,4′-DDD	0.013	0.0067	0.0079	0.0063
4,4′-DDD	0.015	0.0091	0.010	0.0085
4,4′-DDE	0.0065	0.0019	0.00093	0.000034
4,4′-DDT	nd	nd	nd	nd
Aldrin	0.0059	0.0040	0.0037	0.0030
alpha-Chlordane	0.0026	0.0017	0.0015	0.0013
gamma-Chlordane	0.0013	0.000077	0.00011	nd
Dieldrin	0.033	0.023	nd	0.020
Endrin	0.014	0.0089	0.0064	nd

CSO – combined sewer overflow
nd – not detected
SPMD – semipermeable membrane device
WQA – water quality assessment

4.2.7.5 Seeps

The only LDW seep water samples analyzed for organochlorine pesticides were those collected as part of the RI (Windward 2004b). Heptachlor epoxide (a degradation product of heptachlor) was detected in both the filtered and unfiltered seep water samples collected at Seep 39, at concentrations of 0.0090 and 0.0076 µg/L, respectively. No other organochlorine pesticides were detected in seep water samples.

4.2.7.6 Summary

Organochlorine pesticides were analyzed in a subset of surface sediment, subsurface sediment, tissue, and seep water samples. Analytical interference in the organochlorine pesticide analyses from the presence of PCBs in the samples resulted in high uncertainty and high bias in some of the organochlorine pesticide data, particularly samples with high PCB concentrations.

4,4′-DDD, 2,4′-DDT, 4,4′-DDT, and gamma-chlordane were the organochlorine pesticides most frequently detected in surface sediment, in 34, 31, 21, and 17% of the samples in which they were analyzed, respectively. The highest total DDT concentration in surface sediment (2,900 µg/kg dw) was detected in a sample collected from Slip 4; this sample also had a relatively high total PCB concentration (7,000 µg/kg dw). Organochlorine pesticides were infrequently detected in subsurface sediment; 4,4′-DDE was most frequently detected (in 15% of the samples in which it was analyzed), at a maximum concentration of 18 µg/kg dw.

DDT and chlordane compounds were frequently detected in tissue samples, but there is high uncertainty in these data because of analytical interference by PCBs. Organochlorine pesticide concentrations in surface water were estimated from SPMDs deployed at two locations in the LDW. Calculated concentrations of organochlorine pesticides in surface water ranged from 0.000034 to 0.033 ng/L. Organochlorine pesticides were detected in one seep location sampled as part of the RI (Seep 39 near RM 4.3 on the western shoreline); heptachlor epoxide was detected in both the filtered and unfiltered samples from this location at concentrations of 0.0090 and 0.0076 µg/L, respectively.

4.2.8 VOCs

This section summarizes the nature and extent of VOC contamination in surface sediment, subsurface sediment, seep water, and porewater samples collected from the LDW.

4.2.8.1 Surface sediment

VOCs were analyzed in 36 surface sediment samples collected throughout the LDW as part of the 1998 EPA SI, in 3 surface sediment samples collected at RM 3.3 as part of the 1995 Boeing Plant 2 RFI, and in 2 surface sediment samples collected near RM 0.5 as part of the 1991 Harbor Island RFI.

The highest VOC concentration (dichloromethane at 1,600 µg/kg dw) was detected in sample DR008 collected near RM 0.5 (Table 4-54). The second highest VOC concentration (acetone at 1,000 µg/kg dw) was detected in sample DR047 collected near RM 0.9 south of Kellogg Island. The remaining detected VOC concentrations were less than 160 µg/kg dw.

Table 4-54. Summary of surface sediment data for detected VOCs

Chemical	Detection Frequency		Concentration (µg/kg dw)		
	Ratio	%	Minimum Detect	Maximum Detect	Range of RLs[a]
Acetone	3/41	7	110 J	1,000 J	11 – 21,000
Carbon disulfide	12/41	29	0.84 J	4.0 J	1.4 – 1,100
p-Cymene	3/36	8	1.6 J	25	1.5 – 530
Dichloromethane	1/41	2	1,600	1,600	2.8 – 21
Methyl ethyl ketone	15/41	37	5.3	35	3.0 – 1,100
Tetrachloroethene	2/41	5	0.21 J	0.52 J	1.4 – 530
Toluene	3/41	7	1.0 J	2.2 J	1.4 – 530

[a] RLs based only on non-detect samples.

dw – dry weight

RL – reporting limit

VOC – volatile organic compounds

4.2.8.2 Subsurface sediment

VOCs were analyzed in subsurface core samples collected from eight locations between RM 3.3 and RM 3.5 as part of four sampling events. As part of the RI, one subsurface core was analyzed for VOCs because of elevated readings from the photoionization detector during core processing. VOCs were not analyzed in other cores because the detector did not indicate that they were present at elevated concentrations. This core was collected at LDW-SC49 between RM 3.5 and RM 3.6 and analyzed at 1-ft intervals to a depth of 12 ft below the sediment surface.

Detected VOC concentrations in subsurface sediment ranged from 1.5 to 200,000 µg/kg dw (Table 4-55). The most frequently detected VOCs were acetone, carbon disulfide, and methyl ethyl ketone, with detection frequencies of 81%, 50%, and 44% of the samples in which they were analyzed, respectively. With the exception of these chemicals, individual VOCs were detected in less than 25% of samples.

Table 4-55. Summary of subsurface sediment data for detected VOCs

Chemical	Detection Frequency		Concentration (µg/kg dw)		
	Ratio	%	Minimum Detect	Maximum Detect	Range of RLs[a]
1,1-Dichloroethene	2/32	6	120	320	0.90 – 9.6
1,2,4-Trimethylbenzene	3/24	13	14 J	580 J	0.90 – 39
1,3,5-Trimethylbenzene	4/24	17	1.9	350 J	0.90 – 39
Acetone	26/32	81	8.20	640	6.6 – 37
Benzene	5/32	16	4.4	62	0.90 – 9.6
Carbon disulfide	16/32	50	1.30	18	0.90 – 9.6
Chloroform	1/32	3	1.8	1.8	0.90 – 9.6

Chemical	Detection Frequency		Concentration (µg/kg dw)		
	Ratio	%	Minimum Detect	Maximum Detect	Range of RLs[a]
cis-1,2-Dichloroethene	7/32	22	2.2	200,000	0.90 – 9.6
p-Cymene	5/24	21	4.1 J	130 J	0.90 – 39
Dichloromethane	10/32	31	1.5 J	5.90	1.90 – 11
Ethylbenzene	3/58	5	27	360 J	0.75 – 9.6
Isopropylbenzene	3/24	13	3.9 J	87 J	0.90 – 39
Methyl ethyl ketone	14/32	44	9.60	100	4.60 – 28
n-Butylbenzene	2/24	8	58	78 J	0.90 – 39
n-Propylbenzene	3/24	13	1.8 J	100 J	0.90 – 39
sec-Butylbenzene	2/24	8	57	57	0.90 – 39
tert-Butylbenzene	1/24	4	6.1	6.1	0.90 – 39
Toluene	7/32	22	1.8	8,300	0.90 – 9.6
trans-1,2-Dichloroethene	3/32	9	22	1,700 J	0.90 – 9.6
Trichloroethene	4/58	7	3.6 J	23	0.37 – 9.6
Vinyl chloride	5/32	16	29	60,000	0.90 – 11
Xylene (ortho)	3/52	6	31	610 J	0.90 – 9.6
Xylene (meta and para)	3/41	7	74 J	4,700	1.1 – 9.6
Total xylenes (calc'd)	3/52	6	105 J	5,300 J	nc

[a] RLs based only on non-detect samples.

dw – dry weight

J – estimated concentration

RL – reporting limit

VOC – volatile organic compounds

The highest concentrations of VOCs were detected in the 9-to-10-ft, 10-to-11-ft, and 11-to-12-ft intervals of core LDW-SC49b, collected in the navigation channel at RM 3.55. The VOCs with the highest concentrations in this core were cis-1,2-dichloroethene (200,000 µg/kg dw and 140,000 µg/kg dw in the 10-to-11-ft and 11-to-12-ft intervals, respectively) and vinyl chloride (60,000 µg/kg dw in the 11-to-12-ft interval). These two VOCs were not detected in samples collected from the top 6 ft of the core.

4.2.8.3 Seeps

VOCs have been analyzed in seep water samples collected at three sites: Boeing Plant 2 (in 1995), Rhône-Poulenc (in 1995), and Great Western International (Linda to recheck) (during 10 sampling events from 1994 to 1999). In addition, 16 seep water samples collected from throughout the LDW were analyzed for VOCs as part of the RI (Map 4-11). Table 4-56 summarizes data for detected VOCs in seep water samples from the LDW investigations listed above. No VOCs were detected in seep water samples collected from Rhône-Poulenc.

Table 4-56. Concentrations of VOCs with at least one detected value in seep samples

Chemical	Detection Frequency		Concentration (µg/L)		
	Ratio	%	Minimum Detect	Maximum Detect	RL or Range of RLs[a]
Boeing Plant 2 – 1995					
Benzene	1/18	6	2.2	2.2	1
cis-1,2-Dichloroethene	6/18	33	1.2	40	1
Trichloroethene	3/18	17	1.9	13	1
Vinyl chloride	4/18	22	3.1	36	0.01 – 2
Great Western – April 1994					
1,1-Dichloroethane	3/6	50	1.0	2.0	1.0
1,1-Dichloroethene	1/6	17	1.0	1.0	1.0
Total 1,2-dichloroethene	3/6	50	7.0	110	1.0
Tetrachloroethene	2/6	33	3.0	760	1.0
Trichloroethene	3/6	50	1.0	370	1.0
Vinyl chloride	1/6	17	1.0	1.0	1.0
Great Western – July 1994					
1,1-Dichloroethane	1/2	50	1.0	1.0	1.0
1,2-Dichloroethene (total)	2/2	100	3.0	40	na
Tetrachloroethene	2/2	100	2.0	220	na
Trichloroethene	1/2	50	94	94	1.0
Vinyl chloride	1/2	50	3.0	3.0	1.0
Great Western – November 1994					
1,2-Dichloroethene (total)	2/7	29	1	50	1.0
Tetrachloroethene	2/7	29	1	440	1.0
Trichloroethene	1/7	14	130	130	1.0
Great Western – May 1995					
1,1,1-Trichloroethane	1/7	14	1.0	1.0	1.0
1,1-Dichloroethane	2/7	29	1.0	2.0	1.0
1,2-Dichloroethane	1/7	14	1.0	1.0	1.0
1,2-Dichloroethene (total)	3/7	43	3.0	150	1.0
Tetrachloroethene	2/7	29	6.0	710	1.0
Trichloroethene	2/7	29	8.0	480	1.0
Vinyl chloride	2/7	29	2.0	2.0	1.0
Great Western – 1995 annual monitoring					
1,1-Dichloroethane	2/7	29	1.0	1.0	1.0
1,2-Dichloroethane	1/7	14	1.0	1.0	1.0

Chemical	Detection Frequency		Concentration (µg/L)		
	Ratio	%	Minimum Detect	Maximum Detect	RL or Range of RLs[a]
1,2-Dichloroethene (total)	4/7	57	1.0	160	1.0
Tetrachloroethene	2/7	29	4.0	610	1.0
Trichloroethene	2/7	29	4.0	360	1.0
Vinyl chloride	2/7	29	1.0	1.0	1.0
Great Western – 1996 annual monitoring					
1,1-Dichloroethane	1/5	20	1.0	1.0	1.0
1,2-Dichloroethane	1/5	20	8.0	8.0	1.0
1,2-Dichloroethene (total)	3/5	60	8.0	310	1.0
1,2-Dichloropropane	1/5	20	1.0	1.0	1.0
Tetrachloroethene	2/5	40	13	79	1.0
Toluene	1/5	20	2.	2.0	1.0
Trichloroethene	2/5	40	7.0	58	1.0
Vinyl chloride	1/5	20	3.0 J	3.0 J	1.0
Total xylenes	1/5	20	1.0	1.0	1.0
Great Western – 1997 annual monitoring					
1,1-Dichloroethane	1/4	25	1.0	1.0	1.0
1,2-Dichloroethene (total)	2/4	50	37	470	1.0
Tetrachloroethene	2/4	50	19	150	1.0
Trichloroethene	2/4	50	7.0	200	1.0
Great Western – 1998 annual monitoring					
1,1-Dichloroethane	2/9	22	4.1	53	1.0
1,1-Dichloroethene	1/9	11	18	18	1.0
1,2-Dichloroethane	1/9	11	8.5	8.5	1.0
1,2-Dichloropropane	1/9	11	2.7	2.7	1.0
Benzene	1/9	11	28	28	1.0
Chlorobenzene	1/9	11	4.1	4.1	1.0
cis-1,2-Dichloroethene	3/9	33	41	3,300	1.0
Tetrachloroethene	4/9	44	1.3	290	1.0
trans-1,2-Dichloroethene	2/9	22	3.5	46	1.0
Trichloroethene	3/9	33	4	170	1.0
Vinyl chloride	2/9	22	1.0	760	1.0
Xylene (ortho)	1/9	11	1.4	1.4	1.0
Total xylenes	1/9	11	1.4	1.4	nc
Great Western – 1999 annual monitoring					
1,1-Dichloroethane	1/5	20	58	58	1.0 – 2.0

Chemical	Detection Frequency		Concentration (µg/L)		
	Ratio	%	Minimum Detect	Maximum Detect	RL or Range of RLs[a]
Benzene	1/5	20	28	28	1.0 – 2.0
cis-1,2-Dichloroethene	3/5	60	5.6	3,200	1.0
Tetrachloroethene	2/5	40	3.7	190	1.0 – 20
trans-1,2-Dichloroethene	2/5	40	12	27	1.0
Trichloroethene	2/5	40	2.1	270	1.0 – 20
Vinyl chloride	1/5	20	3,500	3,500	1.0 – 2.0
Great Western – 1998 Embayment Study					
1,1-Dichloroethane	5/10	50	1.5	62	1.0
1,1-Dichloroethene	3/10	30	1.0	27	1.0 – 10
1,2,4-Trimethylbenzene	1/4	25	4.1	4.1	10
1,2-Dichloroethane	2/10	20	6.6	27	1.0 – 10
1,2-Dichloropropane	1/10	10	16	16	1.0 – 10
Acetone	3/8	38	1.0	6.4	5.0 – 11
Benzene	2/10	20	21	36	1.0 – 10
Chlorobenzene	1/10	10	8.8	8.8	1.0 – 10
cis-1,2-Dichloroethene	4/4	100	29	5,400	na
Tetrachloroethene	4/10	40	2.6	180	1.0 – 10
Toluene	1/10	10	2.7	2.7	1.0 – 10
trans-1,2-Dichloroethene	3/10	30	16	110	1.0
Trichloroethene	3/10	30	6.0	110	1.0 – 10
Vinyl chloride	4/10	40	3.8	1,600	1.0 – 1.0
Ortho-xylene	1/10	10	2.8	2.8	1.0 – 10
Total xylenes	1/10	10	2.8	2.8	nc
LDW RI – 2004					
Carbon disulfide	2/16	13	1.1	2.4	1.0
Chlorobenzene	1/16	6	6.5	6.5	1.0

[a] RLs based only on non-detect samples.

J – estimated concentration

LDW – Lower Duwamish Waterway

na – not applicable

nc – not calculated (RLs not available)

nr – not reported (only detected chemicals reported in the data report; RLs for nondetected results not available)

RI – remedial investigation

RL – reporting limit

VOC – volatile organic compound

At Boeing Plant 2, four VOCs were detected in seep samples. The highest VOC concentration (40 µg/L of cis-1,2-dichloroethene) was detected in seep sample SE-11001.

At GWI, numerous VOCs were detected during the various sampling events. The VOCs with the highest detected concentrations were cis-1,2-dichloroethene (5,400 µg/L in S-13), tetrachloroethene (760 µg/L in S-2), trichloroethene (480 µg/L in S-2), and vinyl chloride (3,500 µg/L in S-13). The seep locations where VOCs were most frequently detected and where the highest concentrations were found were S-1, S-2, S-4, and S-13.

As part of the RI, two VOCs were detected in seep water samples. Carbon disulfide was detected in seep water from two locations (SP- 80 at 1.1 µg/L and SP-54 at 2.4 µg/L), and chlorobenzene was detected in seep water from location SP-54 at a concentration of 6.5 µg/L.

4.2.8.4 Porewater

VOCs were analyzed in porewater samples collected from GWI and Boeing Plant 2/Jorgensen Forge as part of the RI in 2005 and from Rhône-Poulenc during a sampling event in 2004 (Map 4-12). VOCs analyzed in the nine porewater samples collected during the Rhône-Poulenc sampling event were benzene, toluene, ethylbenzene, and xylene; none of these VOCs was detected.

Table 4-57 summarizes data for detected VOCs in porewater samples collected using peepers in shallow nearshore areas offshore of GWI and Boeing Plant 2/Jorgensen Forge as part of the RI. Porewater samples were also collected from deeper, offshore areas using piezometers (see Map 4-12); VOCs were not detected in any porewater samples collected using piezometers at either site.

Table 4-57. Summary of VOC data for porewater samples collected from locations using peepers

Chemical	Detection Frequency[a]		Concentration (µg/L)		
	Ratio	%	Minimum Detect	Maximum Detect	RL or Range of RLs[b]
Great Western					
1,1-Dichloroethane	8/10	80	0.4	16	0.2
1,1-Dichloroethene	3/10	30	0.3	4.9	0.2 – 0.4
1,2-Dichlorobenzene	3/10	30	0.5	1.2	0.2 – 0.4
1,2-Dichloroethane	2/10	20	7.4	15	0.2 – 0.4
1,2-Dichloropropane	2/10	20	1.7	2.5	0.2 – 0.4
1,4-Dichlorobenzene	2/10	20	0.3	0.3	0.2 – 0.4
Benzene	1/10	10	9.4	9.4	0.2 – 4.7
Carbon disulfide	5/10	50	0.2	0.7	0.2 – 0.4
Chlorobenzene	4/10	40	0.3	1.4	0.2 – 0.4
cis-1,2-Dichloroethene	10/10	100	0.5	2,900	na
Isopropylbenzene	2/10	20	0.2	0.3	0.2 – 0.4

Chemical	Detection Frequency[a]		Concentration (µg/L)		
	Ratio	%	Minimum Detect	Maximum Detect	RL or Range of RLs[b]
Tetrachloroethene	2/10	20	0.4	1.1	0.2 – 0.4
Toluene	5/10	50	0.3	3.5	0.2 – 0.4
trans-1,2-Dichloroethene	7/10	70	0.3	21 J	0.2 – 0.4
Trichloroethene	4/10	40	0.4	2.5	0.2 – 0.4
Vinyl chloride	10/10	100	0.4	2,500	na
Boeing Plant 2/Jorgensen Forge					
1,1-Dichloroethane	1/10	10	0.3	0.3	0.2
cis-1,2-Dichloroethene	7/10	70	0.2	1.7	0.2
Trichloroethene	2/10	20	0.2	0.2	0.2
Vinyl chloride	2/10	20	1.1	13	0.2

[a] Two of the 10 samples from GWI were field replicates collected at PE-08 and two of the 10 samples from Boeing Plant 2 were field replicates collected at PE-10.

[b] RLs are based only on non-detect samples.

J – estimated concentration

na – not applicable

nd – not detected

RL – reporting limit

VOC – volatile organic compound

The highest concentrations of VOCs were detected in samples collected from GWI. Vinyl chloride and cis-1,2-dichloroethene were detected in all samples collected using peepers and had the highest concentrations. Maximum concentrations of vinyl chloride and cis-1,2-dichloroethene were 2,500 and 2,900 µg/L, respectively, at location PE-06 (Map 4-12).

At Boeing Plant 2/Jorgensen Forge, four VOCs were detected in porewater samples (Table 4-57). Cis-1,2-dichloroethene was the most frequently detected VOC, in 7 of the 10 samples collected using peepers, at concentrations ranging from 0.2 to 1.7 µg/L. The highest VOC concentration (13 µg/L of vinyl chloride) was detected in a porewater sample collected from location PE-11 (Map 4-12).

4.2.8.5 Summary

Seven VOCs were detected in surface sediment samples collected from the LDW. The highest concentrations were 1,600 µg/kg dw for dichloromethane and 1,000 µg/kg dw for acetone. All other VOCs were detected at concentrations less than 35 µg/kg dw. In subsurface sediment collected between RM 3.5 and RM 3.6 as part of the RI, high concentrations of cis-1,2-dichloroethene and vinyl chloride (200,000 and 60,000 µg/kg dw, respectively) were detected at depth (between 10 and 12 ft). These VOCs were not detected within the top 6 ft of sediment in this core.

Few VOCs were detected in seep water samples collected from Boeing Plant 2, Rhône-Poulenc, and at locations throughout the LDW as part of the RI; concentrations

were low (≤ 40 µg/L). Numerous VOCs were frequently detected in seep water samples from GWI. VOCs with the highest concentrations were cis-1,2-dichloroethene (5,400 µg/L) and vinyl chloride (3,500 µg/L).

Porewater samples were collected and analyzed for VOCs at Boeing Plant 2/Jorgensen Forge and GWI as part of the RI. At GWI, cis-1,2-dichloroethene and vinyl chloride were detected in all 10 nearshore samples, at maximum concentrations of 2,900 and 2,500 µg/L, respectively. At Boeing Plant 2/Jorgensen Forge, VOCs were detected at low concentrations in porewater samples (≤ 13 µg/L).

4.3 CHEMICAL PATTERNS AT DEPTH AND SEDIMENT TRANSPORT

This section combines information on depth patterns in sediment chemistry (from sediment cores) with net sedimentation and erosion estimates from the STM to determine whether vertical patterns of total PCB concentrations are consistent with the predictions of the STM and the CSM. This analysis is focused on total PCBs because PCBs are persistent, have a relatively clear history of use, and are widespread.

The CSM for sediment transport was refined based on results of the STM, which generates estimates of net sedimentation and net erosion for each LDW model cell (Maps 3-8 and 3-11).[75] Results of the STM indicate that the LDW is net depositional over annual time scales, although rates of net sedimentation vary across the LDW from < 0.5 cm/yr to approximately 145 cm/yr (the latter value only in the Upper Turning Basin), with typical rates for most of the LDW between 0.5 and 3 cm/yr. The STM also identified areas of the LDW where erosion could occur during high-flow events.[76] According to the STM, high-flow events are unlikely to erode sediment to depths greater than approximately 15 cm and net erosion up to this depth is expected in relatively small areas (Map 3-8). Other physical events that could potentially expose subsurface sediments deeper than approximately 15 cm include stationary tugboats using high thrust, dredge events, or construction events; these events are expected to be highly localized and of short duration. Erosive forces from tugboats transiting the navigation channel tend to repeatedly mix the upper 1 to 2 cm of sediment in the areas frequented by these boats.

Much of the sediment contamination in the LDW, and particularly PCB contamination, is believed to have originated from historical sources in the LDW (see

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[75] The STM models conditions within 2,000 horizontal grid cells, with seven lateral (i.e., from bank to bank) grid cells at most locations in the LDW. Cell lengths (north-south axis) ranged from 20 to 150 m. For each model cell, the STM estimated net sedimentation rates on annual time scales for a 30-year simulation and net erosion depths over the course of a high-flow event.

[76] High-flow events include above-average flows with return periods of 2 years (8,400 cfs), 10 years (10,800 cfs), and 100 years (12,000 cfs). The average LDW flow is 1,350 cfs. The results from the three high-flow event simulations were similar.

Section 9). Peak PCB use was recorded in Puget Sound sediment cores between 1960 and 1970 (Van Metre and Mahler 2004; Lefkovitz et al. 1997); the use of PCBs was banned in 1978 and they were phased out over the next few years. Thus, peak total PCB concentrations in LDW sediments should be buried except in areas with low net sedimentation, localized disturbances, or ongoing or recent secondary sources (e.g., erosion of contaminated soil). In depositional areas, the depth within a core of the peak total PCB concentration should be a function of both the time since sources have been controlled and the estimated rate of net sedimentation.

To investigate this CSM, the expected depth of peak total PCB concentration was estimated for each core. This depth within a core was calculated as described in Equation 4-1 below:

$$D = (Tc - Tm) \times S \qquad \textbf{Equation 4-1}$$

Where:

- D = depth of peak total PCB concentration (cm)
- Tc = year of core collection
- Tm = assumed year of maximum concentration in surface sediment, corresponding to the assumed peak in PCB use and releases to the LDW
- S = net sedimentation rate (cm/yr) estimated from the STM for the cell containing the core

A range of estimated depths of the peak total PCB concentration was calculated for each core to reflect the uncertainty in the year of maximum historical PCB releases to the LDW (estimated depths within each core were calculated for peak PCB releases in 1960, 1965, and 1974). The latter year was selected because it was the year of the PCB spill in Slip 1 and PCBs were still likely in use. These depth estimates were then compared to the depth of the peak total PCB concentration in each core. If the depth of the peak total PCB concentration in a core was at or deeper than the estimated depth, then the core was considered to be consistent with the predictions of the STM and the CSM. Detailed results of this analysis for each individual core are presented in Appendix F.

Of the 366 cores available in the RI database, 219 cores could not be used in this analysis because those cores did not contain the type of information needed for the analysis (Table 4-58). Cores were excluded if they had only one interval analyzed for total PCBs, if the depth of the expected peak could not be analyzed (e.g., if only discontinuous intervals were analyzed), or if the core did not have detected concentrations of total PCBs.

Table 4-58. Results of the analysis comparing expected and actual depths of peak total PCB concentrations in subsurface sediment

DETERMINATION	REASON	NUMBER OF CORES
Consistent with STM/CSM	Peak was as deep or deeper than expected depth range.	113
Inconsistent with STM/CSM	Peak was shallower than expected depth range.	34[a]
Core information inconclusive with respect to STM/CSM	Only one interval was analyzed for total PCBs, **or** no interval was analyzed within the depth range of the expected peak	210
	No PCBs were detected in the core.	6
	Total PCB concentrations were similar throughout the core.	3
Total		**366[c]**

[a] All 34 of these cores had actual peak total PCB concentrations that were greater (i.e.,> 25%) than the concentration at the depth expected.

[b] Sediment accumulation for 4 of the 34 cores may have been affected by dredging.

[c] Four of the samples were collected outside of the modeling area (RM 4.9).

PCB – polychlorinated biphenyl

CSM – conceptual site model

STM – sediment transport model

Of the 147 cores included in the analysis, 113 cores (77%) had peak total PCB concentrations at depths equal to or greater than the estimated depths, consistent with the predictions of the STM of burial in depositional areas. Thirty-four cores had maximum total PCB concentrations that were shallower than the estimated depth range based on net sedimentation rates from the STM, assumed maximum PCB releases to the LDW in the 1960s and 1970s, and minimal localized disturbances. Fourteen of these 34 cores were located in EAAs with either very low net sedimentation rates or in areas where source investigations have suggested the potential for ongoing releases of PCBs. Some of these cores were also in areas of localized disturbances, such as dredging. The results of this analysis are discussed in more detail in the following sections for each of the three reaches defined in the CSM: Reach 1 – from RM 0.0 to RM 2.2; Reach 2 – from RM 2.2 to RM 4.0; and Reach 3 – from RM 4.0 to RM 4.8.

4.3.1 Reach 1 – RM 0.0 to RM 2.2

Based on the results of the STM, Reach 1 was characterized as net depositional on annual time scales in both the navigation channel and the adjacent bench areas. The sedimentation rates in the navigation channel and bench areas were generally classified as intermediate net depositional (1 to 2 cm/yr) within the first RM, and as intermediate to high depositional (1 to 3 cm/yr) in the remaining portion of this reach. Some relatively small areas are also present in this reach that were classified as lower net depositional (0 to 1 cm/yr) (Map 3-11).

In general, the potential for re-exposing buried sediments in this reach as a result of scour during high-flow events is considered minimal. The STM identified the potential for relatively low bed scour (< 2 cm) during a 100-yr event in a small area between RM 0.8 and RM 0.9 and in several areas between RM 1.8 and RM 2.2 (Map 3-8). Ship-induced mixing of the surficial bed layer was estimated to extend to average depths of about 1 to 2 cm in the bench areas and less than 1 cm in the navigation channel.

Map 4-59 presents an overlay of sediment core locations on a map depicting estimated average annual net sedimentation rates in the LDW. Core data are presented to indicate whether the depth of the peak total PCB concentration is consistent with (the same depth or deeper than) or shallower than the expected depth.

Most patterns of total PCB concentrations as a function of depth within the cores in this reach were consistent with the net sedimentation rates estimated by the STM and estimated period of peak PCB releases to the LDW. Of the 60 cores in this reach with data sufficient to assess consistency with the predictions of the STM, 44 (73%) had peak total PCB concentrations as deep or deeper than expected (Table 4-59). Examples of cores in which peak total PCB concentrations were consistent with the STM can be found at locations throughout the reach, including cores along the eastern bench, along the western bench, and in the navigation channel.

Table 4-59. Results of the analysis comparing expected and actual depths of peak total PCB concentrations in subsurface sediment by river reach

River Reach	Area Within Reach	Number of Cores		
		Consistent with STM/CSM	Inconsistent with STM/CSM	Inconclusive with Respect to STM/CSM
RM 0 to RM 2.2	EAA 1 (Duwamish/Diagonal)	13	0	0
	Rest of reach	31	16	47
	Subtotal	**44**	**16**	**47**
RM 2.2 to RM 4	EAA 2 (RM 2.2)	0	1	0
	EAA 3 (Slip 4)	6	0	0
	Boeing Plant 2/Jorgensen Forge EAA	44	5	83
	EAA 5 (Terminal 117)	7	7	6
	EAA 6 (RM 3.8)	0	1	0
	Rest of reach	11	4	33
	Subtotal	**68**	**18**	**122**
RM 4 to RM 4.8	Entire reach	1	0	46
	Subtotal	**1**	**0**	**46**
Total		**113**	**34**	**215**

CSM – conceptual site model
EAA – Early Action Area
PCB – polychlorinated biphenyl
RM – river mile
STM – sediment transport model

There were 16 cores (27%) in which the peak total PCB concentration was in a shallower interval than expected. Nine of the cores were collected on the eastern bench, and seven were collected on the western bench. Potential explanations for the differences between the expected depth of the peak total PCB concentration and the actual depth include: 1) localized net sedimentation or scour depths different than those estimated by the STM or just outside the modeling area boundary, 2) presence of localized, ongoing sources of PCBs, or 3) presence of localized disturbance events sufficient to expose subsurface sediment.

Eleven of the 16 cores not fitting the expected pattern had relatively high PCB concentrations (> 212 µg/kg dw) in the uppermost core interval. These 11 cores were located throughout the reach, mostly close to the shoreline. Most of the 11 cores were near an outfall of some kind, two were outside of the modeling area, and five were near berthing areas. Potential sources of PCBs have not been characterized in detail in these areas.

Five of the 16 cores had relatively low total PCB concentrations (< 100 µg/kg dw) throughout the core profile. These cores were located just upstream of Slip 1, at the mouth of Slip 3, in the embayment on the west side of the LDW between RM 1.4 and RM 1.5, and just downstream of Slip 2 at RM 1.6. Therefore, although the expected and actual peak total PCB concentrations were not at the same depth, differences in concentrations among the intervals within cores were relatively small.

In summary, data from the cores in Reach 1 suggest that historically contaminated sediment is being buried by relatively less-contaminated sediment in most of this reach except in areas that may be affected by localized disturbances or ongoing sources.

4.3.2 Reach 2 – RM 2.2 to RM 4.0

Reach 2 was characterized as net depositional, with high spatial variability in net sedimentation rates ranging from 0 to approximately 11 cm/yr (Map 4-59). Some of the benches within the reach were characterized as having high net sedimentation rates (> 3 cm/yr), including both eastern and western benches between RM 2.0 and RM 2.6 and the western bench between RM 3.3 and RM 3.6. This reach also includes several areas with the lowest net annual sedimentation rates (0 to 0.5 cm/yr). Estimated scour depths were also variable, with a maximum net erosion depth of 14 cm in a small area near RM 3.6. Scour depths were generally higher in the navigation channel than in the bench areas. Ship-induced mixing of the surficial bed layer is expected to be minimal, potentially extending to average depths of less than 1 cm in the bench areas and less than 0.1 cm in the navigation channel.

As in Reach 1, most patterns of total PCB chemical concentrations as a function of depth within the cores in Reach 2 were consistent with the net sedimentation rates estimated by the STM and the assumed peak usage of PCBs (Map 4-59). Of the 86

cores in this reach with data sufficient to assess consistency with the STM, 68 (79%) had peak total PCB concentrations as deep or deeper than expected based on the estimated net sedimentation rate over the time period since the expected peak use of PCBs (Table 4-59). These cores were distributed throughout the reach.

There were 18 cores (21%) that had peak total PCB concentrations in a shallower interval than expected. Seven of the cores were collected near T-117 and five of the cores were collected in the Boeing Plant 2/Jorgensen Forge EAA near RM 3.6. The remaining six cores were located near RM 2.2 (two cores), one of which was in EAA 2 at this location), RM 2.7, RM 2.8, RM 3.8, and RM 3.9. Most of these cores were also located near outfalls. Potential sources have been characterized in the EAAs, but not in the other areas.

Seventeen of the 18 cores in Reach 2 that did not fit the expected pattern had relatively high total PCB concentrations (> 212 µg/kg dw) in the uppermost core interval. The core that did not have a total PCB concentration above the 75th percentile was located near the inlet at RM 2.2.

In summary, data suggest that historically contaminated sediment is being buried by relatively less-contaminated sediment in much of the reach. At 18 of the core locations with data sufficient to assess consistency with the predictions of the STM, it appears that either net sedimentation is lower than expected, localized disturbances are occurring, or ongoing sources may be present. Fourteen of these 18 cores were located in EAAs and all but one of the remaining cores were near outfalls.

4.3.3 Reach 3 – RM 4.0 to RM 4.8

This reach was characterized as net depositional with relatively high net sedimentation rates: 2 to 3 cm/yr in Slip 6; 4 to 23 cm/yr between RM 4.0 and RM 4.5, and up to 145 cm/yr in the Upper Turning Basin, which is designed to act as a trap for much of the coarser fraction of the bed load entering the LDW from upstream. Episodic erosion may occur during high-flow events in some areas of the reach (up to approximately 14 cm in one small area at RM 4.6), but most of the reach is not expected to experience scour during high-flow events. Ship-induced mixing of the surficial bed layer is expected to be minimal.

Of the 47 cores in this reach, only one core (located in Slip 6) was analyzed for total PCBs at more than one subsurface interval (Map 4-59), and, therefore, only one core had sufficient data to assess consistency with the predictions of the STM. In the core from Slip 6, peak total PCB concentrations were at a depth in the core consistent with expectations based on the STM and peak PCB usage.

4.3.4 Summary

A total of 147 cores from throughout the LDW were evaluated for consistency with the CSM. Most (> 77%) of the cores were clearly consistent with the assumptions of the

analysis and showed that peaks in total PCB concentrations in subsurface sediments are being buried by cleaner sediment. The remaining 23% of the cores had peak concentrations that were shallower than expected assuming net sedimentation rates from the STM, peak PCB usage in the 1960s and 1970s, and minimal localized disturbances. Therefore, one or more of these assumptions were not valid at these particular locations. Because most of these locations were either in areas just outside the STM modeling areas, in dredged areas, near berthing areas, or near locations with the potential for ongoing sources, the CSM for sediment deposition was validated based on the total PCB patterns as a function of depth.

4.4 SUMMARY

This section provides a summary of the nature and extent of contamination in the LDW based on a comprehensive chemical dataset for surface sediment, subsurface sediment, tissue, and surface water, seep water, and porewater.

Surface sediment samples from 1,484 locations within the LDW were used to evaluate the nature and extent of contamination. Chemical contamination of surface sediment in the LDW is characterized by a discrete set of localized areas with high chemical concentrations (hot spots) separated by relatively large areas with much lower contamination.

Table 4-60 presents summary statistics for selected risk driver chemicals identified in the risk assessments and also lists locations with the highest concentrations. The SWAC provides an indication of average concentrations within the LDW that is not biased by sampling density, whereas the area-based 95th percentile identifies areas with the highest concentrations in the waterway (95% of the LDW, by area, has lower concentrations). As shown on Maps 4-60a through 4-60c, areas with the highest chemical concentrations of risk driver chemicals tend to be co-located. Many of the locations with the highest concentrations are within areas that have been identified as EAAs (see Section 10).

Table 4-60. Summary of LDW surface sediment data

CHEMICAL OR CHEMICAL GROUP	AREA-BASED 95TH PERCENTILE[a]	50TH PERCENTILE[a]	SWAC	AREAS WITH HIGHEST CONCENTRATIONS[b]
Total PCBs	808 µg/kg dw	115 µg/kg dw	342 µg/kg dw	Duwamish/Diagonal EAA RM 1.0 in navigation channel EAA 2 Slip 4 EAA Boeing Plant 2/Jogensen Forge EAA T-117 EAA Norfolk EAA

Chemical or Chemical Group	Area-Based 95th Percentile[a]	50th Percentile[a]	SWAC	Areas with Highest Concentrations[b]
Arsenic	23.6 mg/kg dw	11 mg/kg dw	14.5 mg/kg dw	RM 3.8 EAA near RM 0.1 on the east side Duwamish/Diagonal EAA Slip 1 RM 1.3 to RM 1.5 on the west side Slip 3 Boeing Plant 2/Jogensen Forge EAA Between EAA4 and RM 3.8 EAA on the east side near RM 5.0 on the west side
cPAHs	1,100 µg/kg dw	227 µg/kg dw	356.9 µg/kg dw	T-117 EAA Slip 4 EAA RM 3.8 EAA near RM 0.6 on the west side RM 0.0 near Harbor Island Duwamish/Diagonal EAA
Dioxins and furans	524 ng/kg dw	10 ng/kg dw	nc	RM 1.4 to RM 1.5 on the west side EAA 2 Duwamish/Diagonal EAA near RM 1.0 in the navigation channel Boeing Plant 2/Jogensen Forge EAA
Metals				
Cadmium	1.8 mg/kg dw	0.39 mg/kg dw	nc	Boeing Plant 2/Jogensen Forge EAA Duwamish/Diagonal EAA Slip 1 Slip 4 RM 1.3 to RM 1.5 on the west side near RM 0.1 on the east side near RM 1.0 in the navigation channel EAA 2
Chromium	70 mg/kg dw	29 mg/kg dw		
Copper	147 mg/kg dw	52 mg/kg dw		
Lead	210 mg/kg dw	36 mg/kg dw		
Mercury	0.41 mg/kg dw	0.14 mg/kg dw		
Nickel	43 mg/kg dw	22 mg/kg dw		
Silver	1.7 mg/kg dw	0.3 mg/kg dw		
Vanadium	81 mg/kg dw	57 mg/kg dw		
Zinc	407 mg/kg dw	113 mg/kg dw		
TBT	251 µg/kg dw	22 µg/kg dw	nc	near RM 1.3 on the west side between RM 0.0 and RM 0.4
HPAH	13,300 µg/kg dw	1,985 µg/kg dw	nc	T-117 EAA Between RM 0.5 and RM 0.6 in the navigation channel and on the west side RM 3.8 EAA RM 2.7 on the west side near RM 1.0 on the east side Boeing Plant 2/Jogensen Forge EAA Duwamish/Diagonal EAA Slip 1 Slip 4
LPAH	2,300 µg/kg dw	230 µg/kg dw	nc	RM 2.7 on the west side T-117 EAA near RM 1.0 on the east side Boeing Plant 2/Jogensen Forge EAA Duwamish/Diagonal EAA
BEHP	2,407 µg/kg dw	230 µg/kg dw	nc	Duwamish/Diagonal EAA EAA 2 Slip 4 EAA Boeing Plant 2/Jogensen Forge EAA Norfolk EAA

[a] The SWAC was calculated on an area basis for total PCBs, arsenic, and cPAHs and was based on a strict percentile of data in the surface sediment dataset for the remaining chemicals.

[b] Areas with the highest concentrations are listed but may not include all areas with concentrations greater than the 95th percentile; see individual maps for all locations.

BEHP – bis(2-ethylhexyl) phthalate
cPAH – carcinogenic polycyclic aromatic hydrocarbon
CSO – combined sewer overflow
dw – dry weight
EAA – Early Action Area
HPAH – high-molecular-weight polycyclic aromatic hydrocarbon
LDW – Lower Duwamish Waterway
LPAH – low-molecular-weight polycyclic aromatic hydrocarbon
nc – not calculated
PCB – polychlorinated biphenyl
RM – river mile
SWAC – spatially weighted average concentration
T-117 – Terminal 117
TBT – tributyltin

Subsurface sediment contamination was also characterized by many studies within the LDW. Many of the areas with high chemical concentrations in surface sediment also had high concentrations in subsurface sediment, although some areas with less contaminated surface sediment had contamination in the subsurface, particularly for PAHs.

A detailed analysis of the locations of peak concentrations of total PCBs, arsenic, and cPAHs within individual subsurface cores was conducted. At many locations, subsurface sediment at deeper intervals contained higher total PCB and cPAH concentrations than did sediment from the uppermost intervals, with some exceptions.[77]

The two highest dioxin and furan TEQs (194 and 136 ng/kg dw) in subsurface sediment were detected in samples collected deeper than 4 ft at locations LDW-SC20 (RM 1.0 in the navigation channel) and LDW-SC26 (RM 1.4 W), respectively. The next highest concentration (54.1 ng/kg dw) was detected in the uppermost interval at LDW-SC29 (the embayment at RM 1.5 W) that also had a very high concentration (2,100 ng/kg dw) in surface sediment. Concentrations decreased with depth at this location.

Chemical patterns in sediment cores collected throughout the site were evaluated for consistency with the CSM. Based on the CSM, peak total PCB concentrations in subsurface sediment should be buried except in areas with low net sedimentation, localized disturbances, or ongoing or recent secondary sources. Available core data collected throughout the LDW are generally consistent with this expected pattern: most (77%) of the cores showed that peak total PCB concentrations are being buried. The remaining 23% of the cores had peak concentrations that were shallower than expected. Most of these locations were either in areas just outside the STM modeling areas, in dredged areas, near berthing areas, or near locations with the potential for

[77] The exceptions for total PCBs were near RM 1.0 in the navigation channel, EAA 2, the Slip 4 EAA, the T-117 EAA, and Boeing Plant 2/Jorgensen. The exceptions for cPAHs were Slip 6, the Norfolk EAA, the Duwamish/Diagonal EAA, and near RM 0.5 on the west side. In all but one of the five areas with high arsenic concentrations in subsurface sediment, peak concentrations were at depth within the cores.

ongoing sources, so the CSM for sediment deposition was considered validated based on the total PCB patterns as a function of depth.

A large number of tissue composite samples have been collected in the LDW for various fish, crab, and invertebrate species. The locations and number of the four sampling areas for fish and crabs were selected primarily to reflect the longitudinal distribution of PCB concentrations in the sediment characterized by a rolling average PCB sediment concentration. In whole-body composite samples of fish, mean sampling area total PCB concentrations ranged from 71 µg/kg ww in juvenile chinook salmon in Area T1 to 4,300 µg/kg ww in shiner surfperch collected in 2004 in Area T2. The highest mean total PCB concentration was in crab hepatopancreas tissue (5,500 µg/kg ww in T4 in 2004); mean concentrations in edible meat were much lower, ranging from non-detect to 270 µg/kg ww. In general, total PCB concentrations in invertebrates (except crab hepatopancreas samples) tended to be lower than in fish, and concentrations in juvenile chinook salmon were lower than concentrations in English sole and shiner surfperch.

The LDW-wide mean total PCB concentrations in whole-body English sole and shiner surfperch were higher in 2004 than in 2005. When the differences between years were evaluated separately for each area, the results showed that for English sole the differences were significant in all areas, and for shiner surfperch, the differences were significant in area T3 only. A second analysis was conducted using whole-body English sole and shiner surfperch data from Area T1, the area for which data span several years. For both species, total PCB concentrations from Area T1 were significantly greater in 2004 than in other years with available data (2005 and 2006 for English sole; and 2004, 2005, and 1997 for shiner surfperch).

Total PCB concentrations in tissues of both English sole and shiner surfperch were significantly different among areas (sampling area effect significant, $p < 0.001$). The lowest mean tissue concentrations for both species occurred in Area T4 in both 2004 and 2005. In 2004, mean concentrations in Area T4 were significantly lower than concentrations in the two areas with the highest means (Areas T1 and T2 for English sole and Areas T2 and T3 for shiner surfperch). In 2005, concentrations in Area T4 were significantly lower than in Areas T2 for English sole, and there were no significant differences among areas for shiner surfperch.

Mean total arsenic concentrations were highest in Dungeness crab edible meat (5.5 mg/kg ww), and mean inorganic arsenic concentrations, which are more relevant from a risk perspective, were highest in soft-shell clams (1.24 mg/kg ww). Clam tissue had a very high percentage of inorganic arsenic (10 to 68%), much higher than fractions detected in tissue from other studies in Puget Sound. Regression analyses showed a statistically significant positive correlation between arsenic concentrations in benthic invertebrate tissue and co-located sediment; this relationship was used to calculate area-wide tissue concentrations for use in the ERA.

The highest cPAH concentrations in tissue were detected in clams, mussels, and benthic invertebrates at mean concentrations ranging from 15 to 44 µg/kg ww. Mean cPAH concentrations in English sole (whole-body tissue) were significantly lower in Area T4 than in Areas T1 and T2. For shiner surfperch, mean concentrations were significantly lower in Area T4 than in Area T3. The cPAH SWAC in sediment was also significantly lower in Area T4 than in any other areas.

The highest mean concentrations of metals (including TBT) were detected in invertebrates (i.e., clams, crabs, benthic invertebrates, and amphipods). Other than PAHs, which were detected in most tissue samples, SVOCs were infrequently detected in tissue. DDT and chlordane compounds were detected in tissue samples, but there is high uncertainty in these data because of analytical interference from PCBs.

To determine concentrations of various chemicals in surface water, grab samples were collected and SPMDs were deployed in the mid 1990s as part of the WQA by King County. In 2005, grab samples were also analyzed for PCB congeners. Total PCB congener concentrations in grab samples collected from two locations in the LDW on four sampling dates in 2005 ranged from 132 to 3,211 pg/L. Concentrations were lowest at all locations in November 2005, when water flow was the highest.

In grab surface water samples, mean dissolved and total arsenic concentrations were higher at RM 1.1 and RM 1.9 than at RM 4.9. Mean concentrations of chromium, copper, lead, nickel, vanadium, and zinc in unfiltered samples and copper, lead, and zinc in filtered samples were higher in samples from RM 4.9 than in samples from RM 1.1 and RM 1.9. Concentrations in filtered samples did not exceed Washington State acute or chronic marine water quality standards for dissolved metals. SVOCs were detected infrequently in surface water samples collected from the LDW. Calculated PAH and organochlorine pesticide concentrations in surface water based on the SPMD data were low, ranging from 0.0029 to 30 ng/L for PAHs and from 0.000034 to 0.033 ng/L for pesticides.

Seep water chemical data have been collected at four sites along the LDW, including Rhône-Poulenc, Boeing Plant 2, T-117, and Great Western International as well as a LDW-wide seep investigation conducted for the RI. In seep water samples, PCBs were detected in a filtered sample collected from one of the RI locations (in the inlet at RM 2.2). The highest dissolved arsenic concentrations in seep water (72.4 and 253 µg/L) were detected in the embayment near RM 1.5 on the west side and in Slip 1, respectively. These areas also had some of the highest arsenic concentrations in surface sediment. An unfiltered seep water sample collected near Slip 6 had the highest concentrations of cadmium, chromium, copper, nickel, and vanadium. SVOCs were detected infrequently in seep water samples collected from the LDW. One organochlorine pesticide (heptachlor epoxide) was detected at one seep location in both filtered and unfiltered samples.

Few VOCs were detected in seep water samples collected from Boeing Plant 2, from Rhône-Poulenc, and at locations throughout the LDW; concentrations were low (≤ 40 µg/L). VOCs were frequently detected in seep water samples collected during various GWI investigations. VOCs with the highest concentrations were cis-1,2-dichloroethene (5,400 µg/L) and vinyl chloride (3,500 µg/L).

Porewater samples collected from non-dredged sediment have been analyzed in four separate sampling events (EPA SI, PSDDA 1999, Rhône-Poulenc, and the RI). As part of the EPA SI, 15 unfiltered porewater samples were collected from throughout the LDW and analyzed for metals; seven metals, including arsenic, were detected in these samples. TBT was also analyzed in porewater as part of the EPA SI and the PSDDA 1999 sampling event. One of the 35 porewater samples had a TBT concentration (0.55 µg/L) that was greater than the DMMP SL (0.15 µg/L). Most SVOCs were detected infrequently, except BEHP in the Rhône-Poulenc study, which was detected in 56% of porewater samples analyzed. Organochlorine pesticides and PCBs were not analyzed in porewater.

Porewater samples were collected and analyzed for VOCs at Boeing Plant 2/Jorgensen Forge and GWI. At GWI, cis-1,2-dichloroethene and vinyl chloride were detected in all 10 nearshore samples, at maximum concentrations of 2,900 and 2,500 µg/L, respectively. At Boeing Plant 2/Jorgensen Forge, VOCs were detected at low concentrations in porewater samples (≤ 13 µg/L).

5 Summary of the Baseline Ecological Risk Assessment

The baseline ERA estimated risks for the benthic invertebrate, fish, and wildlife species that may be exposed to chemicals in sediment, water, and aquatic biota in the LDW. This assessment was based on historical data (used in the Phase 1 ERA) (Windward 2003a) and sediment and tissue chemistry data collected as part of the RI. This section summarizes the problem formulation, exposure assessment, effects assessment, risk characterization, uncertainty analysis, and selection of risk drivers for ecological receptors. The complete ERA is included as Appendix A to this RI.

5.1 PROBLEM FORMULATION

The problem formulation of the ERA established the overall scope of the assessment. The problem formulation included a description of the data available for conducting the ERA, the suitability of the data for risk assessment purposes, and a risk-based screening process that allowed the risk assessment to focus on chemicals of potential concern (COPCs) and eliminate chemicals that did not pose risks to ecological receptors.

Data used in the ERA consisted largely of:

- Surface sediment (uppermost 0 to 15 cm) chemistry data
- Tissue chemistry data for benthic invertebrates (including benthic infauna and epifauna, crabs, clams, and mussels), English sole, Pacific staghorn sculpin, shiner surfperch, and juvenile chinook salmon
- Site-specific sediment toxicity test results
- Site-specific evaluation of imposex in field-collected gastropods
- Sediment porewater chemistry data

Because it is generally considered impractical to evaluate risks for every potentially exposed species, it is standard ERA practice and consistent with the SMS to focus on representative receptor species that typify groups of organisms with specific exposure pathways. One objective of selecting representative receptors is to choose species for which the risk conclusions will be protective or representative of other species that were not explicitly evaluated. For example, an assessment of risks to great blue herons would be assumed to be protective of all wading birds that eat fish because of the higher exposure potential of great blue herons. Receptors may also be selected for risk analysis because the species is highly valued by society (e.g., endangered or threatened species). Representative receptors of concern (ROCs) selected for the baseline ERA were the benthic invertebrate community, crabs, English sole, Pacific staghorn sculpin, great blue heron, spotted sandpiper, osprey, river otter, and harbor seal (Table 5-1). In addition, juvenile chinook salmon was selected as an ROC because this species is federally protected and uses the LDW during outmigration to Puget Sound.

Table 5-1. ROCs selected for the LDW and a summary of the considerations for selection

ROC	Exposure Route	Ecological Significance	Societal Significance	Site Use	Exposure Data Availability	Sensitivity
Benthic invertebrate community	direct contact, diet, sediment ingestion	food source for other invertebrates, fish, and mammals; nutrient cycling	target community for protection in the development of numerical sediment quality criteria	present year-round; multiple life stages	abundant surface sediment data available	because of the diversity of organisms in this ROC group, the range of sensitivities is represented
Crabs	direct contact, diet, incidental sediment ingestion	higher-trophic-level benthic invertebrate	some recreational and commercial value	present year-round; multiple life stages	site-specific tissue data available	effects data are available for decapods
Pacific staghorn sculpin	incidental sediment ingestion; preys on both invertebrates and other fish; thus potential for elevated exposure via bioaccumulation because of trophic position	serves as prey for higher-trophic-level species such as birds and mammals	considered to be a nuisance by anglers	adults and juveniles present year-round; may spawn in the LDW	site-specific tissue data and prey tissue data available	Effects data are available for other fish species; unknown relative sensitivity of sculpin
English sole	direct contact, diet, incidental sediment ingestion	important prey item for birds and fish; key benthic predator	some recreational and commercial value	juveniles present year-round; adults present except during spawning migrations to Puget Sound	site-specific tissue and prey tissue data available	NMFS data suggest that they are as sensitive as other flatfish species (Myers et al. 1998b)
Juvenile chinook salmon	diet	important prey item for birds and fish; seasonally one of the most abundant juvenile salmonids in the LDW	threatened and endangered species;[a] returning adults important to commercial, sport, and tribal fisheries	generally present April to July; most estuary-dependent juvenile salmonid	site-specific tissue and prey tissue data available	believed to be sensitive to a wide range of COPCs
Great blue heron	preys on fish; thus potential for elevated exposure via bioaccumulation because of trophic position	high on food chain	valued by society	present year-round; feed in LDW	site-specific data available for some food resources	effects data available for other bird species; unknown relative sensitivity of heron
Osprey	preys on fish; thus potential for elevated exposure via bioaccumulation because of trophic position	top of food chain	valued by society; protected under migratory bird treaty	nests along the LDW and forages at shallow depths in the LDW	site-specific data available for food resources	effects data available for other bird species; unknown relative sensitivity of osprey
Spotted sandpiper	diet, incidental sediment ingestion	important role as an intermediate predator	protected under migratory bird treaty	present June to September; nests along the LDW	site-specific data available for food resources	effects data available for other bird species; unknown relative sensitivity of sandpiper.

ROC	Exposure Route	Ecological Significance	Societal Significance	Site Use	Exposure Data Availability	Sensitivity
River otter	preys on fish; thus, potential for elevated exposure via bioaccumulation because of trophic position	top of food chain	valued by society	present year-round	site-specific data available for food resources	some mustelids shown to be highly sensitive to some chemicals (e.g., PCBs)
Harbor seal	preys on fish; thus potential for elevated exposure via bioaccumulation because of trophic position	top of food chain	protected under Marine Mammal Act	infrequent	site-specific data available for food resources	pinnipeds suspected to be sensitive to some chemicals (e.g., PCBs)

[a] Species listed under the Endangered Species Act.

COPC – chemical of potential concern

LDW – Lower Duwamish Waterway

NMFS – National Marine Fisheries Service

PCB – polychlorinated biphenyl

ROC – receptor of concern

For each ROC selected, COPCs were identified through a conservative, risk-based screening process using maximum exposure concentrations and SMS sediment quality standards (SQS) for benthic community or no-observed-adverse-effect levels (NOAELs) from the scientific literature for other receptors. COPCs identified included:
46 chemicals for the benthic invertebrate community (including tributyltin [TBT], metals, and PCBs and other organic compounds) (Table 5-2); 2 chemicals for crabs (total PCBs and zinc); 6 chemicals for at least one fish ROC (arsenic, cadmium, copper, total PCBs, TBT, and vanadium) (Table 5-3), and 12 chemicals for at least one wildlife ROC (arsenic, cadmium, chromium, cobalt, copper, lead, mercury, nickel, selenium, total PCBs, zinc, and vanadium) (Table 5-4). COPCs were evaluated only in the uncertainty analysis if they were never detected in tissue but RLs were above the screening criteria or if they did not have effect-level toxicity information. Risks from organochlorine pesticides were also assessed in the uncertainty analysis. Detected concentrations of organochlorine pesticides in some sediment samples and in all tissue samples collected in 2004 were qualified as estimates (JN-qualified) because of probable analytical interference from PCBs, resulting in a high bias in concentration and a tentative identification (Windward 2005a, b).

Table 5-2. COPCs evaluated in the risk characterization for the benthic invertebrate community and for crabs

COPC	Benthic Invertebrate Community	Crabs
Metals		
Arsenic	X	
Cadmium	X	
Chromium	X	
Copper	X	
Lead	X	
Mercury	X	
Nickel	X	
Silver	X	
Zinc	X	X
Organometals		
TBT	X	
PAHs		
Acenaphthene	X	
Anthracene	X	
Benz(a)anthracene	X	
Benzo(a)pyrene	X	
Benzo(g,h,i)perylene	X	
Chrysene	X	
Dibenzo (a,h)anthracene	X	
Fluoranthene	X	
Fluorene	X	
Indeno (1,2,3,-c,d)pyrene	X	
Naphthalene	X	
Phenanthrene	X	

Pyrene	X	
Total benzofluoranthenes	X	
HPAH	X	
LPAH	X	
Phthalates		
BBP	X	
BEHP	X	
Dimethyl phthalate	X	
Other SVOCs		
1,2,4-Trichlorobenzene	X	
1,2-Dichlorobenzene	X	
1,4-Dichlorobenzene	X	
2-Methylnaphthalene	X	
4-Methylphenol	X	
2,4-Dimethylphenol	X	
Benzoic acid	X	
Benzyl alcohol	X	
Dibenzofuran	X	
Hexachlorobenzene	X	
n-Nitrosodiphenylamine	X	
Pentachlorophenol	X	
Phenol	X	
VOCs		
cis-1,2-Dichloroethene	X	
PCBs		
Total PCBs	X	X
Organochlorine Pesticides		
Total DDTs	X	
Total chlordane	X	

BBP – butyl benzyl phthalate
BEHP – bis(2-ethylhexyl) phthalate
COPC – chemical of potential concern
HPAH – high-molecular-weight polycyclic aromatic hydrocarbon
LPAH – low-molecular-weight polycyclic aromatic hydrocarbon
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
SVOC – semivolatile organic compound
TBT – tributyltin
VOC – volatile organic compound

Table 5-3. COPCs selected for fish ROCs

COPC	Juvenile Chinook Salmon	English Sole	Pacific Staghorn Sculpin
Arsenic	X	X	X
Cadmium	X	X	X
Copper	X	X	X
Vanadium	X	X	X
TBT			X
Total PCBs		X	X

COPC – chemical of potential concern
PCB – polychlorinated biphenyl

ROC – receptor of concern

TBT – tributyltin

Table 5-4. COPCs selected for wildlife ROCs

COPC	Spotted Sandpiper	Great Blue Heron and Osprey	River Otter	Harbor Seal
Arsenic	X		X	
Cadmium	X			
Chromium	X	X		
Cobalt	X		X	
Copper	X			
Lead	X	X		
Mercury	X	X	X	X
Nickel	X			
Selenium	X		X	
Vanadium	X			
Zinc	X			
Total PCBs	X	X	X	X

COPC – chemical of potential concern

PCB – polychlorinated biphenyl

No quantitative ecological risks were estimated for dioxins and furans because tissue samples from the LDW were not analyzed for dioxins and furans.[78] Tissues were not analyzed because of the difficulties associated with assessing site-specific risks from dioxins and furans, the need for a large background dataset with which to compare site-specific data, and the paucity of background tissue data in the Puget Sound area. Risk management decisions to address dioxin and furan contamination in LDW sediment will be based on relevant guidance and regulations, including those specifically related to the role of background-level contamination in the CERCLA process (EPA 2002a). Remedial decisions to address dioxin and furan contamination in sediment will be made by EPA and Ecology as part of the FS process.

The problem formulation also presented the conceptual site models for the ROCs (Figures 5-1 and 5-2). Conceptual site models identify and describe pathways in which ROCs may be exposed to COPCs associated with sediment within the LDW. The pathways evaluated in the ERA included both direct exposure through sediment and water and indirect exposure through the ingestion of prey from the LDW. The potential exposure pathways of sediment-associated chemicals to upper-trophic-level ROCs in the LDW were presented in a generalized food web diagram (Figure 5-3).

[78] In the human health risk assessment, it was assumed that risks associated with exposure to dioxins and furans are unacceptable, although tissue data were not available for specific exposure estimates. Dioxins and furans were assumed to be a risk driver based on human health risks associated with both the seafood consumption and direct contact pathways.

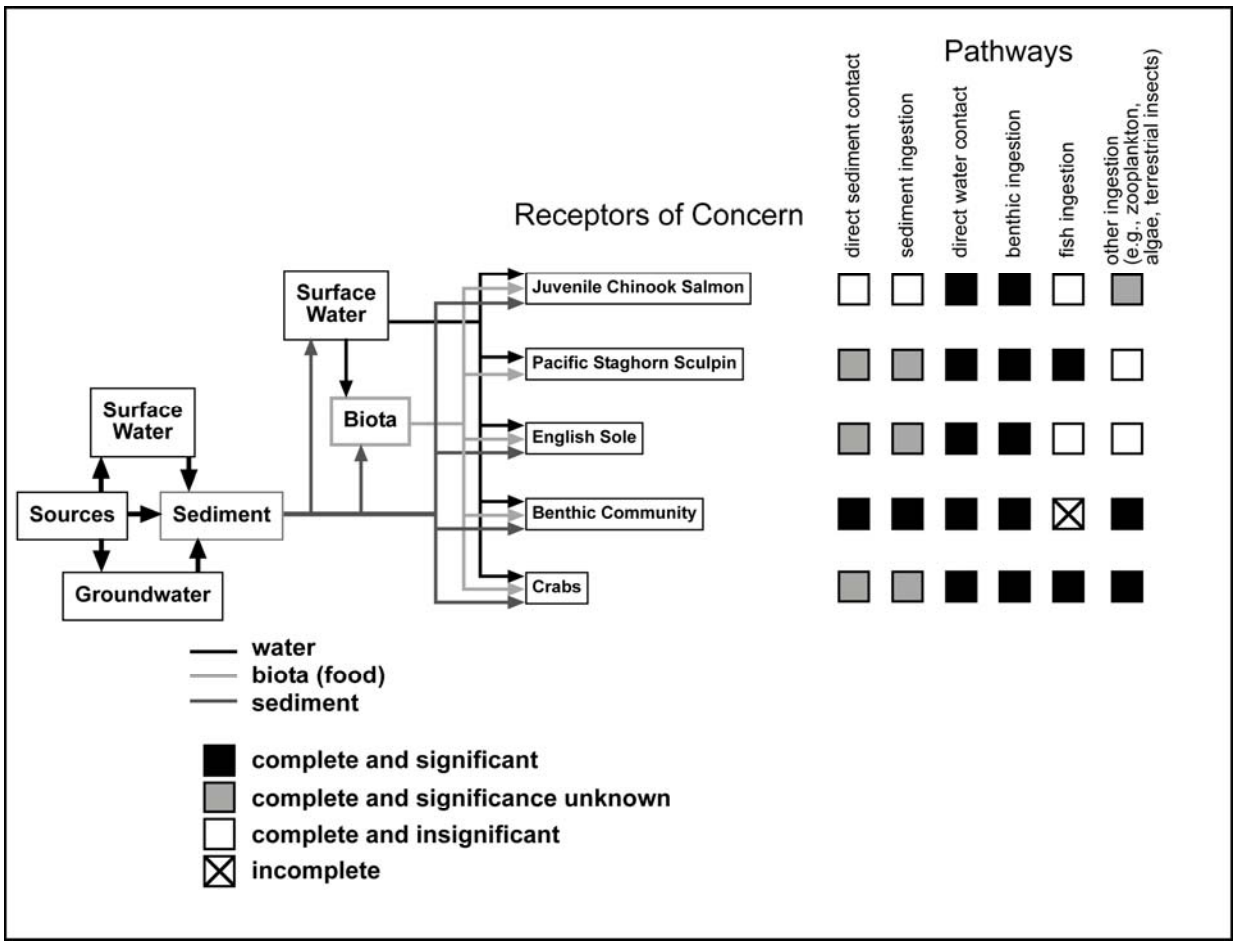


Figure 5-1. Conceptual site model for fish and the benthic invertebrate community

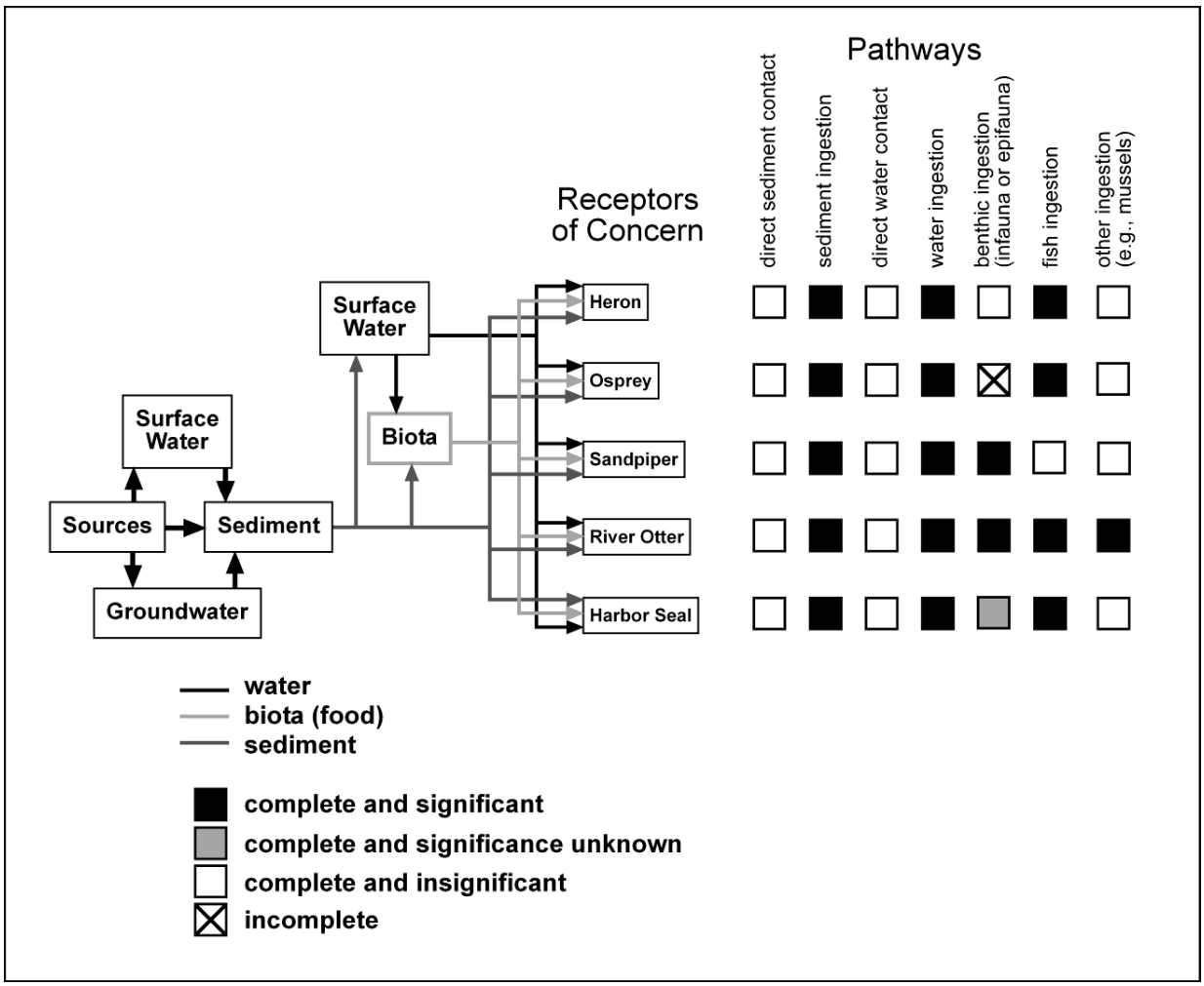


Figure 5-2. Conceptual site model for wildlife

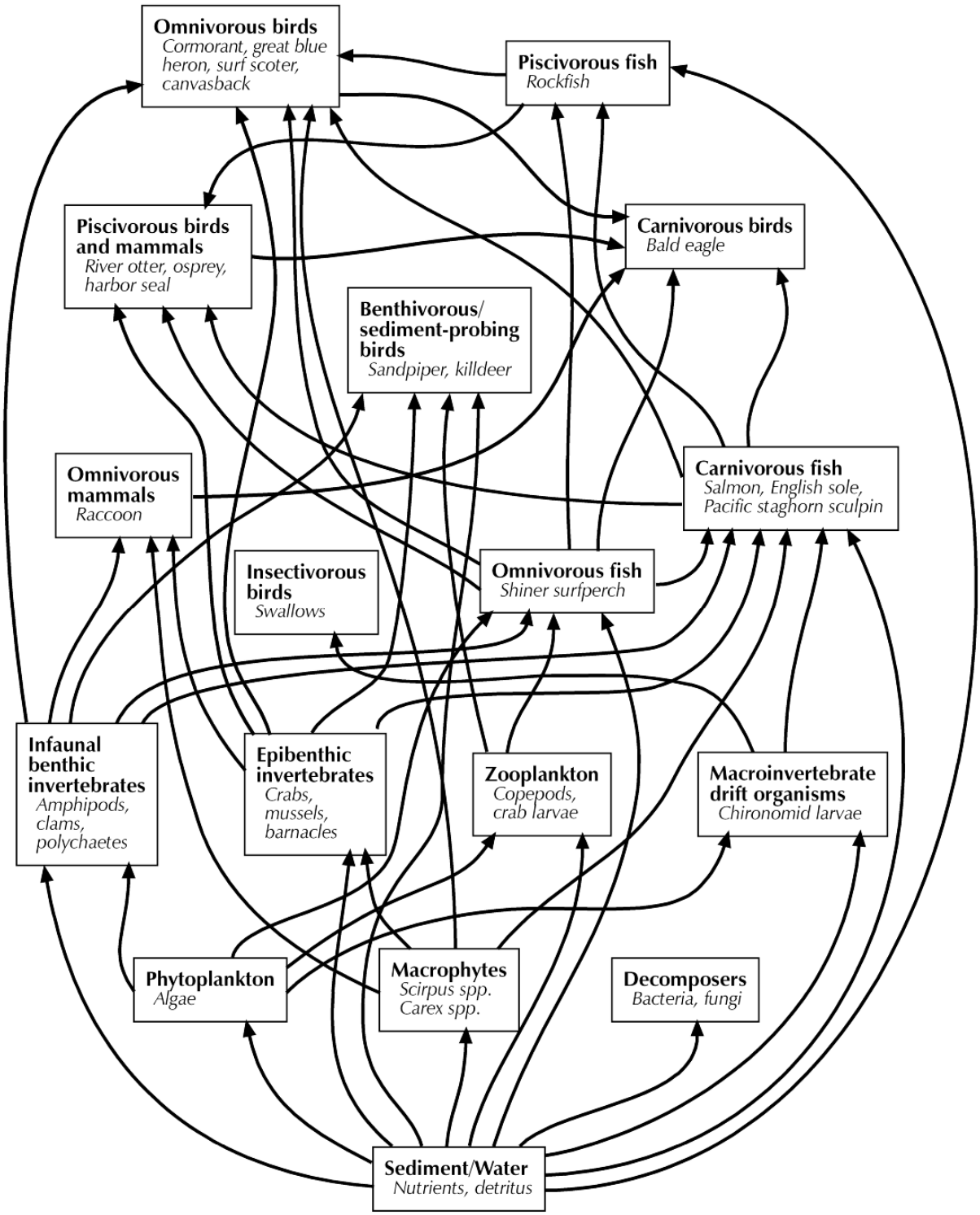


Figure 5-3. Generalized food web diagram for the LDW

Assessment and measurement endpoints were also identified in the problem formulation (Table 5-5). Survival, growth, and reproduction were included as endpoints for all ROCs except juvenile chinook salmon. Assessment endpoints for juvenile chinook salmon included survival and growth; reproduction was not included because chinook salmon are not exposed to chemicals in the LDW during reproductive life phases. Biomarker, behavioral, and histological endpoints were not included as assessment endpoints. Typically, ERAs focus on ecological effects at the individual level or higher (i.e., population level). In this way, the emphasis is placed on endpoints that integrate an overall response by an organism, rather than indicators of a biochemical response that may or may not result in an ecologically relevant effect. The assessment scale for each ROC was identified based on the potential exposure area for the ROC (Table 5-5).

Table 5-5. Assessment endpoints for ROCs and measures of effect and exposure

ROC	Assessment Endpoint	Assessment Scale	Measures of Effect	Measures of Exposure
Benthic				
Benthic invertebrate community	survival, growth, reproduction	potential exposure area: small exposure areas for individuals assessment scale: small exposure areas throughout the LDW	SMS and toxicologically based sediment guidelines or TRVs[a]	chemical concentrations in sediment
			water-based TRVs for VOCs	VOC concentrations in porewater
			site-specific toxicity tests	chemical concentrations in sediment samples co-located with toxicity test samples
			tissue-based TRVs for TBT (excluding imposex in gastropods)	TBT concentrations in sediment samples co-located with benthic invertebrate tissue collection
			assessment of imposex in field-collected gastropods	TBT concentrations in sediment samples co-located with gastropod collection
Crabs	survival, growth, reproduction	potential exposure area: crabs may forage throughout the LDW assessment scale: LDW-wide	tissue-based TRVs for decapods	chemical concentrations in crab tissue collected from four tissue sampling areas located throughout the LDW
Fish				
Juvenile chinook salmon	survival and growth	potential exposure area: juvenile salmonids migrate throughout the LDW and forage in shallow areas assessment scale: intertidal areas throughout the LDW	tissue-based TRVs for chemicals evaluated using a critical tissue-residue approach	chemical concentrations in juvenile chinook salmon tissue collected from middle and lower segments of the LDW
			dietary-based TRVs for chemicals evaluated using a dietary approach	chemical concentrations in juvenile chinook salmon prey collected from intertidal habitat throughout the LDW, stomach contents collected from juvenile chinook salmon captured throughout the LDW, and sediment collected from intertidal habitats throughout the LDW
English sole	survival, growth, reproduction	potential exposure area: English sole may forage throughout the LDW assessment scale: LDW-wide	tissue-based TRVs for chemicals evaluated using a critical tissue-residue approach	chemical concentrations in English sole tissue collected from four tissue sampling areas located throughout the LDW
			dietary-based TRVs for chemicals evaluated using a dietary approach	chemical concentrations in English sole prey and sediment collected throughout the LDW
Pacific staghorn sculpin	survival, growth, reproduction	potential exposure area: sculpin may forage throughout the LDW or small segments of LDW assessment scale: LDW-wide and four modeling areas	tissue-based TRVs for chemicals evaluated using a critical tissue-residue approach	chemical concentrations in sculpin tissue collected from four tissue sampling areas located throughout the LDW
			dietary-based TRVs for chemicals evaluated using a dietary approach	chemical concentrations in sculpin prey and sediment collected throughout the LDW and divided into four modeling areas

ROC	Assessment Endpoint	Assessment Scale	Measures of Effect	Measures of Exposure
Wildlife				
Great blue heron	survival, growth, reproduction	potential exposure area: herons may forage in areas of shallow water depths throughout the LDW assessment scale: LDW-wide intertidal	dietary-based TRVs for birds	chemical concentrations in heron prey collected throughout the LDW and in sediment collected from intertidal habitats throughout the LDW
Osprey	survival, growth, reproduction	potential exposure area: osprey may forage from the top meter of water throughout the LDW assessment scale: LDW-wide	dietary-based TRVs for birds	chemical concentrations in osprey prey collected throughout the LDW and in sediment collected from intertidal habitats throughout the LDW
Spotted sandpiper	survival, growth, reproduction	potential exposure area: sandpipers predominantly forage within small home range segments of the LDW assessment scale: three intertidal modeling areas	dietary-based TRVs for birds	chemical concentrations in sandpiper prey and sediment collected from intertidal habitats throughout the LDW
River otter	survival, growth, reproduction	potential exposure area: river otters may forage throughout the LDW assessment scale: LDW-wide	dietary-based TRVs for mammals	chemical concentrations in river otter prey and sediment collected throughout the LDW
Harbor seal	survival, growth, reproduction	potential exposure area: harbor seals may forage throughout the LDW assessment scale: LDW-wide	dietary-based TRVs for mammals	chemical concentrations in harbor seal prey and sediment collected throughout the LDW

[a] A DMMP SL guideline is available for TBT; however, this guideline was not used in this ERA because it is based on an interstitial water concentration. TBT was included as a COPC based on the results of the Phase 1 ERA (Windward 2003b); the potential for adverse effects associated with exposure to TBT was evaluated using benthic invertebrate tissue data consistent with EPA (1999b) and Meador et al. (2002), and through a direct assessment of effects (i.e., imposex in gastropods collected from the LDW).

DMMP – Dredged Material Management Program
COPC – chemical of potential concern
LDW – Lower Duwamish Waterway
ROC – receptor of concern
SMS – Washington State Sediment Management Standards
SL – screening level
TBT – tributyltin
TRV – toxicity reference value
VOC – volatile organic compound

5.2 BENTHIC INVERTEBRATE ASSESSMENT

This section presents an overview of the risk assessment for the benthic invertebrate community and crabs. The complete assessment is presented in Appendix A, Sections A.3 and A.6.1.

5.2.1 Exposure and effects assessment

This section presents the exposure and effects assessment for benthic invertebrates. The following approaches were used in the assessment for the benthic invertebrate community and crabs:

- Risks to the benthic invertebrate community were evaluated by comparing surface sediment chemical concentrations and site-specific sediment toxicity test results to Washington State SMS; when these were not available, Dredged Material Management Program (DMMP) guidelines and toxicity reference values (TRVs) were used.
- Risks to the benthic invertebrate community from VOC exposure were evaluated by comparing VOC concentrations in porewater to TRVs from the scientific literature.
- Risks to the benthic invertebrate community from TBT exposure were evaluated using results from a site-specific imposex study with gastropods and by comparing TBT concentrations in LDW benthic invertebrate tissue to TRVs from the scientific literature.
- Risks to crabs were evaluated by comparing COPC concentrations in LDW crab tissue to TRVs from the scientific literature.

Each of these approaches is described in the following sections.

5.2.1.1 Sediment data and toxicity test results

The exposure assessment for the benthic invertebrate community summarized the available surface sediment chemistry data for each COPC. The potential for effects on the benthic invertebrate community was assessed by comparing the COPC concentrations in LDW surface sediment to SQS and cleanup screening levels (CSLs) of the Washington State SMS (WAC 173-204). These numerical chemical criteria are based on apparent effects thresholds (AETs) developed for four different benthic endpoints by the Puget Sound Estuary Program (PSEP) (Barrick et al. 1988). An AET is the highest "no effect" chemical-specific sediment concentration above which a significant adverse biological effect always occurred among the several hundred samples used in its derivation. In general, the lowest of the four AETs for each chemical was identified as the SQS; the second lowest AET was identified as the CSL. According to the SMS (WAC 173-204), locations with all chemical concentrations less than or equal to the SQS are defined as having no acute or chronic adverse effects on

biological resources, locations with any chemical concentrations between the SQS and CSL are defined as having minor adverse effects, and locations with any chemical concentration greater than the CSL are defined as having more pronounced adverse effects.

Three COPCs identified during the screening process did not have SMS criteria: nickel, total DDTs, and total chlordane. TRVs for nickel were based on toxicologically based DMMP guidelines for nickel. Because the DMMP guidelines for total DDTs and total chlordane were not toxicologically based, TRVs for these COPCs were selected from the scientific literature.

The effects assessment for the benthic invertebrate community also included the results of site-specific toxicity tests conducted using LDW sediment samples during the RI. Three toxicity tests were conducted with surface sediments (0 to 10 cm) collected at each of 48 locations (Windward 2005c, d). Locations were selected for toxicity testing if one or more chemicals exceeded SMS at that location; the suite of locations selected for testing was selected in consultation with EPA and Ecology. The toxicity tests included:

- Acute 10-day amphipod (*Eohaustorius estuarius*) survival test
- Acute 48-hr bivalve larvae (*Mytilus galloprovincialis*) normal survival test
- Chronic 20-day juvenile polychaete (*Neanthes arenaceodentata*) survival and growth test

Biological responses in these tests were evaluated with respect to the biological criteria from the SMS. In addition to the toxicity tests conducted as part of the RI, the effects assessment included results from toxicity tests conducted by King County on sediment samples collected from seven locations at the Duwamish/Diagonal CSO/SD site.

5.2.1.2 VOCs in porewater

Cis-1,2-dichloroethene was the only VOC with a maximum detected concentration in porewater greater than the no-observed-effect concentration (NOEC) in the problem formulation, and thus was the only COPC identified in porewater for the benthic invertebrate community. The concentrations of cis-1,2-dichloroethene detected in porewater at the two sites sampled as part of the RI (GWI and Boeing Plant 2/Jorgensen Forge) were reported in the exposure assessment. The effects assessment identified the NOEC and lowest-observed-effect concentration (LOEC) that were selected for cis-1,2-dichloroethene in water, based on a review of the scientific literature.

5.2.1.3 TBT

The exposure assessment for the benthic invertebrate community for TBT included the results of the co-located benthic invertebrate tissue and sediment samples collected

and analyzed as part of the RI from 20 locations throughout the LDW. There was a significant non-linear regression relationship between the TBT concentrations in benthic invertebrate tissue and co-located sediment samples. This relationship was used to estimate the maximum TBT concentration in benthic invertebrates tissue. TBT exposure concentrations included both the maximum concentration estimated from the regression and the maximum detected TBT concentration in benthic invertebrates tissue.

The effects assessment identified the NOAEL and lowest-observed-adverse-effect level (LOAEL) for TBT in tissue from the scientific literature. The effects assessment also discussed the results of a site-specific evaluation of imposex (a condition known to be associated with exposure to TBT) in gastropods collected from the LDW. Imposex analysis was performed on multiple individuals of three neogastropod species (*Nassarius mendicus, Astyris gausapata*, and *Olivella baetica)* and on a few individuals (two to five) of four mesogastropod species (*Natica* sp., *Polinices* sp., *L. vincta*, and *Melanella* sp.). Signs of imposex (Stage 2 and maximum relative penis size [RPS] index of 3.4) were observed in *N. mendicus,* but no signs of imposex were observed in the other gastropod species. Stage 2 imposex is not considered to adversely affect the ability of gastropods to reproduce.

5.2.1.4 Crab tissue-residue

The exposure assessment for crabs presented UCL on the mean concentrations of the two COPCs (zinc and total PCBs) in crab tissue from the LDW. The effects assessment for crab identified NOAELs and LOAELs for zinc and total PCBs in tissue, based on a review of the scientific literature.

5.2.2 Risk characterization and uncertainty analysis

The exposure and effects data for the benthic invertebrate community and for crabs were evaluated in the risk characterization to estimate the potential for adverse effects. The ERA also evaluated uncertainties associated with these risk estimates.

5.2.2.1 Sediment data and toxicity test results

The SQS and CSL were exceeded by 39 chemicals at one or more locations; 2 additional chemicals exceeded the SQS only (Table 5-6). Total DDTs, nickel, and total chlordane also exceeded their DMMP guidelines or literature-based TRVs at one or more locations. Total PCBs and BEHP were the chemicals with the highest number of detected concentrations exceeding the SQS and CSL criteria.

Table 5-6. Benthic invertebrate community COPCs with detected concentrations in LDW surface sediments greater than SMS criteria, DMMP guidelines, or TRVs

COPC	Number of Detected Concentrations > SQS and ≤ CSL	Number of Detected Concentrations > CSL
Total PCBs	301	173
BEHP	48	58
Mercury	14	23
Lead	2	19
Zinc	26	16
Total chlordane[a]	19	14
Copper	0	12
Cadmium	2	11
Silver	0	10
Fluoranthene	31	8
BBP	69	8
Indeno(1,2,3-cd)pyrene	15	8
Chromium	1	8
Arsenic	5	8
Phenol	18	7
Benzo(g,h,i)perylene	9	7
Benzoic acid	0	7
Dibenzo(a,h)anthracene	15	4
Nickel[b]	9	4
Total benzofluoranthenes	5	4
4-Methylphenol	0	4
Phenanthrene	24	3
Total HPAH	21	3
Acenaphthene	16	3
Fluorene	11	3
Benzo(a)anthracene	9	3
Dibenzofuran	7	3
Benzo(a)pyrene	5	3
Total LPAH	3	3
Pyrene	1	3
1,4-Dichlorobenzene	0	3
1,2-Dichlorobenzene	0	3
2-Methylnaphthalene	0	3
Dimethyl phthalate	0	2
Naphthalene	0	2
N-Nitrosodiphenylamine	0	2

COPC	Number of Detected Concentrations > SQS and ≤ CSL	Number of Detected Concentrations > CSL
Hexachlorobenzene	4	2
Benzyl alcohol	2	2
Chrysene	23	1
Total DDTs[a]	1	1
1,2,4-Trichlorobenzene	0	1
2,4-Dimethylphenol	0	1
Anthracene	2	0
Pentachlorophenol	1	0

[a] SMS criteria do not exist for these chemicals; the number of exceedances was based on a comparison of sediment chemical concentrations to a TRV.

[b] SMS criteria do not exist for nickel. The DMMP SL and ML values were used for the comparison.

BBP – butyl benzyl phthalate

BEHP – bis(2-ethylhexyl) phthalate

COPC – chemical of potential concern

CSL – cleanup screening level

DMMP – Dredged Material Management Program

HPAH – high-molecular-weight polycyclic aromatic hydrocarbon

LPAH – low-molecular-weight polycyclic aromatic hydrocarbon

ML – maximum level

PCB – polychlorinated biphenyl

SL – screening level

SMS – Washington State Sediment Management Standards

SQS – sediment quality standard

TRV – toxicity reference value

The SMS criteria for biological effects were applied to interpret results of site-specific sediment toxicity tests: 18 of the 48 sediment samples (37.5%) did not exceed the SQS biological effects criteria, 11 sediment samples (22.9%) exceeded the SQS biological effects criteria but not the CSL biological effects criteria, and 19 samples (39.6%) exceeded the CSL biological effects criteria. Because these tests are direct measures of effects at a particular location, the uncertainty in their interpretation was considered to be low at the specific locations tested compared to locations with chemistry data but no site-specific toxicity data. The potential for adverse effects was considered to be uncertain at locations where no detected chemicals exceeded the chemical criteria, guidelines, or TRVs but RLs were greater than criteria and guidelines. RLs were greater than criteria for some chemicals because the sample was diluted or because analysis of the chemical posed analytical challenges in the laboratory.

Thiessen polygons were used to estimate the areal extent of potential effects based on combined toxicity test results and surface sediment chemistry data. Using this approach, no adverse effects to benthic invertebrates living in intertidal and subtidal sediments were estimated for 75% of the LDW area (i.e., the area in which chemical concentrations were less than or equal to chemical SQS criteria and where sediments were nontoxic according to biological SQS criteria). There was a higher likelihood for adverse effects in approximately 7% of the LDW area, which was designated as having chemical concentrations or biological effects in excess of CSL criteria. The remaining 18% of the LDW area had chemical concentrations or biological effects between the

SQS and CSL criteria, indicating that risks to benthic invertebrate communities were considered to be less certain in these areas than in areas with concentrations greater than one or more CSL values. Some uncertainty was associated with these area estimates because areas were estimated by interpolating from individual points at which sediments were sampled.

5.2.2.2 VOCs in porewater

Risks to the benthic invertebrate community from exposure to cis-1,2-dichloroethene were estimated by calculating hazard quotients (HQs) as the ratio of the concentration in porewater to the NOEC and LOEC concentrations. LOEC-based HQs were less than 1.0 at all locations where cis-1,2-dichloroethene was detected in porewater. At two locations at the GWI site, NOEC-based HQs were greater than 1.0 (4.9 and 21); NOEC-based HQs were less than 1.0 at all other locations. These results indicate it is uncertain whether exposure to cis-1,2-dichloroethene within this small area of the LDW is sufficiently high to result in adverse effects. The limited amount of relevant toxicity data available in the literature was identified as a primary uncertainty in the risk characterization for benthic invertebrates exposed to VOCs in porewater.

5.2.2.3 TBT

Risks to the benthic invertebrate community from exposure to TBT were estimated by calculating HQs as the ratio of the TBT concentration in benthic invertebrate tissue collected from the LDW to the selected NOAEL and LOAEL for benthic invertebrate tissue. Risks to benthic invertebrates from TBT were considered to be very low based on NOAEL-based HQs of less than 1.0. The largest source of uncertainty associated with the critical tissue-residue evaluation of risk to benthic invertebrates was the TRV value. The selected TRV was based on the response of a single species, the polychaete *Armandia brevis,* in a spiked sediment bioassay.

Risks to the benthic invertebrate community from TBT exposure were also considered to be low based on the absence of imposex in all gastropods, except one neogastropod species with imposex characterized as Stage 2, a stage that is not expected to impact reproduction. There was uncertainty in the imposex assessment because: 1) neo- and mesogastropod species may already have been affected by TBT and therefore were no longer present in the LDW, and 2) relatively low numbers of gastropods were collected in the LDW.

5.2.2.4 Crab tissue-residue

Risks to crabs from exposure to total PCBs and zinc were estimated by calculating HQs as the ratio of the COPC concentrations in LDW crab tissue to the selected NOAELs and LOAELs for crab tissue. Total PCB concentrations in crab tissue were equal to the lowest concentrations associated with adverse effects in crabs, indicating the potential for adverse effects (Table 5-7). Zinc concentrations in crab tissue were greater than concentrations associated with no effects but less than those associated

with adverse effects, indicating that there is uncertainty as to whether exposure within the LDW would result in adverse effects. The primary uncertainty in the crab risk characterization was the limited number of tissue-based TRVs for total PCBs and zinc available in the literature.

Table 5-7. HQs for crabs using a critical tissue-residue approach

COPC	NOAEL-Based HQ	LOAEL-Based HQ
Total PCBs	**10**	**1.0**
Zinc	**2.5**	0.91

COPC – chemical of potential concern

HQ – hazard quotient

LOAEL – low-observed-adverse-effect level

NOAEL – no-observed-adverse-effect level

PCB – polychlorinated biphenyl

Bold identifies NOAEL-based HQs greater than 1.0 or LOAEL-based HQs greater than or equal to 1.0.

Risks to crabs from exposure to total DDTs and methoxychlor were evaluated in the uncertainty section because of the uncertainty associated with JN-qualified tissue organochlorine pesticide data. These data were considered to be highly uncertain because of analytical interference from PCBs, likely resulting in overestimates of organochlorine pesticide concentrations. The LOAEL-based HQs were less than 1.0 for both methoxychlor and total DDTs. The NOAEL-based HQ was 3.6 for methoxychlor and equal to 1.0 for total DDTs. The TRVs were based on a survival endpoint following acute exposures. Overall, risks to crabs from these pesticides are uncertain but likely to be low.

5.3 Fish Assessment

This section presents an overview of the risk assessment for fish. The complete assessment is presented in Appendix A, Sections A.4 and A.6.2. Three ROCs were selected in the problem formulation to represent fish that use the LDW and may be exposed to sediment-associated chemicals. The three ROCs and COPCs identified for those species were as follows:

- Juvenile chinook salmon—arsenic, cadmium, copper, and vanadium
- English sole—arsenic, cadmium, copper, vanadium, and PCBs
- Pacific staghorn sculpin—arsenic, cadmium, copper, vanadium, TBT, and PCBs

Risks to fish were evaluated using two approaches, depending on the COPC. For TBT and PCBs, concentrations in LDW fish tissue were compared to concentrations in fish tissue associated with adverse effects. For arsenic, cadmium, copper, and vanadium, concentrations in the diet of LDW fish were compared to concentrations associated

with adverse effects. A dietary approach was used for these metals because they are metabolically regulated by fish.

5.3.1 Exposure and effects assessment

Estimated concentrations of arsenic, cadmium, copper, and vanadium in fish diets were presented in the exposure assessment for all three fish ROCs. Dietary concentrations of cadmium, copper, and vanadium were estimated using mean UCL concentrations of COPCs in prey tissue (i.e., benthic invertebrates for all three ROCs plus shiner surfperch for Pacific staghorn sculpin) and incidentally ingested sediment. For arsenic, the linear regression relationship between benthic invertebrate tissue and co-located sediment was used to estimate the arsenic concentration in benthic invertebrate tissue; the mean UCL of this estimated concentration in tissue was used as the exposure concentration for estimating risk. For juvenile chinook salmon, dietary concentrations were also estimated based on chemical analysis of stomach contents of juvenile chinook salmon collected from the LDW.

To evaluate risk using a tissue-residue approach, the exposure assessment presented the mean UCL concentration of total PCBs in English sole tissue and total PCBs and TBT in Pacific staghorn sculpin tissue analyzed in whole-body tissue samples collected from the LDW.

For Pacific staghorn sculpin, exposure was evaluated for two foraging scenarios because the foraging range for this species is unknown and may be smaller than the entire LDW. The two scenarios were: 1) foraging throughout the LDW, and 2) foraging in smaller segments of the LDW corresponding to modeling areas M1, M2, M3, and M4 (Map 5-1). For English sole, data from the entire LDW was combined to estimate the exposure concentrations.

In the effects assessment, toxicological studies from the scientific literature reporting dietary and fish tissue concentrations associated with potential adverse effects on growth, survival, and reproduction of fish were used to derive NOAELs and LOAELs. Studies reporting reproductive endpoints were not used for deriving juvenile chinook salmon TRVs because of their life stage at the time of exposure (i.e., migrating juveniles) and because their exposure to LDW sediments as adults is limited.

5.3.2 Risk characterization and uncertainty analysis

The exposure and effects data were compared in the risk characterization to assess the potential for sediment-associated COPCs to cause adverse effects to juvenile chinook salmon, English sole, and Pacific staghorn sculpin, as discussed in the following subsections.

5.3.2.1 Juvenile chinook salmon

In the risk characterization for juvenile chinook salmon, HQs were calculated as the ratio of the COPC concentrations in the diet (based on either benthic invertebrate data

or juvenile chinook stomach content data) to the dietary NOAELs and LOAELs for arsenic, cadmium, copper, and vanadium. Dietary exposure estimates based on benthic invertebrate tissue data from the LDW resulted in higher HQs for all COPCs than did juvenile chinook salmon stomach content data.

No COPC had a LOAEL-based HQs that exceeded 1.0 (Table 5-8). The cadmium LOAEL-based HQs were 1.0 and 0.92 based on concentrations in benthic invertebrates collected from the LDW and in stomach contents, respectively. These HQs are uncertain because the cadmium TRVs were based on juvenile rockfish, and several toxicity studies using salmonids were available with TRVs that were orders of magnitude higher. Because the salmonid-specific studies likely better represent the sensitivity of juvenile chinook salmon, the selected NOAEL and LOAEL TRVs applied in this ERA may overestimate the sensitivity of juvenile chinook salmon to cadmium.

Table 5-8. Summary of risk characterization for juvenile chinook salmon

COPC	Highest NOAEL-Based HQ[a]	Highest LOAEL-Based HQ[a]
Arsenic	**1.1**	0.73
Cadmium	**5.0**	**1.0**
Copper	**1.9**	0.93
Vanadium	**4.0**	0.79

[a] Dietary exposure based on benthic invertebrate tissue data resulted in higher HQs for all COPCs than did juvenile chinook salmon stomach content data. Thus, the HQs presented in this table are based on benthic invertebrate tissue data.

COPC – chemical of potential concern

HQ – hazard quotient

LOAEL – lowest-observed-adverse-effect level

NOAEL – no-observed-adverse-effect level

Bold identifies NOAEL-based HQs greater than 1.0 and LOAEL-based HQs greater than or equal to 1.0.

NOAEL-based HQs for arsenic, copper, and vanadium were greater than 1.0, whereas LOAEL-based HQs for these COPCs were less than 1.0. Therefore, risks from these COPCs were low and uncertain. The vanadium HQ is based on a NOAEL TRV extrapolated from the only available TRV, which was an unbounded LOAEL. Therefore, risks from vanadium are uncertain and could be higher or lower than the HQs indicate.

An evaluation of risks to juvenile chinook salmon from exposure to endrin was presented in the uncertainty section because of PCB interference in the organochlorine pesticide analyses and resulting uncertainty in the chemistry data. The NOAEL- and LOAEL-based HQs were 3.7 and 0.37, respectively, indicating a low risk with some uncertainty. The HQs are uncertain because: 1) the selected TRVs were based on survival and may have underestimated the potential for sublethal effects, and 2)

analytical interference from PCBs was likely to have overestimated actual organochlorine pesticide concentrations in LDW tissue samples.

Risk from chromium was evaluated in the uncertainty section because a LOAEL TRV was not identified. The NOAEL-based HQ was 2.1 based on benthic invertebrate tissue data and 0.18 based on stomach contents data, indicating low risk with some uncertainty.

A number of uncertainties were evaluated in addition to uncertainties in the TRVs discussed above. In the exposure section, there were uncertainties associated with ROC selection, assessment endpoints, foraging habitat, dietary composition, water exposure to PAHs and metals, benthic invertebrate tissue data, and future habitat changes resulting from restoration. Risks to juvenile chinook salmon from exposure to vanadium would be slightly higher if juvenile chinook salmon were assumed to forage throughout the LDW rather than in intertidal areas only. It is not known if juvenile chinook salmon forage in deeper waters, such as the navigation channel, of the LDW, but it is generally considered less likely. In the effects assessment, there were uncertainties associated with exclusion of field studies in TRV selection and interpretation of regional studies.

5.3.2.2 English sole

In the risk characterization for English sole, HQs were calculated as the ratio of the COPC concentrations in the diet to the dietary NOAELs and LOAELs for arsenic, cadmium, copper, and vanadium. For the critical tissue-residue approach, HQs were calculated as the ratio of the total PCB concentration in English sole tissue to the tissue-based NOAEL and LOAEL TRVs.

Dietary concentrations for two of the six COPCs for fish (cadmium and vanadium) were greater than concentrations associated with adverse effects for English sole (LOAEL-based HQs for cadmium and vanadium were both 1.2); therefore, there is a potential for adverse effects (Table 5-9). Effects data for cadmium were highly variable, and there was uncertainty associated with the selected TRV because effects were partially attributed to reduced feeding. Thus, risks from cadmium were low but uncertain. The vanadium LOAEL-based HQ was based on an unbounded LOAEL reported in the only dietary toxicity study identified. The NOAEL TRV was estimated from the unbounded LOAEL TRV using a safety factor of 5. Because of the very limited effects data, risk from vanadium was uncertain and could have been higher or lower than the HQs indicated.

Table 5-9. Summary of risk characterization for English sole

APPROACH	COPC	NOAEL-BASED HQ	LOAEL-BASED HQ
Tissue residue	total PCBs	**4.9 – 25**[a]	0.98 – **5.0**[a]
Dietary	arsenic	**1.2**	0.80
	cadmium	**6.1**	**1.2**
	copper	**1.9**	0.93
	vanadium	**5.9**	**1.2**

[a] Because of uncertainty in the LOAEL, LOAEL-based HQs were calculated from a range of effects concentrations reported in Hugla and Thome (1999). The NOAEL TRV range was estimated by dividing the LOAEL TRV range by an uncertainty factor of 5.

COPC – chemical of potential concern

HQ – hazard quotient

LOAEL – lowest-observed-adverse-effect level

NOAEL – no-observed-adverse-effect level

Bold identifies NOAEL-based HQs greater than 1.0 and LOAEL-based HQs greater than or equal to 1.0.

For PCBs, the LOAEL-based HQs ranged from 0.98 to 5.0 based on toxicity information presented in one study reporting the lowest TRVs (Hugla and Thome 1999). A NOAEL was not available from this study, and no NOAELs were reported for any other study below the selected LOAEL range. There is uncertainty in the risk estimates because of uncertainties with the study reporting the lowest TRVs. If the study reporting the next higher LOAEL TRV had been selected, exposures would have been below the effects thresholds (LOAEL-based HQ of 0.28). Risks were also estimated from PCB congener concentrations in English sole tissue using a TEQ approach; those risks were determined to be very low.

Estimated dietary exposures of English sole to two additional COPCs (arsenic and copper) were greater than their respective no-effects levels but lower than the adverse effect levels associated with survival, growth, or reproduction, indicating that there is uncertainty as to whether exposure to these metals within the LDW is sufficiently high to result in adverse effects.

An evaluation of risks to English sole from exposure to endosulfan and endrin was presented in the uncertainty section because of PCB interference in the organochlorine pesticide analyses. The LOAEL-based HQs were less than 1.0 for endosulfan and 1.2 for endrin. These HQs are uncertain because: 1) the selected TRVs were based on survival and may have underestimated the potential for sublethal effects, and 2) analytical interference from PCBs is likely to have overestimated actual organochlorine pesticide concentrations in LDW tissue samples.

LOAEL TRVs were not available for chromium and benzoic acid, so risks from these chemicals were evaluated in the uncertainty analysis. Comparison of dietary exposure concentrations to NOAEL TRVs resulted in NOAEL-based HQs of 1.1 and 1.5,

respectively, for chromium and benzoic acid, indicating low risk with some uncertainty.

Regional studies, which included samples collected from the LDW, indicated an increased risk of adverse effects on English sole reproduction in the LDW. Potential causal factors included PAHs and PCBs; however, these effects cannot be conclusively associated with exposure to specific chemicals or chemical mixtures in the LDW because, among other factors, of the complex mixtures of chemicals in the field and uncertainties in the English sole home range.

A number of uncertainties were evaluated in addition to uncertainties in the TRVs and additional chemicals discussed above. Uncertainties associated with ROC selection, incidental sediment ingestion, dietary composition, foraging range, water exposure to PAHs and metals, benthic invertebrate tissue data, and future habitat changes resulting from restoration were not expected to have an effect on risk conclusions. Uncertainties associated with the use of field studies were also discussed.

5.3.2.3 Pacific staghorn sculpin

In the risk characterization for Pacific staghorn sculpin, HQs were calculated as the ratio of the COPC concentrations in the diet to the dietary NOAELs and LOAELs for arsenic, cadmium, copper, and vanadium. For the critical tissue-residue approach, HQs were calculated as the ratio of the total PCB or TBT concentrations in Pacific staghorn sculpin tissue to the tissue-based NOAEL and LOAEL TRVs.

Both the NOAEL- and LOAEL-based HQs were greater than 1.0 for vanadium in modeling areas M2 and M3 (NOAEL- and LOAEL-based HQs of 1.2 and 5.9, respectively, in both areas) (Table 5-10). In modeling areas M1 and M4 and LDW-wide, NOAEL-based HQs were greater than 1.0, but LOAEL-based HQs were less than or equal to 1.0. As discussed above for English sole, because of the limited amount of effects data available, risks from vanadium were uncertain and could have been higher or lower than the HQs indicated.

Table 5-10. Summary of risk characterization for Pacific staghorn sculpin

APPROACH	COPC	FORAGING ASSUMPTION	NOAEL-BASED HQ OR RANGE OF HQs[a]	LOAEL-BASED HQ OR RANGE OF HQs[a]
Tissue residue	total PCBs	LDW-wide	**2 – 11**[b]	0.42 – **2.1**[b]
		modeling areas	**1.5** – **19**	0.30 – **3.8**
	TBT	LDW-wide	**2.0**	0.23
		modeling areas	**1.6 – 2.9**	0.18 – 0.33
Dietary	arsenic	LDW-wide	0.75	0.50
		modeling areas	0.60 – 0.80	0.40 – 0.53
	cadmium	LDW-wide	**3.8**	0.76
		modeling areas	**3.0 – 5.2**	0.60 – **1.0**
	copper	LDW-wide	**1.1**	0.56
		modeling areas	0.90 – **1.5**	0.45 – 0.77
	vanadium	LDW-wide	**4.0**	0.79
		modeling areas	**3.2 – 5.9**	0.65 – **1.2**

[a] Range of HQs for the four modeling areas evaluated for Pacific staghorn sculpin.

[b] Because of uncertainty in the LOAEL, LOAEL-based HQs were calculated from a range of effects concentrations reported in Hugla and Thome (1999). The NOAEL TRV range was estimated by dividing the LOAEL TRV range by an uncertainty factor of 5.

COPC – chemical of potential concern

HQ – hazard quotient

LDW – Lower Duwamish Waterway

LOAEL – lowest-observed-adverse-effect level

NOAEL – no-observed-adverse-effect level

PCB – polychlorinated biphenyl

TBT – tributyltin

TRV – toxicity reference value

Bold identifies NOAEL-based HQs greater than 1.0 and LOAEL-based HQs greater than or equal to 1.0.

For PCBs, the LOAEL-based HQs ranged from 0.30 to 3.8 based on toxicity information presented in one study reporting the lowest TRVs (Hugla and Thome 1999) and exposure data from discrete areas in the LDW as well as LDW-wide. A NOAEL was not available from Hugla and Thome (1999), and no NOAELs below the selected LOAEL range were reported for any other study. There is uncertainty in the risk estimates because of uncertainties with the study reporting the lowest TRVs. If the study reporting the next higher LOAEL TRV had been selected, the LOAEL-based HQ would have been 0.12 for LDW-wide exposure instead of ranging from 0.42 to 2.1.

The LOAEL-based HQ for cadmium was equal to 1.0 in modeling area M3 but lower than 1.0 in all other modeling areas and LDW-wide. As discussed above for English sole, there was some uncertainty associated with the selected cadmium TRV because effects were partially attributed to reduced feeding. In addition, data used to represent crabs in Pacific staghorn sculpin dietary exposure calculations were uncertain; the LOAEL-based HQ for cadmium would have been 1.9 if crab data had been used

instead of benthic invertebrate data. LOAEL-based HQs were less than 1.0 for TBT and copper. NOAEL-based HQs for TBT and copper were greater than 1.0 LDW-wide and in one or more modeling areas, indicating that the potential for adverse effects was low but uncertain.

An evaluation of risks to Pacific staghorn sculpin from exposure to endosulfan and endrin was presented in the uncertainty section because the presence of PCBs interfered in the organochlorine pesticide analyses. This evaluation indicated a low risk from endosulfan and the potential for adverse effects from endrin (LOAEL-based HQ of 3.1). These HQs were uncertain because: 1) the selected TRVs were based on survival and may have underestimated the potential for sublethal effects, and 2) analytical interference from PCBs is likely to have overestimated actual organochlorine pesticide concentrations in LDW tissue samples.

LOAEL TRVs were not available for chromium and benzoic acid, so risks from these chemicals were evaluated in the uncertainty analysis. Comparison of dietary exposure concentrations to NOAEL TRVs resulted in NOAEL-based HQs of 0.85 and 2.1, respectively, for chromium and benzoic acid, indicating low risk with some uncertainty.

A number of uncertainties were evaluated in addition to uncertainties in the TRVs discussed above. Uncertainties associated with the COPC screen, ROC selection, water exposure to PAHs and metals, PCB analysis approach (total PCBs versus PCB TEQ), and future habitat changes resulting from restoration were not expected to have an effect on risk conclusions.

In the uncertainty analysis, uncertainty associated with the benthic invertebrate portion of Pacific staghorn sculpin diet was evaluated. Use of large crab tissue data versus benthic invertebrate tissue samples (which included crabs and shrimps < 20 mm) resulted in slight changes in LOAEL-based HQs for cadmium, copper, and vanadium. For arsenic, no HQs changed from less than 1.0 to greater than 1.0 or vice versa, based on this data substitution.

5.4 WILDLIFE ASSESSMENT

This section presents an overview of the risk assessment for wildlife. The complete assessment is presented in Appendix A, Sections A.5 and A.6.3. Five ROCs were selected in the problem formulation to represent wildlife that use the LDW and may be exposed to sediment-associated chemicals. The five ROCs and associated COPCs were:

- Spotted sandpiper – arsenic, cadmium, chromium, cobalt, copper, lead, mercury, nickel, selenium, vanadium, zinc, and PCBs
- Great blue heron – chromium, lead, mercury, and PCBs
- Osprey – arsenic, chromium, lead, mercury, and PCBs

- River otter – arsenic, cobalt, mercury, selenium, and PCBs
- Harbor seal – mercury and PCBs

Risks to wildlife ROCs were evaluated by comparing estimated COPC concentrations as a dietary dose for each ROC to dietary doses associated with adverse effects from the scientific literature.

5.4.1 Exposure and effects assessment

For wildlife ROCs, the daily exposure doses of COPCs through the ingestion of prey, water, and incidental sediment were estimated in the exposure assessment. Exposure doses were calculated as follows:

$$\text{Exposure dose} = \frac{[(\text{FIR} \times C_{food}) + (\text{WIR} \times C_{water}) + (\text{SIR} \times C_{sed})] \times \text{SUF}}{\text{BW}} \qquad \textbf{Equation 5-1}$$

Where:

Exposure dose	=	COPCs ingested per day via food, water, and sediment (mg COPC/kg body weight/day)
FIR	=	food ingestion rate (kg food dw/day)
C_{food}	=	concentration in prey items (mg COPC/kg food dw)
WIR	=	water ingestion rate (L water/day)
C_{water}	=	concentration in water (mg COPC/L water)
SIR	=	sediment ingestion rate (kg sediment dw/day)
C_{sed}	=	concentration in sediment (mg COPC/kg dw)
SUF	=	site use factor (unitless); fraction of time that a receptor spends foraging in the LDW relative to the entire home range
BW	=	ROC species body weight (kg ww)

Body weights, and food, water, and sediment ingestion rates for each receptor were derived from the literature. Site use factors for great blue heron, osprey, and harbor seal were less than 1.0 because site-specific data showed that in addition to foraging in the LDW, these receptors also foraged outside the LDW. Spotted sandpiper and river otter were assumed to forage only in the LDW. Within the LDW, the foraging areas for all wildlife ROCs except spotted sandpiper, were assumed to be the entire LDW. Spotted sandpiper foraging areas were identified based on a field survey conducted as part of the RI in 2004 (Windward 2004i). The field survey identified potential nesting and foraging habitat. Assuming a foraging range of about 1 mile from the nesting habitat, three hypothetical exposure areas were created. Within each area, two foraging scenarios were evaluated: one in which spotted sandpipers forage in only high-quality habitat, and another in which they forage in both high- and poor-quality habitats. These two foraging scenarios in each of the three exposure areas resulted in a total of six exposure scenarios for spotted sandpiper (Map 5-2).

Concentrations of COPCs in prey, sediment, and water were calculated as the UCL on the mean of the data within the relevant exposure area for each ROC, with a few exceptions. Total PCBs in sediment were calculated as the 95% UCL of the SWAC within relevant exposure areas. A linear regression relationship between benthic invertebrate tissue and co-located sediment was used to estimate UCL on the mean benthic invertebrate tissue concentrations of arsenic and total PCBs in each exposure area.

The effects assessment presented a review of the scientific literature for dietary exposure doses associated with adverse effects on survival, growth, and reproduction in birds and mammals. NOAELs and LOAELs were selected for each COPC based on these toxicity data.

5.4.2 Risk characterization and uncertainty analysis

The exposure and effects data were compared in the risk characterization to assess the potential for sediment-associated COPCs to have adverse effects on the ROCs. For each ROC, HQs were calculated as the ratio of the COPC concentration in the daily exposure dose to the NOAELs and LOAELs. The following sections present the risk conclusions for each wildlife receptor.

5.4.2.1 Spotted sandpiper

The following COPCs had LOAEL-based HQs greater than or equal to 1.0 for spotted sandpiper: chromium, copper, lead, mercury, and vanadium (Table 5-11). LOAEL-based HQs ranged from 1.0 to 5.5 for these COPCs. LOAEL-based HQs were equal to or greater than 1.0 in Exposure Area 2 for chromium (1.8) and lead (5.5), in Exposure Area 3 for copper (1.1) and mercury (1.0), and in all three exposure areas for vanadium (1.0 to 1.4) (see Map 5-2 for exposure areas). Risks to spotted sandpiper from copper and vanadium may be under- or overestimated because the selected TRVs were based on subchronic growth endpoints. The ingested dose was primarily from sediment for copper and lead in Exposure Area 3 and for vanadium in all areas. Bioavailability of metals in sediment is unlikely to be 100%, so the LOAEL-based HQs (ranging from 1.0 to 1.5) for copper, lead, and vanadium may have been overestimated. Overall, these findings indicated risks to spotted sandpiper in some areas of the LDW from exposure to chromium, copper, lead, mercury, and vanadium, but these risks are expected to be low.

Table 5-11. Summary of risk characterization for spotted sandpiper

COPC	Range of NOAEL-Based HQs[a]	Range of LOAEL-Based HQs[a]
Arsenic	0.34 – 0.58	< 0.1 – 0.15
Cadmium	< 0.1 – 0.21	< 0.1
Chromium	**1.3 – 8.8**	0.26 – **1.8**
Cobalt	0.22 – 0.27	< 0.1

COPC	RANGE OF NOAEL-BASED HQs[a]	RANGE OF LOAEL-BASED HQs[a]
Copper	0.62 – **1.5**	0.45 – **1.1**
Lead	0.58 – **19**	0.17 – **5.5**
Mercury	**1.1** – **5.3**	0.21 – **1.0**
Nickel	< 0.1 – 0.22	< 0.1– 0.11
Selenium	0.62 – 0.90	0.38 – 0.55
Vanadium	**2.0** – **2.7**	**1.0** – **1.4**
Zinc	0.41 – 0.78	0.27 – 0.52
Total PCBs	0.51 – **2.0**	0.18 – 0.71
PCB TEQs	**1.9** – **10**	0.18 – **1.5**

[a] Range of HQs for the six spotted sandpiper exposure scenarios for each COPC.

COPC – chemical of potential concern

HQ – hazard quotient

LOAEL – lowest-observed-adverse-effect level

NOAEL – no-observed-adverse-effect level

PCB – polychlorinated biphenyl

Bold identifies NOAEL-based HQs greater than 1.0 and LOAEL-based HQs greater than or equal to 1.0.

For total PCBs, the NOAEL-based HQs were greater than 1.0 in some of the spotted sandpiper exposure areas, but LOAEL-based HQs in all areas were less than 1.0, indicating low risk with some uncertainty because the true threshold of effects between the NOAEL and the LOAEL is not known. For PCB TEQs, the LOAEL-based HQs were less than 1.0, except in Exposure Area 2 under the high-quality foraging habitat scenario, in which the LOAEL-based HQ was 1.5. Uncertainty was considered to be high for PCB TEQ risk estimates because the TRVs were based on a study that used an acute weekly dose via intraperitoneal (IP) injection, and TEFs derived from studies of toxicity to eggs were applied to dietary exposure estimates for adults. Therefore, risks to spotted sandpiper from PCBs were considered to be low in most areas of the LDW, with some uncertainty.

For the remaining COPCs (arsenic, cadmium, cobalt, nickel, selenium, and zinc), all NOAEL- and LOAEL-based HQs were less than 1.0 in all sandpiper exposure areas. Exposure of spotted sandpipers to these COPCs was not expected to result in adverse effects.

An evaluation of risks from exposure to total DDT was conducted in the uncertainty section because of PCB interference in the organochlorine pesticide analyses. The NOAEL-based HQs were greater than 1.0 in some of the spotted sandpiper exposure areas, but LOAEL-based HQs were less than 1.0, indicating low risk. It is likely that HQs would have been lower if there had been no analytical interferences from PCBs because the interference is known to bias the DDT concentrations high. Therefore, risks to spotted sandpiper from DDT are expected to be very low in most areas of the LDW, with some uncertainty.

A number of additional uncertainties were evaluated in the uncertainty section. Uncertainties in COPC selection, food ingestion rate, direct sediment contact, incidental sediment ingestion, dietary composition, and site use were expected to have minimal or no effect on risk conclusions. Risks were considered to be uncertain for cadmium, cobalt, copper, nickel, vanadium, and zinc because TRVs were based on subchronic exposure and growth endpoints. Uncertainties in toxicity data for chromium, lead, mercury, selenium, and PCBs (based on a total PCB approach) could also have affected risk estimates. Risks could have been underestimated for arsenic because of the lack of arsenite toxicity data.

5.4.2.2 Great blue heron

The NOAEL- and LOAEL-based HQs for great blue heron were all less than 1.0, indicating very low risks to great blue herons were expected from exposure to chromium, lead, mercury, and PCBs in the LDW (Table 5-12). For chromium and for PCB TEQs, there was high uncertainty in the TRVs, but the LOAEL TRVs would have needed to be lower by a factor of 14 for the LOAEL-based HQs to equal 1.0. There was also some uncertainty in the PCB TEQ risk estimate because TEFs derived from studies of toxicity to eggs were applied to dietary exposure estimates for adults.

Table 5-12. Summary of risk characterization for great blue heron

COPC	NOAEL-Based HQ	LOAEL-Based HQ
Chromium	< 0.1	< 0.1
Lead	< 0.1	< 0.1
Mercury	0.17	< 0.1
Total PCBs	0.35	0.12
PCB TEQs	0.66	< 0.1

COPC – chemical of potential concern
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
NOAEL – no-observed-adverse-effect level
PCB – polychlorinated biphenyl
TEQ – toxic equivalent

Uncertainties in incidental sediment ingestion, dietary composition, and site use were evaluated and expected to have minimal or no effect on risk conclusions. The uncertainty analysis section also evaluated risks from the exposure of great blue heron to total DDTs. The NOAEL-based HQ was less than 1.0, indicating very low risk.

5.4.2.3 Osprey

The NOAEL- and LOAEL-based HQs for osprey for chromium, lead, and mercury were less than 1.0, indicating that there are very low risks to osprey from exposure to these COPCs in the LDW (Table 5-13). There was high uncertainty in the chromium

TRV, but the LOAEL TRV would have needed to be lower by a factor of 50 for the LOAEL-based HQ to equal 1.0.

Table 5-13. Summary of risk characterization for osprey

COPC	NOAEL-BASED HQ	LOAEL-BASED HQ
Chromium	0.11	< 0.1
Lead	< 0.1	< 0.1
Mercury	0.28	< 0.1
Total PCBs	0.65	0.23
PCB TEQs	**1.6**	0.16

COPC – chemical of potential concern

HQ – hazard quotient

LOAEL – lowest-observed-adverse-effect level

NOAEL – no-observed-adverse-effect level

PCB – polychlorinated biphenyl

TEQ – toxic equivalent

Bold identifies NOAEL-based HQs greater than 1.0.

NOAEL- and LOAEL-based HQs for total PCBs were less than 1.0. The LOAEL-based HQ for PCBs using the PCB TEQ approach was less than 1.0, but the NOAEL-based HQ was 1.6, indicating a low risk with uncertainty. There were no PCB congener data for juvenile chinook salmon, one of the three species assumed to be ingested by osprey in this assessment, so the portion of prey ingestion that had been assigned to juvenile chinook salmon (45%) was divided proportionally among shiner surfperch and English sole. Shiner surfperch had higher PCB concentrations than did juvenile chinook salmon, so is it possible that PCB TEQ HQs would have been lower if PCB congener data had been available for juvenile salmon. There was also high uncertainty in the PCB TEQ TRV because: 1) birds in the study were given an acute weekly dose via IP injection rather than via the diet, which may result in an overestimate of exposure and resulting toxicity, and 2) TEFs derived from studies of toxicity to eggs were applied to dietary exposure estimates for adults.

The uncertainty assessment included an estimate of total PCB concentrations in osprey eggs using a biomagnification factor relating total PCB concentrations in fish to concentrations in bird eggs. The comparison of the estimated total PCB concentration in eggs to the LOAEL TRV resulted in an HQ of 1.4. This approach was highly uncertain, resulting in potential over- or underestimates of risk. Thus, for total PCBs, the ingested dose approach with its associated uncertainties indicated that risks to osprey were low, while risks calculated using estimated concentrations in eggs indicated that risks were low with a high level of uncertainty.

Uncertainties in incidental sediment ingestion and dietary composition were evaluated and expected to have minimal or no effect on risk conclusions. The

uncertainty section also evaluated risks from exposure of osprey to total DDTs. The NOAEL-based HQ was less than 1.0, indicating very low risk.

5.4.2.4 River otter

The NOAEL- and LOAEL-based HQs for PCBs of 5.8 and 2.9, respectively, using the total PCB approach, indicated a risk to river otters from PCB exposure (Table 5-14). The PCB TRVs were based on an 18-month study with mink in which reduced growth of offspring was reported. The NOAEL-based HQ of 4.5 using PCB congener data and the PCB TEQ approach indicated that there could be some risk to river otters from PCB exposure, although the LOAEL-based HQ was less than 1.0 using this approach. The PCB TEQ TRVs were based on a 90-day study with guinea pigs in which reduced body weights were reported. There were some uncertainties associated with the PCB TRVs and with the TEF approach, but it was not known whether these uncertainties would result in an under- or overestimate of risk. Therefore, risks to river otter from PCBs were considered to be low, with some uncertainty.

Table 5-14. Summary of risk characterization for river otter

COPC	NOAEL-Based HQ	LOAEL-Based HQ
Arsenic	0.12	< 0.1
Cobalt	0.16	< 0.1
Mercury	**2.8**	0.57
Selenium	0.58	0.40
Total PCBs	**5.8**	**2.9**
PCB TEQs	**4.5**	0.59

COPC – chemical of potential concern

HQ – hazard quotient

LOAEL – lowest-observed-adverse-effect level

NOAEL – no-observed-adverse-effect level

PCB – polychlorinated biphenyl

TEQ – toxic equivalent

Bold identifies NOAEL-based HQs greater than 1.0 and LOAEL-based HQs greater than or equal to 1.0.

For mercury, the LOAEL-based HQ was less than 1.0 (0.57), but the NOAEL-based HQ was greater than 1.0 (2.8). There was some uncertainty in the TRV, which was based on a chronic study of rats resulting in effects on growth, because the NOAEL was estimated from the LOAEL using an uncertainty factor of 5. Therefore, risks to river otter from mercury were considered to be low, with some uncertainty.

The NOAEL- and LOAEL-based HQs for river otter for arsenic, cobalt, and selenium were less than 1.0, indicating very low risks to river otters from exposure to these COPCs in the LDW.

Uncertainties in food and incidental sediment ingestion rates and dietary composition were evaluated and were expected to have minimal or no effect on risk conclusions.

5.4.2.5 Harbor seal

The NOAEL- and LOAEL-based HQs for harbor seal for mercury and PCBs were less than 1.0, indicating very low risk to harbor seals from exposure to these COPCs in the LDW (Table 5-15). Uncertainties in incidental sediment ingestion rates and dietary composition were expected to have minimal or no effect on risk conclusions.

Table 5-15. Summary of risk characterization for harbor seal

COPC	NOAEL-Based HQ	LOAEL-Based HQ
Mercury	0.19	< 0.1
Total PCBs	0.44	0.22
PCB TEQs	0.35	< 0.1

COPC – chemical of potential concern
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
NOAEL – no-observed-adverse-effect level
PCB – polychlorinated biphenyl
TEQ – toxic equivalent

5.5 Summary and Selection of Ecological Risk Drivers

In summary, risk estimates for PCBs indicated a potential for adverse effects to the benthic invertebrate community, crabs, spotted sandpiper, and river otter (Table 5-16). There is a potential for adverse effects to English sole, and Pacific staghorn sculpin; however, the risk estimates for PCBs for these two fish species are uncertain both because the exposure concentrations were in between the concentrations selected as the LOAEL TRV range and because of the uncertainty in the study that served as the basis for the LOAEL TRV range itself. There is also a potential for adverse effects for osprey from PCBs based on the PCB TEQ, which was highly uncertain. Other COPCs with exposures greater than or equal to levels associated with adverse effects for at least one fish or wildlife receptor were cadmium, chromium, copper, lead, mercury, and vanadium. Numerous additional chemicals pose a risk to the benthic invertebrate community, as indicated in Table 5-6. No quantitative ecological risk estimates were calculated for dioxins and furans.

Table 5-16. List of COPCs and ROCs with NOAEL- or LOAEL-based HQs ≥ 1.0

COPC	ROC	NOAEL-Based HQ	LOAEL-Based HQ
COPCs with LOAEL-based HQs ≥ 1.0[a]			
Total PCBs	English sole	**4.9 – 25[b]**	**0.98 – 5.0[b]**

COPC	ROC	NOAEL-BASED HQ	LOAEL-BASED HQ
	Pacific staghorn sculpin	**3.8 – 19**[b]	**0.76 – 3.8**[b]
	crabs	**10**	**1.0**
	river otter	**5.8**	**2.9**
PCB TEQs	spotted sandpiper	**1.9 – 15**	0.18 – **1.5**
Cadmium	juvenile chinook salmon	**5.0**	**1.0**
	English sole	**6.1**	**1.2**
	Pacific staghorn sculpin	**3.0 – 5.2**	0.60 – **1.0**
Chromium	spotted sandpiper	**1.3 – 8.8**	0.26 – **1.8**
Copper	spotted sandpiper	0.62 – **1.5**	0.45 – **1.1`**
Lead	spotted sandpiper	0.58 – **19**	0.17 – **5.5**
Mercury	spotted sandpiper	**1.1 – 5.3**	0.21 – **1.0**
Vanadium	English sole	**5.9**	**1.2**
	Pacific staghorn sculpin	**3.2 – 5.9**	0.65 **– 1.2**
	spotted sandpiper	**2.0 – 2.7**	**1.0 – 1.4**
COPCs with NOAEL-based HQs ≥ 1.0 and LOAEL-based HQs < 1.0[c]			
Total PCBs	English sole	**1.4**	0.28
	Pacific staghorn sculpin	0.42 – **1.1**	< 0.1 – 0.22
	spotted sandpiper	0.51 – **2.0**	0.18 – 0.71
PCB TEQs	osprey	**1.6**	0.16
	river otter	**4.5**	0.59
Arsenic	juvenile chinook salmon	**1.1**	0.73
	English sole	**1.2**	0.80
	crabs	**3.9**	na
Benzoic acid	English sole	**1.5**	na
	Pacific staghorn sculpin	**2.1**	na
Cadmium	Pacific staghorn sculpin	**3.0 – 4.9**	0.60 – 0.98
Chromium	juvenile chinook salmon	**2.1**	na
	English sole	**1.1**	na
Copper	juvenile chinook salmon	**1.9**	0.93
	English sole	**1.9**	0.93
	Pacific staghorn sculpin	0.9 – **1.5**	0.45 – 0.77
Mercury	river otter	**2.8**	0.57
TBT	Pacific staghorn sculpin	**1.6** – **2.9**	0.18 – 0.33
Vanadium	juvenile chinook salmon	**4.0**	0.79
Zinc	crabs	**2.5**	0.91

Note: HQs reported for fish are the highest HQs in cases where more than one approach was used.

[a] The LOAEL-based HQs for endrin were 1.2 and 3.1 for English sole and Pacific staghorn sculpin, respectively, based on risk calculations discussed in the uncertainty analysis. These calculations were discussed only in the uncertainty analysis because of analytical interferences from PCB Aroclors in the organochlorine pesticide analyses, resulting in uncertainties in pesticide identification and a high bias in pesticide concentrations.

[b] LOAEL-based HQs were calculated from a range of effects concentrations reported in Hugla and Thome (1999) because of uncertainty in the LOAEL. The NOAEL TRV range was estimated by dividing the LOAEL TRV range by an uncertainty factor of 5. Ranges reported for Pacific staghorn sculpin also included the range in exposure estimates for areas smaller than the entire LDW.

[c] The NOAEL-based HQs were greater than 1.0 for the following COPC/ROC pairs based on risk calculations discussed in the uncertainty analysis: 1) total DDTs and spotted sandpiper (2.6 to 4.3), 2) endrin and juvenile chinook salmon (3.6), 3) alpha-endosulfan and English sole (6.8) and Pacific staghorn sculpin (2.3), 4) beta-endosulfan and English sole (29) and Pacific staghorn sculpin (6.6), 5) endrin and juvenile chinook salmon (3.6), and 6) methoxychlor and crabs (3.6). These calculations were discussed in the uncertainty analysis because of analytical interferences from PCB Aroclors in the organochlorine pesticide analyses, resulting in uncertainties in pesticide identification and a high bias in pesticide concentrations.

COPC – chemical of potential concern
HQ – hazard quotient
LOAEL – low-observed-adverse-effect level
na – not available
NOAEL – no-observed-adverse-effect level
PCB – polychlorinated biphenyl
ROC – receptor of concern
TBT – tributyltin
TEQ – toxic equivalent

Bold identifies NOAEL-based HQs greater than 1.0 or LOAEL-based HQs greater than or equal to 1.0.

Chemicals were identified as risk drivers for ecological receptors based on the risk estimates, uncertainties discussed in the ERA (Appendix A), natural background concentrations, and residual risks following planned early actions in the LDW. The risk drivers from both the ERA and the HHRA will be the focus of remedial analyses in the FS.

In consultation with EPA and Ecology, PCBs were identified as a risk driver for river otter because estimated exposures of river otter were greater than the LOAEL by a factor of 2.9 and uncertainties in the risk estimate were relatively low. In addition, 41 chemicals were selected as risk drivers for benthic invertebrates because detected concentrations of these 41 chemicals exceeded the SQS of the Washington SMS in one or more locations.

Other chemicals that exceeded risk thresholds (LOAEL-based HQ ≥ 1.0) but were not selected as risk drivers may be addressed through focused evaluation in the FS, as part of the 5-year review, or included in the post-remedial monitoring program, as appropriate.

The ERA was based on the baseline surface sediment dataset, which included sediment data collected prior to early actions in the LDW. Since these data were collected, early actions in the LDW have been conducted at two locations (Duwamish/Diagonal and the Boeing Developmental Center south SD in the Norfolk area). Therefore, the risks discussed in the ERA may represent an overestimate of current risks for areas where remediation has already occurred.

6 Summary of the Baseline Human Health Risk Assessment

The baseline HHRA estimated risks for people who may be exposed to COPCs present in sediment, water, and aquatic biota in the LDW. The dataset used in the baseline HHRA consisted of historical data used in the Phase 1 HHRA and sediment and tissue chemistry data collected from the LDW as part of the RI, as summarized in Appendix B, Section B.2.1. This section summarizes the exposure assessment (including the conceptual site model and selection of exposure pathways, the screening and evaluation of COPCs, and the selection of exposure parameters), toxicity assessment, risk characterization, uncertainty analysis, and selection of risk drivers for human health. The complete HHRA is included as Appendix B of this RI.

6.1 EXPOSURE ASSESSMENT

This section summarizes the conceptual site model, selection of exposure pathways, screening and evaluation of COPCs, and selection of exposure parameters used in the baseline HHRA.

6.1.1 Conceptual site model and selection of exposure pathways

The conceptual site model developed in the baseline HHRA describes scenarios in which people could be exposed to COPCs associated with sediment in the LDW (Figure 6-1). The primary exposure scenarios were identified through input from site users, including local residents and members of the Muckleshoot and Suquamish Tribes, and through the review of prior risk assessments of the LDW. Exposure pathways consisted of direct contact with sediments during commercial netfishing, beach play, and clam harvesting in the LDW, as well as indirect exposure through consumption of seafood from the LDW. Exposures associated with swimming in the LDW were evaluated through the inclusion of risk estimates developed previously as part of the King County water quality assessment HHRA (1999c).

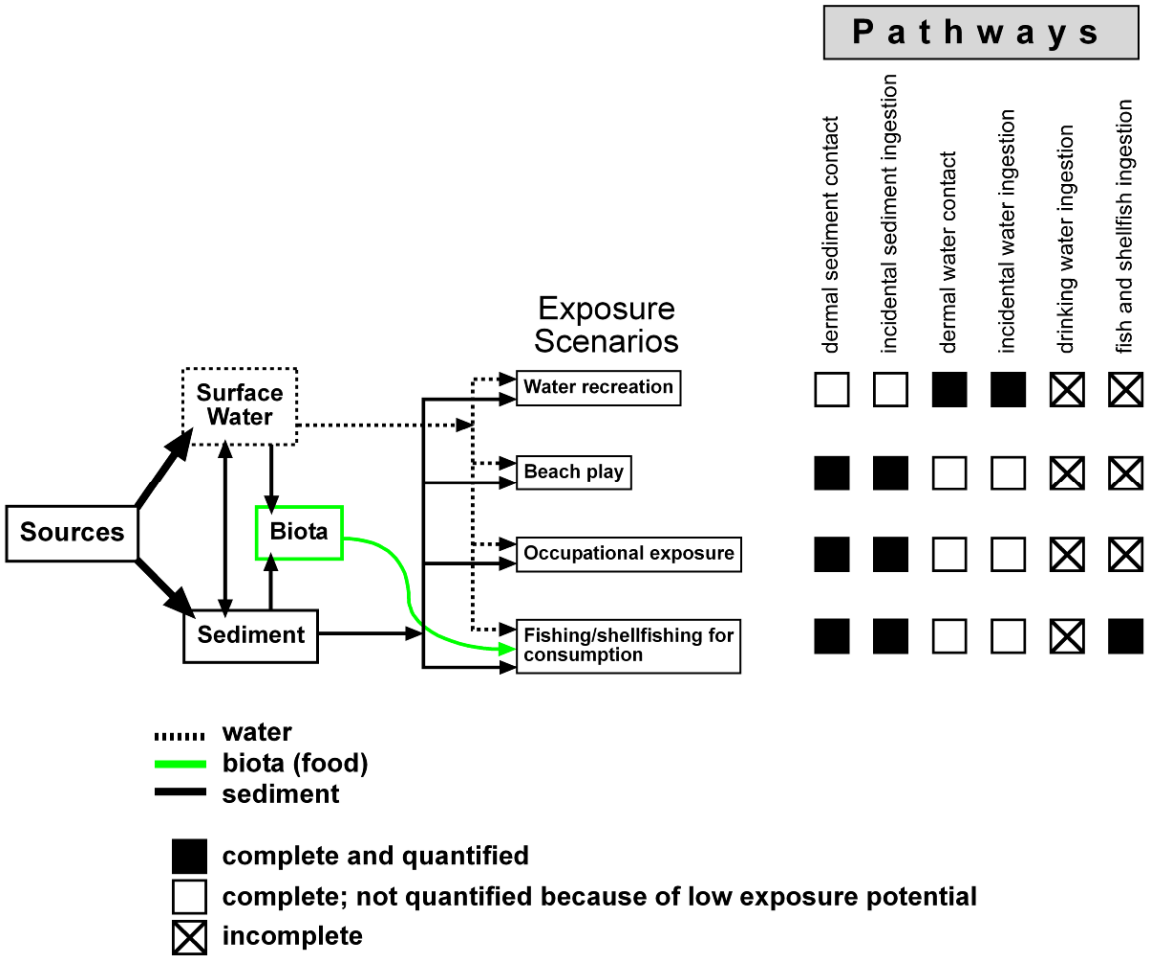


Figure 6-1. Conceptual site model for baseline human health risk assessment

6.1.2 Screening and evaluation of chemicals of potential concern

This section provides a review of the COPC screening process, including the data evaluation.

6.1.2.1 Data evaluation

Sediment chemistry data used in the baseline HHRA consisted of chemical concentrations in the uppermost 15 cm of the sediment (i.e., surface sediment). The dataset included approximately 600 surface sediment samples from intertidal locations and approximately 750 surface sediment samples from subtidal locations in the LDW. The dataset consisted of data collected prior to early removal actions in the LDW in order to reflect baseline conditions. In addition, some chemistry data for tissue and sediment samples from outside the LDW were identified from other datasets or collected as part of the RI for comparison to LDW data.

Tissue data were available for English sole, starry flounder, crabs, clams, mussels, and perch caught within the LDW. Seafood consumption rates applied to these tissue data

were based largely on consumption studies representative of seafood harvest from other areas of Puget Sound. Many of the species that may be consumed from these other areas (e.g., speckled sanddab, Pacific cod, rockfish, spiny dogfish, walleye pollock) are rarely found in the LDW, and no LDW tissue chemistry data were available for those species. Therefore, data from a representative trophic group were used as a surrogate. In addition, although salmon are a highly preferred and consumed fish from the LDW and tissue data were available for salmon, human health risks were not calculated for the consumption of adult salmon. Their exposure to chemicals in LDW sediment is not anticipated to significantly influence chemical concentrations in their tissues, primarily because of the very small portion of their lives spent in the LDW. Although risks from salmon consumption were not included in the baseline HHRA, risks to juvenile salmon were addressed in the ERA (Appendix A).

Sediment and tissues samples were analyzed for numerous chemicals, including metals, SVOCs, organochlorine pesticides, and PCBs. Organochlorine pesticides in seafood tissues collected in 2004 were qualified as estimated concentrations with uncertain presence. The tissue concentrations used to determine the intake of these chemicals were thus highly uncertain. There is uncertainty as to whether these chemicals were present; however, if present, their concentrations were likely biased high because of PCB interferences. Therefore, organochlorine pesticide data may not accurately represent exposures to these chemicals from seafood consumption within the LDW. Uncertainties in the data used in this assessment are summarized in Section 4.4 and in Appendix B, Section B.6.1.

6.1.2.2 COPC screening

Using the dataset defined for the baseline HHRA, a risk-based screen was performed following EPA guidance to identify COPCs to be evaluated. Briefly, the maximum concentrations in sediment were screened against risk-based concentrations (RBCs) for soil provided by EPA Region 9 (EPA 2004). The maximum seafood tissue concentrations were screened against EPA Region 3 RBCs (EPA 2005c) for fish tissue, which had been adjusted to account for tribal consumption rates (97.5 g/day) (EPA 2006a) and a lower target HQ, as required by EPA Region 10 (EPA 1996). Sixty-five chemicals were identified as COPCs in sediment or tissue for one or more exposure scenarios. Many of these 65 chemicals were identified for more detailed analysis using both seafood consumption and direct sediment exposure scenarios (Table 6-1). Of the 65 COPCs, 6 chemicals were never detected in tissue (but were detected in sediment), and 20 were never detected in either sediment or tissue.[79] These 26 chemicals were

[79] N-nitroso-di-n-propylamine was not included in the risk characterization but was discussed in the uncertainty analysis despite being detected in one tissue sample. This chemical was never detected in sediment samples, and had high RLs in tissue samples. The detection in tissue was qualified as JN (estimated concentration, tentative identification).

included as COPCs because analytical RLs were above the screening criteria for tissue, sediment, or both. The undetected COPCs were evaluated in the uncertainty analysis: 26 of these COPCs were evaluated for seafood consumptions scenarios and 5 were evaluated for direct contact scenarios.

Table 6-1. COPCs identified for direct sediment exposure and seafood consumption scenarios

Chemical	Netfishing Scenarios	Beach Play and Clamming Scenarios	Seafood Consumption Scenarios
Chemicals detected in both tissue and sediment			
Inorganic			
Aluminum[a]	X	X	
Antimony	X	X	X
Arsenic	X	X	X[b]
Barium[a]	X	X	
Cadmium	X	X	X
Chromium	X	X	X
Copper	X	X	X
Iron[a]	X	X	
Lead	X	X	X[c]
Manganese[a]	X	X	
Mercury		X	X
Molybdenum		X	
Nickel			X
Silver		X	
TBT			X
Thallium	X	X	
Vanadium	X	X	X
Zinc		X	X
Organic			
4-methylphenol			X
Aldrin			X
alpha-BHC			X
beta-BHC			X
gamma-BHC			X
BEHP			X
BBP			X
Carbazole			X
cPAHs	X	X	X
Total chlordane			X
Total DDTs		X	X
Dieldrin	X	X	X

Chemical	Netfishing Scenarios	Beach Play and Clamming Scenarios	Seafood Consumption Scenarios
Dioxin and furan TEQ[a]	X	X	X[d]
Endrin			X
Endrin aldehyde			X
Heptachlor			X
Heptachlor epoxide			X
Hexachlorobenzene			X
PCB TEQ	X	X	X
Total PCBs	X	X	X
Pentachlorophenol			X
Chemicals detected in sediment, but not detected in tissue			
1,3-Dichlorobenzene			X_{ND}
1,4-Dichlorobenzene			X_{ND}
Aniline[e]			X_{ND}
Hexachlorocyclopentadiene			X_{ND}[e]
n-Nitrosodiphenylamine			X_{ND}
Toxaphene	X	X	X_{ND}
Chemicals not detected in sediment or tissue			
1,2-Diphenylhydrazine			X_{ND}
2,4,6-Trichlorophenol			X_{ND}
2,4-Dichlorophenol			X_{ND}
2,4-Dinitrophenol			X_{ND}
2,4-Dinitrotoluene			X_{ND}
2,6-Dinitrotoluene			X_{ND}
2-Chlorophenol			X_{ND}
3,3'-Dichlorobenzidine			X_{ND}
3-Nitroaniline			X_{ND}
4,6-Dinitro-o-cresol		X_{ND}	X_{ND}
4-Chloroaniline			X_{ND}
4-Nitroaniline			X_{ND}
Benzidine	X_{ND}	X_{ND}	X_{ND}
Bis(2-chloroethyl)ether		X_{ND}	X_{ND}
Bis(2-chloroisopropyl)ether			X_{ND}
Hexachlorobutadiene			X_{ND}
Hexachloroethane			X_{ND}
Nitrobenzene			X_{ND}
n-Nitrosodimethylamine	X_{ND}	X_{ND}	X_{ND}
n-Nitroso-di-n-propylamine		X_{ND}	X_{ND}[f]

[a] This chemical was not analyzed in tissue.

[b] For the seafood consumption scenarios, the arsenic assessment was based on inorganic arsenic exposure and toxicity data.

[c] No RBC was available for lead. An alternative toxicity evaluation method recommended by EPA was used instead (see Appendix B, Section B.3.4.4, for more information on lead models).

[d] Dioxin and furan TEQ was identified as a COPC in sediment and tissue; although dioxins and furans were not analyzed in tissue, they were identified as potential bioaccumulative compounds by EPA (USEPA 2000).

[e] This chemical was detected in one sediment sample.

[f] One composite sample of whole-body English sole contained a detected concentration of n-nitroso-di-n-propylamine (270 µg/kg ww). However, this result was qualified as JN (estimated concentration, tentative identification). Given the uncertain quantification for this single result (all other results were undetected), the risks for this chemical were discussed in the uncertainty analysis.

BBP – butyl benzyl phthalate

BEHP – bis(2-ethylhexyl) phthalate

BHC – benzene hexachloride

COPC – chemical of potential concern

cPAH – carcinogenic polycyclic aromatic hydrocarbon

PAH – polycyclic aromatic hydrocarbon

PCB – polychlorinated biphenyl

TEQ – toxic equivalent

X indicates that the chemical was selected as a COPC and was detected in the exposure scenario media.

X_{ND} indicates that the chemical was selected as a COPC but was not detected in the exposure scenario media.

6.1.3 Selection of exposure parameters

Human exposures to COPCs were estimated for several seafood consumption and direct contact sediment exposure scenarios, as described below. Exposure estimates were based on the concentration data for each COPC and health-protective assumptions regarding the rates and frequency of seafood intake and the rates, frequency, and duration of direct sediment contact associated with incidental ingestion and dermal exposure.

6.1.3.1 Seafood consumption scenarios

Several different seafood consumption scenarios were evaluated, as summarized in Table 6-2. Seafood consumption rates that were assumed for the seafood consumption scenarios with multiple species were developed by EPA based on data collected from several surveys. Specifically, for the adult tribal RME seafood consumption scenario based on Tulalip data, a consumption rate of 97.5 g of resident seafood per day (which equals approximately three 227-g [8-oz] meals per week)[80] was assumed, based on EPA's analysis (*Framework for Selecting and Using Tribal Fish and Shellfish Consumption Rates for Risk-Based Decision Making at CERCLA and RCRA Cleanup Sites in Puget Sound and the Strait of Georgia* (EPA 2007d)) of a survey of the Tulalip Tribes' consumption of resident species from the Puget Sound region (Toy et al. 1996). This rate was intended to reflect the 95th percentile of Puget Sound non-anadromous seafood consumption and was designated as a reasonable maximum exposure (RME) scenario. For some

[80] Rate does not include consumption of anadromous fish. Total consumption rate including anadromous fish is 194 g/day (EPA 2007d).

pathways, both RME exposure estimates and central tendency (CT) exposure estimates were developed to describe the range of possible exposures and risks. EPA (1989) defines the RME as "the highest exposure that is reasonably expected to occur at a site." CT exposure estimates are intended to reflect average exposures. The scenarios developed in the baseline HHRA are consistent with RME guidelines. The CT scenarios were intended to describe more typical average exposures. Specific assumptions for the CT scenarios were developed in consultation with EPA (EPA 2007d; Kissinger 2005).

Table 6-2. Summary of seafood consumption scenarios

Scenario	Ingestion Rate (g/day)					Exposure Duration (years)
	Pelagic Fish	Benthic Fish	Crabs	Other Shellfish	Total[a]	
Adult tribal RME (Tulalip data)	8.1	7.5	43.4	38.5	97.5	70
Adult tribal CT (Tulalip data)	1.3	1.2	6.6	5.9	15.0	30
Child tribal RME (Tulalip data)	3.2	3.0	17.4	15.4	39.0	6
Child tribal CT (Tulalip data)	0.52	0.48	2.64	2.34	6.0[b]	6
Adult tribal (Suquamish data)	56	29.2[b]	55[b]	444[b]	584.2[b]	70
Adult API – RME	4.9	2.4	10.6	33.6	51.5	30
Adult API – CT	0.5	0.24	1.1	3.5	5.3	12
Adult one meal per month[c]	7.5	7.5	7.5	7.5	na	30

[a] Rates do not include consumption of anadromous fish.

[b] Totals of benthic and shellfish categories (and overall total) differ slightly from values provided by EPA (2007d) because of significant figure and rounding issues when total consumption was allocated to more specific consumption categories. Overall total listed here is slightly higher than the corresponding value provided by EPA (EPA 2007d).

[b] Adult one-meal-per-month consumption was evaluated for individual seafood categories independently to reflect different fishing and consumption practices.

API – Asian and Pacific Islander

CT – central tendency

na – not applicable

RME – reasonable maximum exposure

The consumption rate for the adult tribal RME scenario based on Tulalip data was assumed to represent the consumption of seafood caught from the LDW and was further divided into seafood categories as shown in Table 6-2. It is likely that current seafood consumption rates within the LDW are lower than those documented in the Tulalip tribal study because of existing seafood consumption advisories. There is considerable uncertainty about the applicability of some of the seafood consumption rates to the LDW, particularly for clams, given the size of the LDW and the availability of suitable habitat.

EPA's Superfund risk assessment guidance (EPA 1989) requires that exposure estimates be protective of future uses. Tribes with treaty rights to obtain seafood from the LDW may increase their seafood consumption rate in the future as chemical contamination in the LDW decreases. Habitat improvements may also increase the harvestable population of fish and shellfish to some degree. Consequently, the seafood consumption rates evaluated in the baseline HHRA were intended to be protective of both current and future uses.

Health risks were also quantified for several seafood consumption scenarios[81] in addition to the adult tribal RME scenario based on Tulalip data:

- Two Asian and Pacific Islander (API) scenarios (5.3 and 51.5 g/day of resident seafood for CT and RME, respectively (Kissinger 2005)
- An adult tribal CT scenario based on Tulalip data (15 g/day of resident seafood) (Hiltner 2007)
- Two child tribal scenarios based on Tulalip data (8.6 and 55.9 g/day of resident seafood for CT and RME, respectively) (EPA 2006b, 2007c; Hiltner 2007)
- A second adult tribal scenario based on Suquamish data (584.2 g/day of resident seafood) (EPA 2007d)
- Four adult one-meal-per-month consumption scenarios based on individual seafood categories (7.5 g/day)

Table 6-2 presents the ingestion rates and exposure durations associated with the various seafood consumption categories.

"Based on policy considerations, EPA is intending to use the Tulalip Tribes' seafood consumption rate as the principal rate to compute health protective tribal seafood consumption risks" for the LDW (EPA 2006b). The Suquamish seafood consumption scenario was included at the request of the Suquamish and Muckleshoot Indian Tribes to assist in characterizing the potential range of seafood consumption risks, although EPA (2005a) has questioned whether the high Suquamish rate could be sustained within the LDW.

The adult tribal seafood consumption scenarios based on Tulalip data, child tribal seafood consumption scenarios based on Tulalip data, adult tribal seafood consumption scenario based on Suquamish data, and adult API seafood consumption scenarios included a combination of all the seafood categories. For the adult one-meal-per-month scenario, risk was evaluated based on a consumption rate of one meal per month of pelagic fish (such as perch), benthic fish fillets (such as English sole and

[81] Rates do not include consumption of anadromous fish. Total consumption rates for tribal and API scenarios were higher than values presented here when anadromous fish were included (EPA 2007d; Kissinger 2005).

starry flounder), crab edible meat, or clams. The one-meal-per-month scenario was included for risk communication to enable individuals to estimate the risk associated with specific behaviors, rather than to describe the behavior of any particular group. Consistent with EPA risk assessment guidance, all assumptions regarding the amount of seafood ingested were selected to be health-protective to avoid underestimating risks. Consequently, individual risk estimates are more likely to be overestimates for most chemicals.

6.1.3.2 Direct sediment exposure scenarios

Risks associated with direct sediment exposure were quantified in the baseline HHRA for adults engaging in clamming or tribal netfishing and for children playing in intertidal sediment areas (hereafter referred to as "beach play"). Exposure frequency and duration assumptions for the evaluation of direct sediment exposure under the commercial netfishing scenario were based on site use information collected from the Muckleshoot Indian Tribe, which conducts commercial netfishing for adult salmon within the LDW. Exposure parameter values for the beach play and clam harvesting scenarios were based primarily on EPA guidance, a King County lakes recreational survey (applied to beach play scenario only) (Parametrix 2003), a recent EPA memo on tribal clamming (Kissinger 2007), and best professional judgment because site-specific data on exposure frequency and duration for these scenarios were not available. Table 6-3 provides a summary of the exposure parameters for these scenarios.

Table 6-3. Summary of sediment exposure scenarios

Scenario	Incidental Sediment IR (g/day)	Exposure Frequency (days/year)	Exposure Duration (years)	Skin Surface Area Exposed (cm^2)
Netfishing RME	0.05	119	44	3,600
Netfishing CT	0.05	63	29	3,600
Child beach play RME[a]	0.2	65	6	varies with age
Clamming, 7 days per year	0.1	7	30	6,040
Tribal clamming RME (120 days per year)	0.1	120	64	6,040
Tribal clamming, 183 days per year	0.1	183	70	6,040

[a] For the beach play scenarios, the LDW was divided into eight areas to assess risks associated with different parts of the LDW. Children were evaluated from birth through 6 years of age.

CT – central tendency

IR – ingestion rate

RME – reasonable maximum exposure

6.1.3.3 Exposure point concentrations

Exposure point concentrations (EPCs) for COPCs are intended to represent health-protective estimates of long-term exposure concentrations for a given scenario. In the baseline HHRA, the EPC was one of the following:

- If there were at least six detected concentrations, the EPC was the UCL on the mean concentration (calculated using EPA ProUCL software, Version 4).
- If there were one to five detected concentrations, the EPC was the higher of one-half the maximum RL or the maximum detected value.
- If the chemical was not detected in any samples, one half the maximum RL was selected as the EPC.

All concentrations qualified as estimates (i.e., J-qualified data) were assumed to indicate positive identification of the chemical and were used without modification in subsequent calculations. Some J-qualified data, most notably the 2004 pesticide results, were also N-qualified, indicating a tentative identification of the chemical. In the case of pesticides, the N-qualifier was necessary because of the high potential for analytical interference from PCBs (Windward 2005b). JN-qualified data were used in the risk assessment, but the uncertainty associated with these results was higher than the uncertainty associated with J-qualified results. Accordingly, in the risk characterization (Appendix B, Section B.5), the JN-qualified organochlorine pesticide data were presented separately from data for chemicals that were not JN-qualified. Analytical results for organochlorine pesticides used in the baseline HHRA from events prior to 2004 were likely to also have been complicated by interference from PCBs; however, the detected results were not all JN-qualified.

The EPCs for the direct sediment exposure scenarios (i.e., netfishing, beach play, and clam harvesting) were calculated for the sediment area over which exposure could potentially occur. The netfishing scenario assumed that tribal members who engage in commercial netfishing could be exposed to both intertidal and subtidal sediment adhering to their nets; consequently, the entire baseline surface sediment dataset was used for EPC calculation for this scenario.

For the beach play scenario, EPCs were based on data for intertidal sediment (i.e., sediment periodically exposed to air during low tides) from areas that are accessible to the public. The LDW was divided into eight areas for the beach play scenarios, corresponding to eight contiguous areas where access to intertidal sediments is relatively easy. Separate risk estimates were made for each of those eight areas. The areal extent of each beach, EPCs, and risks associated with the beach play scenario are presented in Map 6-1. The exposure frequency selected for the beach play scenario (i.e., 65 days per year) was based on King County's survey of parks immediately adjacent to lakes (Parametrix 2003) and represents the 95th percentile of exposure frequency for children up to 6 years old who play in sand near the water. This behavior is consistent with the behavior that is assumed for the beach play scenario in the LDW.

The clam harvesting scenarios assumed that people are exposed to COPCs in sediment as they dig for clams. The two tribal clamming scenarios (RME and 183 days per year) included all potential clam habitat areas (as identified during a 2004 survey) that could

be accessed either by boat or on foot from the bank. Another clamming scenario used a lower exposure frequency (7 days per year) and included only potential clam habitat areas that could be accessed from the bank (Map 6-2).

EPCs for the seafood consumption scenarios were calculated separately for various types of seafood, or consumption categories. Seven consumption categories were developed based on available seafood tissue types: pelagic fish, benthic fish—fillet, benthic fish—whole body, crab—edible meat, crab—whole body, clams, and mussels. In some cases, chemistry data for more than one species were combined within a single consumption category (e.g., data for Dungeness crab and slender crab edible meat were combined in the edible crab meat category). A COPC intake rate was then calculated for each consumption category using the COPC tissue dataset and the consumption rate for each category. The chemical intakes for each consumption category were then summed within each seafood consumption scenario (except the adult one-meal-per-month scenario) to yield an overall COPC intake for that scenario.

6.2 TOXICITY ASSESSMENT

EPA toxicity values (i.e., slope factors [SFs] for the evaluation of carcinogenic risks or reference doses [RfDs] for the evaluation of effects other than cancer) were identified for all COPCs. Toxicity values for each COPC have been established by EPA and other agencies and are based on either laboratory experiments using animals or epidemiological studies of human populations unintentionally exposed in the workplace or the environment. The SFs provide a health-protective means to evaluate risks because they represent upper bound estimates of carcinogenic potency. Similarly, non-cancer toxicity values (i.e., RfDs) are health-protective in that they are typically derived based on the most sensitive endpoint and population for which adequate data are available and include uncertainty factors or extrapolations to account for sensitive sub-populations or other limitations of the toxicity study data on which they were based.

6.3 RISK CHARACTERIZATION

This section presents the risk estimates calculated using the exposure parameters and toxicity values discussed in Sections 6.1 and 6.2, respectively

6.3.1 Risk estimate calculations

Carcinogenic risks and non-carcinogenic health effects were evaluated separately in the baseline HHRA because of fundamental differences in assumptions about the mechanism of these toxic effects. Carcinogenic risk estimates were calculated by multiplying the estimated chemical intake by the SF. Excess cancer risk estimates were compared to EPA's acceptable risk range of 10^{-6} to 10^{-4} established in the National Contingency Plan for Superfund sites (40 CFR 300). The lifetime risk of developing cancer in the US population is one in two (i.e., 5×10^{-1}) for men and one in three (i.e.,

3×10^{-1}) for women (American Cancer Society 2006). A 1×10^{-6} excess cancer risk represents an additional one-in-one-million probability that an individual may develop cancer over a 70-yr lifetime as a result of exposure to chemicals in LDW sediments and surface water (either through direct exposure or indirect exposure through the consumption of seafood).

Carcinogenic risks associated with PCBs may be assessed based on total PCBs or as PCB TEQ. The latter relates the toxicity of coplanar PCBs to the toxicity of 2,3,7,8-tetrachlorodibenzo-*p*-dioxin (TCDD). Risk estimates from the two approaches may be partially overlapping in terms of both exposure and toxicity. There is currently no consensus approach for considering these two risk approaches together. Therefore, two different total excess cancer risk estimates were provided: one estimate included all COPC risks except those for PCB TEQ (i.e., including risks from all non-PCB COPCs plus total PCBs); the other estimate included all COPCs except total PCBs (i.e., including risks from all non-PCB COPCs plus PCB TEQ).

Chemicals with non-carcinogenic health effects are generally not toxic below a certain threshold; a critical chemical dose must be exceeded before adverse health effects are observed. The potential for non-carcinogenic health effects is represented by the ratio of the estimated chemical intake to the critical chemical dose (called a reference dose) and is expressed as an HQ. Exposures resulting in an HQ less than or equal to 1 are unlikely to result in non-cancer adverse health effects. For chemicals sharing a common toxicity endpoint, the HQs were summed to develop a hazard index (HI) for that endpoint.

6.3.2 Risk characterization results

Tables 6-4 to 6-6 summarize the excess cancer risk estimates and non-cancer hazards for chemicals that exceeded 1×10^{-6} excess cancer risk or an HQ of 1 for at least one scenario. All other risk estimates are presented in Appendix B, Section B.5.

Estimated excess cancer risk estimates were highest for the seafood consumption scenarios (Table 6-4). The cumulative risk for all carcinogenic chemicals was 3×10^{-3} for the adult tribal RME seafood consumption scenario based on Tulalip data; the primary contributors were PCBs (2×10^{-3}) and inorganic arsenic (1×10^{-3}). The cumulative excess cancer risk for the seafood consumption scenarios did not include dioxins and furans, which were not analyzed in tissue samples. The majority of the excess cancer risk for PCBs was attributable to consumption of pelagic fish and clams. The excess cancer risk estimates from inorganic arsenic were largely attributable to consumption of clams. Inorganic arsenic concentrations in fish and crabs were much lower and were very similar to background tissue concentrations.

Cumulative excess cancer risk estimates for the other seafood consumption scenarios (except for the adult tribal scenario based on Suquamish data) were lower (approximately 1.3 to 33% of cancer risk estimates for the adult tribal RME scenario

based on Tulalip data) (Table 6-4). Specifically, the total excess cancer risks were 2×10^{-5} to 1×10^{-3} for the API consumption scenarios; 1×10^{-4} for the adult tribal CT scenario based on Tulalip data; 6×10^{-5} to 8×10^{-4} for the child tribal scenarios based on Tulalip data; 3×10^{-2} for the adult tribal scenario based on Suquamish data; and 2×10^{-4} (pelagic fish), 1×10^{-4} (benthic fish fillets), 4×10^{-5} (crab edible meat), and 2×10^{-4} (clams) for individuals who consumed those seafood categories at a rate of one meal per month. The risks for the adult tribal scenario based on Suquamish data were 10 times higher than risks for the adult tribal RME scenario based on Tulalip data, reflecting the much higher seafood consumption rate (almost 3 meals per day) used in the adult Suquamish scenario estimates. In the evaluation of non-cancer hazards, arsenic and PCBs had HQs greater than 1 for one or more of the RME seafood consumption scenarios, indicating some potential for adverse effects other than cancer. In addition, HQs were greater than 1 for the child tribal RME scenario based on Tulalip data for TBT and vanadium (Table 6-5). For the adult tribal scenario based on Suquamish data, two additional chemicals had HQs greater than 1: chromium and mercury. HQs for these chemicals were less than 1 for all other seafood consumption scenarios. For the purpose of brevity, chemical-specific HQ estimates are provided in Table 6-5 only for chemicals exceeding an HQ of 1 for any scenario.

Excess cancer risk estimates for the direct sediment exposure scenarios were much lower than those for the seafood consumption scenarios. With the exception of the tribal clamming RME scenario and the tribal clamming 183-day-per-year scenario, all excess cancer risk estimates for direct sediment exposure scenarios were less than or equal to 5×10^{-5} (i.e., only 1.7% of the risks from the adult tribal RME seafood consumption scenario based on Tulalip intake data) (Table 6-6). Total excess cancer risk from the tribal clamming RME scenario was 1×10^{-4} (excluding PCB TEQ), which is 3.3% of the risks from the adult tribal RME seafood consumption scenario based on Tulalip data and 2×10^{-4} (excluding PCB TEQ) for the tribal clamming 183-day-per-year scenario. Total excess cancer risk estimates were greater than 1×10^{-6} for the netfishing scenarios and between 5×10^{-6} and 5×10^{-5} for the beach play RME scenarios. Risk estimates associated with specific beach play areas are also presented in Map 6-1. The chemicals contributing the most to direct sediment excess cancer risk estimates were dioxins and furans, PCBs, arsenic, and cPAHs. No HQs were greater than 1 for any of the direct sediment exposure scenarios.

Table 6-4. Summary of estimated excess cancer risks for the seafood consumption scenarios

Chemical	Adult Tribal RME (Tulalip Data)	Adult Tribal CT (Tulalip Data)	Child Tribal RME (Tulalip Data)	Child Tribal CT (Tulalip Data)	Adult Tribal (Suquamish Data)	Adult API RME	Adult API CT	Adult One Meal per Month			
								Benthic Fish	Clam	Crab	Pelagic Fish
Arsenic (inorganic)[a]	1×10^{-3}	6×10^{-5}	3×10^{-4}	3×10^{-5}	2×10^{-2} [b]	7×10^{-4}	1×10^{-5}	4×10^{-7}	1×10^{-4}	3×10^{-6}	6×10^{-6}
BEHP	6×10^{-6}	2×10^{-7}	1×10^{-6}	7×10^{-8}	4×10^{-5}	2×10^{-6}	3×10^{-8}	8×10^{-7}	8×10^{-8}	8×10^{-8} [c]	1×10^{-6}
cPAHs[d]	7×10^{-5}	4×10^{-6}	7×10^{-5}	8×10^{-6}	8×10^{-4}	3×10^{-5}	8×10^{-7}	2×10^{-7}	7×10^{-6}	2×10^{-7}	3×10^{-7}
Dioxin and furan TEQ[e]	na	na	na	na	na	na	na	na	na	na	na
PCB TEQ	1×10^{-3}	6×10^{-5}	2×10^{-4}	2×10^{-5}	7×10^{-3}	4×10^{-4}	7×10^{-6}	8×10^{-5}	2×10^{-5}	2×10^{-5}	2×10^{-4}
Total PCBs	2×10^{-3}	6×10^{-5}	3×10^{-4}	3×10^{-5}	1×10^{-2} [b]	5×10^{-4}	8×10^{-6}	1×10^{-4}	5×10^{-5}	2×10^{-5}	2×10^{-4}
Pentachlorophenol[a]	9×10^{-5} [f]	2×10^{-6} [f]	2×10^{-5} [f]	7×10^{-7} [f]	5×10^{-4} [f]	2×10^{-5}	3×10^{-7}	2×10^{-5} [c]	1×10^{-6} [c]	2×10^{-6} [c]	1×10^{-5}
Subtotal (excluding PCB TEQ)	**3×10^{-3}**	**1×10^{-4}**	**7×10^{-4}**	**7×10^{-5}**	**3×10^{-2}**	**1×10^{-3}**	**2×10^{-5}**	**1×10^{-4}**	**2×10^{-4}**	**3×10^{-5}**	**2×10^{-4}**
Subtotal (excluding total PCBs)	**2×10^{-3}**	**1×10^{-4}**	**6×10^{-4}**	**6×10^{-5}**	**3×10^{-2}**	**1×10^{-3}**	**2×10^{-5}**	**1×10^{-4}**	**1×10^{-4}**	**3×10^{-5}**	**2×10^{-4}**
Tentatively Identified Chemicals (JN-qualified)											
Aldrin	5×10^{-5} [f]	1×10^{-6} [f]	9×10^{-6} [f]	6×10^{-7} [f]	2×10^{-4}	1×10^{-5}	2×10^{-7}	3×10^{-6} [c]	8×10^{-7} [c]	3×10^{-6} [c]	3×10^{-6}
alpha-BHC	2×10^{-5} [f]	5×10^{-7} [f]	3×10^{-6} [f]	2×10^{-7} [f]	6×10^{-5}	3×10^{-6}	6×10^{-8}	1×10^{-6}	1×10^{-7}	1×10^{-6} [c]	1×10^{-6}
beta-BHC	6×10^{-6} [f]	3×10^{-7} [f]	1×10^{-6} [f]	1×10^{-7} [f]	3×10^{-5}	1×10^{-6}	3×10^{-8}	3×10^{-7}	1×10^{-7}	3×10^{-7} [c]	6×10^{-7}
Carbazole	5×10^{-5}	1×10^{-6}	8×10^{-6}	4×10^{-7}	2×10^{-4}	1×10^{-5}	8×10^{-8}	1×10^{-6} [c]	9×10^{-8} [c]	1×10^{-6} [c]	1×10^{-5}
Total chlordane	6×10^{-6}	2×10^{-7}	1×10^{-6}	9×10^{-8}	3×10^{-5}	2×10^{-6}	3×10^{-8}	3×10^{-7}	7×10^{-8}	7×10^{-8}	1×10^{-6}
Total DDTs	2×10^{-5}	1×10^{-6}	4×10^{-6}	4×10^{-7}	1×10^{-4}	6×10^{-6}	1×10^{-7}	1×10^{-6}	2×10^{-7}	4×10^{-7}	4×10^{-6}
Dieldrin	1×10^{-4}	3×10^{-6}	2×10^{-5}	1×10^{-6}	1×10^{-3}	5×10^{-5}	4×10^{-7}	3×10^{-6} [c]	9×10^{-6}	3×10^{-6}	3×10^{-6} [c]
gamma-BHC	6×10^{-6}	1×10^{-7}	1×10^{-6}	5×10^{-8}	3×10^{-5}	1×10^{-6}	1×10^{-8}	2×10^{-7} [c]	1×10^{-7}	2×10^{-7}	1×10^{-7}
Heptachlor	1×10^{-5} [f]	4×10^{-7} [f]	3×10^{-6} [f]	2×10^{-7} [f]	6×10^{-5}	3×10^{-6}	4×10^{-8}	7×10^{-7} [c]	1×10^{-7} [c]	7×10^{-7} [c]	2×10^{-6}
Heptachlor epoxide	3×10^{-5}	1×10^{-6}	6×10^{-6}	5×10^{-7}	2×10^{-4}	9×10^{-6}	1×10^{-7}	1×10^{-6} [c]	6×10^{-7}	9×10^{-7}	4×10^{-6}
Hexachlorobenzene	1×10^{-5}	2×10^{-7}	2×10^{-6}	1×10^{-7}	4×10^{-5}	2×10^{-6}	3×10^{-8}	6×10^{-7}	6×10^{-8}	6×10^{-7}	9×10^{-7}
Subtotal	**3×10^{-4}**	**9×10^{-6}**	**6×10^{-5}**	**4×10^{-6}**	**2×10^{-3}**	**1×10^{-4}**	**1×10^{-6}**	**1×10^{-5}**	**1×10^{-5}**	**1×10^{-5}**	**3×10^{-5}**
Total excess cancer risk (excluding PCB TEQ)	**3×10^{-3}**	**1×10^{-4}**	**8×10^{-4}**	**7×10^{-5}**	**3×10^{-2}**	**1×10^{-3}**	**2×10^{-5}**	**1×10^{-4}**	**2×10^{-4}**	**4×10^{-5}**	**2×10^{-4}**
Total excess cancer risk (excluding total PCBs)	**2×10^{-3}**	**1×10^{-4}**	**7×10^{-4}**	**6×10^{-5}**	**3×10^{-2}**	**1×10^{-3}**	**2×10^{-5}**	**1×10^{-4}**	**1×10^{-4}**	**4×10^{-5}**	**2×10^{-4}**

[a] No mussel data were available for this chemical. When calculating the chronic daily intake and risk values, the portion of seafood consumption that had been assigned to mussels was divided proportionally among the remaining consumption categories.

[b] Because the excess cancer risk is greater than or equal to 0.01, risk was calculated using the exponential equation in EPA (1989).

[c] No detected values in this seafood category. Chronic daily intake and risk estimate are based on one-half the maximum reporting limit.

[d] cPAH concentrations are based on benzo(a)pyrene equivalents. Data used in the risk characterization are from only 2004 because of high reporting limits in historical data. All cPAH data are analyzed in the uncertainty analysis (Appendix B, Section B.6). Because of the potential for increased susceptibility of children to carcinogens with mutagenic activity, as described in EPA guidance (EPA 2005e), the risk estimate for children for cPAHs is based on dose adjustments across the 0-6 year age range of children. See Appendix B, Section B.5.1, for more information.

[e] Tissue data for dioxins and furans were not collected. Thus, the calculated risk total, which does not include risks from dioxins and furans, is underestimated to an unknown degree.

[f] Greater than 50% of the risk associated with this chemical is derived from seafood categories with no detected values.

API – Asian and Pacific Islander

BEHP – bis(2-ethylhexyl) phthalate

BHC – benzene hexachloride

cPAH – carcinogenic polycyclic aromatic hydrocarbon

CT – central tendency

PCB – polychlorinated biphenyl

RME – reasonable maximum exposure

TEQ – toxic equivalent

Table 6-5. Summary of estimated non-cancer hazards for the seafood consumption scenarios

Chemical	Adult Tulalip RME (Tulalip Data)	Adult Tribal CT (Tulalip Data)	Child Tribal RME (Tulalip Data)	Child Tribal CT (Tulalip Data)	Adult Tribal (Suquamish Data)	Adult API RME	Adult API CT	Adult One Meal per Month			
								Benthic Fish	Clam	Crab	Pelagic Fish
Arsenic[a, b]	3	0.3	7	0.7	38	3	0.2	0.002	0.7	0.01	0.03
Chromium	0.2	0.02	0.3	0.04	2	0.1	0.01	0.002	0.03	0.006	0.007
Mercury	0.5	0.07	1	0.1	2	0.3	0.02	0.06	0.02	0.07	0.04
Total PCBs	40	4	86	8	274	29	2	6	3	1	10
TBT (as ion)	1	0.2	3	0.3	15	1	0.1	0.002	0.3	0.02	0.06
Vanadium	0.8	0.1	2	0.2	9	0.8	0.07	0.01	0.2	0.01	0.06
Subtotal[c]	**46**	**5**	**101**	**10**	**344**	**35**	**2**	**6**	**4**	**1**	**10**
Hazard Indices by Effect											
HI for cardiovascular endpoint[d]	**4**	**0.4**	**9**	**0.9**	**47**	**4**	**0.3**	**0.01**	**0.9**	**0.02**	**0.09**
HI for developmental endpoint[e]	**41**	**4**	**87**	**8**	**276**	**29**	**2**	**6**	**3**	**1**	**10**
HI for hematologic endpoint[f]	**0.2**	**0.03**	**0.5**	**0.05**	**2**	**0.2**	**0.01**	**0.006**	**0.03**	**0.01**	**0.009**
HI for immunological endpoint[g]	**41**	**4**	**89**	**8**	**289**	**30**	**2**	**6**	**3**	**1**	**10**
HI for kidney endpoint[h]	**0.4**	**0.05**	**1.0**	**0.1**	**2**	**0.3**	**0.02**	**0.03**	**0.03**	**0.04**	**0.04**
HI for liver endpoint[i]	**1**	**0.1**	**3**	**0.3**	**7**	**0.8**	**0.05**	**0.1**	**0.1**	**0.09**	**0.3**
HI for neurological endpoint[j]	**41**	**4**	**87**	**8**	**276**	**29**	**2**	**6**	**3**	**1**	**10**
HI for dermal endpoint[k]	**3**	**0.3**	**7**	**0.7**	**38**	**3**	**0.2**	**0.01**	**0.7**	**0.02**	**0.06**
Total risk across all exposure routes/ pathways	**47**	**5**	**103**	**10**	**348**	**35**	**2**	**6**	**4**	**1**	**10**

[a] No mussel data were available for this chemical. When calculating the risk values, the portion of seafood consumption that had been assigned to mussels was divided proportionally among the remaining consumption categories.

[b] Arsenic risk estimates are based on inorganic arsenic.

[c] Total and subtotal risk values include the risks associated with all COPCs. However, only those COPCs with a hazard quotient greater than or equal to 1 for at least one scenario are listed in this table.

[d] Cardiovascular endpoint is for arsenic and vanadium.

[e] Developmental endpoint is for PCBs and mercury.

[f] Hematologic endpoint is for antimony and zinc.

[g] Immunological endpoint is for PCBs and TBT.

[h] Kidney endpoint is for 4-methylphenol, cadmium, copper, gamma-BHC, and pentachlorophenol.

[i] Liver endpoint is for 4-methylphenol, aldrin, alpha-BHC, beta-BHC, bis(2-ethylhexyl) phthalate, butyl benzyl phthalate, chlordane, copper, total DDTs, dieldrin, endrin, endrin aldehyde, gamma-BHC, heptachlor, heptachlor epoxide, hexachlorobenzene, and pentachlorophenol.

[j] Neurological endpoint is for 4-methylphenol, mercury, and total PCBs.

[k] Dermal endpoint is for 4-methylphenol and arsenic.

API – Asian and Pacific Islander

BHC – benzene hexachloride

COPC – chemical of potential concern

cPAH – carcinogenic polycyclic aromatic hydrocarbon

CT – central tendency

HI – hazard index

PCB – polychlorinated biphenyl

RME – reasonable maximum exposure

TBT – tributyltin

Table 6-6. Summary of estimated excess cancer risks for direct sediment exposure scenarios

Chemical	Netfishing		Beach Play RME								Clamming		
	RME	CT	Area 1	Area 2	Area 3	Area 4	Area 5	Area 6	Area 7	Area 8	Tribal – 183 Days per Year	Tribal RME	7 Days per Year
Arsenic	6×10^{-6}	1×10^{-6}	5×10^{-6}	7×10^{-6}	4×10^{-6}	4×10^{-6}	3×10^{-6}	3×10^{-6}	4×10^{-6}	3×10^{-6}	3×10^{-5}	2×10^{-5}	3×10^{-7}
cPAHs[a]	1×10^{-6}	2×10^{-7}	1×10^{-5}	4×10^{-5}	3×10^{-5}	8×10^{-6}	5×10^{-6}	5×10^{-6}	1×10^{-6}	4×10^{-6}	8×10^{-6}	5×10^{-6}	1×10^{-7}
Dioxin and furan TEQ	2×10^{-5}	4×10^{-6}	na	na	na	1×10^{-5}	8×10^{-8}	na	6×10^{-8}	na	2×10^{-4}	1×10^{-4}	8×10^{-7}
PCB TEQ	4×10^{-6}	6×10^{-7}	4×10^{-9}	3×10^{-7}	na	9×10^{-6}	1×10^{-7}	3×10^{-7}	3×10^{-8}	8×10^{-8}	5×10^{-5}	3×10^{-5}	2×10^{-7}
Total PCBs	2×10^{-6}[b]	3×10^{-7}[b]	7×10^{-8}	1×10^{-7}	1×10^{-7}	6×10^{-6}	1×10^{-7}	5×10^{-7}	1×10^{-7}	1×10^{-7}	1×10^{-5}[b]	8×10^{-6}[b]	9×10^{-8}
Subtotal (excluding PCB TEQ)	**3×10^{-5}**	**6×10^{-6}**	**2×10^{-5}**	**5×10^{-5}**	**3×10^{-5}**	**3×10^{-5}**	**8×10^{-6}**	**9×10^{-6}**	**5×10^{-6}**	**7×10^{-6}**	**2×10^{-4}**	**1×10^{-4}**	**1×10^{-6}**
Subtotal (excluding total PCBs)	**3×10^{-5}**	**6×10^{-6}**	**2×10^{-5}**	**5×10^{-5}**	**3×10^{-5}**	**3×10^{-5}**	**8×10^{-6}**	**8×10^{-6}**	**5×10^{-6}**	**7×10^{-6}**	**3×10^{-4}**	**2×10^{-4}**	**1×10^{-6}**
Tentatively Identified Chemicals (JN-qualified)													
Toxaphene	2×10^{-6}	3×10^{-7}	1×10^{-8}	7×10^{-9}	2×10^{-8}	2×10^{-7}	1×10^{-7}	1×10^{-8}	1×10^{-8}	1×10^{-8}	9×10^{-6}	6×10^{-6}	2×10^{-8}
Total risk (excluding PCB TEQ) across both exposure routes)[c]	**3×10^{-5}**	**5×10^{-6}**	**2×10^{-5}**	**5×10^{-5}**	**3×10^{-5}**	**3×10^{-5}**	**8×10^{-6}**	**9×10^{-6}**	**5×10^{-6}**	**7×10^{-6}**	**3×10^{-4}**	**1×10^{-4}**	**1×10^{-6}**
Total risk (excluding total PCBs) across both exposure routes)[c]	**3×10^{-5}**	**6×10^{-6}**	**2×10^{-5}**	**5×10^{-5}**	**3×10^{-5}**	**3×10^{-5}**	**8×10^{-6}**	**8×10^{-6}**	**5×10^{-6}**	**7×10^{-6}**	**3×10^{-4}**	**2×10^{-4}**	**1×10^{-6}**

[a] cPAH concentrations are based on benzo(a)pyrene equivalents. Because of the potential for increased susceptibility of children to carcinogens with mutagenic activity, as described in EPA guidance (2005e), the risk estimate for beach play RME for cPAHs is based on dose adjustments across the 0-to-6-year age range of children. See Appendix B, Section B.5.1, for more information.

[b] The EPC used for this risk estimate was based on an arithmetic UCL, which is expected to overestimate exposure because of spatially biased sampling. The arithmetic mean was greater than the spatially-weighted mean (developed using Thiessen polygons) by a factor of approximately 5.

[c] Total risk values include the risks associated with all COPCs. However, only those COPCs with an excess cancer risk greater than or equal to 1×10^{-6} for at least one scenario are listed in this table.

COPC – chemical of potential concern

cPAH – carcinogenic polycyclic aromatic hydrocarbon

CT – central tendency

EPC – exposure point concentration

na – not available

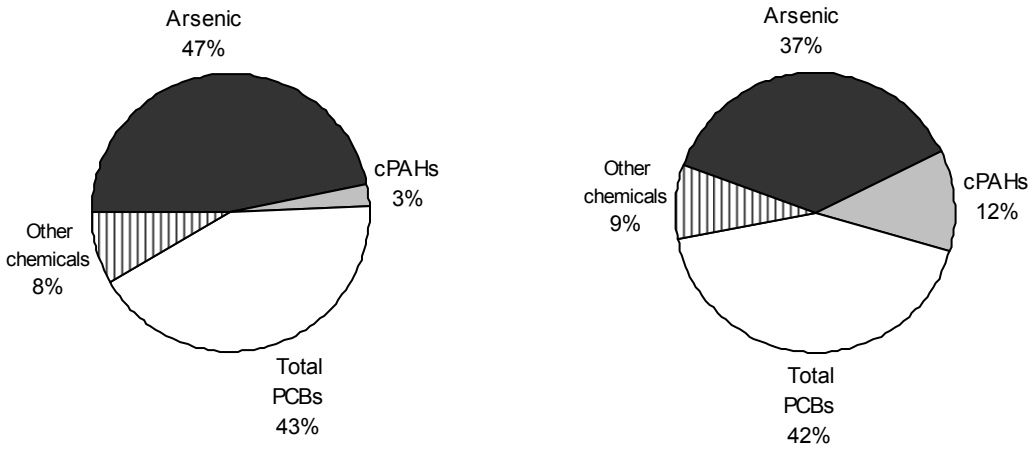
PCB – polychlorinated biphenyl

RME – reasonable maximum exposure

SWAC – spatially weighted average concentration

TEQ – toxic equivalent

A small number of chemicals were responsible for the majority of the estimated health risks, and the highest risk estimates were associated with seafood consumption. The distributions of cancer risks by chemical are shown in Figure 6-2 for both the adult and child seafood consumption scenarios. The average percent contribution of different chemicals is shown for the adult scenarios because there was little variability in relative chemical contribution among the different scenarios (i.e., adult tribal RME based on Tulalip data, adult API RME, and adult tribal based on Suquamish data). Approximately 85 to 90% of health risks were associated with only two or three chemicals (i.e., arsenic, PCBs, or cPAHs, and almost all the risks in the "other chemicals" category were attributed to tentatively identified pesticides. Note that Figure 6-2 does not show risks for dioxins and furans, which were not analyzed in tissue. The majority of the non-cancer hazards associated with seafood consumption were contributed by total PCBs (> 80% of the total developmental, neurological, and immunological hazard indices). The total PCB HQ for all seafood consumption scenarios, except the adult one-meal-per-month crab scenario, exceeded 1.



a) Average total cancer risk (excluding PCB TEQ) for adult seafood consumption scenarios

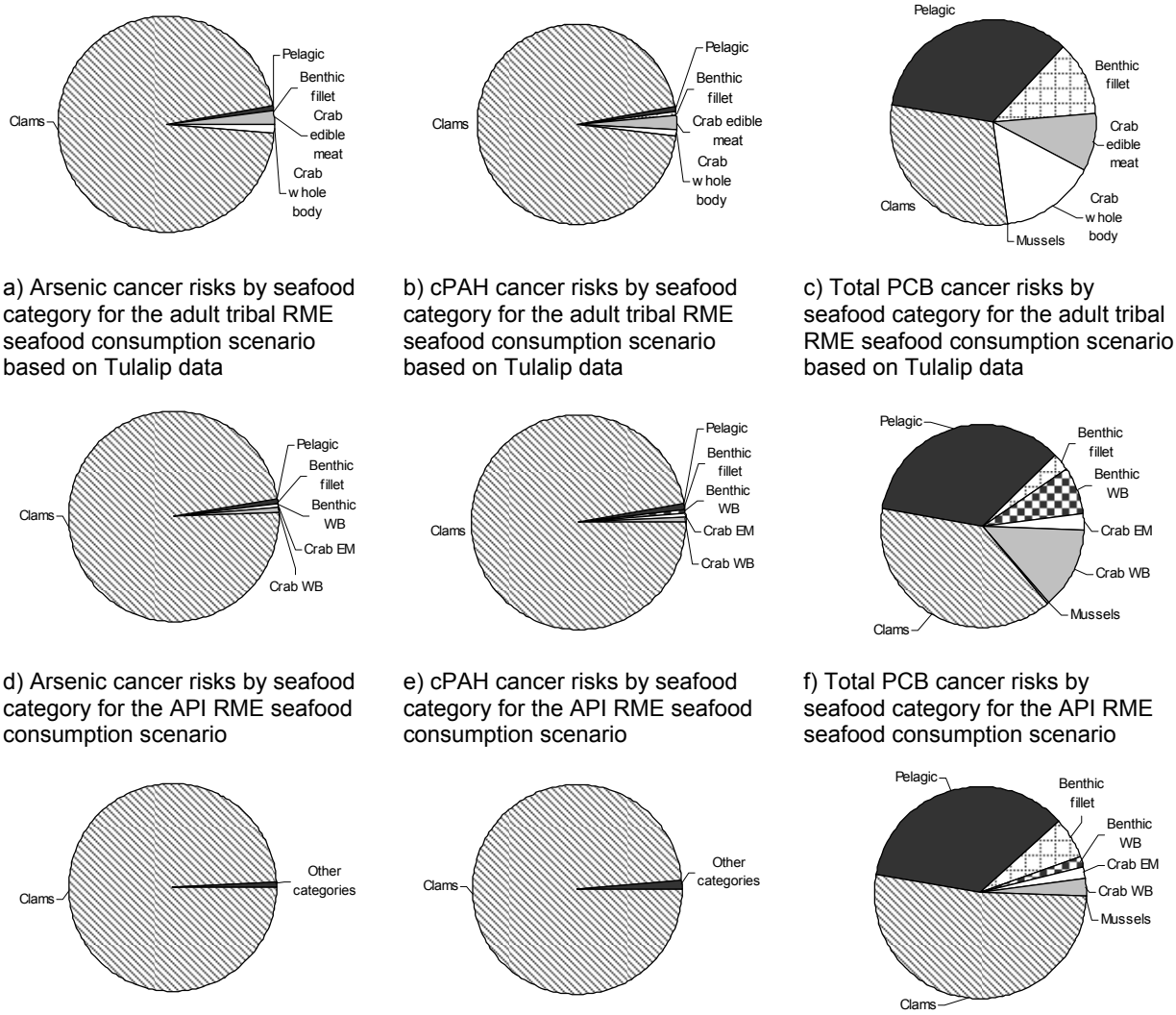
b) Average total cancer risk (excluding PCB TEQ) for child seafood consumption scenarios

Figure 6-2. Seafood consumption scenario risks by chemical

An incremental risk evaluation was performed for arsenic in tissue because arsenic is a an element found in the sediments of central and northern Puget Sound as result of natural sources (see Appendix B, Section B.5.5.1). Tissue samples for several species were collected from background locations and analyzed for arsenic. To estimate the incremental arsenic risk associated with the consumption of LDW seafood, arsenic risk calculated for consumption of seafood from background areas was subtracted from arsenic risk calculated for consumption of seafood from the LDW. Consideration of arsenic background risk did not appreciably affect LDW risk estimates associated with

clam consumption. However, the incremental arsenic risk associated with the LDW was negligible for all other seafood categories assessed.

Elevated risk estimates associated with cPAHs and arsenic in seafood were largely attributed to consumption of clams (Figure 6-3). In contrast, risks from total PCBs through the consumption of seafood were more evenly divided among the seafood consumption categories (Figure 6-3, Panels c, f, and i), with pelagic fish and clams as dominant contributors. There were relatively small differences between the three scenarios (i.e., adult tribal RME based on Tulalip data, adult API RME, and adult tribal based on Suquamish data) for arsenic and cPAHs. For total PCB risks, the proportional contributions of all fish compared to all shellfish were similar across the three scenarios. However, within the shellfish group, the types of shellfish contributing the most to total PCB excess cancer risk differed across the three groups, reflecting differences in shellfish consumption patterns across the scenarios.



a) Arsenic cancer risks by seafood category for the adult tribal RME seafood consumption scenario based on Tulalip data

b) cPAH cancer risks by seafood category for the adult tribal RME seafood consumption scenario based on Tulalip data

c) Total PCB cancer risks by seafood category for the adult tribal RME seafood consumption scenario based on Tulalip data

d) Arsenic cancer risks by seafood category for the API RME seafood consumption scenario

e) cPAH cancer risks by seafood category for the API RME seafood consumption scenario

f) Total PCB cancer risks by seafood category for the API RME seafood consumption scenario

g) Arsenic cancer risks by seafood category for the adult tribal seafood consumption scenario based on Suquamish data

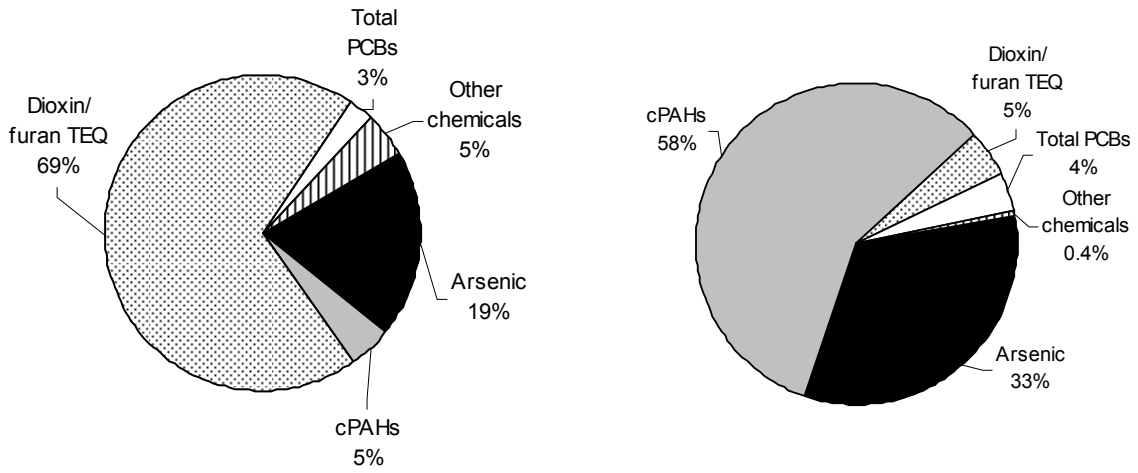
h) cPAH cancer risks by seafood category for the adult tribal seafood consumption scenario based on Suquamish data

i) Total PCB cancer risks by seafood category for the adult tribal seafood consumption scenario based on Suquamish data

Figure 6-3. Excess cancer risks by seafood category for the adult seafood consumption scenarios

The risk contribution by chemical was different for direct sediment exposure scenarios than for seafood consumption exposure scenarios. The direct sediment exposure risk estimates were also quite different between the adult (i.e., netfishing and clamming) and child (i.e., beach play) scenarios. Dioxin and furan risks contributed the majority (average of 69%) of the risks for the adult sediment scenarios, followed by arsenic (average of 19%) (Figure 6-4, Panel a). For the beach play scenarios, cPAHs were responsible for the majority of the estimated risks (average of 58%) and dioxins and furans were much less important to the overall risk estimate (average of 5%), primarily

because the highest concentrations of dioxins and furans in the LDW were not in beach play areas, and there were fewer data available, so risks associated with these chemicals could not be evaluated for some areas (Figure 6-4, Panel b). In areas where dioxin and furan data were lacking, estimates of their contribution to risk could not be made.



a) Average total cancer risk estimate for the two netfishing and three clamming scenarios

b) Average total cancer risk estimate for the eight beach play RME scenarios

Figure 6-4. Excess cancer risks by chemical for direct sediment exposure scenarios

The sediment risk estimates provided do not consider what portion of the total dioxin and furan risk in the LDW could be attributable to exposures expected throughout the greater Seattle area. Dioxin and furan concentrations in sediment from the LDW were compared to dioxin and furan concentrations in sediments from other urban areas (see Appendix B, Section B.5.5.2.1). This assessment indicated that most of the locations that have been sampled in the LDW are within the range of dioxin and furan concentrations in sediments from urban background areas. However, concentrations of dioxins and furans in surface sediments at a few locations within the LDW are more than 100 times higher than in sediments at urban background locations.

In addition to estimating the risks for each exposure scenario separately, risks for multiple scenarios were summed to represent possible exposure of the same individuals to LDW chemicals from different activities, as shown in Table 6-7. The sum of excess cancer risk estimates for each of these four sets of scenarios is the same as the estimates for their seafood consumption components alone after rounding to one significant figure, as recommended by EPA (1989). This analysis demonstrates that the contributions of netfishing, clamming, beach play, and swimming are relatively small in comparison to seafood consumption risk estimates and highlights the significance of the seafood-consumption exposure pathways for all users of the

LDW. Summing child beach play and swimming increased the risk estimate only slightly over that of beach play alone. Overall, swimming was the least important exposure pathway.

Table 6-7. Excess cancer risk estimates across related scenarios

ACTIVITY	EXCESS CANCER RISK[a]
Adult Tribal Scenarios	
Adult tribal netfishing RME[b]	3×10^{-5}
Swimming[c]	$< 1 \times 10^{-6}$
Adult tribal RME seafood consumption (Tulalip data)	3×10^{-3}
Total	**3×10^{-3}**
Child Scenarios[d]	
Beach play RME – Area 2[e]	5×10^{-5}
Swimming[c]	$< 1 \times 10^{-6}$
Subtotal for beach play and swimming	**5×10^{-5}**
Child tribal RME seafood consumption (Tulalip data)	8×10^{-4}
Total	**9×10^{-4}**
Adult Low-End Clamming Scenarios	
Clamming – 7 days per year	1×10^{-6}
Swimming[c]	$< 1 \times 10^{-6}$
Clam consumption – one meal per month	2×10^{-4}
Total	**2×10^{-4}**
Adult Tribal RME Clamming Scenarios	
Tribal clamming RME (120 days per year)	1×10^{-4}
Swimming[c]	$< 1 \times 10^{-6}$
Adult tribal RME seafood consumption based on Tulalip data	3×10^{-3}
Total	**3×10^{-3}**

[a] All non-swimming risk estimates are presented in Appendix B; total excess cancer risk estimates excluding PCB TEQ were used because these were equal to or higher than total excess cancer risk estimates excluding total PCBs.

[b] Although EPA guidance generally discourages summing risk estimates from multiple RME scenarios, risks for the netfishing RME scenario, rather than the netfishing CT scenario, were added to the seafood consumption RME scenario to account for the fact that RME seafood consumption and RME netfishing may be practiced by tribal members simultaneously.

[c] Adult and child swimming risk estimates as reported by King County for Elliott Bay and the Duwamish River for medium exposure assumptions (12 events per year for adults or children aged 1 to 6) (King County 1999c). Exposure pathways consisted of dermal contact and incidental sediment ingestion of water during swimming. Risks were estimated based on total PCB concentrations of 14.4 ng/L in the LDW originally modeled by King County (King County 1999c). PCB congener data from samples collected from the LDW by King County in 2005 indicate that this modeled estimate is likely an overestimate of actual total PCB concentrations, which were no greater than 3.14 ng/L during low-flow sampling conducted in August 2005 (Mickelson and Williston 2006). These results indicate that the risk estimates for the swimming scenario presented by King County in the water quality assessment (King County 1999c) are also likely overestimated.

[d] Child scenarios include the child tribal RME seafood consumption estimate based on 40% of the total adult tribal RME consumption based on Tulalip data, which is considered protective of non-tribal children.

[e] Beach play Area 2 is included because it had the highest risk estimate of the beach play scenarios.

CT – central tendency

PCB – polychlorinated biphenyl

RME – reasonable maximum exposure

6.4 UNCERTAINTY ANALYSIS

There are many uncertainties associated with the risk estimates for each exposure scenario in the baseline HHRA. To be health-protective of all members of the general public, the risk estimates presented in the baseline HHRA were intended to avoid underestimation of risks for the RME individual, and thus are likely to overestimate risks for most individuals for the chemicals that were evaluated. Although risk estimates were highest for the seafood consumption scenarios, the uncertainties associated with those risk estimates were also very high. There is considerable uncertainty about the applicability of some of the seafood consumption rates to this HHRA, particularly for clams, given the size of the LDW and the availability of suitable habitat. These risk estimates were intended to provide information to risk managers involved in remedial planning at the site but may not reflect actual risks to people currently consuming LDW seafood.

Another important uncertainty is in the methods used to characterize the excess cancer risks associated with exposures to PCBs. Two methods were used in the baseline HHRA, one method based on total PCB data and the cancer SF for total PCBs and a second method based on dioxin-like PCB congener data and the cancer SF for TCDD. Because both of these methods account for the potential carcinogenic effects of PCBs, the excess cancer risk estimates from these two methods were not summed in estimating cumulative risks in order to avoid double-counting PCB excess cancer risks. Hence, the risk estimates for the two methods were presented separately in the baseline HHRA. Although this approach avoids the double-counting of PCB-related excess cancer risks, it is possible that each method for quantifying PCB excess cancer risks on its own underestimates the overall PCB cancer health risk. The issues associated with assessing risks posed by environmental PCB mixtures, various approaches for addressing double-counting, and quantitative risk estimates derived using these approaches are discussed in detail in the baseline HHRA uncertainty analysis section (Appendix B, Section B.6.3.1).

The excess cancer risk estimates related to inorganic arsenic in the seafood consumption scenarios were almost entirely driven by elevated concentrations of arsenic in clams. These risk estimates are uncertain because they were based on eight composite clam tissue samples. These eight samples were collected from locations in the LDW where sediment arsenic concentrations were generally similar to or lower

than both the average arsenic concentration in LDW sediments and natural background arsenic concentrations.

Dioxins and furans were not analyzed in seafood samples. This lack of data contributes to an underestimation of risk because these chemicals were not included in the risk assessment for the seafood consumption scenarios.

The final risk estimates also reflect uncertainties associated with using data and assumptions from multiple sources; the combined effect of those uncertainties on risk estimates cannot be quantified. However, the risk assessment tended to overestimate risks more than underestimate them, consistent with the health-protective nature of risk assessment. All or any of the uncertainties are relevant to the risk estimates. In spite of these uncertainties, the baseline risk characterization for the LDW site is considered to be health-protective and sufficient to support risk management decisions.

6.5 SUMMARY AND SELECTION OF HUMAN HEALTH RISK DRIVERS

The baseline HHRA identified risk driver chemicals (referred to as indicator hazardous substances in the MTCA program [WAC 173-340-703]). The risk drivers from both the HHRA and the ERA will be the focus of remedial analyses in the FS. The starting point for designating risk drivers was to identify COCs. Chemicals with excess cancer risk estimates greater than 1×10^{-6} or an HQ greater than 1 for any RME exposure scenario were designated COCs. Nineteen chemical were identified as seafood consumption COCs.[82] Eighteen chemicals were identified as seafood consumption COCs based on an exceedance of one or both of these risk thresholds for at least one seafood RME consumption scenario. Dioxins and furans were also included as a COC for seafood consumption, even though no quantitative risk estimates were made. It was assumed that dioxin and furan excess cancer risk estimates for seafood consumption scenarios would have been greater than 1×10^{-6}. Five chemicals were identified as direct sediment exposure COCs[83] based on excess cancer risk estimates greater than 1×10^{-6} for at least one RME scenario. Note that none of the direct sediment contact scenarios had HQs greater than 1; thus, no chemicals were designated as COCs based on non-cancer risk for the direct contact scenarios.

Organochlorine pesticides were identified as COCs for several seafood consumption and direct sediment scenarios. However, as discussed previously, there may have been analytical interference from PCBs that affected their identification and

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[82] The COCs based on seafood consumption scenarios were PCBs, arsenic, cPAHs, dioxins and furans, aldrin, BEHP, alpha-BHC, beta-BHC, carbazole, total chlordane, total DDTs, dieldrin, gamma-BHC, heptachlor, heptachlor epoxide, hexachlorobenzene, pentachlorophenol, TBT, and vanadium.

[83] The COCs based on direct sediment exposure were PCBs, arsenic, cPAHs, dioxins and furans, and toxaphene.

quantification. Because there is no known source of organochlorine pesticides in the LDW, health risks associated with organochlorine pesticides in the LDW would not be expected to differ from those for other, similar urban or industrial waterway environments.

Chemicals selected as risk drivers were a subset of the COCs. The relative percentage of the total human health risk posed by these COCs was considered in designating a risk driver. Other factors that were also considered included the criteria identified in WAC 173-340-703 for designating an indicator hazardous substance: (a) toxicological characteristics that influence its ability to adversely affect human health or the environment relative to the concentration of the hazardous substance at the site, (b) tendency to persist in the environment, (c) tendency to move into and through environmental media, (d) natural background concentrations, (e) thoroughness of testing, (f) detection frequency, and (g) chemicals that readily break down into less toxic byproducts (see Appendix B, Section B.7, for more details). Based on this analysis, four risk drivers were identified for both seafood consumption and direct sediment exposure: PCBs, arsenic, cPAHs, and dioxins and furans. The direct contact risk estimate for dioxins and furans was driven by high concentrations in surface sediments at a small number of sampling locations within the LDW. Outside of these areas, dioxin and furan concentrations were similar to concentrations in background sediment locations in the greater Seattle area.

COCs not selected as risk drivers in the baseline HHRA may be further evaluated in subsequent steps in the cleanup process, in consultation with EPA and Ecology. This evaluation may include:

- Assessment of reductions in sediment concentrations or residual risks from these chemicals following the selection of the preferred alternative in the FS
- Review of any new toxicological effects data, as part of the 5-year review that is conducted once a CERCLA cleanup is completed
- Inclusion of these chemicals as part of the post-cleanup monitoring program

7.0 Background Concentrations of Risk Driver Chemicals

This section presents concentrations of selected risk driver chemicals (i.e., PCBs, arsenic, cPAHs, and dioxins and furans) in sediment and tissue collected from locations outside of the LDW. Understanding background levels for these risk driver chemicals is helpful because it provides a context for both the ecological and human health RBTCs, which are discussed in Section 8. The background data presented in this section may also be used in the FS or by EPA and Ecology in their evaluation of cleanup levels. Background concentrations in sediment are not relevant for those chemicals that are risk drivers solely because they exceed SMS criteria (i.e., SQS and CSL); there is no provision in the SMS for consideration of background in remedial decision-making.

According to MTCA (WAC 173-340-200), natural background concentrations are concentrations of hazardous substances that are consistently present in an environment that has not been influenced by localized human activities. Natural background concentrations are used in the evaluation of sediment cleanup levels as a lower limit below which cleanup levels cannot be achieved (WAC 173-340-705(6)). MTCA also discusses area background, which is a term used to represent those concentrations that are consistently present in the environment in the vicinity of the site as a result of human activities unrelated to releases from the site. When cleanup levels are less than area background concentrations, MTCA recognizes that recontamination could result in future exceedance of cleanup levels at a site. In such cases, MTCA allows that portion of the cleanup action to be delayed until offsite sources are controlled.

CERCLA uses different definitions for background. Under CERCLA guidance, anthropogenic background refers to natural and human-made substances that are present in the environment as a result of human activities, but that are not specifically related to the CERCLA release in question (EPA 2002d). Natural background refers to substances that are naturally present in the environment in forms that have not been influenced by human activity (e.g., naturally occurring metals). CERCLA generally does not require clean up to concentrations below anthropogenic background levels (EPA 2002a). CERCLA's anthropogenic background concentrations are generally higher than MTCA's natural background concentrations because anthropogenic background is more inclusive of urban environments. Application of anthropogenic background in risk management decision-making is recognized by EPA as a means to minimize risk of recontamination from sources outside of those related to the site.

This section discusses natural background concentrations for sediment and tissue, as well as information relevant to area or anthropogenic background concentrations in sediment within the greater Seattle metropolitan area. For natural background, sediment data were compiled from areas that have not been influenced by localized

human activities. Preliminary[84] natural background concentrations were calculated for each of the risk driver chemicals using statistical guidance from MTCA (Ecology 1992).

Area and anthropogenic background concentrations in sediment have not formally been defined for the LDW. To address these background concepts, the term "urban background" is used in this report when evaluating relevant data.

For urban background, it is necessary to evaluate data sets that include the effect of ongoing, non-point sources of chemicals to the LDW. Although some of these sources may be controlled or abated in the near-term, many of them will require long-term source control efforts. Because the methods and appropriate data sets for calculating these background values have not yet been approved by EPA or Ecology, an assessment of various data that could be used to inform urban background concentrations was included. Relevant data include upstream data (from the Green River) as well as sediment data from Elliott Bay and other urban bays in Puget Sound, Lake Washington, and Lake Sammamish.

For tissue, most of the available data for total PCBs from Puget Sound have been characterized as either near-urban or non-urban (West et al. 2001). Available arsenic data were categorized according to whether or not the sampling location was likely to have been impacted by emissions from the former Asarco smelter located in Ruston, Washington. Tissue data are presented to show the general range of concentrations in areas outside the influence of site-related activities.

7.1 NATURAL BACKGROUND SEDIMENT DATA

This section describes which sediment data were selected to derive preliminary natural background concentrations, and which statistical methods were applied to the data.

7.1.1 Data Sources

Preliminary natural background concentrations of total PCBs, arsenic, cPAHs, and dioxins and furans in sediment were derived using methods presented in WAC 173-340-709 and in Ecology's *Statistical Guidance for Ecology Site Managers and Supplement S-6* (Ecology 1992). Data from 10 areas in Puget Sound identified as reference areas by the Battelle Marine Research Laboratory (Battelle) (1986) or WSDOH (1995) were used in natural background calculations for total PCBs, arsenic, and cPAHs (Table 7-1; Map 7-1).

[84] The final determination of natural background numbers and their application with respect to remedial decision-making will be determined by EPA and Ecology.

Table 7-1. Reference areas for natural background sediment calculations for total PCBs, arsenic, and cPAHs

Reference Area	Sources Used to Identify Reference Areas
Carr Inlet	WSDOH (1995)
Case Inlet	Battelle (1986)
Dabob Bay	WSDOH (1995); Battelle (1986)
Discovery Bay	WSDOH (1995)
Rich Passage	WSDOH (1995)
Samish Bay	WSDOH (1995); Battelle (1986)
Saratoga Passage	WSDOH (1995)
Sequim Bay	WSDOH (1995); Battelle (1986)
West Beach	WSDOH (1995)
Wollochet Bay	WSDOH (1995)

Battelle – Battelle Marine Research Laboratory

cPAH – carcinogenic polycyclic aromatic hydrocarbon

PCB – polychlorinated biphenyl

WSDOH – Washington State Department of Health

Sediment data for total PCBs, arsenic, and cPAHs from these 10 reference areas were obtained from four sources: 1) a study by NOAA and Ecology on sediment quality in Puget Sound conducted from 1997 to 1999 (NOAA and Ecology 1999, 2000, 2002), 2) Washington State's SEDQUAL database (Ecology 2004c), 3) a Battelle survey of eight bays (1986), and 4) Ecology's Environmental Information Management (EIM) System database (Ecology 2007c). Data were processed using data management procedures described in Appendix E.

Sediment data for dioxins and furans were not available from the 10 reference areas, with the exception of one sample collected from Carr Inlet by NOAA (Steinhoff 2007). Therefore, a search was conducted to identify additional background sediment data for dioxins and furans. The search resulted in the identification of data for 11 locations in Freshwater Bay and 11 locations in Dungeness Bay, which were collected for the RI at the Rayonier, Inc., mill site near Port Angeles, Washington (Map 7-1) (Malcolm Pirnie 2007). Because these data were collected from locations outside the influence of localized human activities, they were included in the derivation of a natural background value for dioxins and furans, along with the sample from Carr Inlet.

7.1.2 Calculation of preliminary natural background concentrations in sediment

The background datasets created for each chemical or group of chemicals were analyzed using MTCAStat, a statistical analysis tool developed by Ecology for MTCA site cleanup work (Ecology 1992). Consistent with guidance provided by Ecology (1992) and requirements of WAC 173-340-709, natural background values were calculated according to the distribution of available background data as follows:

- For datasets that are log-normally distributed, background should be defined as the true upper 90th percentile or four times the true 50th percentile, whichever is lower.
- For datasets that are normally distributed, background should be defined as the true upper 80th percentile or four times the true 50th percentile, whichever is lower.
- For datasets that are neither log-normally nor normally distributed, an alternative approach to calculating background sediment concentrations should be recommended to and reviewed by Ecology.

This section presents the preliminary values derived using the above criteria; the application of these background data to setting cleanup levels will be determined by EPA and Ecology and documented in the Record of Decision (ROD).

The approach and resulting natural background concentrations for PCBs, arsenic, cPAHs, and dioxins and furans in sediment are discussed in detail below. Data distributions for these chemicals are summarized in Table 7-2.

Table 7-2. Natural background concentrations calculated using reference area data

Chemical	Unit	n	Detection Frequency (%)	Distribution	Maximum Detected Conc.	80th Percentile	90th Percentile	4 x 50th Percentile
Total PCBs	µg/kg dw	70	7.1	na	19	na	na	na
Arsenic	mg/kg dw	124	79	log-normal	27	8.8	12	21
cPAHs[a]	µg/kg dw	119	71	log-normal	274	54	92	77
Dioxin and furan TEQ	ng/kg dw	23	100[b]	non-parametric	1.0	0.32	0.60	0.52

[a] Total cPAHs were calculated by summing the products of concentrations and compound-specific PEFs for individual cPAH compounds, as discussed in detail in Appendix E, Section E.3.

[b] At least one dioxin or furan congener was detected in every sample.

cPAH – carcinogenic polycyclic aromatic hydrocarbon

dw – dry weight

n – number of samples

na – not available (distribution could not be determined because of the large number of non-detects in the total PCBs dataset)

PCB – polychlorinated biphenyl

TEQ – toxic equivalent

Total PCBs were not detected in a high number of the sediment samples in this dataset (only 5 of 70 samples had detected concentrations) (Table 7-2). Because the high number of non-detects would result in a calculated background value based largely on reporting limits, no distribution was assigned to total PCBs.

In an earlier evaluation of total PCBs in surface sediment by Ecology (1997), a low number of detected concentrations was reported from five Puget Sound reference areas (i.e., Carr Inlet, Dabob Bay, Holmes Harbor, Samish Bay, and Sequim Bay). Only 6 out of 60 samples in that study had detected concentrations of total PCBs. Ecology conducted a statistical analysis using six different approaches. Ecology's recommended approach included a statistical analysis of 26 of the 60 data points using a combination of detected and undetected data. Details regarding these statistical analyses and the selection of these 26 data points were not available. The 90th percentile PCB concentration for those 26 values was 31 µg/kg dw, and the corresponding OC-normalized PCB concentration was 1.2 mg/kg OC (Ecology 1997). This value was calculated using OC-normalized concentrations for each data point based on sample-specific TOC data. This OC-normalized value has been cited by EPA in the Puget Sound Naval Shipyard Complex ROD (EPA 2000a) and is generally recommended by Ecology (1997).

Because Ecology conducted extensive statistical analyses on total PCB data from reference areas (Ecology 1997) and recommends a background concentration of 1.2 mg/kg OC, this concentration (and its corresponding dry weight concentration of 31 µg/kg dw) was selected as the preliminary natural background value for total PCBs without further statistical analysis.

Background sediment data for arsenic and for cPAHs derived from Battelle (1986) or WSDOH (1995) were log-normally distributed (Table 7-2). Arsenic and at least one cPAH compound were detected in 79% and 71%, respectively, of the samples from the 10 reference areas. Consistent with MTCA guidance, the lower of two values, the upper 90th percentile or four times the 50th percentile, was selected as the preliminary natural background value for each chemical. These values were 12 mg/kg dw for arsenic and 77 µg/kg dw for cPAHs. [85]

To identify a natural background concentration for dioxins and furans, the distribution of the TEQs was evaluated; it did fit any of the standard distributions (normal, log-normal, or geometric) (Table 7-2). When this occurs, an alternative approach is recommended by Ecology (1992). However, because the fit criterion for the log-normal distribution was almost achieved ($R^2 = 0.89$ compared to the required $R^2 = 0.90$), the approach for calculating background when the data are log-normally

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[85] Data for cPAHs and dioxins and furans are presented as TEQs. TEQs for cPAHs were based on concentrations of seven individual cPAHs and their toxicity relative to benzo(a)pyrene, and TEQs for dioxins and furans were based on concentrations of 17 individual dioxin and furan congeners and their toxicity relative to 2,3,7,8-tetrachlorodibenzo-*p*-dioxin. These TEQs were calculated as the sum of each individual cPAH or dioxin and furan congener concentration and then multiplied by the corresponding PEF or TEF value, as described in Appendix E. When the individual cPAH compound or dioxin and furan congener concentration was reported as non-detected, the PEF or TEF was multiplied by one-half the RL for that compound or congener. If no individual compounds or congeners were detected, the TEQ was flagged as non-detected.

distributed was used. That approach involved selecting the lower of the upper 90th percentile or four times the 50th percentile as the preliminary natural background value. The calculated dioxin and furan TEQ value was 0.52 ng/kg dw (Table 7-2).

A summary of the preliminary natural background concentrations for risk driver chemicals in surface sediment is presented in Table 7-3.

Table 7-3. Preliminary natural background concentrations for risk driver chemicals in surface sediment

Risk Driver Chemical	Unit	Natural Background Concentration	Basis
Arsenic	mg/kg dw	12	90th percentile
cPAHs[a]	µg/kg dw	77	4 x 50th percentile
Dioxin and furan TEQ	ng/kg dw	0.52	4 x 50th percentile
Total PCBs	µg/kg dw	31	Ecology (1997)
	mg/kg OC	1.2	Ecology (1997)

[a] Total cPAHs were calculated by summing the products of concentrations and compound-specific PEFs for individual cPAH compounds, as discussed in detail in Appendix E, Section E.3.

cPAH – carcinogenic polycyclic aromatic hydrocarbon

dw – dry weight

Ecology – Washington State Department of Ecology

OC – organic carbon

PCB – polychlorinated biphenyl

TEQ – toxic equivalent

7.2 Urban Background Sediment Data

This section presents information on urban background concentrations in sediments for PCBs, arsenic, cPAHs and dioxins and furans. Various data are presented because no single specific study exists to document typical concentrations of chemicals found in urban sediments in the greater Seattle area or Puget Sound.

Determining urban background in a highly urbanized waterway is difficult because no single data source defines this value. Therefore, multiple lines of evidence were evaluated in order to bracket a range of potential values.

7.2.1 PCBs, Arsenic, and cPAHs

For PCBs, arsenic, and cPAHs, an array of studies conducted for different purposes were evaluated as lines of evidence to assess the range of urban background concentrations. Four general lines of evidence were considered:

- Concentrations associated with suspended solids in Green/Duwamish River inflow, based on upstream water quality monitoring data
- Surface sediment samples collected upstream of RM 5.0

- USACE dredged material characterization cores collected from RM 4.0 to RM 4.75
- Surface sediment concentrations in Puget Sound area urban bays outside of defined cleanup or disposal areas.

To provide a consistent basis for comparison, all data are expressed on a dry weight basis. The 90th percentile, mean, and median concentrations were calculated using MTCAStat for all datasets evaluated following the same data rules described in Section 7.1.2. cPAHs were calculated as TEQs in terms of benzo(a)pyrene equivalents using the method described in Section 7.1.2.

7.2.1.1 Green River Upstream Water Quality

King County water quality data from the Green/Duwamish River were reviewed to evaluate the concentrations of PCBs as Aroclors, arsenic, and cPAHs associated with suspended solids in samples collected upstream of the LDW (Herrera 2005, 2004b). These data potentially represent urban inputs to the LDW from the Green/Duwamish River system.

Water quality samples routinely collected in the Green River (at Fort Dent) have been analyzed for PCBs as Aroclors, PCBs, and PAHs (Herrera 2005, 2004b). Arsenic was detected in most samples. All PCB and PAH results to date have been reported as non-detects. PCB congeners have also been analyzed by King County using methods with lower reporting limits. PAHs have not been re-analyzed using methods with lower reporting limits. The PCB congener and arsenic water data are discussed below.

King County Water Data – PCB Congeners

King County analyzed water samples collected four times in 2005 (August to December) and five times in 2007 (March to June) for PCB congeners using methods with low detection limits (Mickelson and Williston 2006; Williston 2007[in prep]). These samples were collected from the Fort Dent location in the Green River (labeled as TGS 1 at RM 10.9) as described by Williston (2007 [in prep.]); TSS was also determined. Samples were also collected from the Duwamish River at the E. Marginal Way S Bridge at South 155th Street (Station 0309 at RM 6.3) four times in 2007. Thus, a total of 13 water samples were analyzed for PCB congeners and TSS (Williston 2007 [in prep.]).

To estimate total PCB concentrations in particulates at locations upstream of the LDW, each total PCB concentration[86] was normalized to the sample-specific TSS concentration and expressed as µg/kg dw TSS. Based on this analysis, a mean total PCB concentration of 60 µg/kg dw TSS was calculated, with a range of 3 to 367 µg/kg

[86] Total PCBs were calculated as the sum of detected PCB congeners.

dw TSS. The median and 90th percentile concentrations were 36 and 113 µg/kg dw TSS, respectively.

King County Water Data – Arsenic

King County water quality data from the Green River were used to calculate arsenic concentrations associated with suspended solids (Herrera 2005, 2004b). Samples with total and dissolved arsenic and TSS data were used for the analysis. Particulate arsenic was calculated by subtracting (on a per sample basis) dissolved arsenic from total arsenic. The resulting particulate arsenic value was then normalized to the sample-specific TSS concentration to estimate the arsenic concentrations in suspended solids that could potentially settle in the LDW. Water quality data from King County monitoring station A310 (Green River at Fort Dent at RM 10.9) were used for this analysis (Herrera 2005, 2004b). While other water quality data are available in the Green River for arsenic, the other data were collected from locations further upstream, and were thus not used in this assessment.

Data from a total of 101 water samples[87] analyzed between 2001 to 2006 from Station A310 were included in the analysis. The mean and 90th percentile concentrations of arsenic normalized to suspended solids were 37 and 78 mg/kg dw TSS, respectively.

7.2.1.2 Upstream Sediment Sampling

Total PCBs, arsenic, and cPAHs data for sediment samples collected upstream of the LDW (RM 5.0 to RM 7.0) were compiled and analyzed to evaluate the quality of sediment potentially being transported to the LDW from the Green/Duwamish River system (RETEC 2007b). The datasets included in this analysis are presented in Table 7-4 and on Map 7-2.

[87] More than 101 arsenic samples were collected during these monitoring events, but only the samples with both total and dissolved arsenic concentrations were used in the analysis. These data were used to calculate the concentration of particulate arsenic (difference between total and dissolved) in each sample. The particulate arsenic concentration was then normalized to TSS.

Table 7-4. Surface sediment sampling events conducted upstream of the LDW

Sampling Event (Event Code)	Year Conducted	River Mile	No. of Samples Analyzed for PCBs (No. of Detections)	No. of Samples Analyzed for Arsenic (No. of Detections)	No. of Samples Analyzed for cPAHs (No. of Detections)	Source
LDW RI: surface sediment sampling for chemical analyses and toxicity testing (LDWRI-SurfaceSedimentRound2)	2005	5.1 – 5.8	6 (0)	4 (4)	6 (4)	Windward (2005d)
LDW RI: surface sediment sampling for chemical analyses and toxicity testing (LDWRI-SurfaceSedimentBackground)[a]	2005	5.3 (PCBs) 5.3 – 7.0 (arsenic)	1 (0)	8 (8)	0	Windward (2005g)
EPA Site Inspection: Lower Duwamish River (EPA SI)	1998	5.3 – 5.5	5 (0)	5 (5)	5 (0)	Weston (1998)
Duwamish Waterway Phase 1 site characterization (Boeing SiteChar; upstream reference samples)[a]	1997	6.1	3 (0)	3 (3)	3 (2)	Exponent (1998)
Duwamish Waterway sediment characterization study (NOAA SiteChar)	1997	5.2 – 6.0	20 (18)	0	0	NOAA (1997; 1998)
Norfolk CSO sediment cleanup study – Phase 1 (Norfolk-cleanup1)	1994	5.4 – 5.5	2 (0)	2[a]	2 (1)	King County (1996)
Total Samples			**37 (18)**	**22 (22)**	**16 (7)**	

[a] Data not included in summary statistics. Only arsenic surface sediment data used for background in the HHRA were used in this evaluation.

CSO – combined sewer overflow
EPA – US Environmental Protection Agency
LDW – Lower Duwamish Waterway
NOAA – National Oceanographic and Atmospheric Administration
PCB – polychlorinated biphenyl
RI – remedial investigation
SI – site investigation

Arsenic was detected in all of the 20 surface sediment samples analyzed. Using MTCAStat, the mean, median, and 90th percentile arsenic concentrations were calculated to be 6, 5, and 9 mg/kg dw, respectively.

PCBs were analyzed in 37 surface sediment samples from six events conducted upstream of RM 5.0. PCBs were detected in the 1997 NOAA event only (18 samples had detected concentrations of PCBs; PCBs were not detected in 2 samples). Total PCB concentrations for the 20 samples in the 1997 NOAA event ranged from nondetected to 140 µg kg dw. The mean, median, and 90th percentile concentrations were 21, 6, and 45 µg/kg dw, respectively.

cPAHs were analyzed in 16 surface sediment samples from four events conducted upstream of RM 5.0. cPAH concentrations were calculated using the TEQ methodology discussed in Section 7.1.2. The mean, median, and 90th percentile cPAH concentrations were 106, 56, and 246 µg/kg dw, respectively.

The upstream surface sediment samples analyzed for PCBs, arsenic, and cPAHs were relatively coarse-grained and may not represent the full spectrum of material, especially the fine-grained fraction, likely to settle in the LDW from upstream sources.

7.2.1.3 USACE Dredged Material

The Seattle District USACE has been characterizing sediments in the Upper Turning Basin and in the navigation channel as part of their frequent, on-going navigation maintenance dredging responsibilities. For the purposes of dredged sediment characterization, the USACE has been compositing sediment cores for characterization both vertically (generally 0-to-4-ft intervals but occasionally as deeper intervals, up to 10 ft), and in some cases, horizontally (compositing two or more cores collected across the basin or navigation channel).

The Upper Turning Basin at the head of the LDW serves as an engineered catch basin, collecting much of the Green River bedload solids and some suspended solids that settle in the upstream end of the LDW. Different sediment particle sizes, ranging from clay to coarse sand, are transported as either suspended load or bedload transport. The transport of suspended load corresponds to the movement of sediment suspended in the water column (primarily clay, silt, and fine sand). Bedload transport corresponds to the movement of sand and gravel in a thin layer (i.e., about 1 mm to 1 cm thick) just above the sediment bed. Twenty-four percent of the total sediment load entering the upstream end of the STM study area (defined as RM 4.75) is bedload, which primarily settles in the Upper Turning Basin (RM 4.5 to RM 4.75) (QEA 2007). Therefore, the majority of the LDW sediment load settling in the lower reaches of the LDW is from suspended material, which carries finer fractions. These finer fractions may have higher chemical concentrations than the coarser bed load.

Therefore, these USACE core data provide a measure of the mean chemical concentrations associated with mixture of fine and coarse grained material from the

Green River. The STM estimates that the Upper Turning Basin bed sediment is primarily composed of sediment from the Green River (75 to 99% of load in 30 yrs).

USACE provided a subset of the Dredged Analysis Information System (DAIS) database queried for LDW chemical and physical data from 1989 to 2003 (Fox 2007). Figures 7-1, 7-2, and 7-3 present the total PCBs, arsenic, and cPAHs data, respectively, as a function of time relative to dredging events. The percent fines (sum of silt and clay fractions) associated with these samples ranged from about 10 to 20% fines near the Upper Turning Basin between RM 4.5 and RM 4.75 (dominated by coarser bed-load fractions) and ranged from about 60 to 80% fines near RM 4.0 (a settled mixture of suspended and bedload material).

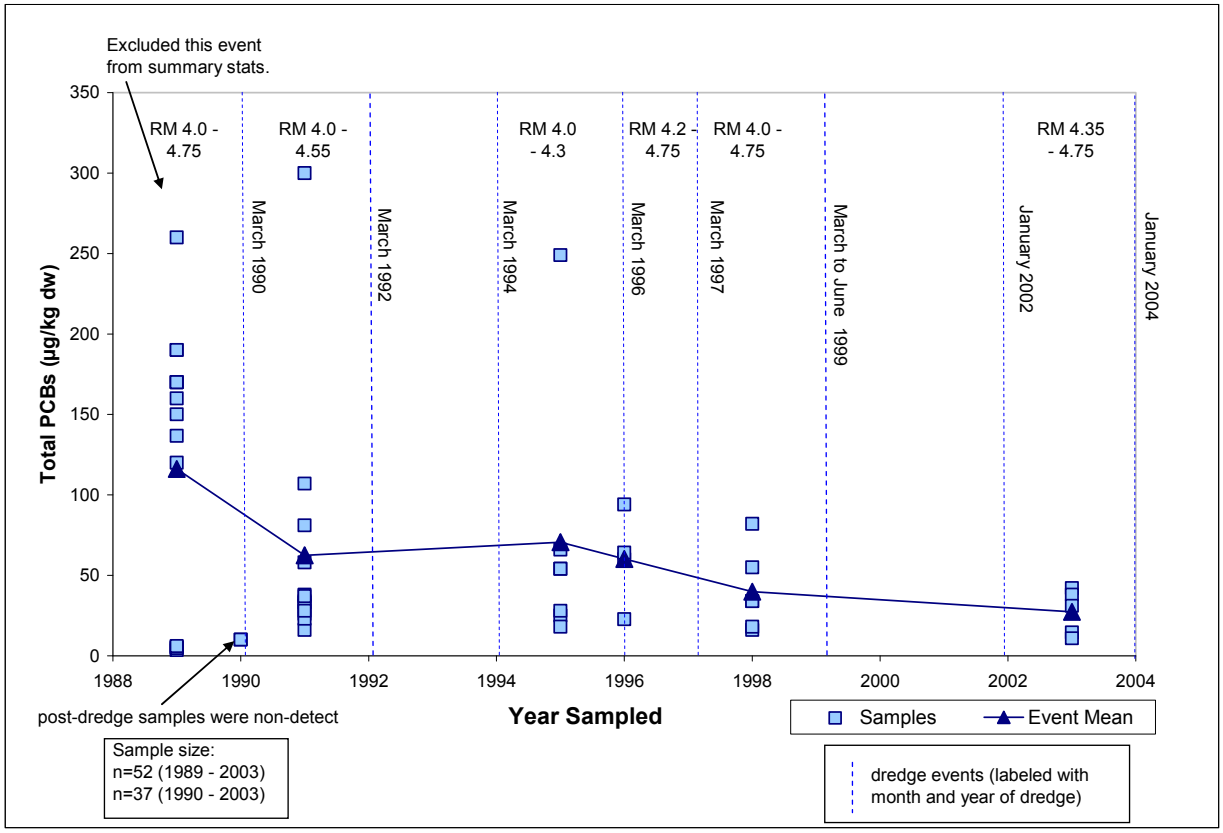


Figure 7-1. Total PCB concentrations in cores collected for dredged material characterization from the Upper Turning Basin over time

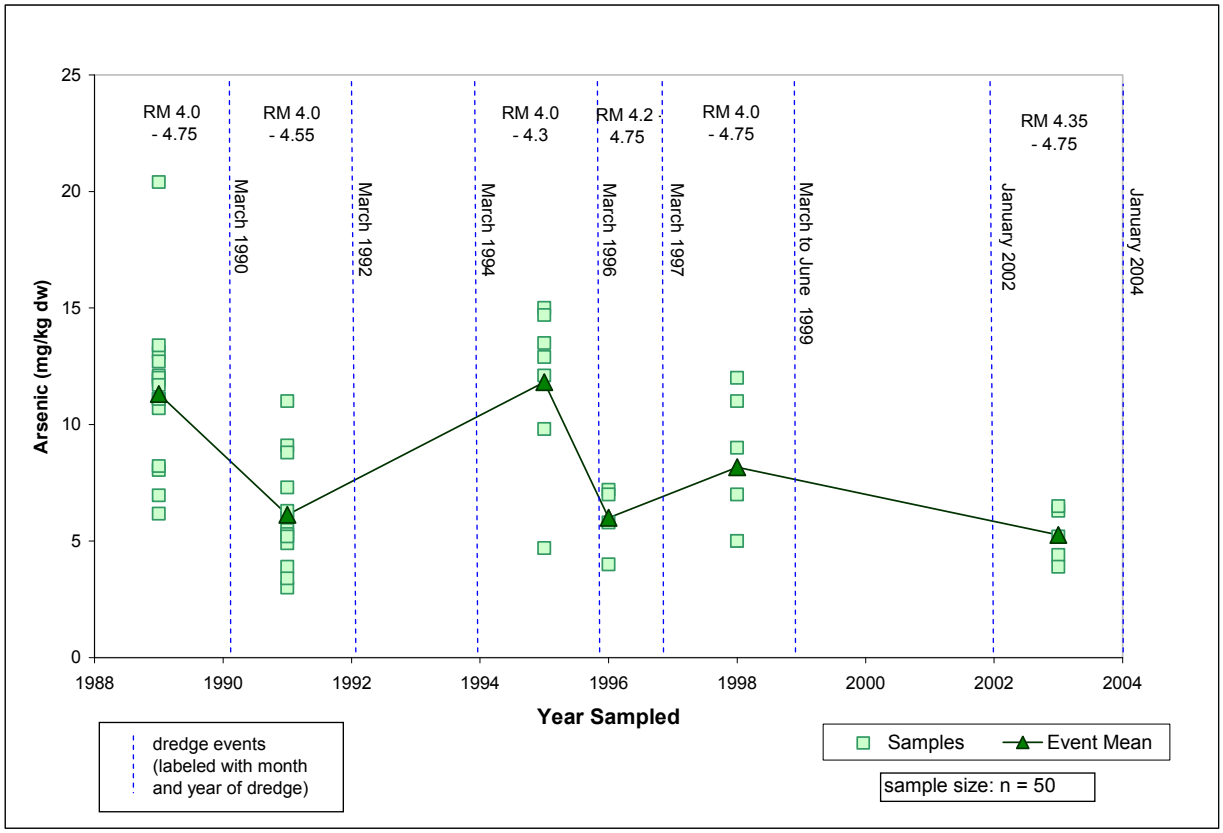
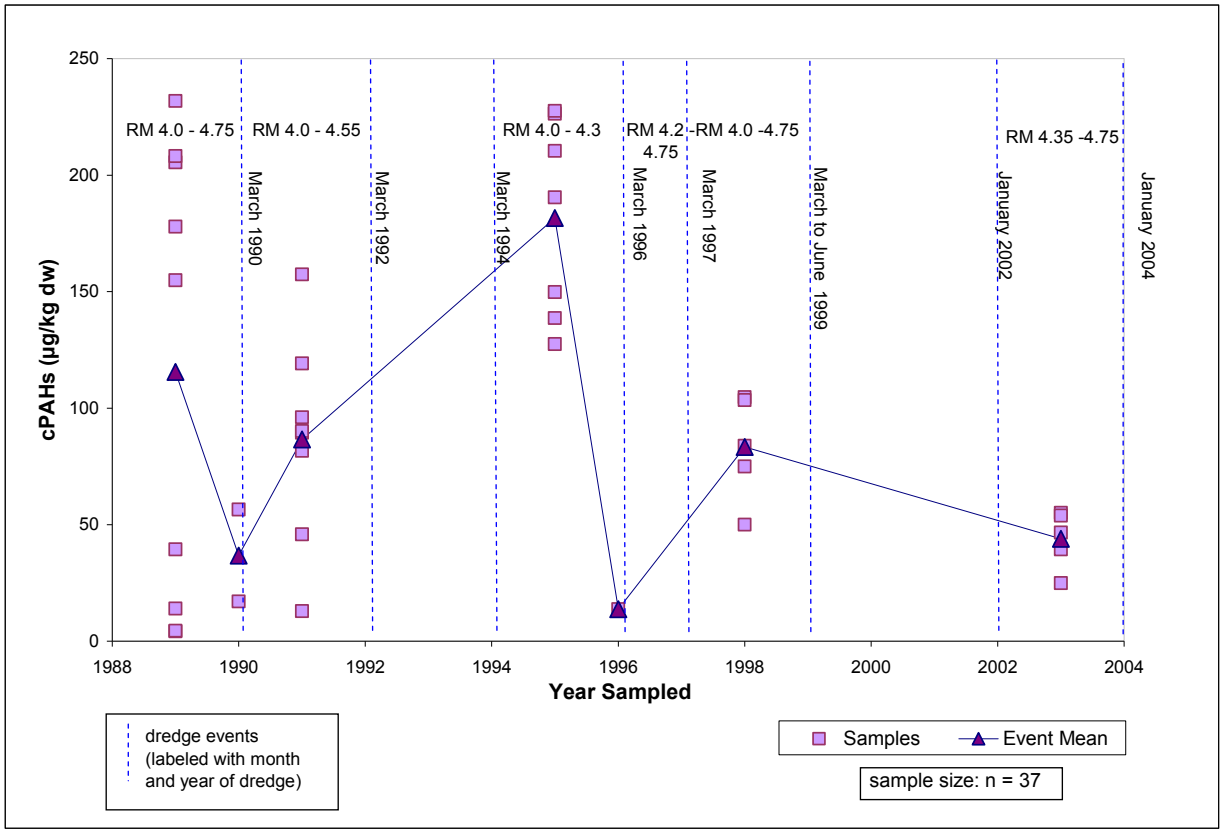


Figure 7-2. Arsenic concentrations in cores collected for dredged material characterization from the Upper Turning Basin over time



Note: Total cPAHs were calculated by summing the products of concentrations and compound-specific PEFs for individual cPAH compounds, as discussed in detail in Appendix E, Section E.3.

Figure 7-3. cPAH concentrations in cores collected for dredged material characterization from the Upper Turning Basin over time

The mean and 90th percentile total PCB concentrations for data collected between 1990 and 2003 were 61 and 109 µg/kg dw, respectively. For PCBs, the 1989 sampling event was excluded from the statistical calculations because it had much higher total PCB concentrations than those measured in the other sampling events. However, this event was retained for the arsenic and cPAH data summaries. The mean and 90th percentile arsenic concentrations were 9 mg/kg dw and 14 mg/kg dw, respectively. The mean and 90th percentile cPAH concentrations were 101 and 298 µg/kg dw, respectively. One concentration (795 µg/kg dw) was excluded from the cPAH dataset because it was an outlier (3 times the next highest sample value). The cPAH TEQs were calculated using five of the seven carcinogenic compounds; benzo(b)fluoranthene and benzo(k)fluoranthene data were not available in the DAIS database.

7.2.1.4 Sediment data from Urban Areas

Arsenic and total PCB concentrations in surface sediments in urban water bodies outside of the LDW were also evaluated. cPAH data were not evaluated because the data were not as readily available.

Surface sediment data for arsenic and total PCBs within the following major urban bays and lakes were extracted from Ecology's EIM system and evaluated within the following major urban bays and lakes: Elliott Bay, Bellingham Bay, Commencement Bay, Lake Washington, and Lake Sammamish. The details of this evaluation were presented in RETEC (2007a). Prior to the calculation of statistics, these data were screened to exclude the following types of samples:

- Samples collected as part of a CERCLA or MTCA cleanup study prior to any sediment remediation (post-remediation monitoring data from any such sites were retained)
- Samples collected as part of routine monitoring of the DMMP open-water dredged material disposal sites

The summary statistics for each urban bay or lake were evaluated individually; the data were then combined and summary statistics for the full dataset were calculated (Tables 7-5 and 7-6). The data were statistically evaluated using MTCAStat and the criteria presented in Section 7.1.2. The mean total PCB concentrations in urban bays and lakes ranged from 40 to 163 µg/kg dw, with an overall mean of 135 µg/kg dw. The 90th percentile of the total PCB data ranged from 64 to 501 µg/kg dw. The mean arsenic concentrations in urban bays and lakes ranged from 7 to 15 mg/kg dw, with an overall mean of 11 mg/kg dw. The 90th percentiles of the arsenic data ranged from 13 to 38 mg/kg dw.

Table 7-5. Total PCB concentrations surface sediment samples collected in urban bays and lakes, excluding cleanup and disposal sites

Variable	No. of Samples	Detection Frequency	PCB Concentration (µg/kg dw)								Data Distribution
			Min	Max	Mean	95% UCL	50TH Percentile	80TH Percentile	90TH Percentile	4X 50TH Percentile	
Elliott Bay	367	224/367	2	1,295	163	251	40	259	501	160	other
Bellingham Bay	61	6/61	8	425	76	164	30	74	320	120	other
Commencement Bay	134	87/134	2	1,104	43	78	21	50	64	170	other
Lake Sammamish	25	25/25	16	88	40	49	34	56	73	136	log normal
All data combined	587	342/587	2	1,295	135	204	35	200	427	138	other

Note: Lake Washington data were not included because of the 17 samples available, total PCBs were detected in one sample.

dw – dry weight

PCB – polychlorinated biphenyl

UCL – upper concentration limit

Table 7-6. Arsenic concentrations surface sediment samples collected in urban bays and lakes, excluding cleanup and disposal sites

Variable	No. of Samples	Detection Frequency	Arsenic Concentration (mg/kg dw)								Data Distribution
			Min	Max	Mean	95% UCL	50TH Percentile	80TH Percentile	90TH Percentile	4 X 50TH Percentile	
Elliott Bay	345	282/345	0.5	15	7	12.	6	10	13	26	lognormal
Bellingham Bay	162	160/162	1.5	19	9	10	9	12	13	37	normal
Commencement Bay	134	132/134	1.4	54	10	12	9	13	15	35	other
Lake Washington	29	25/29	2.0	27	7	9	6	10	13	25	lognormal
Lake Sammamish	29	29/29	1.8	72	15	59	9	23	38	35	lognormal
All data combined	699	628/699	0.5	72	11	12	9	14	21	34	other

dw – dry weight

UCL – upper concentration limit

7.2.2 Dioxins and furans

Surface sediment sampling for dioxins and furans in the greater Seattle metropolitan area was conducted as part of the RI sampling event in 2005 (Windward 2005d). Individual dioxin and furan congener data were converted to a dioxin and furan TEQ using concentrations of detected congeners, one-half the RLs for congeners that were not detected in a given sample, and mammalian TEFs (Van den Berg et al. 2006). The TEQs for the nine locations are presented in Table 7-7. TEQs ranged from 2.59 ng/kg dw (upstream of the LDW) to 187 ng/kg dw (in Salmon Bay). Most of the TEQs were less than 30 ng/kg dw.

Table 7-7. Dioxin and furan TEQs in surface sediment samples collected from the greater Seattle area

Location	Location ID	Dioxin and Furan TEQ (ng/kg dw)	Values Included in Calculation of Mean (ng/kg dw)
Elliott Bay (Terminal 91)[a]	EB-SS2a	13.7 J	16.3
	EB-SS2b	18.9 J	
Lake Union (I-5 bridge)[a]	LU-SS9a	5.46 J	15.8
	LU-SS9b	26.1 J	
Lake Washington (Bothell)	LW-SS3	13.2 J[b]	13.2
Lake Washington (Bellevue)	LW-SS4	14.7 J	14.7
Lake Washington (Renton)[a]	LW-SS5a	14.1 J	14.3
	LW-SS5b	14.5 J	
Ship Canal (Salmon Bay)[a]	SC-SS1a	187 J	
	SC-SS1b	63.1 J	
Union Bay (Laurelhurst)	UB-SS8	53.4 J	
Mean for greater Seattle area locations (excluding samples from the Ship Canal and Union Bay)			**14.9**
Springbrook Creek (Tukwila)	SB-SS6	2.23 J	2.23
Duwamish River (Tukwila)	DRD-SS7	2.59 J	2.59
Mean for upstream locations			**2.41**

[a] Two samples were collected, one approximately 30 to 50 ft from the outfall and the other approximately 100 to 120 ft from the outfall.

[b] Reported concentration is the average of two field samples.

dw – dry weight

ID – identification

na – not applicable

TEQ – toxic equivalent

These Seattle-area samples were collected to represent ambient dioxin and furan concentrations associated with typical urban sources. Dioxins and furans from these sources are distributed throughout urban areas via atmospheric deposition followed by stormwater runoff. The criteria used to select sampling areas representative of

LDW background were as follows: 1) the area must receive drainage from basins with land uses similar to the LDW, 2) the area must not be located near known industrial point sources of dioxins and furans, 3) the area must represent a range of receiving water environments, and 4) the area must represent a range of stormwater discharge frequencies, volumes, and types similar to those in the LDW. The nine sampling locations that met these four criteria are shown on Map 7-3. In order to assess the potential influence of outfall proximity on dioxin and furan concentrations in sediment, two samples, one between 30 and 50 ft from the outfall and a second between 100 and 120 ft from the outfall, were collected at each of four of these locations: Elliott Bay (Terminal 91), Lake Union (I-5 bridge), Lake Washington (Renton), and the Ship Canal (Salmon Bay).

Data were reviewed to determine the appropriate dataset to represent urban conditions expected for areas with stormwater runoff, such as the LDW. Data from the Ship Canal and Union Bay were excluded because the TEQs were higher than the national range of 0.12 to 16.3 ng/kg dw[88] (EPA 2000b). Data from the three remaining locations with samples collected at different distances from the outfall were averaged to better represent area concentrations and not overemphasize individual locations. Also, data from the two locations upstream of the LDW were averaged separately to represent upstream sediment. The mean dioxin and furan TEQ for the resulting dataset was 14.9 ng/kg dw. The mean TEQ of the two upstream samples was 2.41 ng/kg dw (Table 7-7).

7.3 BACKGROUND TISSUE DATA

This section presents the range of total PCB and arsenic concentrations in tissue samples from background areas.

Relatively few tissue data were available for dioxins and furans and cPAHs. Dioxins and furans were identified as a seafood consumption risk driver based on an assumption of unacceptable risk (no LDW tissue data were available); thus, the limited background tissue data available are not presented. For cPAHs, limited background tissue data were available from King County (King County 1999e). However, cPAHs were rarely detected.[89] Thus, these cPAH data are not presented.

7.3.1 PCBs

A search was conducted to identify available PCB tissue data from background areas in the greater Puget Sound area. The first step included a general online search,

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[88] The range of dioxin and furan TEQs in sediments of 11 lakes and reservoirs throughout the US, which were selected to represent background conditions in areas removed from known sources, was 0.12 to 16.3 ng/kg dw (EPA 2000b), with an arithmetic mean of 5.3 ng/kg dw

[89] The detection limit for the available background samples (King County 1999e) was higher than that achieved in the LDW, further reducing the benefit of using such samples to characterize background.

including the Web sites for WDFW, the King County Department of Natural Resources (KCDNR), Ecology, and WSDOH. A search was also conducted of the EIM database on Ecology's Web site (Ecology 2007c). In addition, both WDFW and DOH were contacted to confirm that all available data had been located.

A summary of available tissue data is provided in Table 7-8. Total PCBs were calculated as the sum of detected Aroclors, unless total PCB concentrations were provided in the individual data sources, in which case the reported value for total PCBs was used. If no individual Aroclors were detected in a sample, the total PCB concentration was reported as undetected at the highest RL for an individual Aroclor.

Table 7-8. Total PCB concentrations in fish, crab, and clam tissue collected from non-urban and near-urban areas of the Puget Sounds

Species	Tissue Type	Sampling Location	Sampling Year(s)	Detection Frequency	Average Individuals per Composite	Total PCB Concentration (µg/kg ww)			Source
						Mean[a]	Minimum	Maximum	
Rockfish									
Copper rockfish	fillet	PSAMP – non-urban[b]	1989 – 1999	1/1	5	4.5	4.5 J	4.5 J	West et al. (2001)
Copper rockfish	fillet	PSAMP – near-urban[b]	1989 – 1999	17/17	3.6	9.9	3.9 J	22.5 J	West et al. (2001)
Quillback rockfish	fillet	PSAMP – non-urban[b]	1989 – 1999	42/79	2.6	12.2	2.0 J	124.4 J	West et al. (2001)
Quillback rockfish	fillet	PSAMP – near-urban[b]	1989 – 1999	42/42	2.4	40.3	4.1 J	140.4 J	West et al. (2001)
Yelloweye rockfish	fillet	PSAMP non-urban[b]	1989 – 1999	2/2	1	32.3	16.3 J	48.2 J	West et al. (2001)
Sole									
English sole	fillet	Case Inlet/ Dana Passage	2005	2/3	4.67	7.2	5.6 J	13.2 J	Era-Miller (2006)
English sole	fillet	Pickering Passage	2005	0/2	5	2.8	5.5 U	5.6 U	Era-Miller (2006)
English sole	fillet	South Puget Sound	2005	2/2	20	6.5	6.1 J	6.8 J	Era-Miller (2006)
English sole	fillet	PSAMP – non urban[b]	1989 – 1999	117/189	15.2	11.6	1.3 J	50.8 J	West et al. (2001)
English sole	fillet	PSAMP – near urban[b]	1989 – 1999	66/78	15.1	16.2	2.0 J	79.9 J	West et al. (2001)
English sole	na	Discovery Bay	1982	1/1	na	13	13	13	Gahler et al. (1982), as cited in cited in Cubbage (1992)
Flathead sole	na	Discovery Bay	1982	0/1	na	5	10 U	10 U	Gahler et al. (1982), as cited in cited in Cubbage (1992)
Rock sole	fillet	Budd Inlet	2005	0/1	5	2.7	5.4 U	5.4 U	Era-Miller (2006)
Rock sole	fillet	Carr Inlet	2005	0/2	5	2.3	5.5 U	5.5 U	Era-Miller (2006)
Rock sole	fillet	Case Inlet/ Dana Passage	2005	0/1	5	2.3	5.5 U	5.5 U	Era-Miller (2006)
Rock sole	fillet	Hale Passage	2005	0/2	5	2.7	5.1 U	5.5 U	Era-Miller (2006)

Species	Tissue Type	Sampling Location	Sampling Year(s)	Detection Frequency	Average Individuals per Composite	Total PCB Concentration (µg/kg ww)			Source
						Mean[a]	Minimum	Maximum	
Crab									
Dungeness crab	hepato-pancreas	Dungeness Bay[c]	2006	7/7	1	25.0	13.1	49.5	Malcolm Pirnie (2007)[d]
Dungeness crab	hepato-pancreas	Freshwater Bay[c]	2006	8/8	1	17.8	8.80	32.3	Malcolm Pirnie (2007)[d]
Dungeness crab	edible meat	Dungeness Bay[c]	2006	7/7	1	1.02	0.46	1.92	Malcolm Pirnie (2007)[d]
Dungeness crab	edible meat	Freshwater Bay[c]	2006	8/8	1	0.62	0.43	0.99	Malcolm Pirnie (2007)[d]
Dungeness crab	edible meat	Padilla/Fidalgo Bay	1999	1/3	5	1.2	1.2J	2.6U	Ecology (2000)
Dungeness crab	edible meat	Samish Island	1999	1/1	5	1.4	1.4J	1.4J	Ecology (2000)
Dungeness crab	calculated whole-body[e]	Dungeness Bay[c]	2006	7/7	1	8.44	4.39	16.05	Malcolm Pirnie (2007)[d]
Dungeness crab	calculated whole-body[e]	Freshwater Bay[c]	2006	8/8	1	5.96	3.03	10.71	Malcolm Pirnie (2007)[d]
Clams									
Geoduck	tissue	Freshwater Bay[c]	2006	8/8	1	0.64	0.24	1.43	Malcolm Pirnie (2007)[d]
Geoduck	visceral cavity	Freshwater Bay[c]	2006	5/5	1	1.35	0.92	2.10	Malcolm Pirnie (2007)[d]
Horse clam	tissue	Dungeness Bay[c]	2006	8/8	1	0.12	0.09	0.14	Malcolm Pirnie (2007)[d]
Horse clam	tissue	Freshwater Bay[c]	2006	8/8	1	0.14	0.10	0.23	Malcolm Pirnie (2007)[d]
Horse clam	visceral cavity	Dungeness Bay[c]	2006	5/5	1	1.06	0.35	1.49	Malcolm Pirnie (2007)[d]
Horse clam	visceral cavity	Freshwater Bay[c]	2006	5/5	1	1.66	1.35	2.14	Malcolm Pirnie (2007)[d]
Butter clam	soft parts	Samish Island	1999	0/1	50	1.25	2.5 U	2.5 U	Ecology (2000)

Species	Tissue Type	Sampling Location	Sampling Year(s)	Detection Frequency	Average Individuals per Composite	Total PCB Concentration (µg/kg ww)			Source
						Mean[a]	Minimum	Maximum	
Butter clam	soft parts	various locations[f]	1994 – 1998	1/66	na	na	3.5[g]	3.5[g]	King County
Littleneck clam	soft parts	Padilla/Fidalgo Bay	1999	1/3	50	2.8	2.3 U	3.6	Ecology (2000)

[a] Mean concentrations were calculated using one-half the highest RL for an individual Aroclor for samples in which no Aroclors were detected.

[b] PSAMP data are from various sites around Puget Sound (Maps 7-4 and 7-6).

[c] Dungeness Bay and Freshwater Bay were the reference sites used in the Rayonier Mill RI near Port Angeles, Washington (Malcolm Pirnie 2007).

[d] The total PCB concentrations in this study were calculated as the sum of all detected PCB congeners.

[e] Data from composite hepatopancreas samples were mathematically combined with data from composite samples of edible meat to form composite samples of edible meat plus hepatopancreas. Total PCB concentrations in whole-body (i.e., edible meat plus hepatopancreas) crab were calculated assuming 69% (by weight) edible meat and 31% hepatopancreas, based on the relative weights of these tissues in a 16.6-cm Dungeness crab dissected by Windward in 2004 (unpublished data). Detection frequencies were not calculated for these samples because they do not represent individually analyzed samples.

[f] Locations include Edmonds, Point Wells, Carkeek Park, Golden Gardens, West Point, Alki Point, Vashon Island, and Normandy Park. Data for clams collected by King County were compiled from five King County reports (1995; 2000e; 2001c; 2005e; 2006d).

[g] One detected concentration was reported for a single Aroclor (1254) in a single sample. Other Aroclors were not detected, and RLs were not reported for the other samples.

J – estimated concentration

na – not available

PCB – polychlorinated biphenyl

PSAMP – Puget Sound Ambient Monitoring Program

U – not detected

ww – wet weight

Background tissue data were obtained from numerous locations around Puget Sound (Maps 7-4, 7-5, and 7-6). The primary sources of background data for fish were a PSAMP report (West et al. 2001) and a study conducted by Ecology (Era-Miller 2006). The PSAMP study area encompassed the Puget Sound region from Georgia Basin to the South Sound, including over 2,300 km^2 of aquatic habitat and over 2,100 km of shoreline (West et al. 2001) (Map 7-4). Locations throughout the study area were sampled intermittently over a 10-yr period from 1989 through 1999 such that all locations were sampled at least once, but not all locations were sampled every year. The species analyzed included English sole, copper rockfish, quillback rockfish, yelloweye rockfish, and adult salmon[90] (both chinook and coho species).

The study conducted by Ecology (Era-Miller 2006) was designed to augment the PSAMP data, and thus some of the locations were co-located with those in the PSAMP monitoring effort. The sampling locations included eight sites in south Puget Sound (Map 7-4). The species sampled during this investigation were English sole and rock sole (see Table 7-8). Limited information on two fish samples was also found in a report by Cubbage (1992) regarding a study conducted by Gahler et al. (1982). This study presented total PCB data from samples collected in Discovery Bay.

The primary source of crab and clam data was the remedial investigation (RI) for the Rayonier mill site near Port Angeles (Malcolm Pirnie 2007). This investigation included tissue sampling at Freshwater and Dungeness Bays; the locations were selected to represent background conditions for that RI (Map 7-5). Species collected in background locations for the Rayonier Mill RI included Dungeness crabs, horse clams, and geoducks.

As part of another study, conducted by Ecology (2000), shellfish tissue samples, including Dungeness crabs, butter clams, and littleneck clams, were collected and analyzed. In addition, King County analyzed shellfish samples for PCBs as part of their water quality assessment surveys (King County 1995, 2000e; 2001c, 2005e, 2006d). Out of the 66 samples collected by King County from various locations around Puget Sound (including Edmonds, Point Wells, Carkeek Park, Golden Gardens, West Point, Alki Point, Vashon Island, Normandy Park), PCBs (Aroclor 1254) were detected in only one sample.

In general, total PCB concentrations ranged from non-detect to 140.4 µg/kg ww in fish tissue samples (Table 7-8; Figure 7-4). Rock sole had the lowest total PCB concentrations, followed by copper rockfish, English sole, yelloweye rockfish, and quillback rockfish. Total PCB concentrations in whole-body Dungeness crabs ranged from 3.03 to 16.05 µg/kg ww. For all species in which both near and non-urban data were available, the non-urban samples had lower mean total PCB concentrations than

[90] Adult salmon data were not included in Table 7-5 because their exposure to chemicals in LDW sediment is not anticipated to significantly influence the concentrations in their tissues, primarily because of the very small portion of their lives spent in the LDW.

near-urban samples. Total PCB concentrations in clam tissue samples ranged from non-detect to 3.6 µg/kg ww, and thus were the lowest of the background tissue types measured.

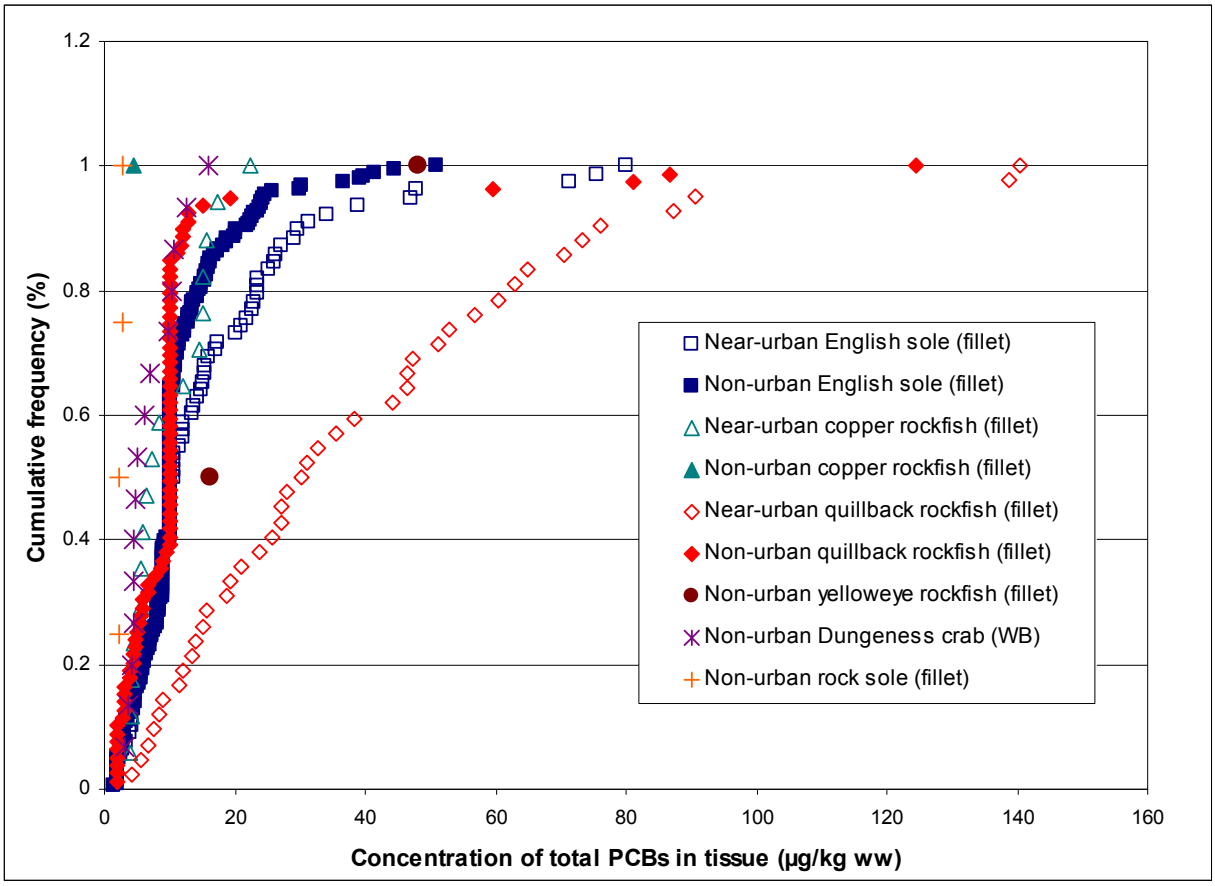


Figure 7-4. Comparison of total PCB concentrations in fish samples collected at non-urban and near-urban locations in Puget Sound

PCBs have been detected in many types of food other than seafood, such as butter, chicken, steak, eggs and baked goods (WSDOH 2006). The concentrations reported in supermarket foods were similar to those detected in background fish samples, and ranged from non-detect to 70 µg/kg.

7.3.2 Arsenic

Data were collected as part of the RI to assess background concentrations of arsenic (both total and inorganic) in tissue (Windward 2005a, b, 2006b). Samples were collected from locations both within and outside of the region thought to be impacted by aerial emissions from the former Asarco smelter in Tacoma, Washington (Area-Wide Soil Contamination Task Force 2003), as noted in Table 7-9 and shown on Map 7-7. This smelter is known have been an important source of arsenic to areas within the smelter plume and has been the focus of an extensive soil study conducted by Ecology in south King County (Ecology 2001b).

Table 7-9. Total and inorganic arsenic concentrations in clam, fish, and crab tissue composite samples collected from Puget Sound locations outside of urban areas

Area	Tissue Type	Sampling Year	Detection Frequency	Inorganic Arsenic (mg/kg ww)			Source
				Mean[a]	Min	Max	
Clams (various species)[b]							
Vashon Island (Asarco influenced)	whole-body	2005	6/6	0.143	0.0930	0.227	Windward (2006b)
Seahurst Park (Asarco influenced)	whole-body	2004	6/6	0.443	0.0980 J	0.616 J	Windward (2005a)
Dungeness NWR (non-Asarco influenced)	whole-body	2005	6/6	0.0645	0.0470	0.117	Windward (2006b)
Bainbridge Island (non-Asarco influenced)	whole-body	2004	6/6	0.201	0.0440 J	0.485 J	Windward (2005a)
Perch (shiner surfperch, pile perch, and striped perch)							
East Passage (Asarco influenced)	whole-body	2004	2/3	0.008	0.005 U	0.01 J	Windward (2005b)
Blake Island (non-Asarco influenced)	whole-body	2004	6/6	0.02	0.01	0.03	Windward (2005b)
Crabs (Dungeness crab and slender crab)							
East Passage (Asarco influenced)	edible meat	2004	6/6	0.018	0.010 J	0.040	Windward (2005b)
	hepatopancreas	2004	2/2	0.08	0.08	0.08	Windward (2005b)
	calculated whole-body[c]	2004	6/6	0.037	0.032 J	0.052	Windward (2005b)
Blake Island (non-Asarco influenced)	edible meat	2004	6/6	0.023	0.020	0.030	Windward (2005b)
	hepatopancreas	2004	2/2	0.31	0.27	0.34	Windward (2005b)
	calculated whole-body[c]	2004	6/6	0.11	0.10	0.13	Windward (2005b)
Benthic fish (English sole and starry flounder)							
East Passage (Asarco influenced)	fillet	2004	1/6	0.002	0.002 U	0.004 J	Windward (2005b)
	whole-body	2004	6/6	0.01	0.007 J	0.02	Windward (2005b)
Blake Island (non-Asarco influenced)	fillet	2004	2/6	0.003	0.002 U	0.005 U	Windward (2005b)
	whole-body	2004	6/6	0.02	0.01	0.03	Windward (2005b)

[a] Means were based on one-half RL for ND.

[b] Composite clam tissue samples from Seahurst Park and Bainbridge Island included multiple species (*Saxidomus giganteus, Clinocardium nuttallii, Macoma nasuta, Macoma secta, Tresus capax*, and *Protothaca staminea*). Composite clam tissue samples from Vashon Island and the Dungeness NWR included only *Mya arenaria*, the same species as in the LDW composite clam tissue samples.

[c] Data from composite hepatopancreas samples were mathematically combined with data from composite samples of edible meat to form composite samples of edible meat plus hepatopancreas. Arsenic concentrations in whole-body (i.e., edible meat plus hepatopancreas) crab were calculated assuming 69% (by weight) edible meat and 31% hepatopancreas, based on the relative weights of these tissues in a 16.6-cm Dungeness crab dissected by Windward in 2004 (unpublished data). Detection frequencies were not calculated for these samples because they do not represent individually analyzed samples.

NWR – National Wildlife Refuge

J – estimated concentration

U – not detected

ww – wet weight

The concentrations of inorganic arsenic are of greater interest because inorganic arsenic is more toxic and because EPA has derived a cancer slope factor for inorganic arsenic but not for total or organic arsenic. Therefore, only inorganic arsenic data are summarized. Clams were collected from background areas in both 2004 and 2005 and fish and crabs were collected in 2004 as part of the RI (Table 7-9 and Map 7-7). The only other study in Puget Sound that analyzed inorganic arsenic in tissue was conducted by Ecology (2002a), but the results from that study were not relevant because samples were only collected from urban bays.

In 2004, fish and crabs were collected from the East Passage (Asarco-influenced) and Blake Island (non-Asarco-influenced) and combined to create 61 composite tissue samples, as shown in Table 7-9 (Windward 2005b). The 2004 tissue dataset also included 12 composite clam tissue samples, 6 from a beach in Seahurst Park (Asarco-influenced) and 6 from a beach on Bainbridge Island (non-Asarco-influenced) (Windward 2005a). Because the clams collected in 2004 were not *Mya arenaria* (the type of clam predominantly found in the LDW) further sampling was conducted in 2005 to collect *Mya arenaria* from other background locations for the analysis of inorganic and total arsenic. In the 2005 sampling event, 12 additional composite *Mya arenaria* samples were formed, 6 from beaches on Vashon Island (Asarco-influenced) and 6 from the Dungeness National Wildlife Refuge (NWR) (non-Asarco-influenced) (Windward 2006).

Inorganic arsenic concentrations in clam, fish, and crab tissue samples ranged from 0.002 to 0.616 mg/kg ww. Inorganic arsenic concentrations were generally higher in clams (0.0440 to 0.616 mg/kg ww) than in the other tissue types (0.002 to 0.13 mg/kg ww). In clams, most of the inorganic arsenic concentrations in Asarco-influenced samples were higher than concentrations in non-Asarco-influenced samples, although there was some overlap in concentrations and two samples from Bainbridge Island (non-Asarco-influenced) had relatively high concentrations (Figure 7-5). No statistical analyses were conducted to evaluate whether the difference was significant.

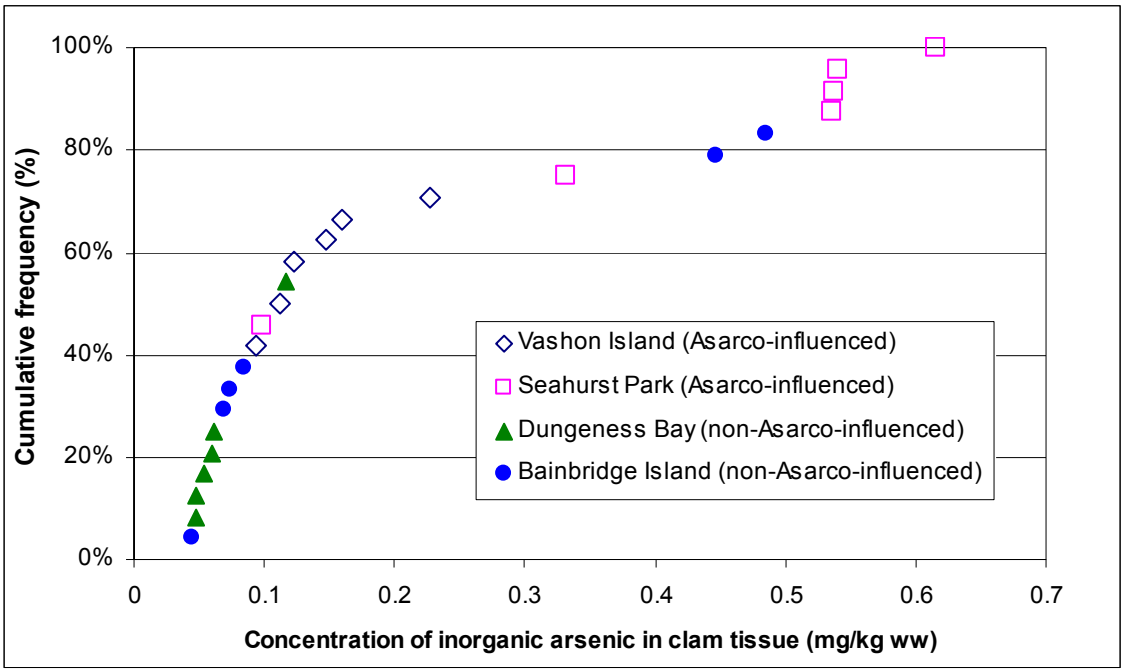


Figure 7-5. Comparison of inorganic arsenic concentrations in clam tissues collected from background areas

7.4 SUMMARY

This section presented available data for characterizing natural and urban background concentrations of selected chemicals in sediment. Concentrations of total PCBs and arsenic in tissue samples collected from the greater Puget Sound area were also presented.

Statistical analyses were conducted using MTCA guidance to determine preliminary natural background sediment concentrations for PCBs, arsenic, cPAHs, and dioxins and furans (31 µg/kg dw, 12 mg/kg dw, 77 µg/kg dw, and 0.52 ng/kg dw, respectively).

Data from various sources were also compiled to assess the range of urban background sediment concentrations. Table 7-10 summarizes the mean, median, and 90th percentile calculated using MTCAStat for each dataset that was analyzed. The range of mean values calculated from the individual datasets for arsenic was 6 mg/kg dw to 37 mg/kg dw. The low end of the range is associated with relatively coarse textured surface sediment samples collected from upstream of the site, whereas the high end of the range is associated with particulates in the water column collected from upstream of the site. Fine particles in the water column may contain higher arsenic concentrations in the Duwamish watershed as a result atmospheric deposition of arsenic-rich fine particles from the historic Asarco smelter in Tacoma. Mean arsenic urban background calculated from urban bay surface sediment data ranged from 7 to 15 mg/kg dw, with an overall mean of 11 mg/kg dw.

For PCBs, the range of mean values was 21 µg/kg dw to 135 µg/kg dw. For total PCBs, there was generally good agreement between the estimated total PCBs in the Green River and in urban bays and lakes. The mean urban background values for urban bays and lakes ranged from 40 to 163 µg/kg dw, with an overall mean of 135 µg/kg dw.

Upstream surface sediment data and core data from the Turning Basin were similar for cPAHs. The range of mean values calculated for cPAHs from the two data sources was 101 µg/kg dw to 106 µg/kg dw.

There were fewer data available for dioxins and furans. TEQs ranged from 2.59 ng/kg dw (upstream of the LDW) to 187 ng/kg dw (in Salmon Bay). Most of the TEQs were less than 30 ng/kg dw. The mean dioxin and furan TEQ for the samples that most closely represent the urban background environment in the LDW was 14.9 ng/kg ww.

Total PCB and inorganic arsenic concentrations in tissue samples collected outside of urban areas were presented to provide a context for PCB and arsenic tissue RBTCs presented in Section 8. Total PCB concentrations in fish and crab tissue ranged from non-detect to 140.4 µg/kg ww among the species. Total PCB concentrations in clams were generally lower than in fish and crab, ranging from non-detect to 3.6 µg/kg ww. Inorganic arsenic concentrations in clam tissue ranged from 0.0440 to 0.616 mg/kg ww. Inorganic arsenic concentrations in fish and crabs were generally lower (0.002 to 0.13 mg/kg ww).

Background concentrations can be used both to assess the feasibility of cleanup actions and to provide context for the tissue RBTCs that are presented in Section 8. In addition, the background concentrations in sediment may be used by EPA and Ecology to inform remedial decisions in the ROD.

Table 7-10. Range of total PCB, arsenic, and cPAH concentrations for Green River inflow and urban sediments

Study/Source	No. of Samples	Concentration			Comments (Source)
		Mean	Median	90th Percentile	
Arsenic (mg/kg dw)					
Green River inflow (upstream water quality and RM 4.0 to RM 7.0 sediment)					
King County 2001 to 2006 Green River water quality data	101	37	29	78	Normalized to TSS (Herrera 2004b, 2005; Mickelson and Williston 2006; Williston 2007 [in prep.])
LDW upstream sediment samples (RM 5.1 to RM 7.0)	20	6	5	9	MTCAStat calculation of upstream data (1998; 1998; Windward 2005d, g)
USACE Upper Turning Basin cores (RM 4.0 to RM 4.75)	50	9	8	14	MTCAStat calculation of 1989 – 2003 DAIS data (Fox 2007)
Urban bay sediment values					
Puget Sound urban bays	699	11	9	21	RETEC (2007a)
Total PCBs (µg/kg dw)					
Green River inflow (upstream water quality and RM 4.0 to RM 6.0 sediment)					
King County PCB water quality data	13	60	36	113	Upstream stations TGS/1 and 0309 normalized to TSS, re-analysis with lower detection limits (Mickelson and Williston 2006; Williston 2007 [in prep.])
LDW upstream sediment samples (RM 5.2 to RM 6.0)	20	21	6	45	MTCAStat calculation of upstream data; using events with detected data NOAA (1997; 1998)
USACE Upper Turning Basin cores RM 4.0 to RM 4.75	37	61	36	109	MTCAStat calculation of 1990 – 2003 DAIS data; (Fox 2007)
Urban bay sediment values					
Puget Sound urban embayments	587	135	35	427	(RETEC 2007a)

Study/Source	No. of Samples	Concentration			Comments (Source)
		Mean	Median	90th Percentile	
Carcinogenic PAHs (µg/kg dw)[a]					
Green River inflow (upstream water quality and RM 4.0 to RM 6.1 sediment)					
King County PCB water quality data	15	nc	nc	nc	All 15 samples were non-detect and below the MDL. 2002 and 2003 data only - (Mickelson and Williston 2006; Williston 2007 [in prep.])
LDW upstream sediment samples (RM 5.0 to RM 7.0)	16	106	56	246	MTCAStat calculation of data above RM 5.0 (1996; 1998; 1998; Windward 2005d)
USACE Upper Turning Basin cores (RM 4.0 to RM 4.75)	37	101	89	298	MTCAStat calculation of 1989 – 2003 DAIS data. Calculated excluding 1 outlier (4 x 50th = 273); only 5 PAHs included in the report (Fox 2007)
Dioxin and Furan TEQ (ng/kg dw)					
Urban bay sediment values					
Greater Seattle area	8	14.9	nc	nc	Windward (2005d)
Greater Seattle area (upstream locations)	2	2.41	nc	nc	Windward (2005d)

[a] Total cPAHs were calculated by summing the products of concentrations and compound-specific PEFs for individual cPAH compounds, as discussed in detail in Appendix E, Section E.3.

cPAH – carcinogenic polycyclic aromatic hydrocarbon
DAIS – Dredged Analysis Information System
dw – dry weight
LDW – Lower Duwamish Waterway
MDL – maximum detection limit
MTCA – Model Toxics Control Act
na – not applicable
nc – not calculated
PAH – polycyclic aromatic hydrocarbon
RI – remedial investigation
RM – river mile
TEQ – toxic equivalent
TSS – total suspended solids
USACE – US Army Corps of Engineers

Section 8 Risk-Based Threshold Concentrations

This section presents the derivation of RBTCs, which represent the concentrations that equate to specific risk thresholds. RBTCs for human health were developed for excess cancer risk thresholds of 1×10^{-4}, 1×10^{-5} and 1×10^{-6} and HQs equal to 1. These thresholds are important in both the CERCLA and MTCA programs, as discussed in the HHRA (Appendix B). RBTCs have been estimated for exposure through direct sediment contact and ingestion of seafood items. RBTCs were developed for chemicals that were identified as risk driver chemicals in either the ERA (Appendix A) or the HHRA (Appendix B). RBTCs are an important component in the derivation of preliminary remediation goals (PRGs) in the FS. The RBTCs derived in this section are not the same as cleanup levels or cleanup goals. EPA and Ecology will select final cleanup values in the ROD.

In the HHRA (Appendix B), PCBs, arsenic, cPAHs, and dioxins and furans were identified as risk driver chemicals for both seafood consumption and direct sediment contact scenarios (Table 8-1). In the ERA (Appendix A), PCBs were identified as a risk driver chemical for river otter based on risks associated with ingestion of PCBs in prey.[91] In addition, 41 chemicals were identified as risk driver chemicals for the benthic invertebrate community based on detected exceedances of SMS criteria.

Table 8-1. Summary of risk driver chemicals from the HHRA and ERA

Exposure Scenario	PCBs	Arsenic	Dioxins and Furans	cPAHs	Other Chemicals
Human health – direct sediment contact	X	X	X	X	
Human health – seafood consumption	X	X	X	X	
River otter prey ingestion	X				
Benthic invertebrate sediment exposure	X	X			X[a]

[a] In addition to arsenic and total PCBs, risk driver chemicals identified for the benthic invertebrate community, based on one or more surface sediment samples with exceedances of the SQS and/or CSL, include cadmium, chromium, copper, lead, mercury, silver, zinc, acenaphthene, anthracene, benz(a)anthracene, benzo(a)pyrene, benzo(g,h,i)perylene, chrysene, dibenzo(a,h)anthracene, fluoranthene, fluorene, indeno(1,2,3,-c,d)pyrene, naphthalene, phenanthrene, pyrene, total benzofluoranthenes, HPAH, LPAH, bis(2-ethylhexyl) phthalate, butyl benzyl phthalate, dimethyl phthalate, 1,2-dichlorobenzene, 1,4-dichlorobenzene, 1,2,4-trichlorobenzene, 2-methylnaphthalene, 4-methylphenol, 2,4-dimethylphenol, benzoic acid, benzyl alcohol, dibenzofuran, hexachlorobenzene, n-nitrosodiphenylamine, pentachlorophenol, and phenol.

cPAH – carcinogenic polycyclic aromatic hydrocarbon

CSL – cleanup screening level

[91] In the ERA, direct exposure to sediments was explicitly evaluated for fish and wildlife receptors as a component of the overall diet (i.e., incidental sediment ingestion) (Windward 2007a). No risk drivers were identified for fish or wildlife receptors based solely on direct contact with sediments because the majority of the risk is associated with ingestion of dietary items, not direct contact with sediment. Therefore, the PCB RBTC for river otter is based only on ingestion of prey items.

ERA – ecological risk assessment
HHRA – human health risk assessment
PCB – polychlorinated biphenyl
SQS – sediment quality standards

Various approaches were used to derive the RBTCs for the risk driver chemicals in Table 8-1. Mathematical equations based on risk assessment assumptions were used to derive sediment RBTCs for the human health direct sediment contact scenarios (Section 8.1) and tissue RBTCs for the seafood ingestion scenarios (human [Section 8.2] and otter [Section 8.4]). Sediment RBTCs for the benthic invertebrate community were based on SMS criteria (Section 8.5).

To develop RBTCs for sediment for seafood consumption scenarios, a relationship was needed between chemicals concentrations in tissues and sediment. Different types of models were used to estimate this relationship, depending on data type. For PCBs, a FWM was developed to relate total PCB concentrations in sediment to concentrations in fish, crabs, and clams, which were all relevant for RBTC development (Section 8.3.1). Co-located sediment and clam tissue data were used to evaluate regression models for arsenic and cPAHs to determine if the derivation of sediment RBTCs was appropriate (Sections 8.3.2 and 8.3.3).

The FWM developed for PCBs could not be used to relate arsenic concentrations in clam tissue and sediment and was not used to relate cPAHs in sediment and clam tissue. The FWM could not be used for arsenic because the Gobas-type FWM was designed to model bioaccumulation of hydrophobic organic chemicals. The FWM was not used for cPAHs because application of the model to a toxicity equivalent mixture of compounds is complicated, and would have required certain input parameters that are not readily available (such as a K_{ow} value weighted to account for differences in toxicity of the various cPAH congeners and cPAH concentrations in water). Also the FWM was not calibrated for PAH compounds.

8.1 SEDIMENT RBTCS FOR HUMAN DIRECT SEDIMENT CONTACT

Sediment RBTCs were determined for direct sediment contact scenarios evaluated in the HHRA (Appendix B). RBTCs are important in the evaluation of remedial action objectives (RAOs); RBTCs developed for RME scenarios will be used in evaluating cleanup alternatives.[92] RBTCs for non-RME direct sediment contact scenarios were also estimated to aid in risk management and risk communication, but the RBTCs for non-RME scenarios will not be used in the remedial decision-making process.

The RME scenarios for direct sediment contact evaluated in the HHRA (Appendix B) included beach play, netfishing, and tribal clamming (120 days per year). Non-RME

[92] In accordance with EPA guidance (Section 6.1.2 of EPA 1989), RME scenarios are used to formulate RAOs and evaluate cleanup alternatives at Superfund sites.

scenarios included netfishing (central tendency) and clamming scenarios with lower (i.e., 7 days per year) and higher (i.e., 183 days per year) exposure frequencies than the RME clamming scenario. An RBTC was not developed for the tribal clamming (183 days per year) scenario because that scenario represents an exposure that is greater than the RME. EPA has stated that this high-end scenario will not be used for identifying risk drivers or developing RBTCs. Key parameters used to estimate risks associated with direct sediment contact are summarized in Table 8-2. More detailed descriptions of the direct sediment contact scenarios and risk results are presented in the HHRA (Appendix B) in Sections B.3 and B.5, respectively.

Table 8-2. Summary of input parameters for direct sediment contact exposure scenarios

Direct Sediment Contact Scenario for Which RBTCs Were Derived	Incidental Sediment IR (g/day)	Exposure Frequency (days/year)	Exposure Duration (years)	Skin Surface Area Exposed (cm^2)
Netfishing RME	0.05	119	44	3,600
Beach play RME[a]	0.2	65	6	varies with age
Tribal clamming RME (120 days per year)	0.1	120	64	6,040
Netfishing CT	0.05	63	29	3,600
Clamming (7 days per year)	0.1	7	30	6,040

[a] For the beach play scenarios, the LDW was divided into eight sections to assess risks associated with different areas of the LDW. Exposure of children was evaluated from the time of birth to 6 years of age.

CT – central tendency

IR – ingestion rate

RME – reasonable maximum exposure

8.1.1 Methods for calculating sediment RBTCs for human direct sediment contact

Sediment RBTCs for direct sediment contact scenarios were derived for each of the risk driver chemicals according to Equation 8-1 (for carcinogenic effects).

Equation 8-1

$$RBTC = \frac{TR}{\left[\left[\left(\frac{ABS \times SA \times AF \times FC \times EF \times ED \times CF_1}{BW \times AT_C}\right) + \left(\frac{IR \times FC \times EF \times ED \times CF_2}{BW \times AT_C}\right)\right] \times SF\right]}$$

Where:

RBTC = risk-based threshold concentration in sediment (mg/kg dw)
TR = target risk (i.e., 1×10^{-6}, 1×10^{-5}, 1×10^{-4}) (unitless)
ABS = absorption factor (unitless)
SA = skin surface area (cm^2)
AF = adherence factor (mg/cm^2-day)
FC = fraction from contaminated site (unitless)
EF = exposure frequency (days/year)

ED	=	exposure duration (years)
CF_1	=	conversion factor, kg to mg (0.000001)
BW	=	body weight (kg)
AT_c	=	averaging time, cancer (days)
SF	=	cancer slope factor $(mg/kg\text{-}day)^{-1}$
AT_n	=	averaging time, non-cancer (days)
IR	=	incidental sediment ingestion rate (g/day)
CF_2	=	conversion factor, kg to g (0.001)

RBTCs were not estimated for non-cancer hazards for direct contact scenarios because none of the RME scenarios had HQs for an individual chemical greater than 1 or generated endpoint-specific HIs in excess of 1 (Appendix B, Section B.5.6).
Equation 8-1 is simply a rearrangement of the equation used in the HHRA to estimate excess cancer risk. In this case, rather than inserting a maximum or 95th percentile UCL sediment concentration into the equation and solving for risk, as was done in the HHRA (Appendix B), the target risk is set and the equation is solved for a corresponding sediment concentration.

8.1.2 Results of sediment RBTC calculations for human direct sediment contact

Sediment RBTCs for the direct sediment contact scenarios are shown in Table 8-3. The RBTCs for the tribal clamming RME scenario were lower than RBTCs for the other two RME scenarios at a given target risk level, except for cPAHs. Beach play RBTCs were lower than clamming or netfishing RBTCs for cPAHs at a given target risk level. Considerations relevant to the application of these RBTCs are discussed in Section 8.6.

Table 8-3. Sediment RBTCs for direct sediment contact human health scenarios

Risk Driver	Target Risk	Netfishing RME	Beach Play RME	Tribal Clamming RME	Netfishing CT	Clamming (7 days/ year)
Arsenic (mg/kg dw)	1×10^{-6}	3.7	2.8	1.3	11	43
	1×10^{-5}	37	28	13	110	430
	1×10^{-4}	370	280	130	1,100	4,300
cPAHs[a] (µg/kg dw)	1×10^{-6}	380	90	150	1,100	4,700
	1×10^{-5}	3,800	900	1,500	11,000	47,000
	1×10^{-4}	38,000	9,000	15,000	110,000	470,000
Dioxin/furan TEQ (ng/kg dw)	1×10^{-6}	37	28	13	110	220
	1×10^{-5}	370	280	130	1,100	2,200
	1×10^{-4}	3,700	2,800	1,300	11,000	22,000
Total PCBs (µg/kg dw)	1×10^{-6}	1,300	1,700	500	9,500	16,000
	1×10^{-5}	13,000	17,000	5,000	95,000	160,000
	1×10^{-4}	130,000	170,000	50,000	950,000	1,600,000

[a] cPAH concentrations are given in terms of benzo(a)pyrene equivalents.

cPAH – carcinogenic PAH
CT – central tendency
dw – dry weight
PCB – polychlorinated biphenyl
RBTC – risk-based threshold concentration
RME – reasonable maximum exposure
TEQ – toxic equivalent

8.2 TISSUE RBTCS FOR HUMAN SEAFOOD CONSUMPTION

Several different human seafood consumption scenarios were evaluated in the HHRA (Appendix B). EPA's tribal framework for conducting seafood consumption risk assessments states that the tribal scenarios based on Tulalip data are the most appropriate scenarios for representing RME exposure in the LDW (EPA 2005a). Therefore, RBTCs associated with the tribal adult and child RME seafood consumption scenarios based on Tulalip consumption data will be used in the evaluation of cleanup alternatives based on both the tribal framework document (EPA 2005a) and more recent communication from EPA (2007b). Because it provides another RME estimate, the API RME seafood consumption scenario may also be used in the evaluation of cleanup alternatives.

In addition to the RBTCs for the RME scenarios, RBTCs were developed for the other seafood consumption scenarios to aid in risk management and risk communication efforts. Thus, RBTCs were derived for adult and child tribal CT (Tulalip data), API CT, and adult one-meal-per-month scenarios.

As described in the HHRA (Appendix B), the tribal seafood consumption scenario based on Suquamish data was included to "assist in characterizing the range of seafood consumption risks"(EPA 2006b). RBTCs were not derived for the Suquamish scenario because EPA stated that the results from the consumption scenario based on Suquamish data would not be used to develop a list of risk drivers or associated RBTCs.

Key assumptions for each of the seafood consumption scenarios are summarized in Table 8-4. More detailed descriptions of the seafood consumption scenarios and risk results are presented in the HHRA (Appendix B), in Sections B.3 and B.5, respectively.

Table 8-4. Summary of key assumptions in the seafood consumption scenarios

Seafood Consumption Scenario for which RBTCs Were Derived	Ingestion Rate (g/day)					Exposure Duration (years)
	Pelagic Fish	Benthic Fish	Crabs	Clams[a]	Total	
Adult tribal RME (Tulalip data)	8.1	7.5	43.4	38.5	97.5	70
Child tribal RME (Tulalip data)	3.2	3.0	17.4	15.4	39.0	6
API RME	4.9	2.4	10.6	33.6	51.5	30
Adult tribal CT (Tulalip data)	1.3	1.2	6.6	5.9	15.0	30

Seafood Consumption Scenario for which RBTCs Were Derived	Ingestion Rate (g/day)					Exposure Duration (years)
	Pelagic Fish	Benthic Fish	Crabs	Clams[a]	Total	
Child tribal CT (Tulalip data)	0.52	0.48	2.64	2.34	6.0	6
API CT	0.51	0.24	1.1	3.47	5.3	12
Adult one meal per month[b]	7.5	7.5	7.5	7.5	na	30

[a] Clams represent both clam and mussel consumption categories.

[b] Adult one-meal-per-month consumption was evaluated for individual seafood categories independently to reflect different seafood consumption practices.

API – Asian and Pacific Islander

CT – central tendency

na – not applicable

RME – reasonable maximum exposure

Tissue RBTCs were estimated for total PCBs, arsenic, and cPAHs according to the methods described in Section 8.2.1. The primary use of these RBTCs will be to evaluate post-cleanup monitoring data for risk driver chemicals in seafood tissue. It should be noted that sediment RBTCs were not derived directly from tissue RBTCs; sediment RBTCs were derived as described in Section 8.3. Tissue RBTCs were not estimated for dioxins and furans because tissue data for these chemicals have not been collected from the LDW.

8.2.1 Methods for calculating tissue RBTCs for human seafood consumption

Tissue RBTCs associated with the human seafood consumption scenarios were estimated for total PCBs, arsenic, and cPAHs using Equation 8-2 (for carcinogenic effects) or Equation 8-3 (for non-carcinogenic effects; PCBs and arsenic only). The values for these parameters are presented in the HHRA (Section 3, Appendix B).

$$\text{Tissue RBTC} = \frac{TR}{\left[\left[\frac{IR \times FC \times EF \times ED \times CF}{BW \times AT_C}\right] \times SF\right]}$$ **Equation 8-2**

$$\text{Tissue RBTC} = \frac{THQ}{\left[\left[\frac{IR \times FC \times EF \times ED \times CF}{BW \times AT_n}\right] \times \frac{1}{RfD}\right]}$$ **Equation 8-3**

Where:

RBTC = risk-based threshold concentration in seafood items (mg/kg ww)
TR = target risk (i.e., 1×10^{-6}, 1×10^{-5}, 1×10^{-4}) (unitless)
IR = ingestion rate (g/day)
FC = fraction from contaminated site (unitless)

EF	=	exposure frequency (days/year)
ED	=	exposure duration (years)
CF	=	conversion factor, kg to g (0.001)
BW	=	body weight (kg)
AT_c	=	averaging time, cancer (days)
SF	=	cancer slope factor $(mg/kg\text{-}day)^{-1}$
THQ	=	target hazard quotient (1) (unitless)
AT_n	=	averaging time, non-cancer (days)
RfD	=	reference dose (mg/kg-day)

RBTCs based on carcinogenic risks were estimated for three risk threshold levels, 1×10^{-6}, 1×10^{-5}, and 1×10^{-4}; for non-cancer hazards, the RBTCs corresponding to an HQ of 1 were estimated.

8.2.2 Results of tissue RBTC calculations for human seafood consumption

Tissue RBTCs corresponding to the various seafood consumption scenarios are presented in Table 8-5. Tissue RBTCs for the seafood consumption scenarios represent the ingestion-weighted average concentration in tissue that results in a certain risk threshold for each scenario. For example, the RBTC for total PCBs for the adult tribal RME seafood consumption scenario based on Tulalip data was 4.2 µg/kg ww at the 1×10^{-5} excess cancer risk level. Thus, consumption of 97.5 g/day (adult tribal RME daily ingestion rate based on Tulalip data) of any tissue type with a total PCB concentration of 4.2 µg/kg ww for 70 years would result in a 1×10^{-5} excess cancer risk. Consumption of numerous types of seafood, such as crabs, clams, and fish (as specified in the adult tribal RME exposure parameters based on Tulalip data) would also result in a 1×10^{-5} excess cancer risk as long as the ingestion-weighted average of the various tissue concentrations consumed was 4.2 µg/kg ww. For the three risk driver chemicals evaluated, the tissue RBTCs for the adult tribal RME scenario based on Tulalip data were lower than all other scenarios for a given risk threshold (Table 8-5).

Table 8-5. Tissue RBTCs estimated from HHRA seafood consumption scenarios

Chemical	Seafood Consumption Scenario	Unit	Tissue RBTC[a]			
			Excess Cancer Risk			Non-Cancer Hazard
			1×10^{-6}	1×10^{-5}	1×10^{-4}	HQ = 1
Arsenic (inorganic)	Adult tribal RME (Tulalip data)	mg/kg ww	0.00056	0.0056	0.056	0.25
	Child tribal RME (Tulalip data)	mg/kg ww	0.0030	0.030	0.30	0.12
	API RME	mg/kg ww	0.0019	0.019	0.19	0.37
	Adult tribal CT (Tulalip data)[b]	mg/kg ww	0.0085	0.085	0.85	1.6
	Child tribal CT (Tulalip data)[b]	mg/kg ww	0.020	0.20	2.0	0.76
	API CT[b]	mg/kg ww	0.046	0.46	4.6	3.6

Chemical	Seafood Consumption Scenario	Unit	Tissue RBTC[a]			
			Excess Cancer Risk			Non-Cancer Hazard
			1×10^{-6}	1×10^{-5}	1×10^{-4}	HQ = 1
	Adult one-meal-per-month[b, c]	mg/kg ww	0.015	0.15	1.5	2.9
Total PCBs	Adult tribal RME (Tulalip data)	µg/kg ww	0.42	4.2	42	17
	Child tribal RME (Tulalip data)	µg/kg ww	2.3	23	230	7.8
	API RME	µg/kg ww	1.4	14	140	24
	Adult tribal CT (Tulalip data)[b]	µg/kg ww	6.4	64	640	110
	Child tribal CT (Tulalip data)[b]	µg/kg ww	15	150	1,500	51
	API CT[b]	µg/kg ww	35	350	3,500	240
	Adult one-meal-per-month[b, c]	µg/kg ww	11	110	1,100	190
cPAHs[d]	Adult tribal RME (Tulalip data)	µg/kg ww	0.11	1.1	11	na
	Child tribal RME (Tulalip data)	µg/kg ww	0.12	1.2	12	na
	API RME	µg/kg ww	0.39	3.9	39	na
	Adult tribal CT (Tulalip data)[b]	µg/kg ww	1.7	17	174	na
	Child tribal CT (Tulalip data)[b]	µg/kg ww	0.76	7.6	76	na
	API CT[b]	µg/kg ww	9.5	95	950	na
	Adult one-meal-per-month[b, c]	µg/kg ww	3.1	31	310	na

[a] Tissue RBTCs represent ingestion-weighted average concentrations. Individual tissue concentrations may be higher or lower.

[b] The RBTC for this scenario is presented for information purposes only, and will not be used in remedial decision-making.

[c] The adult one-meal-per-month value represents the tissue RBTC corresponding to consumption of any type of seafood at a rate of one meal per month (7.5 g/day).

[d] cPAH concentrations are given in terms of benzo(a)pyrene equivalents.

API – Asian and Pacific Islander

cPAH – carcinogenic polycyclic aromatic hydrocarbon

CT – central tendency

HQ – hazard quotient

PCB – polychlorinated biphenyl

RBTC – risk-based threshold concentration

RME – reasonable maximum exposure

ww – wet weight

8.3 Sediment RBTCs for Human Seafood Consumption

Methods for deriving sediment RBTCs for human seafood consumption varied by risk driver. For PCBs, a mechanistic FWM was developed (Section 8.3.1). Such a model is not appropriate for arsenic and cPAHs, so regression models were evaluated for those risk drivers (Sections 8.3.2 and 8.3.3, respectively).

Dioxins and furans were identified as a risk driver for human seafood consumption based on an assumption of unacceptable risk, although no tissue data were collected. Consequently, sediment RBTCs for the seafood consumption pathway for dioxins and furans could not be estimated (sediment RBTCs for dioxins and furans based on direct contact were derived in Section 8.1). Remedial decision making for dioxins and furans by EPA and Ecology will be based on appropriate guidance and regulations, including those specifically related to the role of background levels in the CERCLA and MTCA processes (EPA 2002a; Ecology 2001a).

8.3.1 PCB sediment RBTCs for human seafood consumption

8.3.1.1 Methods for calculating total PCB sediment RBTCs for human seafood consumption

For PCBs, the FWM was used to estimate sediment RBTCs. Information regarding the specific input parameters, calibration, and performance of the FWM is presented in Appendix D (Section D.8), which describes how the FWM was used to derive sediment RBTCs. A range of sediment RBTCs for total PCBs was estimated for each seafood exposure scenario. The range of RBTCs reflects the uncertainty in the model bounded by the model performance criterion, which required that the estimated concentrations in tissues were within a factor of 2 of the empirical data for all of the target species. In addition, a best-fit RBTC was estimated, which was calculated using the FWM parameter set that most closely fit the empirical PCB tissue data for all species combined.

8.3.1.2 Results of total PCB sediment RBTC calculations for human seafood consumption

RBTCs were identified as the total PCB concentrations in sediment corresponding to a range of excess cancer risk thresholds (1×10^{-4}, 1×10^{-5}, and 1×10^{-6}) for each of the HHRA scenarios evaluated (Table 8-6). Sediment RBTCs could not be calculated at the 1×10^{-5} or 1×10^{-6} target risk levels for either the adult or child tribal RME scenarios based on Tulalip data or for the API RME scenario because the low water concentrations assumed in the model (0.6 to 1.2 ng/L; see Appendix D, Section D.8) resulted in estimated tissue concentrations greater than many of the tissue PCB RBTCs in Table 8-6 even in the absence of any contribution from sediment. At the 1×10^{-4} target risk level, sediment RBTCs ranged from 8.4 to 33 µg/kg dw for the adult tribal RME scenario based on Tulalip data and from 173 to 380 µg/kg dw for the child tribal RME scenario based on Tulalip data (Table 8-6).

Table 8-6. Sediment RBTCs for total PCBs based on excess cancer risks in HHRA seafood consumption scenarios

Excess Risk Level	Seafood Consumption Scenario	Total PCB Sediment RBTC (µg/kg dw)		
		Lower bound	Best Fit	Upper bound
1×10^{-4}	Adult tribal RME (Tulalip data)	8.4	19	33
	Child tribal RME (Tulalip data)	173	320	380
	API RME	100	180	220
	Adult tribal CT (Tulalip data)[a]	> 380	> 380	> 380
	Child tribal CT (Tulalip data)[a]	> 380	> 380	> 380
	API CT[a]	> 380	> 380	> 380
	Adult one meal per month of benthic fish[a]	280	> 380	> 380
	Adult one meal per month of pelagic fish[a]	230	> 380	> 380
	Adult one meal per month of crabs[a]	> 380	> 380	> 380
	Adult one meal per month of clams[a]	> 380	> 380	> 380
1×10^{-5}	Adult tribal RME (Tulalip data)	nc	nc	nc
	Child tribal RME (Tulalip data)	nc	nc	nc
	API RME	nc	nc	nc
	Adult tribal CT (Tulalip data)[a]	26	53	72
	Child tribal CT (Tulalip data)[a]	100	190	230
	API CT[a]	>380	> 380	> 380
	Adult one meal per month of benthic fish[a]	7	20	24
	Adult one meal per month of pelagic fish[a]	na	na	na
	Adult one meal per month of crabs[a]	260	> 380	> 380
	Adult one meal per month of clams[a]	340	> 380	> 380
1×10^{-6}	Adult tribal RME (Tulalip data)	nc	nc	nc
	Child tribal RME (Tulalip data)	nc	nc	nc
	API RME	nc	nc	nc
	Adult tribal CT (Tulalip data)[a]	nc	nc	nc
	Child tribal CT (Tulalip data)[a]	nc	nc	nc
	API CT[a]	12	25	39
	Adult one meal per month of benthic fish[a]	nc	nc	nc
	Adult one meal per month of pelagic fish[a]	nc	nc	nc
	Adult one meal per month of crabs[a]	nc	nc	17
	Adult one meal per month of clams[a]	nc	nc	2.6

[a] The RBTC for this scenario is presented for information only, and will not be used in remedial decision-making.

API – Asian and Pacific Islander

CT – central tendency

dw – dry weight

HHRA – human health risk assessment

nc – not calculated (sediment RBTC could not be calculated because contribution from water led to predicted tissue concentrations greater than the applicable risk level, even in the absence of any contribution from sediment.)

PCB – polychlorinated biphenyl

RBTC – risk-based threshold concentration

RME – reasonable maximum exposure

RBTCs for some of the other seafood consumption scenarios, particularly the one-meal-per-month scenarios, are shown as > 380 µg/kg dw in Table 8-6. A sediment RBTC of > 380 µg/kg dw indicates that even under current conditions in the LDW (i.e., an LDW-wide SWAC of 380 µg/kg dw), risks are already estimated to be less than the target risk level.

Sediment RBTCs for total PCBs corresponding to non-cancer hazards with a target HQ of 1 were also estimated (Table 8-7). Sediment RBTCs could not be calculated for the adult and child tribal RME scenarios based on Tulalip data because the low water concentrations assumed in the model (0.6 to 1.2 ng/L; see Appendix D, Section D.8) resulted in estimated tissue concentrations that corresponded to an HQ of greater than 1 even in the absence of any sediment contribution. Best-fit RBTCs for the other scenarios ranged from not calculable to > 380 µg/kg dw (Table 8-7).

Table 8-7. Sediment RBTCs for total PCBs based on non-cancer hazard quotient of 1 in the HHRA seafood consumption scenarios

Scenario	Sediment RBTCs for Total PCBs (µg/kg dw)		
	Lower Bound	Best Fit	Upper Bound
Adult tribal RME (Tulalip data)	nc	nc	nc
Child tribal RME (Tulalip data)	nc	nc	nc
Adult tribal CT (Tulalip data)[a]	67	100	140
Child tribal CT (Tulalip data)[a]	16	34	51
API RME[a]	nc	nc	nc
API CT[a]	180	340	> 380
Adult one meal per month of benthic fish[a]	31	61	68
Adult one meal per month of pelagic fish[a]	17	41	63
Adult one meal per month of crabs[a]	> 380	> 380	> 380
Adult one meal per month of clams[a]	> 380	> 380	> 380

[a] The RBTC for this scenario is presented for information only, and will not be used in remedial decision-making.

API – Asian and Pacific Islander

CT – central tendency

dw – dry weight

HHRA – human health risk assessment

nc – not calculated (sediment RBTC could not be calculated because contribution from water led to predicted tissue concentrations greater than the applicable risk level, even in the absence of any contribution from sediment)

PCB – polychlorinated biphenyl

RBTC – risk-based threshold concentration

RME – reasonable maximum exposure

8.3.2 Arsenic sediment RBTCs for human seafood consumption

As discussed in the HHRA (Appendix B), 96% of inorganic arsenic exposure, and hence, 96% of arsenic risk associated with seafood consumption is from consumption of clams. Consequently, the relationship between inorganic arsenic concentrations in clams and sediment is of considerable importance. Tissue chemistry data were collected in 2004 for the Eastern soft-shell clam (*Mya arenaria*). This species was selected for sampling because it is the most abundant clam in the LDW within the size range of interest. Also, *M. arenaria* can tolerate lower salinity environments better than other native species, such as *Macoma*, which have very low abundance in the LDW. *M. arenaria* are filter feeders, like many other commercially important Puget Sound clam species, such as butter clams (*Saxidomus giganteus*) and Manila clams (*Tapes philippinarum*). *M. arenaria* can process large quantities of water (4 to 6 L/hr) while feeding on suspended particulates in the water column (Brown et al. 1994).

Given the feeding behavior of *M. arenaria*, water and suspended material in the water are likely to be important sources of bioaccumulated arsenic (Abraham and Dillon 1986). Arsenic was analyzed in hundreds of surface water samples in Elliott Bay, both East and West Waterways, and in the LDW as part of the King County WQA conducted from 1996 to 1997 (King County 1999e). In general, arsenic concentrations were lower in upstream portions of the LDW compared to the more marine waters of the East and West Waterways and Elliott Bay (Table 8-8). The downstream portions of the LDW had arsenic concentrations in water that were similar to, but on average slightly lower than Elliott Bay; arsenic concentrations in the water at RM 5.0 were similar to those in water collected from RM 10.0.

Table 8-8. Arsenic concentrations in water collected from the LDW and surrounding water bodies

Location	Dissolved Arsenic (µg/L)		Total Arsenic (µg/L)	
	Range	Mean	Range	Mean
Elliott Bay				
North end of waterfront	0.79 – 1.5	1.1	0.27 – 1.5	1.1
Outer Elliott Bay	0.96 – 1.5	1.2	0.19 –1.5	1.2
South end of waterfront	0.68 – 1.5	1.2	0.16 – 1.5	1.1
East and West Waterway				
West Waterway	0.30 – 1.5	0.98	0.30 – 1.5	0.97
East Waterway	0.51 – 1.4	1.1	0.29 – 1.5	1.1
LDW				
RM 1	0.24 – 1.5	0.86	0.29 – 1.5	0.93
RM 2	0.22 – 1.4	0.81	0.28 – 1.6	0.88
RM 5	0.18 – 0.46	0.32	0.18 – 0.87	0.49
Upstream of LDW				

Location	Dissolved Arsenic (µg/L)		Total Arsenic (µg/L)	
	Range	Mean	Range	Mean
Tukwila (~ RM 10)	0.15 – 0.44	0.33	0.22 – 0.74	0.47

Source: King County (1999e)

LDW – Lower Duwamish Waterway

RM – river mile

There is no clear relationship between the arsenic concentration in water and the arsenic concentration in clam tissue samples collected throughout the LDW (i.e., total arsenic concentrations in clams were similar throughout the waterway except for samples collected in the embayment between RM 1.4 and 1.5 and in Slip 4, where concentrations were higher in clam tissue). Thus, although filter-feeding clams are likely exposed to arsenic in water (Abraham and Dillon 1986), it is not clear how the arsenic concentration in water contributes to the arsenic concentration in LDW clam tissue.

Co-located sediment samples were collected in 2004 with the *M. arenaria* clam samples (Windward 2005a). The sediment samples provided information on the bedded sediment where the clams were found, but other studies suggest that the relationship of the arsenic concentrations in these sediment samples to the arsenic concentrations in the clams is probably tenuous.

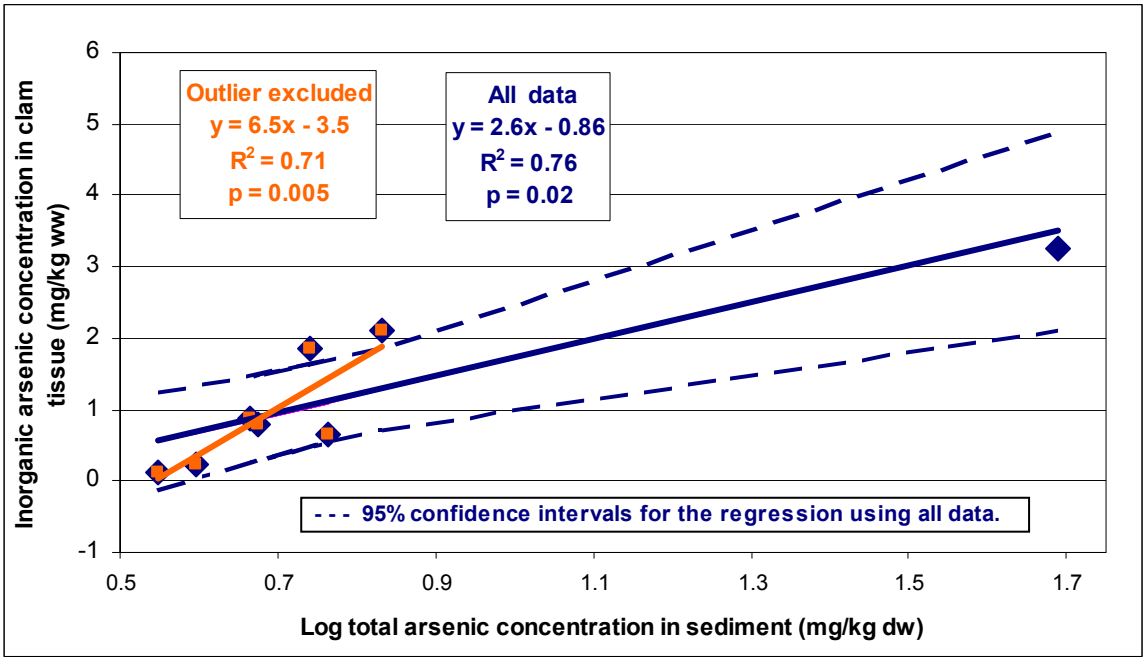
Foster et al. (1987) conducted an experiment using sediment from Chesapeake Bay to evaluate the bioaccumulation of sediment-sorbed chemicals in a filter-feeding clam (*M. arenaria*) and a deposit-feeding clam (*Macoma balthica*). The *Macoma* clams bioaccumulated the chemicals (naphthalene, diphenyl ether, di-n-octyl phthalate, 4,4′-DDT, and chrysene) at measurable rates over the 12-day exposure period. *Mya* did not show bioaccumulation of any of the sediment chemicals (Foster et al. 1987). These data suggest that a significant relationship between chemical concentrations in bedded sediment and *M. arenaria* in the LDW may be unlikely. Nevertheless, the relationship between arsenic in clams and sediment was evaluated using a regression model, as discussed below.

As previously stated, the FWM, which incorporates various exposure pathways, cannot be used to estimate sediment RBTCs for arsenic. Therefore, the utility of a simpler model (a regression model) was evaluated. Because *M. arenaria* are filter feeders, the appropriate areal extent of the source of suspended particulates is likely to be greater than the single co-located composite sediment sample that was collected with the clams. Thus, the relationship between the concentrations of inorganic arsenic in clam tissue and total arsenic in sediment was evaluated in two ways. First, the arsenic concentrations in individual co-located sediment samples were compared to arsenic concentrations in clam tissues. Second, the spatially weighted average arsenic concentrations in sediment from larger areas surrounding the intertidal beach areas where clams were collected were also compared to the concentrations in clam tissue.

For the regression, inorganic arsenic tissue concentrations in clams, rather than total arsenic tissue concentrations, were used to evaluate the relationship between concentrations in tissue and sediment because the excess risk estimates for seafood consumption were based on the toxicity of inorganic arsenic. The regression was based on available co-located sediment and clam tissue data from the LDW. The locations of the co-located clam tissue and sediment samples in the LDW are presented in Map 4-29.

When developing a regression model, certain assumptions must be met for summary statistics (regression coefficients, R^2) and significance results (p values) to be interpreted exactly. These assumptions include: 1) the relationship between the two variables is linear, 2) the residuals around the model (the distribution of the empirical Y data around the regression line) are normally distributed, and 3) variance of the residuals around the model is homogeneous across the range of predicted values. The assumption requiring normality of the residuals is often relaxed as long as the variance of the residuals is homogeneous. Quantitative tests may also be performed to determine the effect of individual data pairs on the regression coefficients (e.g., "Cook's Distance"). When any of the assumptions of a regression are not met, the validity of a linear model should be investigated and inference about regression results should be drawn with care.

Simple visual diagnostics are often adequate for determining if the assumptions are met, and, if they do not appear to be, whether transformation of one or both variables could help the data to meet test assumptions. The arithmetic arsenic data did meet the assumption of residual homogeneity of variance and the distance between the highest data point and the majority of the data was approximately an order of magnitude. Both of these characteristics of the data suggested that a log-transformation would help to linearize the sediment-tissue relationships and equalize variance of the residuals along the line. After a log-transformation was applied, a significant positive relationship ($R^2 = 0.76$, $p = 0.005$) was found between the eight inorganic arsenic concentrations in LDW clam tissue and the total arsenic concentrations in the eight co-located sediment samples (Figure 8-1). However, the regression residuals were only approximately normally distributed (as evidenced by a fit-R^2 of regression residuals vs. expected normal scores = 0.82) and several outlier diagnostics indicated that the highest data pair had a large effect on the regression equation. For example, a perfect fit would be a fit-$R^2 = 1.0$, and MTCAStat (Ecology 2007l) requires a fit-R^2 of 0.90 in order to consider data normally or log-normally distributed.



Note: Total arsenic concentrations are presented as log values. The range of total arsenic concentrations in sediment at co-located clam sampling locations in the LDW was between 3.5 and 49 mg/kg dw.

Figure 8-1. Regression of inorganic arsenic concentrations in LDW clam tissue relative to total arsenic concentrations in co-located sediment using arithmetic tissue and log[sediment] data

The highest data point can be considered an outlier in terms of its sediment concentration (Leverage value = 0.82) and its tissue concentration (Studentized deleted residual = -4.18, p = 0.006). It can also be considered "influential" in its effect on the predicted tissue concentration at its sediment concentration (DFFit = -3.95), its effect on the estimate of the intercept (DFbeta0 = 2.67), its effect on the estimate of the slope (DFbeta1 = 3.9), and its overall influence on the regression coefficients (Cook's Distance = 25.36, p = 0.002). The distance of the highest data point from the next highest sediment and tissue concentrations, and its strong influence on the slope and intercept of the regression equation are much greater than any other point in the data set and could indicate that the co-located sediment data may not reflect the area of clam exposure, or that an important explanatory variable has not been included in the model (e.g., grain size).

When the highest data point was removed, the relationship between the other seven points remained positive and significant (R^2 = 0.71, p = 0.02). The slope of the regression equation was much higher than the slope calculated including the highest point. The difference between the slopes of the two regressions has a large effect on the estimated concentration in clam tissue at the highest co-located sediment concentration of 49 mg/kg dw. If the regression using all data were applied, the estimated tissue concentration would be 3.5 mg/kg ww, while the regression

excluding the high value would estimate a tissue concentration of 7.5 mg/kg ww. While tissue concentrations at lower sediment concentrations might be estimated more accurately when using the regression without the highest data point, tissue concentrations at higher sediment concentrations might be greatly overestimated. Another problem with excluding the highest data point is that doing so would limit the range across which the regression could be applied (from 3.5 and 6.8 mg/kg dw) because extrapolation beyond the range of data used to create a regression is not valid.

The width of the 95% confidence intervals calculated for the arsenic regression affects the confidence in the back calculation of risk-based sediment concentrations from the tissue RBTC. Ideally, the confidence intervals shown on Figure 8-1 could be used to estimate a range of sediment RBTCs at a given tissue RBTC. However, because the sample size is small and the highest sediment concentration is an order of magnitude greater than the mean sediment concentration (sediment concentrations are skewed), the width of the confidence intervals is quite large, especially at higher sediment concentrations. These wide confidence limits, along with the relatively shallow slope of the relationship using all the data, would lead to high uncertainty in back calculations of sediment concentrations from risk-based tissue concentrations.

The relationship between inorganic arsenic concentrations in clam tissue and total arsenic concentrations in sediment was also evaluated on a larger spatial scale by comparing the arsenic concentrations in sediment from broader intertidal areas surrounding the locations where clams were collected. SWACs were calculated based on both the entire extent of the clam intertidal area and an additional area that includes subtidal habitat, as shown on Map B.6-11. The SWACs were regressed against the clam tissue data and the regression statistics (not shown) were similar for both these approaches to the statistics calculated for the co-located samples. In addition, similar regression analyses were conducted using total arsenic concentrations in sediment normalized to percent fines, based on the premise that fine-grained particles would be more likely to be suspended in the water column and potentially consumed by clams. The resulting regression statistics (not shown) were also similar to those for the co-located samples. Overall, these data explorations illustrate the uncertainty in the relationship between arsenic concentrations in clams and arsenic concentrations in various sediment exposure areas.

Thus, using data from either a small scale (co-located composite samples) or a larger scale (intertidal areas), neither arithmetic nor log transformation was able to achieve a truly linear relationship with equal-variance residuals. The lack of homogeneity was caused by a single data point with a sediment concentration more than an order or magnitude greater than the next highest sediment concentration. Because all the data were collected using the same protocol, there is no substantive reason to exclude the high sediment-tissue pair from the data set, and because data were collected randomly, the high data pair provides evidence that other areas with high arsenic

concentrations may exist, even if they are less frequent and cover less area than areas with low concentrations.

Although the sample size was small, the uncertainty found in the relationship between total arsenic in sediment and inorganic arsenic in clam tissue is consistent with the water-driven exposure pathway reported for this clam species (Foster et al. 1987). Without a suitable model to estimate the sediment concentrations that correspond to the tissue RBTCs for inorganic arsenic, arsenic sediment RBTCs for the seafood consumption scenarios were not calculated.

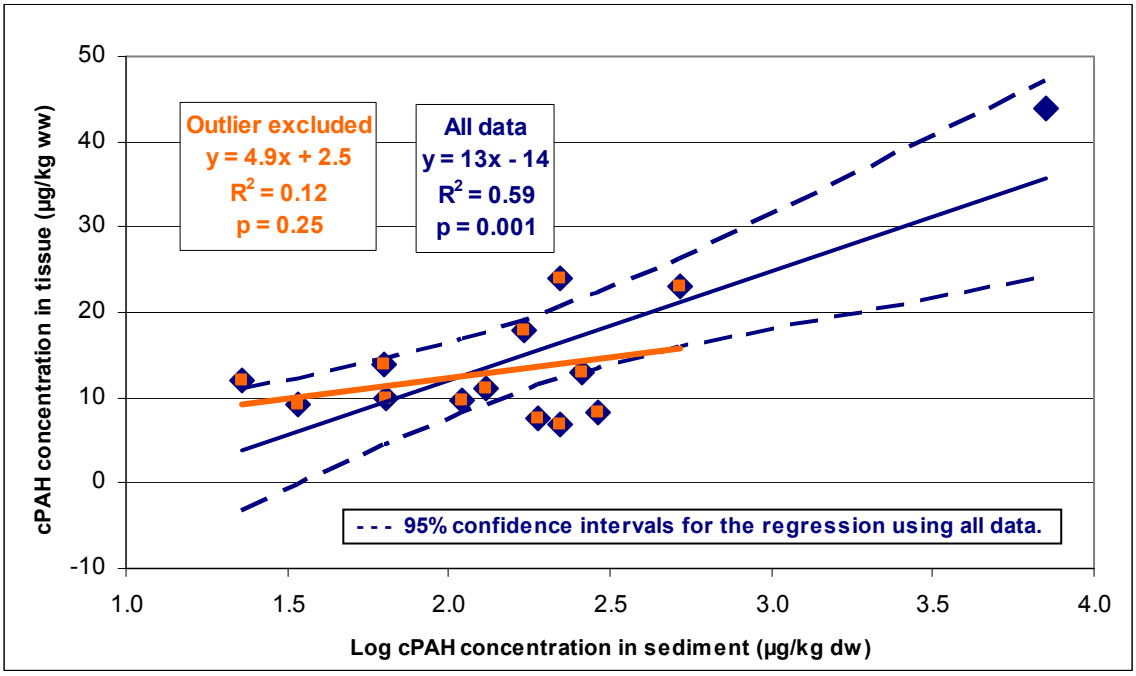
8.3.3 cPAH sediment RBTCs for human seafood consumption

As discussed in the HHRA (Appendix B), 95% of cPAH exposure, and hence, 95% of cPAH risk associated with seafood consumption is from consumption of clams. As discussed in Section 8.3.2, given the feeding behavior of *M. arenaria*, water and suspended particulates in the water are likely to be important sources of bioaccumulated chemicals for this species (Abraham and Dillon 1986). Foster et al (1987) did not report uptake of PAHs by *M. arenaria* in their study, where the clams "appeared to remain isolated from exposure to contaminated sediment." Nonetheless, the clam tissue/sediment relationship was evaluated to assess the potential to calculate sediment RBTCs for cPAHs associated with seafood consumption in the LDW using site-specific data.

Although the FWM (Appendix D) was designed to mechanistically model the relationship between concentrations of hydrophobic organic chemicals in sediment and tissue, it was not used to model cPAHs in tissue as discussed previously. Instead, various regression models were investigated in an attempt to relate site-specific concentrations of cPAHs in composite clam tissue samples and co-located or nearby sediment samples. Multiple spatial scales were evaluated. The performance of the regression model described below was not improved using sediment concentrations averaged over larger spatial areas (data not shown).

The co-located cPAH data (presented on Map 4-36) were assessed to determine whether the basic assumptions for a regression (such as a linear relationship between the variables and a normal distribution of the residuals) were met. The data did not meet the basic assumptions for a regression. Similar to the case with arsenic, the cPAH dataset had one point that was clearly higher than the rest of the data.

After the sediment data were log-transformed, a linear regression was significant ($p = 0.001$) and explained 59% of the variance in tissue concentrations (Figure 8-2). Although the residuals were normally distributed (fit-R^2 of residuals against expected normal scores = 0.95), the variance of the residuals increased with concentration and the pattern of the residuals was not uniform along the range of sediment and tissue concentrations.



Note: cPAH concentrations are presented as log values. The range of cPAH concentrations in sediment at co-located clam sampling locations in the LDW was between 23 and 7,100 µg/kg dw.

Figure 8-2. Regression of cPAH concentrations in LDW clam tissue relative to concentrations in co-located sediment using arithmetic tissue data and log[sediment] data

Several outlier diagnostics indicated that the highest sediment-tissue pair had a strong effect on the regression equation. The high data point can be considered an outlier in terms of its sediment concentration (Leverage value = 0.56) and its tissue concentration (Studentized deleted residual = 22.67, $p < 0.0005$). It can also be considered "influential" in its effect on the estimated tissue concentration at its sediment concentration (DFFit = 14.39), its effect on the estimate of the intercept (DFbeta0 = 16.04), and its effect on the estimate of the slope (DFbeta1 = 7.90). The overall influence of the high data point on the regression coefficients was significant at alpha = 0.10 (Cook's Distance = 3.77, $p = 0.074$). Four other points also met the criteria defining outliers for both sediment and tissue concentrations and influenced the slope and intercept of the regression. Although these points did not have the same degree of influence on fitted values or the overall regression equation as the highest data point, they contributed to the irregular pattern in the residuals.

The width of the confidence intervals of the sediment-tissue relationship (Figure 8-2) would generate considerable uncertainty in back calculations of sediment concentrations from risk-based tissue concentrations. Removal of the high value does not improve the relationship ($R^2 = 0.12$, $p = 0.25$) and leads to a regression that can only be applied for sediment cPAH concentrations between 23 and 520 µg/kg dw.

Several aspects of the regression presented in Figure 8-2 indicate that it may not be appropriate to use the current cPAH dataset to predict tissue concentrations from sediment concentrations. First, the percent of variance explained by the regression relationship (R^2) is low. Second, although the residuals appeared to be normally distributed, their irregular pattern indicates that other factors besides sediment are affecting tissue concentrations. The variance of the residuals is lowest at low sediment concentrations and highest at intermediate concentrations. Data were not sufficient to evaluate variability at the highest sediment concentrations.

When the highest data pair was removed, the regression was no longer significant. As with arsenic, the highest data point had a large influence on the regression relationship (in this case, it changes not only the slope but also the significance of the relationship), and estimates from one regression can differ greatly from estimates based on the other. Given the uncertainty about the true relationship, the variance in predictions from either of the regressions (width of the confidence intervals), and Foster et al (1987) findings that did not report bioaccumulation of PAHs from sediment by *M. arenaria*, it does not appear to be appropriate to use a regression model to estimate sediment RBTCs for cPAHs for the seafood consumption scenarios.

8.4 RBTCs Based on River Otter Prey Ingestion

PCBs were identified as a risk driver chemical for river otter in the ERA (Appendix A). This section presents the derivation of both tissue and sediment RBTCs for this receptor.

8.4.1 Tissue RBTCs for river otter prey ingestion

Tissue RBTCs were estimated for total PCBs for river otter according to Equation 8-4. To estimate a tissue RBTC corresponding to a HQ of 1, the exposure dose was set equal to the LOAEL that was identified in the ERA (0.26 mg/kg body weight/day). Body weight and food ingestion rate values for river otter were derived from the literature, as described in the ERA (Appendix A, Section A.5.1.2.4). The site use factor was assumed to be 1. PCB dose contributions from water ingestion and incidental sediment ingestion were assumed to be negligible.

$$\text{Tissue RBTC} = \frac{ED \times BW}{SUF \times FIR}$$ **Equation 8-4**

Where:

RBTC	=	risk-based threshold concentration in prey tissue (mg PCBs/kg food dw)
ED	=	exposure dose set equal to the LOAEL (mg/kg body weight/day)
BW	=	body weight (kg)
SUF	=	site use factor (unitless); fraction of time that river otter spend foraging in the LDW relative to the entire home range
FIR	=	food ingestion rate (kg food dw/day)

The tissue RBTC for river otter for a LOAEL-based HQ of 1 was 2.7 mg/kg food dw or 540 µg/kg ww (assuming 80% moisture). River otter ingest various fish and shellfish species. The estimated tissue RBTC represents the weighted average concentration of all prey types that could be ingested by river otter.

8.4.2 Sediment RBTCs for river otter prey ingestion

A range of sediment RBTCs for total PCBs was estimated for river otter using the FWM and the same general approach described for the human health consumption scenarios (Section 8.3.1). Section D.8 of Appendix D describes the how the FWM was used to derive sediment RBTCs.

A range of sediment RBTCs for total PCBs was estimated for two dietary scenarios. The range of RBTCs reflects the uncertainty in the model bounded by the model performance criterion, which required that the estimated concentrations in tissues were within a factor of 2 of the empirical data for all of the target species. In addition, a best-fit RBTC was estimated using the model parameter set that most closely fit the empirical PCB tissue data for all modeled species combined.

Two dietary scenarios were investigated because juvenile chinook salmon were included as a portion of the river otter's diet in the ERA risk calculations as a surrogate for adult salmon. The FWM estimated concentrations of total PCBs in juvenile fish in general to represent various species of small fish, as described in more detail in Appendix D. Thus, the assumed river otter diet in the ERA and the FWM output were not consistent. Therefore, RBTCs were estimated using two dietary scenarios for river otter (Table 8-9). The first scenario assumed that juvenile fish modeled in the FWM were representative of juvenile chinook salmon. The second scenario assumed that river otter do not ingest juvenile fish or salmon of any kind.

Table 8-9. Dietary fractions of prey items used in the two river otter dietary scenarios for calculation of RBTCs for total PCBs

Receptor of Concern	Dietary Scenario	Fraction of Prey Item in Diet (by weight)					
		Clams[a]	Juvenile Fish	Crabs	Pacific Staghorn Sculpin	Shiner Surfperch	English Sole
River otter	including juvenile fish	0.02	0.22	0.1	0.22	0.22	0.22
	excluding juvenile fish	0.02	na	0.1	0.29	0.29	0.29

[a] In the ERA, clam and mussels were assumed to be 1% of the river otter's diet. Because mussels were not modeled in the FWM, clams were assumed to contribute 2% of the river otter's diet for RBTC calculations.

ERA – ecological risk assessment

PCBs – polychlorinated biphenyls

RBTC – risk-based threshold concentration

Best-fit sediment RBTCs for river otter corresponding to a LOAEL-based HQ of 1 were 250 and 218 µg/kg dw with and without inclusion of juvenile fish in the diet,

respectively (Table 8-10). When juvenile fish were excluded from the river otter diet, the RBTCs were slightly lower (ranging from 114 to 250 µg/kg dw) than when juvenile fish were included (ranging from 133 to 298 µg/kg dw). RBTCs were lower because total PCB concentrations in juvenile chinook salmon were lower than concentrations in Pacific staghorn sculpin, shiner surfperch, and English sole.

Table 8-10. Sediment RBTCs for total PCBs for prey consumption by river otter

Scenario	RBTCs for Total PCBs in Sediment (µg/kg dw)		
	Lower Bound	Best Fit	Upper Bound
Inclusion of juvenile fish in diet	133	250	298
Exclusion of juvenile fish from diet	114	218	250

dw – dry weight

PCB – polychlorinated biphenyl

RBTC – risk-based threshold concentration

8.5 RBTCs Based on Benthic Invertebrate Exposure

In the ERA, 41 chemicals were identified as risk drivers for the benthic invertebrate community because they had at least one detected exceedance of the SQS in surface sediment. Because SMS are applicable promulgated standards at the site, sediment RBTCs for these 41 chemicals were set equal to the SQS and the CSL numerical chemical criteria (Table 8-11). Toxicity test data for locations with SMS exceedances must also be considered in the application of these RBTCs because toxicity test information overrides chemical data as part of the SMS. The interpretation of SMS exceedances and site-specific toxicity data is discussed in the ERA (Appendix A).

Table 8-11. SMS chemical criteria for risk driver chemicals identified for the benthic invertebrate community

Chemical	Unit	SQS	CSL
Metals			
Arsenic	mg/kg dw	57	93
Cadmium	mg/kg dw	5.1	6.7
Chromium	mg/kg dw	260	270
Copper	mg/kg dw	390	390
Lead	mg/kg dw	450	530
Mercury	mg/kg dw	0.41	0.59
Silver	mg/kg dw	6.1	6.1
Zinc	mg/kg dw	410	960
PAHs			
2-Methylnaphthalene	mg/kg OC	38	64
Acenaphthene	mg/kg OC	16	57

CHEMICAL	UNIT	SQS	CSL
Anthracene	mg/kg OC	220	1,200
Benzo(a)anthracene	mg/kg OC	110	270
Benzo(a)pyrene	mg/kg OC	99	210
Benzo(g,h,i)perylene	mg/kg OC	31	78
Total benzofluoranthenes	mg/kg OC	230	450
Chrysene	mg/kg OC	110	460
Dibenzo(a,h)anthracene	mg/kg OC	12	33
Dibenzofuran	mg/kg OC	15	58
Fluoranthene	mg/kg OC	160	1,200
Fluorene	mg/kg OC	23	79
Indeno(1,2,3-cd)pyrene	mg/kg OC	34	88
Naphthalene	mg/kg OC	99	170
Phenanthrene	mg/kg OC	100	480
Pyrene	mg/kg OC	1,000	1,400
Total HPAH	mg/kg OC	960	5,300
Total LPAH	mg/kg OC	370	780
Phthalates			
bis(2-ethylhexyl) phthalate	mg/kg OC	47	78
Butyl benzyl phthalate	mg/kg OC	4.9	64
Dimethyl phthalate	mg/kg OC	53	53
Other SVOCs			
1,2,4-Trichlorobenzene	mg/kg OC	0.81	1.8
1,2-Dichlorobenzene	mg/kg OC	2.3	2.3
1,4-Dichlorobenzene	mg/kg OC	3.1	9.0
2,4-Dimethylphenol	µg/kg dw	29	29
4-Methylphenol	µg/kg dw	670	670
Benzoic acid	µg/kg dw	650	650
Benzyl alcohol	µg/kg dw	57	73
Hexachlorobenzene	mg/kg OC	0.38	2.3
n-Nitrosodiphenylamine	mg/kg OC	11	11
Pentachlorophenol	µg/kg dw	360	690
Phenol	µg/kg dw	420	1,200
PCBs			
Total PCBs	mg/kg OC	12	65

CSL – cleanup screening level
dw – dry weight
HPAH – high-molecular-weight polycyclic aromatic hydrocarbon
OC –organic carbon
PCB – polychlorinated biphenyl
LPAH – low-molecular-weight polycyclic aromatic hydrocarbon
SMS – Washington State Sediment Management Standards
SQS – sediment quality standard
SVOC – semivolatile organic compound

8.6 APPLICATION OF RBTCS

The RBTCs derived in this section are an important consideration in the derivation of PRGs in the FS. Three issues related to application of RBTCs are discussed here: spatial considerations (Section 8.6.1), consideration of cumulative and aggregate risk (Section 8.6.2), and considerations of assumptions regarding current and future use (Section 8.6.3).

8.6.1 Spatial considerations

Appropriate application of RBTCs requires explicit consideration of spatial scale. Some RBTCs are appropriate for application on a point-by-point basis, while most others should be applied only at a larger scale. For example, RBTCs based on the protection of benthic invertebrates (see Section 8.5) should be applied on a point-by-point basis because of the small home ranges of these animals and the regulatory framework described in the SMS. All other RBTCs (Sections 8.1 through 8.4) are most appropriately applied at larger spatial scales as arithmetic or spatially weighted mean concentrations. RBTCs for seafood consumption should be applied to an LDW-wide spatial scale because risks were evaluated at this scale.

RBTCs derived for direct sediment contact scenarios should be applied to exposure areas identified in the HHRA. The exposure area for the netfishing RME scenario included all of the intertidal and subtidal areas of the LDW. For the beach play RME scenario, eight separate intertidal areas were selected as exposure areas where children are most likely to contact sediment (Map 6-1). The exposure area for the RME clamming included all intertidal areas where clamming could occur because such areas are potentially accessible either from a boat or from shore (Map 6-2).

8.6.2 Consideration of cumulative and aggregate risk

Following both EPA and Ecology guidance for conducting HHRAs, cumulative excess cancer risk estimates were calculated in the HHRA by summing risk estimates for individual COCs. Cumulative excess cancer risk should also be given consideration during the application of RBTCs in the FS. In practice, cumulative carcinogenic risk can be considered in the application of RBTCs in specific areas by summing the risks for individual chemicals within a given exposure area to ensure that they do not result in an unacceptable cumulative target risk. Cumulative risk for non-cancer hazards can only be evaluated if the chemicals affect the same target organs or have the same endpoint effects.

For example, if the target excess cancer risks for four individual risk drivers were all set to 1×10^{-6} for calculation of individual chemical RBTCs, the cumulative target risk would be no greater than 4×10^{-6} (i.e., 4 times 1×10^{-6}). In this example, if the acceptable target cumulative risk was 1×10^{-5}, then meeting each individual RBTC that was based on a target risk of 1×10^{-6} would result in an acceptable cumulative risk estimate for that area because 4×10^{-6} is less than 1×10^{-5}.

8.6.3 Consideration of assumptions regarding current and future LDW use

The human health RBTCs presented in the RI, particularly those for RME scenarios, are based on exposure assumptions that are intended to be health-protective of both current and future uses of the LDW. There is limited site-specific data available on the amount of resident seafood organisms currently being harvested and consumed from the LDW, and no site-specific data on the degree to which people may currently be exposed via beach play or clamming activities. Consequently, the data used to establish the exposure parameters for these RME scenarios were based on other sources, including tribal seafood consumers and clam diggers from other areas of Puget Sound and beach users from King County lakes. The tribal netfishing scenario, on the other hand, reflects exposure conditions that could occur under current tribal fishing activities within the LDW.

The exposure assumptions for the RME scenarios for tribal seafood consumption and clam digging are likely overestimates of current behavior in the LDW. With respect to seafood consumption, WSDOH has issued health advisories for LDW fish and crab, warning the public that any seafood consumption from the LDW is unsafe because of elevated PCB concentrations in these species (WSDOH 2005). WSDOH (Office of Food Safety and Shellfish) and Public Health – Seattle & King County advise against consumption of clams in the LDW because of pathogens from urban sewage releases (WSDOH 2005). Environmental quality, and consequently seafood resources, is expected to improve in the future, but it is not known if future site use will reach the RME exposure levels assumed in the HHRA (Appendix B).

A creel study (in which fishermen were observed and surveyed while fishing) was conducted by the County to assess the amount of fishing activity that occurred in Elliott Bay and the LDW in 1997 (King County 1999c). The study included surveys at eight locations (Table 8-12) within the LDW on 30 different days during the summer of 1997 (between June 22 and August 30), including 10 Saturdays, 10 Sundays, and 10 weekdays. Sites were visited repeatedly between 5 a.m. and 8 p.m. (weekends) and from 5 a.m. to 11 a.m. and 4 p.m. to 10 p.m. (weekdays).

Table 8-12. Locations and number of fishermen identified in the King County creel survey

Fishing Location Within LDW	Number of Individuals Observed Fishing[a]
T-105	10
Diagonal Avenue S	2
Duwamish Waterway Park	1
Kellogg Island	0
Muckleshoot 1st Avenue. boat ramp	0
8th Avenue, street end	0
Boeing office building access	0

Fishing Location Within LDW	Number of Individuals Observed Fishing[a]
8th Avenue S and S Portland Street	0

Source: King County (1999c)

[a] Surveys were conducted within the LDW on 30 different days during the summer of 1997 (between June 22 and August 30), including 10 Saturdays, 10 Sundays, and 10 weekdays. Sites were visited repeatedly between 5 a.m. and 8 p.m. (weekends) and from 5 a.m. to 11 a.m. and 4 p.m. to 10 p.m. (weekdays).

T-105 – Terminal 105

Very few individuals were observed fishing in the LDW compared to Elliott Bay (13 in the LDW and over 1,000 in Elliott Bay), and seafood was caught at only one of the eight locations (two crabs and one sole were caught at T-105). King County (1999d) concluded that the frequency of such recreational activities would be low in the LDW, as compared to Elliott Bay, because of the industrial nature of the waterway and limited access to the water from the shoreline. Although fishing was documented to be infrequent within the LDW, one of the most popular fishing locations in the survey was the Spokane Street Bridge just north of the study area, from which people can fish in the East Waterway. Although this survey did not evaluate the frequency of fishing from boats, people have been observed using boats to fish for salmon in the LDW.

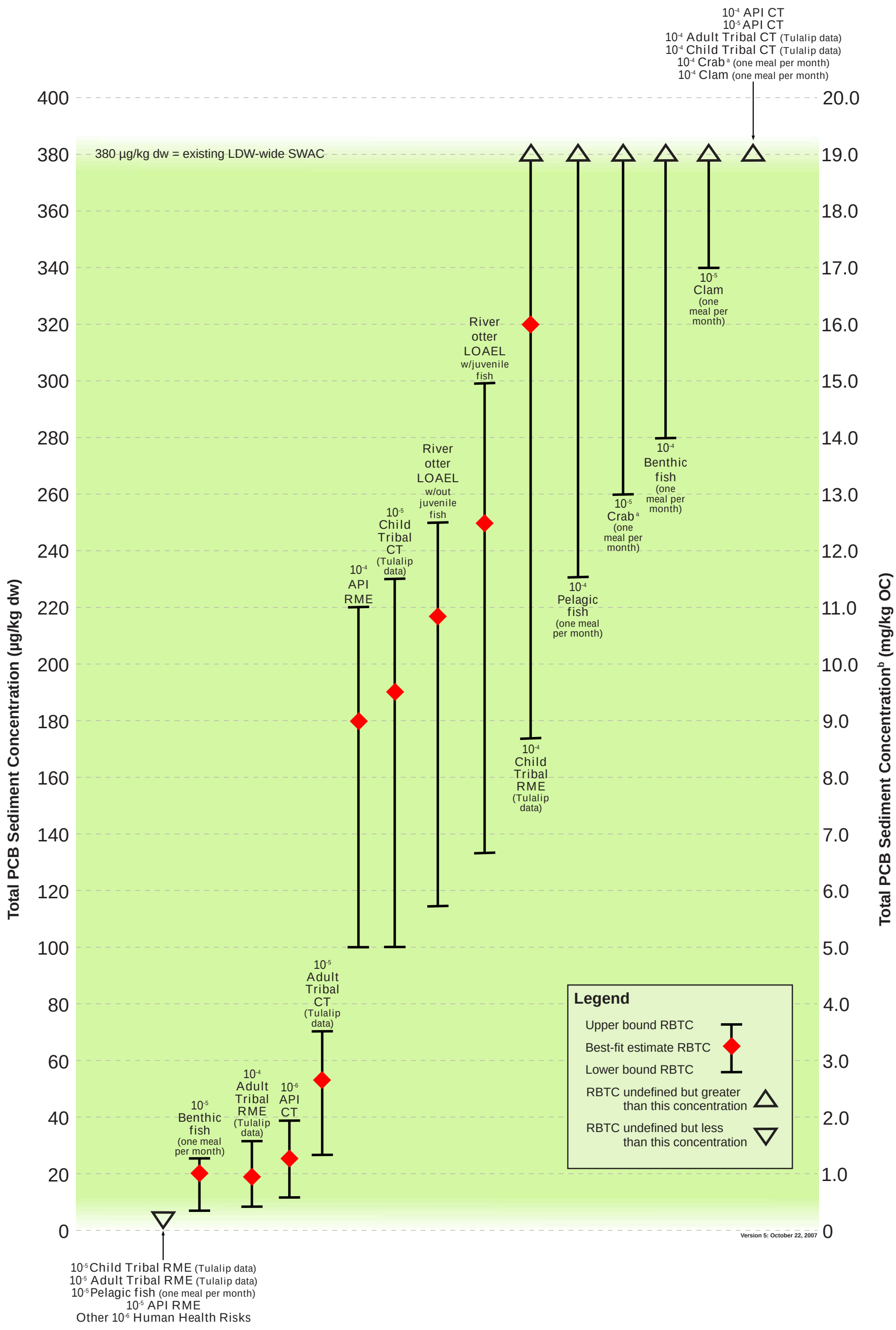
Future remedial and restoration actions could increase the frequency of fishing and other recreational activities, particularly kayaking and canoeing. However, the LDW is expected to remain an industrial corridor for the foreseeable future. Increased public access and habitat restoration activities may not only expand the areas where people currently access the river, but also increase the frequency with which they do so. Consequently, the potential overestimation of current risks in the exposure scenarios is intended to be protective of future uses as well.

8.7 Summary

RBTCs in sediment were derived for consideration in the development of PRGs in the FS. RBTCs were derived for human direct sediment contact scenarios for arsenic, cPAHs, dioxins and furans, and total PCBs (Table 8-13). Sediment RBTCs based on the seafood consumption scenarios were estimated for total PCBs using the FWM. Regression models for relating concentrations of inorganic arsenic and cPAHs in clams to concentrations of arsenic and cPAHs in co-located sediment were evaluated and found to be unsuitable for developing RBTCs. No sediment RBTC based on seafood consumption was estimated for dioxins and furans because there are no available tissue data from the LDW to support such a derivation. RBTCs were also estimated for total PCBs in both tissue and sediment for prey consumption by river otter. Sediment RBTCs for benthic invertebrates were set equal to SMS criteria.

Sediment RBTCs for total PCBs under the various human seafood consumption scenarios and for prey consumption by river otters are compared in Figure 8-3. This figure makes it possible to visually compare the ranges where RBTCs for the different

seafood consumption scenarios overlap. It also shows that across the range of arsenic concentrations of interest in sediment (0 to 380 µg/kg dw), RBTCs range from incalculable to greater than 380 µg/kg dw, depending upon the seafood consumption scenario and target risk level. The appropriate application of RBTCs to sediment in the LDW must consider the specific exposure areas identified in the HHRA for direct contact scenarios and for seafood consumption RBTCs.



Note: Ranges shown incorporate results from the final ERA and draft final HHRA, results from FWM Calibration 2, and the ranges associated with two different diet assumptions for river otter.

[a] Crab one-meal-per-month scenario included consumption of equal portions of slender crabs and Dungeness crabs.

[b] Sediment RBTCs estimated from the FWM were on a dry-weight basis; for the sake of comparison, the dry-weight RBTCs were converted to approximate OC-normalized RBTCs, assuming 2% organic carbon.

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Figure 8-3. Summary of sediment RBTCs for total PCBs for selected human seafood consumption and river otter prey consumption scenarios

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Table 8-13. Summary of sediment RBTCs for risk drivers

Risk Driver Chemical and SMS Criteria (if available)	Risk Level	Sediment RBTCs for Direct Contact Scenarios			Sediment RBTCs for Human Seafood Consumption Scenarios[b] Best fit (lower bound, upper bound)			Sediment RBTCs for Ecological Receptor[a]
		Netfishing RME	Beach Play RME	Tribal Clamming RME	Adult Tribal RME (Tulalip Data)	Child Tribal RME (Tulalip Data)	API RME	River Otter Prey Consumption
Arsenic (mg/kg dw) SQS = 57, CSL = 93	1×10^{-6}	3.7	2.8	1.3	nc[c]	nc[c]	nc[c]	na
	1×10^{-5}	37	28	13	nc[c]	nc[c]	nc[c]	na
	1×10^{-4}	370	280	130	nc[c]	nc[c]	nc[c]	na
	HQ = 1	nc[b]	nc[b]	nc[b]	nc[c]	nc[c]	nc[c]	na
cPAHs (µg/kg dw)	1×10^{-6}	380	90	150	nc[c]	nc[c]	nc[c]	na
	1×10^{-5}	3,800	900	1,500	nc[c]	nc[c]	nc[c]	na
	1×10^{-4}	38,000	9,000	15,000	nc[c]	nc[c]	nc[c]	na
	HQ = 1	nc[b]	nc[b]	nc[b]	nc[c]	nc[c]	nc[c]	na
Dioxin and furan TEQ (ng/kg dw)	1×10^{-6}	37	28	13	nc[d]	nc[d]	nc[d]	na
	1×10^{-5}	370	280	130	nc[d]	nc[d]	nc[d]	na
	1×10^{-4}	3,700	2,8000	1,300	nc[d]	nc[d]	nc[d]	na
	HQ = 1	nc[b]	nc[b]	nc[b]	nc[d]	nc[d]	nc[d]	na
Total PCBs (µg/kg dw) SQS = 12 mg/kg OC, CSL = 65 mg/kg OC	1×10^{-6}	1,300	1,700	500	nc[e]	nc[e]	nc[e]	na
	1×10^{-5}	13,000	17,000	5,000	nc[e]	nc[e]	nc[e]	na
	1×10^{-4}	130,000	170,000	50,000	19 (8.4 – 33)	320 (173, 380)	180 (100, 220)	na
	HQ = 1	nc[b]	nc[b]	nc[b]	nc[e]	nc[e]	nc[e]	218 – 250[f]

[a] RBTCs for the remaining 39 risk driver chemicals identified for the benthic invertebrate community were presented in Table 8-11.

[b] RBTCs were not estimated for non-cancer hazards for direct contact scenarios because none of the RME scenarios had HQs for an individual chemical greater than 1 or generated endpoint-specific HIs in excess of 1 (Appendix B, Section B.5.6).

[c] Regression models for relating concentrations of arsenic and cPAHs in clams to concentrations of these chemicals in co-located sediment were evaluated and found to be unsuitable for developing RBTCs.

[d] RBTCs could not be calculated because there are no dioxin and furan tissue data from the LDW.

[e] RBTC could not be calculated because contribution from water led to predicted tissue concentrations greater than the applicable risk level, even in the absence of any contribution from sediment.

[f] Represents best-fit estimates for two different dietary scenarios, as described in Section 8.4.

cPAH – carcinogenic polycyclic aromatic hydrocarbon

CSL – cleanup screening level

dw – dry weight

HQ – hazard quotient

na - not applicable

nc – not calculated

OC – organic carbon

PCB – polychlorinated biphenyl

RBTC – risk-based threshold concentration

RME – reasonable maximum exposure

SQS – sediment quality standards

TEQ – toxic equivalent

9.0 Potential Pathways, Source Identification, and Source Control Efforts

Source control is the process of identifying and preventing or reducing releases of chemicals into the environment. The goal of an effective source control strategy is to minimize the potential for sediment to become recontaminated to levels of concern after remediation has been completed. Since 2001, source identification and control programs in the LDW have been established by regulatory authorities including the City, the County, Ecology, and EPA. The goal of these efforts has been to identify potential sources of chemicals to the LDW and to implement cleanup plans and source control strategies.

The LDW and its adjacent upland areas have been used for industrial, commercial, and shipping operations for nearly a century. During this time, chemicals associated with those practices have entered the LDW through various pathways; and some, such as PCBs, PAHs, metals, and phthalates, are present in LDW sediment. Many private and agency-led environmental investigations have been conducted both in the LDW and on adjacent upland properties to further characterize the nature and extent of contamination in the LDW, as well as to identify potential sources of chemicals that could continue to enter the LDW.

Ecology, as the lead source control agency, formed the LDW Source Control Work Group (SCWG) in 2002. The SCWG is composed primarily of public agencies responsible for source control, including Ecology, the City of Seattle, the City of Tukwila, the County, the Port, and EPA. The purpose of the SCWG is to share information, discuss strategy, actively participate in developing action plans, jointly implement source control measures, and share progress reports on source control activities within the LDW area. Public agencies with the authority to regulate source control in the LDW use stormwater programs, permits, hazardous waste management and pollution prevention programs, inspection and maintenance programs, water quality compliance and spill response programs, environmental and pathways assessments, and other regulatory programs in an effort to control sources of contamination.

In 2004, Ecology developed a strategic source control plan for the LDW (Ecology 2004b). The basic source control strategy for the LDW is to identify and manage sources of chemicals to LDW sediments in coordination with site-specific sediment investigations and efforts. The plan provides the basis for Ecology to identify priority areas through a tiered process and to develop summary of existing information and data gaps reports (hereafter referred to as Data Gaps Reports) and detailed area-specific source control action plans (SCAPs) for each priority area. The status of the reports for the various areas is discussed in Sections 9.2 and 10.

This section describes the potential migration pathways through which chemicals can enter the LDW (Section 9.1), discusses the overall LDW source control strategy and SCWG efforts (Section 9.2), identifies the various potential historical and ongoing sources to the LDW (Section 9.3), and discusses the ongoing source control efforts and programs being implemented in the LDW (Section 9.4). The information in this section was compiled using readily available data; historical records and documents; and information from EPA, Ecology, the City of Seattle, the City of Tukwila, the Port, and the County. The area included for this discussion (the greater LDW drainage basin) is presented on Map 9-1. The information presented in this section is useful in the development of a CSM for the LDW and will assist in the remedial alternative decision-making process.

9.1 POTENTIAL PATHWAYS

Chemicals released to media such as air, soil, groundwater, surface water, or impervious surfaces may migrate to the LDW through various potential pathways. As shown in Figure 9-1, potential pathways include atmospheric deposition; surface water inputs; direct discharge into the LDW (e.g., CSOs, storm drains); chemical spills and/or leaks to the ground, surface water, or directly into the LDW; groundwater migration; and bank erosion. This section briefly highlights the potential pathways to the LDW. In-depth information is provided in Sections 9.3 and 9.4.

9.1.1 Atmospheric deposition

Chemicals are emitted to the air from both point and non-point sources. Point sources include a variety of industrial facilities (EPA 2001). Non-point sources include mobile emissions from motor vehicles, marine vessels, and trains, as well as common materials (e.g., plastics) through off-gassing. Chemicals emitted to the air may be transported over long distances, generally in the direction of the area's prevailing winds. They can be deposited from the atmosphere to land and water surfaces through wet deposition (precipitation) or dry deposition (as particles).

Air pollutants can be deposited to water bodies through either direct or indirect deposition. Direct deposition occurs when chemicals are deposited onto the surface of a water body. Indirect deposition occurs when chemicals are first deposited on land and then transported to the water body via surface water or stormwater runoff. Many air pollutants deposited in aquatic systems, such as the LDW, have the potential to contaminate sediment because they adhere to sediment particles (PSCAA 2003). The potential sources of chemicals to the LDW through atmospheric deposition are discussed in further detail in Sections 9.3.2.4 and 9.4.2.

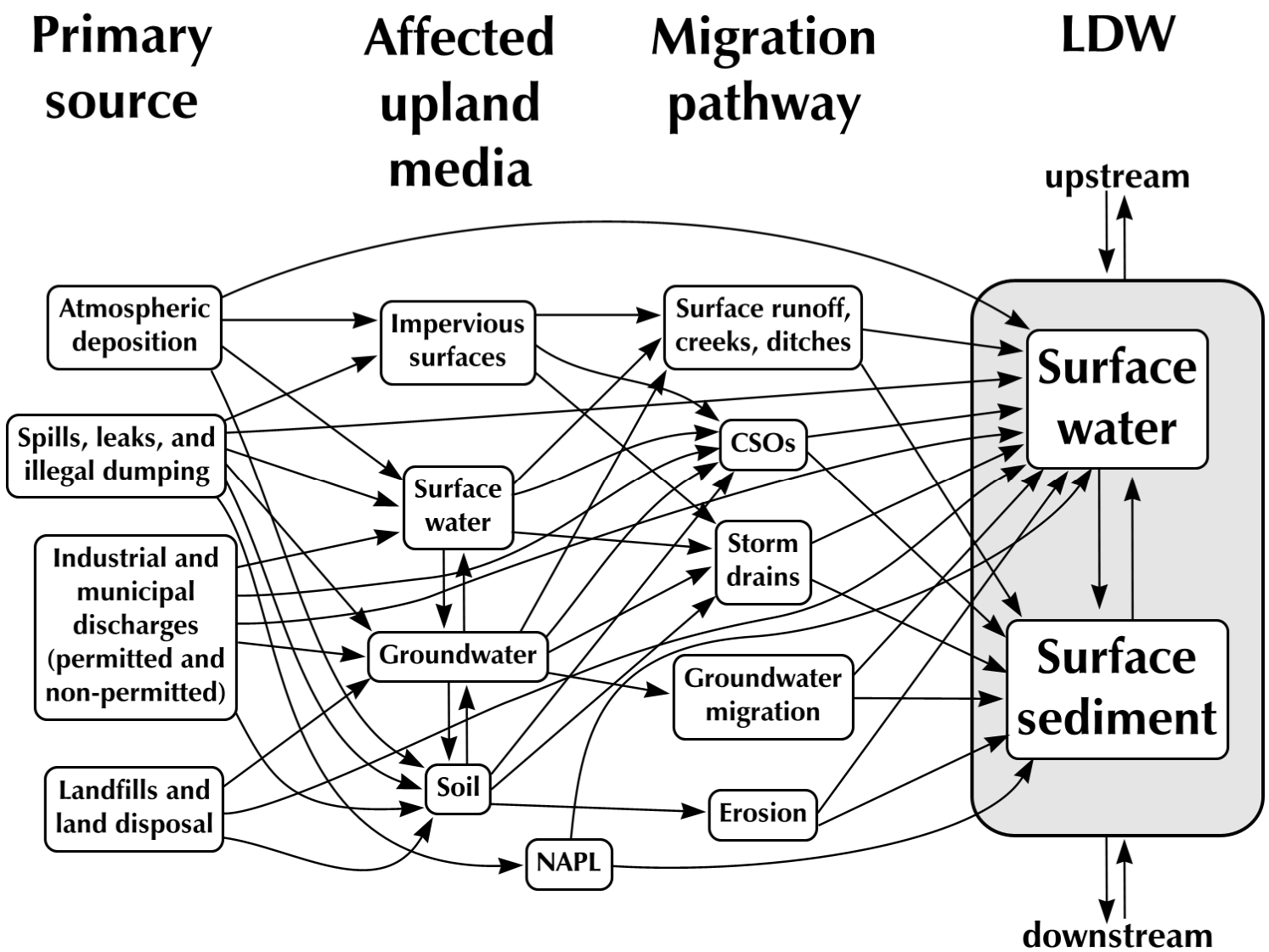

Figure 9-1. Conceptual model of chemical sources and potential pathways to the LDW

9.1.2 Surface water inputs

Potentially contaminated surface waters and suspended sediment may be transported to the LDW from the upper reaches of the Green/Duwamish River, tributary creeks, and through open-channel drainage ditches. The inputs from these sources vary greatly during the year; the upper Green/Duwamish system is variable because it can be influenced by a large area of mixed industrial, commercial, residential, and agricultural lands. Accidental spills and leaks that occur upriver or along its tributaries also have the potential to enter the LDW in areas lacking collection systems, through which spills and leaks can flow directly over impervious surfaces and drain into the LDW. Surface water inputs into the LDW are discussed in further detail in Section 9.4.3.

9.1.3 Direct discharges

In general, direct discharge systems include municipal or other publicly owned drainage systems, privately owned and managed drainage systems, and sanitary/combined sewer systems. Since 2003, over 250 individual points that discharge into the LDW study area have been identified. These individual points, often referred to as outfalls, are defined here as locations of discharge of storm water and/or industrial wastewaters transported via a collection system; though most industrial discharges are now routed through the sanitary sewer and no longer discharge directly to the waterway. The classifications of direct discharges include CSOs, emergency overflows (EOFs), and public and private storm drain systems. Direct discharge systems and the various types of outfalls are discussed in further detail in Section 9.4.4 and Appendix H.

9.1.4 Spills and leaks

Chemicals released through spills and leaks to soil, other ground surfaces (such as roadways), surface water, or groundwater have the potential to migrate to the LDW. Leaks can occur from various sources including pipes and storage tanks, industrial or commercial equipment, and process operations. Spills can occur accidentally during vehicle fueling and maintenance, or purposefully in the case of illegal dumping. The USCG, Ecology, and the City each have programs to record and clean up reported spills and leaks of hazardous materials. These programs are discussed further in Section 9.4.5.

9.1.5 Groundwater migration

Groundwater flow in the greater Duwamish Basin is generally towards the LDW, although the direction varies locally depending on the nature of subsurface materials, hydrostratigraphy, and proximity to the LDW (see Section 2.5). Near the LDW, tidal action can greatly alter groundwater flow directions, rates, and water quality. The determination of whether a chemical identified in groundwater will reach sediment and surface water in the LDW study area is a complex process. Groundwater issues in

the LDW area are discussed further in Sections 9.4.6, and an overview of monitoring is presented in Appendix I.

9.1.6 Bank erosion

Unprotected shoreline banks are susceptible to erosion by both wind and surface water runoff. Shoreline armoring and vegetation reduce bank erosion. Bank slope is also a factor; steeper banks are more susceptible to erosion. Currently, the majority of the shoreline is armored with constructed steel and concrete bulkheads, sheet pile walls, and riprap banks, limiting bank erosion in many areas. Bank erosion is more likely to occur in unarmored areas such as the banks of Kellogg Island, the shoreline east of the island, and areas to the south near the Upper Turning Basin. Much of the bank material in the LDW has not been well characterized, and the importance of this pathway from an LDW-wide perspective is not well known. Erosion of contaminated bank material has been identified as a potential pathway to LDW sediments in a few areas along the LDW shoreline, as discussed in Section 9.4.7.

9.2 LDW Source Control Strategy and the Source Control Work Group

Ecology is the lead agency responsible for source control in the LDW. Ecology works with several other entities, including the City of Seattle, the City of Tukwila, the County, the Port, and EPA to create and implement source control strategies and prioritize cleanup efforts in the LDW. In 2002, these entities formed the LDW SCWG. The SCWG and Ecology's general source control strategy (Ecology 2004b), including the evaluation of priority areas, is discussed in the following sections.

9.2.1 Source Control Work Group strategy and activities

The purpose of the SCWG is to share information, discuss strategies, develop action plans, jointly implement source control measures, and share progress reports on source control activities in the LDW area. Other public entities with related source control responsibilities include PSCAA and Seattle-King County Department of Public Health. Both of these entities actively coordinate with the SCWG.

In January 2004, Ecology and the SCWG issued the final *Lower Duwamish Waterway Source Control Strategy* (Ecology 2004b). This strategy is consistent with sediment source control protocols described in EPA guidance (2002c) and the Washington State SMS (Ecology 1993). The strategy describes the process and timing for implementing source control, the roles of various regulatory agencies responsible for providing enforcement, and Ecology's methods for tracking and documenting source control progress in the LDW.

The focus of Ecology's source control strategy for LDW sediment is to meet sediment cleanup goals and to prevent post-remediation recontamination to levels that exceed the SMS (Ecology 2004b). The strategy was designed to identify and manage sources of chemicals that could contaminate LDW sediments, and it serves as the basis for the

development of a series of detailed, area-specific SCAPs that are being developed for priority action areas.

The source control strategy established the priority for source control activities in the LDW according to the following four tiers:

- **Tier One –** Priority cleanup areas identified through the RI process (Windward 2003d)
- **Tier Two –** Areas requiring source control evaluations as identified by Ecology in consultation with EPA
- **Tier Three –** Additional areas in LDW drainage basins not identified as EAAs or other Tier Two Areas (TTAs), as needed
- **Tier Four –** Areas that have been remediated and were subsequently recontaminated above SMS criteria or LDW cleanup goals based on post-cleanup monitoring

In 2003, LDWG identified seven EAAs, sediment areas that were given priority for investigation and potential cleanup in the LDW because of the relatively high chemical concentrations in sediment (Windward 2003d). These areas were subsequently adopted by Ecology as their Tier One Areas; these areas will be referred to as EAAs in this document. These sites were selected using both risk- and management-based criteria and are discussed in a LDWG technical memorandum (Windward 2003d) that identified the candidate sites for early action. Risk-based criteria were based on CSL exceedances or the top 95th percentile of risk drivers. Management-based criteria included: 1) consistency with methods used by EPA and Ecology to identify contaminated sites requiring remediation, 2) ability to prevent unacceptable recontamination at the site, and 3) consistency between potential early action cleanup methods and remedial alternatives that are likely to be evaluated for the LDW (Windward 2003d). The locations of these EAAs are shown on Map 9-2.

Subsequent to the identification of the EAAs, several tentative TTAs were identified by Ecology, in consultation with EPA, based on the sediment data collected during completion of the draft RI. These areas have been identified as potentially requiring cleanup or additional source control evaluation. The list of these TTAs will likely change after completion of the RI and as source control investigations continue. For the purposes of the RI, these areas are referred to as TTAs (see Map 9-2).

9.2.2 Ecology Source Control Action Plans

Ecology is producing SCAPs for EAAs and TTAs. SCAPs are intended to provide a plan for managing and monitoring potential sources of chemicals within individual EAAs and TTAs. SCAPS are also intended to document when source control activities are sufficient to allow sediment cleanup to begin without an unacceptable likelihood of recontamination (Ecology 2004b) because minimizing and/or preventing recontamination is a goal of the source control program. SCAPs identify potential

sources at each area, describe how elements of the source control strategy are to be implemented, and document the following:

- Status of contaminated media and pathways
- Plans for completing source identification and characterization, including the collection of any additional environmental data
- Actions and tools appropriate for controlling sources, including monitoring requirements to assess source control effectiveness.
- Criteria or other goals necessary for Ecology to determine the effectiveness and completeness of source control
- Target dates for these key milestones

A schedule for the production of SCAPs for EAAs and TTAs is presented in Table 9-1. These areas are discussed further in Section 10.

Table 9-1. EAAs and Ecology's TTA locations and their respective SCAP status

Property Name or Description	Approximate Location	Status of Draft SCAP	Date of Draft SCAP[a]
EAAs (also referred to as Tier One Areas by Ecology)			
Duwamish/Diagonal	RM 0.4 to RM 0.6 E	completed	December 2004
EAA 2 (RM 2.2)	RM 2.2 W	completed	June 2007
Slip 4	RM 2.8 to RM 2.9 E	completed	July 2006
Boeing Plant 2/Jorgensen Forge	RM 2.9 to RM 3.7 E	completed	November 2007
T-117	RM 3.5 to RM 3.7 W	completed	July 2005
EAA 6 (RM 3.8)	RM 3.8 E	scheduled	May 2008
Norfolk	RM 4.8 to RM 5.0 E	completed	September 2007
Tier Two Areas[b]			
Glacier Bay	RM 1.3 to RM 1.55 W	completed	November 2007
Ash Grove Cement[c]	RM 0.0 to RM 0.3 E	scheduled	January 2009
Slip 1/Manson[c]	RM 0.9 to RM 1.0 E	scheduled	February 2009
Glacier NW to Saint-Gobain Glass Company[c]	RM 1.3 to RM 1.7 E	scheduled	January 2009
Slip 3 to Slip 2	RM 1.8 to RM 2.0 E	scheduled	December 2008
Slip 3	RM 2.0 to RM 2.1 E	scheduled	July 2008
Slip 4 to Seattle Boiler Works[c]	RM 2.4 to RM 2.8 E	scheduled	August 2008
Central KCIA[c] discharge (former Slip 5)	see EAA 6 (RM 3.8 E)	see EAA 6 (RM 3.8)	see EAA 6 (RM 3.8)
Slip 6	RM 3.9 to RM 4.4 E	scheduled	July 2008

Source: Cargill (2007a, b, c); Cargill et al. (2007); SCWG (2005-2007)

[a] SCAP schedule taken from draft LDW Source Control Status Report (SAIC 2007b; Ecology 2007g).

[b] List of Tier Two Areas and associated names are from Ecology's LDW webpage.

[c] Company names are used only to designate areas and are not necessarily included in the cleanup area nor are they necessarily responsible for contamination in these areas.

CSO – combined sewer overflow
EAA – Early Action Area
KCIA – King County International Airport
RM – river mile
SCAP – source control action plan
TTA – Tier Two Area

Data Gaps Reports are also being prepared by Ecology to summarize available existing information that may have implications for source control for each of the EAA and TTAs. Elements of these reports include:

- Location and physical characteristics of the EAAs or TTAs
- Documentation of readily available information with implications for sediment recontamination
- Identification of potential COCs[93] and evaluation of potential pathways
- Identification of critical data gaps that need to be addressed prior to sediment remediation in order to reduce the risk of sediment recontamination

Ecology has completed or is in the process of developing detailed SCAPs and summaries of existing information and Data Gaps Reports for the EAAs and TTAs. Once completed, the Data Gaps Reports and SCAPs will be provided to EPA, the SCWG, and stakeholders. After source control plans contained in a SCAP have been implemented, the success of the plan will be evaluated by Ecology through a series of source control effectiveness and completeness determinations (Ecology 2004b). These evaluations will then be provided to EPA for comment and concurrence, at which time a final determination of completeness will be granted. Final decisions on completeness of source control actions will be made by Ecology, and may include a conditional No Further Action (NFA) determination.

SCAPs have been completed or are in draft form for all of the EAAs except EAA 6 (RM 3.8) (Table 9-1). Draft summaries of existing information and Data Gaps Reports have been completed for the following EAAs: EAA 2, Slip 4, Boeing Plant 2/Jorgensen Forge, T-117,[94] and Norfolk. Other than the Glacier Bay TTA, TTAs have not yet been assessed by Ecology through the SCAP and Data Gaps Report process. They are scheduled to be evaluated and investigated over the next few years (Table 9-1). Tier Three Areas will be designated by Ecology in the future and may require SCAPs and Data Gaps Reports, depending on their complexity and specific concerns. If a Tier Four action is required because of sediment recontamination, an amendment to the applicable SCAP for that area will be provided by Ecology (2004b).

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[93] Potential COCs are based on a comparison to applicable state and federal standards per media type and were not based on the results of the LDW baseline risk assessments presented in Appendices A and B.

[94] This document was completed by Windward et al.(2003).

9.3 HISTORICAL SOURCES AND PRACTICES IN THE LDW

In the course of over 100 years of growth and development, industrial practices and waste streams have contributed chemicals to the LDW. Understanding these various historical practices is important to facilitate ongoing characterization, remediation, and source control efforts. The following section provides an overview of the industrial development of the waterway, including the dynamic reshaping of the river and the manufacturing and commercial operations that were established along its shores. The section also discusses some of the manufacturing and waste handling and disposal practices prevalent in the earlier history of the waterway. This type of information is important to consider in source identification and control effects.

Maps 9-3a through 9-3d provide information on some of the historical operations identified in the LDW study area. The majority of source information used to prepare the maps was compiled from Foster (1945) and the Pollution Control Commission (1955). Some historical information, at a more limited level, was obtained from other source documents, including the Data Gaps Reports and SCAPs. The locations of the historical operations presented on the maps are the approximate locations based on available information; in many cases, the exact locations were unknown. These maps are intended to give general locations of historical operations relative to the LDW and should not be used to associate these operations with individual parcels.

Information in this section was based on readily available resources, most notably Ecology's Data Gaps Reports and SCAPs, other summaries of source information, and an Environmental Data Resources, Inc. (EDR), document that was prepared in 2006 for the LDW (EDR 2006a). Some of these documents are known to contain errors and to have omitted data and other information. None of the resource information has been independently verified beyond the citations noted. Furthermore, the primary data themselves have had varying levels of QC and data validation, which have not generally been documented in the literature reviewed. Therefore, information included in this section should not be relied upon as complete and factual. Additional evaluations of the data would be required to use the data quantitatively or as an inference to identify actual sources. It is expected that more thorough evaluations of the data will be conducted in the course of further source control investigations.

9.3.1 Industrial development of the LDW

Historically, land adjacent to the Duwamish was accessed using the river as a transportation route. By 1874, Seattle's first railroad had been established, allowing further access to and development of lands along the river. Between 1901 and 1913, the LDW, the East and West Waterways, and Harbor Island were all developed through dredging, filling, and channelization of the Duwamish River and its delta (Section 2.2). Filling of mudflats and wetlands along the LDW created additional land that was available for development, and dredging and channelization of the LDW allowed oceangoing ships and barges access to the LDW. A steam plant used for electric power generation was built along the LDW in 1906.

Shipyards, airplane manufacturing, cement manufacturing, food processing, and cold storage were early industries along the LDW that became increasingly important with the onset of World War I (Sato 1997). Other common industries at this time included lumber storage and milling yards, metal fabrication, and equipment manufacturing (Sato 1997; Ecology 2006a). Residential areas in Georgetown and South Park began to grow with the increase in job opportunities and improved transportation; the South Park neighborhood area increased by more than 66 ac after channelization of the river, and the new land was claimed by farmers (Wilma 2001b).

The aviation industry continued to develop when Boeing Field (now the King County International Airport [KCIA]), Seattle's first municipal airport, was opened in 1928 (Wilma 2001a). Thousands of B-17 and B-29 bombers were produced at Boeing Plant 2 during World War II (King County 2005c). Boeing has had several other facilities near the LDW at various times. World War II also resulted in an increased need for shipyards and maintenance docks for the US Navy. Harbor Island became home to several ship-building, salvaging, and maintenance companies and it became an important port for economic trade. A lead smelter was also built on the island in 1937 (Tetra Tech 1988b).

Construction and cement companies, slaughterhouses, meat packing facilities, and a cannery were also located along the LDW in the 1930s and 1940s (Foster 1945). By 1945, industrial facilities were prevalent along the LDW. Lumber production continued to develop, including the use of pole treating and wood preserving processes (Foster 1945). Metal working facilities, machine shops, shipyards, cement and asphalt companies, manufacturing facilities, and meat packing companies were all common industries during the World War II era and following years. Lime and concrete plants and chemical manufacturing (including paint, glue, resin, and wood preservatives) were also established in the LDW at this time. Industrial development resulted in an increased need for waste disposal, and many casual dumping sites were created. Eventually, some dumping sites became official landfills that received industrial waste, though most remained small informal landfills used by neighboring properties for their waste disposal.

Continuing through the 1950s and 1960s, metal working facilities along the LDW included metal manufacturers and recyclers, galvanization plants, and foundries. Steel, tin, aluminum, titanium, nickel, and lead (at the Harbor Island smelter) were all used in various metal-working industries. With an increase in demand for fuel products along the LDW, several petroleum storage facilities and storage tank farms were established on Harbor Island. From the 1950s to the 1970s, several smaller companies and landfill sites along the LDW were closed and redeveloped as parking lots or other commercial and industrial facilities.

By the 1980s, much of the land adjacent to the LDW was occupied by industry. Aviation industries and Port operations including cargo storage and transport continued to expand during this time. Harbor Island was still used largely for

shipyard activities and petroleum storage. Other metal-working industries remained on Harbor Island and along the LDW during this time period (Sweet Edwards 1985). Wood preserving, chemical processing (including used oil and antifreeze), and other types of manufacturing continued throughout the 1980s. Industry to the south of Harbor Island near Kellogg Island was primarily related to cement and gypsum manufacturing and included several waste disposal sites. In 1998, the Harbor Island lead smelter was permanently closed.

Many of the industries that were present along the LDW in the past are still in operation today. Currently aviation facilities used for the construction, maintenance, and operation of aircraft, Port-owned and privately owned terminals used for cargo shipping and receiving, motor vehicle and marine vessel maintenance, cement production, manufacturing plants, and various other industries all operate along the LDW.

9.3.2 Historical discharge sources

Historically, chemical disposal practices were not regulated as they are today. In many cases, chemicals and other wastes were discharged directly to the LDW or disposed of on upland properties without sufficient containment (Foster 1945; Seattle-King County Department of Public Health 1984). This section provides an overview of information regarding historical sources to the LDW.

9.3.2.1 Historical industrial discharges

Potential sources of contamination along the LDW were documented in several historical reports (Foster 1945; Sweet Edwards 1985; Tetra Tech 1988b). In the report by Foster (1945), the river was divided into three sections for assessment: the Duwamish Waterway, which received predominantly industrial waste, the Green River below the City of Palmer (south-central King County), which received primarily domestic sewage and agricultural waste, and the upper Green River, which received little contaminated discharge of any type at that time.

Wastes generated during the development and ongoing industrial history of the LDW had the potential to be discharged into the waterway. Wastes were transported to the LDW directly through storm drainage systems, as a component of the discharge from sanitary sewer systems, through atmospheric deposition, or direct disposal along the shoreline. Sanitary sewer systems serviced large industrial areas along the LDW. Information on sewer system discharges are discussed in Section 9.3.2.2.

Historically, many industrial facilities used unlined, open-air waste ponds for pre-treatment of slurries and other waste water (Sweet Edwards 1985; Foster 1945). Often the ponds and lagoons were located along the LDW shoreline and contained chemicals that could enter area soil and/or groundwater and eventually enter the LDW. Waste slurries derived from routine maintenance and cleaning of tanks and equipment were often dumped directly into the LDW or into wastewater ponds on a daily to yearly basis, depending on the industry (Foster 1945).

Several lumber yards were located along the LDW in the past, particularly at the upstream reach of the LDW near the Upper Turning Basin. Prior to 1945, considerable amounts of sawdust were disposed of in the LDW (Foster 1945); after that time, sawdust became valued as a fuel product and was kept by most mills. Waste products from lumber mills and sand and gravel companies were sources of solids (in addition to ongoing inputs from the Upper Duwamish/Green River) to the LDW in the past. Additional wastes associated with lumber processing included creosote used in pole dipping, and arsenic and sulfate salts of copper and zinc used to preserve logs. Creosote is known to contain PAHs. Other wood preservatives and resins may have contained TBT (EPA 2003).

Historically, oil and fuel were discharged to the LDW from various industrial activities including ship-building and repair facilities, ship salvaging companies, brick manufacturing, machine shops, pole treating facilities, and hydraulic equipment manufacturing (Foster 1945). Oily wastes were observed seeping from facilities located along the LDW shoreline in the past (Sweet Edwards 1985). The storage, transport, and use of fuel at aviation, railway, motor vehicle, and fuel storage/tank farm facilities were also historical sources of petroleum product wastes that may have reached the LDW through both direct discharge (spills and dumping), transport through storm drain networks, and indirect discharge (migration through soil or groundwater). Historically, recycled or waste oils sometimes contained PCBs from electrical or hydraulic equipment and other sources directly containing PCBs or due to mixing or cross-contamination with PCB oil from those sources (ATSDR 2000). Recycling of these oils often resulted in oil spills (Sweet Edwards 1985).

In the late 1970s, the EPA issued regulations banning the manufacture of PCBs. The use of PCB-containing equipment began to be phased out at this time. The presence of PCBs in waste oils would have diminished after these regulations were introduced in the late 1970s.

Significant uses of PCBs were as a dielectric fluid in transformers and as a component of hydraulic oils; releases to the environmental associated with both of these uses have been well documented. PCBs have also been used in many other products, such as capacitors, heat transfer systems, inks and carbonless copy paper, the ballasts of some fluorescent light fixtures, and paints. PCB contamination along the LDW has also been documented in connection with the waste oil used as fuel in asphalt manufacturing (Ecology 2005a; Windward et al. 2005), as well as associated with caulking and sealant materials used on pavement and asphalt.

Industrial waste discharges originating from manufacturing and metalworking companies often contained a variety of chemicals. Acidic solutions were among the most widely reported chemical releases associated with these industries (Sweet Edwards 1985; Foster 1945). Other wastes included metal scraps, sodium borate, cyanide, and zinc salts (Foster 1945). Chromium, copper, cadmium, and nickel are associated with electroplating (Sweet Edwards 1985). In addition, the Harbor Island

and Asarco smelters emitted lead, arsenic, and other metals to the atmosphere (Huey 2002; Ecology 2007b; Area-Wide Soil Contamination Task Force 2003) (Section 9.3.2.4).

Shipbuilding and marine vessel maintenance is another industry that has been present in the LDW for many years. Metals (including arsenic, chromium, copper, lead, and zinc) and petroleum products were common waste products produced by shipyards activities (Tetra Tech 1988b). Anti-fouling paint applied to boats often contained TBT, copper, mercury, and other metals (SAIC 2007f). Sandblasting, painting, boat repair, and other maintenance activities also had the potential to contribute metals and various petroleum products to the LDW. In addition, solvents, acids, various types of paint, antifreeze, and other chemicals have also been associated with shipyards (SAIC 2007f).

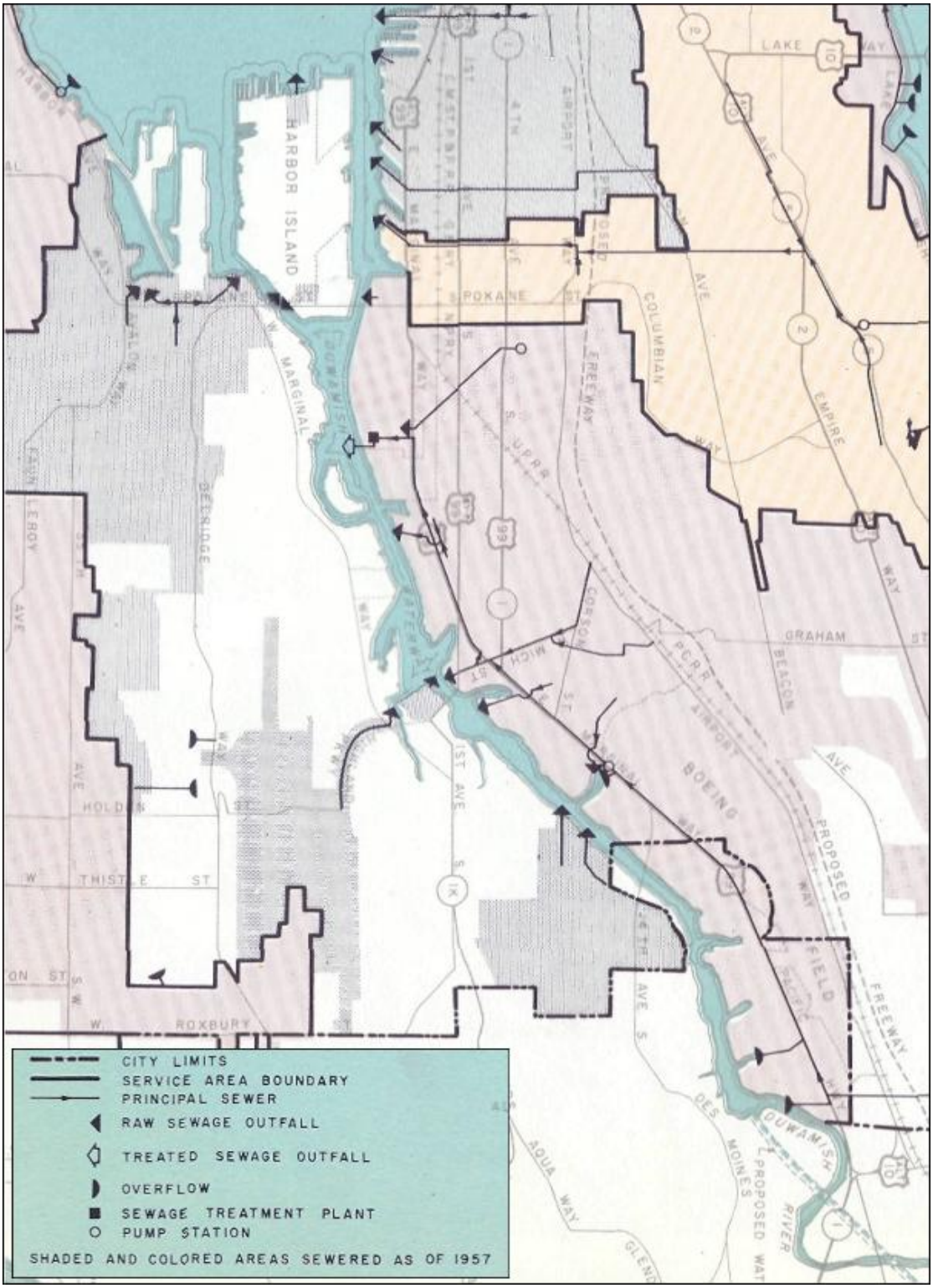
Several waste products are associated with the aviation industry, including alkaline carbide wastes from acetylene generators (Foster 1945), petroleum products used for fuel, and metal wastes produced in the manufacture of aircraft (Foster 1945; Ecology 2006a). Solvents and paints were also associated with aircraft maintenance.

Chemical manufacturing has produced various wastes that entered the LDW through a number of pathways in the past (Sweet Edwards 1985). Used antifreeze is generally known to contain lead and cadmium, and production of adhesives, water-soluble glues, formaldehyde, and wood-preserving resins produced phenol formaldehyde and pentachlorophenol waste (SAIC 2007f). Dioxins and furans are known to be a by-product of pentachlorophenol production. Copper was a waste product associated with the production of vanillin and wood preserving resins (Sweet Edwards 1985). Cement and asphalt manufacturing produced several waste products, such as cement kiln dust, which were often disposed of at casual landfill sites (Sweet Edwards 1985).

Organic wastes originating from various operations were also commonly discharged directly to the LDW in the past. Agricultural operations, slaughterhouses, meat packing facilities, and raw sewage discharges were all sources of organic wastes, including blood, manure, and grease (Foster 1945). Historical sewage discharges are discussed further in Section 9.3.2.2.

9.3.2.2 Historical sewer system discharges

Prior to the creation of Metro (now the King County Wastewater Treatment Division) in 1958, local sewer systems along the waterway discharged raw sewage directly into the LDW. Although the LDW is currently served by a combination of separated, partially separated, and combined sewer systems, historically, most of the LDW area was served by a combined storm/sanitary sewer system. As shown on Figure 9-2, approximately seven combined sewer outfalls and five CSOs discharged to the LDW in 1958. Combined sewer outfalls discharged raw sewage and stormwater directly to the waterway in areas not served by a wastewater treatment plant. CSOs discharged raw sewage and stormwater flows that exceeded the capacity of the treatment plant.



Source: Brown and Caldwell (1958)

Figure 9-2. Outfalls that historically discharged raw sewage to the LDW

The Henderson-E Marginal Way S system collected stormwater runoff and municipal/ industrial wastewater from approximately 5,900 ac (2,388 ha) on the east side of the waterway and conveyed flow to the Diagonal Avenue wastewater treatment plant (WWTP). The Diagonal Avenue WWTP operated from 1938 until approximately 1969 when the plant was closed and flows were diverted to the West Point Wastewater Treatment Plant. In 1958, the total population served by the Diagonal Avenue WWTP was about 30,000 (Brown and Caldwell 1958). Two of the combined raw sewer outfalls and all five of the CSOs in the LDW identified on Figure 9-2 were associated with the Henderson-E Marginal Way S system. The last sewer outfall of this system was connected to the West Point system in 1976.

In 1958, the West Seattle system served an area of about 1,950 ac (789 ha) located along the west side of the LDW (Brown and Caldwell 1958). At that time, none of the raw sewage and stormwater flow from this system was treated. Four raw sewage outfalls discharged to the LDW study area and another three outfalls discharged outside the LDW in the vicinity of West Waterway (see Figure 9-2). In addition to the areas served by the West Seattle system, large areas on the west side of the river, stretching from the southern tip of Harbor Island to approximately RM 2.8, were not connected to the City's sanitary sewer system as of 1958 (Brown and Caldwell 1958). These areas depended on soil leaching systems made up of subsurface tile drains to dispose of septic system tank effluent. However, some private systems used dry wells or directly discharged to roadside ditches or into surface water bodies (Brown and Caldwell 1958).

As mentioned previously, the first WWTP along the LDW, the Diagonal Avenue WWTP, was built by the City of Seattle in 1938 near RM 0.6, approximately 1,200 ft south of the current location of the Diagonal Avenue S CSO/SD (Brown and Caldwell 1958) (Map 9-4). The Diagonal Avenue WWTP provided primary treatment for sanitary wastewater, industrial waste streams, and stormwater collected from the Rainier Beach neighborhood near Lake Washington and the industrial district along E Marginal Way S from the Diagonal Avenue WWTP south to the Upper Turning Basin (the area previously identified as the Henderson-E Marginal Way S system). Industrial facilities along the LDW were not required to redirect industrial waste discharges to this system until the early 1950s.

The plant had a design capacity of 8 million gal. per day (mgd), but up to 90% of the wet weather flow was bypassed directly to the LDW because of limited sewer capacity (Brown and Caldwell 1958). In 1956, the average volume of inflow to the treatment plant ranged from 2.5 to 5.0 mgd. Inflow to the Diagonal Avenue WWTP was limited by regulator structures that bypassed excess flow at three locations (at Slip 6, at the present location of the Norfolk CSO/SD, and at the present location of the E Marginal Way S pump station in Slip 4).

The principal source of organic waste to the LDW was raw sewage that originated from industrial facilities and residential areas along the LDW (Foster 1945). This type

of waste was usually routed through the City's sewer system into the Diagonal Avenue WWTP for example; however, many of the industrial properties along the LDW were too low in elevation for effective use of the gravity fed sewer lines and raw waste streams in this locations were either stored temporarily in open ponds or discharged untreated to the LDW (Brown and Caldwell 1958).

By the late 1950s, industries were required to obtain a wastewater discharge permit from the Pollution Control Commission for management of their industrial and raw sewage waste streams. At that time, 150 industries had industrial discharge permits. A sewerage and drainage survey conducted in 1957 indicated that industrial waste streams that discharged directly into the LDW without pre-treatment could potentially reach a volume of about 33,900 gpd (Brown and Caldwell 1958). The majority of this industrial waste (63%) was associated with metal-related industries. The study indicated that the wastes from metal-plating industries were particularly hazardous and required separate pretreatment.

In the same survey, a total of 14 industries that discharged industrial wastewater (at a rate of approximately 2,500 gpd) into networks that would be proceeded through the Diagonal Avenue WWTP were also identified (Brown and Caldwell 1958). Industrial discharges that passed through the Diagonal Avenue WWTP originated from adhesives and chemical plants, metal plating facilities, beverage bottling operations, cement handling and production, compressed gas operations, food canning, sawmill and wood-working related industries, steel plants, and truck manufacturing operations (Brown and Caldwell 1958).

In 1962, Metro assumed operation of the Diagonal Avenue WWTP until its closure in 1969, when the Elliott Bay Interceptor (EBI) was brought on-line and flow was diverted to the West Point WWTP, which discharges treated wastewater to Puget Sound. The evolution of the wastewater permit programs required industrial properties along the waterway to upgrade their systems and interconnect with the interceptor line.

Starting in 1969, large portions of the public and private sewer lines were connected into the EBI line, and subsequent system overflows were eventually limited to the current County and City CSO locations (see Section 9.4.4.4). Some connections required complex infrastructure improvements; thus, raw sewage was discharged for several years after 1969, including through the Henderson/E Marginal Way S system. The raw sewage outfall (Fox sewer line) into Slip 4 continued to discharge until 1976 (City of Seattle 1998).

9.3.2.3 Significant known historical spills

In 1974, a spill occurred at the GSA dock when a transformer was dropped and cracked while being loaded onto a barge in Slip 1, resulting in the release of approximately 980 L (260 gal.) of high-concentration PCB dielectric fluid (primarily Aroclor 1242) into the LDW (EPA 1975). Two separate dredging operations were conducted to address the sediment contamination that resulted from the spill. An

initial cleanup by EPA in 1974 was estimated to recover approximately 300 L (80 gal.) of PCBs.

A second dredging attempt by EPA and the USACE in 1976 was required to recover PCBs that had spread throughout Slip 1 and into the river channel, in part because of a 20-yr flood that occurred in the winter of 1975-1976. The second cleanup involved hydraulic dredging of sediments containing PCBs in the northwest corner of Slip 1. About 38 million L (10 million gal.) of PCB-contaminated slurry was piped to settling lagoons on the Chiyoda Property (formerly the Diagonal Avenue WWTP property and currently the location of the Port's T-108 (Map 9-4). Most of the slurry was deposited in one of the lagoons located closest to the LDW, while the second lagoon received overflow water from the first lagoon. Water pumped from the lagoons was filtered through a sand and charcoal filter to remove suspended particles and PCBs before discharge to the LDW. It was estimated that the second dredging removed another 640 L (170 gal.) of the 980-L (260-gal.) spill (EPA 1975).

Two other known significant historical spills were documented in the Washington State Pollution Control Commission (WPCC) Progress reports between 1945 and 1957 (WPCC, as cited in Dexter et al. 1985). The Duwamish Manufacturing Company reportedly spilled 5,000 gal. of crude oil from a valve left open on a tank (the exact date and location of the spill is unknown). The site was subsequently cleaned, but most of the leaking oil had already entered the LDW. Another reported incident occurred at the Duwamish/Diagonal Avenue S SD. A large amount (quantity not reported) of oil was found entering the LDW from the SD. The oil had come from the Union Pacific Railroad roundhouse where drained oil had overflowed from a sump, which was intended to prevent the oil from reaching the sewer (WPCC, as cited in Dexter et al. 1985).

9.3.2.4 Atmospheric discharges

Historical atmospheric discharges may have contributed to sediment contamination in the LDW in the past. Sources of these discharges included regional smelter facilities, fossil fuel combustion by motor vehicles and industrial facilities, cement manufacturing plants, and other industrial and urban practices. Air emissions were not commonly monitored or regulated prior to the establishment of the Clean Air Act (CAA) in the 1970s.

Regional Smelter Facilities

The LDW is located in an area that has been designated as having potential soil contamination caused by historical smelter emissions (Area-Wide Soil Contamination Task Force 2003). Two historical regional smelter facilities were potential sources to the LDW. The Asarco copper smelter operated from 1890 to 1985 in Ruston, Washington, northwest of Tacoma. A lead smelter was also located on Harbor Island from the 1930s until 1998, though operations were greatly reduced in the mid-1980s.

Smokestack emissions from the Asarco smelter contained arsenic and lead, and were likely a historical source of these metals to the LDW (Huey 2002). Prevailing winds are a large factor in determining the spread of smelter emissions (Ecology 2001b); for much of the year, prevailing winds in the LDW are from the south (PSCAA 2007c) where the Asarco smelter was located. Heavy metals were distributed through atmospheric deposition to over 1,000 square miles of the Puget Sound basin from the Asarco smelter (Ecology 2007b).

Since the closure of the Asarco smelter, Ecology and several local health departments have initiated studies to outline the extent of the plume "footprint" by testing arsenic and lead concentrations in area soils. Soil samples collected to date show that arsenic concentrations in the LDW drainage basin range from 17 to 30 mg/kg dw (Ecology 2001b). Lead concentrations ranged from 37 to 200 mg/kg dw. MTCA Method A soil cleanup levels for unrestricted land uses for arsenic and lead are 20 and 250 mg/kg dw, respectively. Soil characterization within the plume footprint is still ongoing (Ecology 2007m).

It is estimated that the Harbor Island lead smelter deposited lead through atmospheric deposition over at least 640 ac of land (Area-Wide Soil Contamination Task Force 2003). Emissions from the smelter contributed to lead contamination of the soil on Harbor Island, leading to its listing as an NPL site in 1983 (EPA 1993b). Emissions from the smelter were also likely deposited into the LDW through direct and indirect atmospheric deposition. An air monitoring station formerly located on Harbor Island was used to collect samples to analyze for lead concentrations in the air. Prior to the smelter's closure, lead was often detected in air samples at concentrations above federal standards. Quarterly average lead concentrations over 10.5 $\mu g/m^3$ were reported by PSCAA for the 1981 monitoring year (PSCAA 2006a). By 1998, all operations at the Harbor Island smelter ceased and lead concentrations in air dropped below the Quarterly Federal Standard of 1.5 $\mu g/m^3$. Air monitoring on Harbor Island was discontinued in 1999.

It is likely that both the Asarco smelter and the Harbor Island smelter contributed metals, particularly arsenic and lead, to the LDW through direct and indirect atmospheric deposition in the past. The contribution of metals to the LDW from historical smelter emissions compared to other sources has not been determined and is difficult to assess.

Historical Urban Atmospheric Discharges

In addition to smelter facilities, other historical urban atmospheric discharges have potentially contributed to sediment contamination in the LDW, such as wood burning, industrial processes for heating, burning of fossil fuels including emissions from motor vehicles, power generating facilities, wood treatment, waste incineration, and cement kilns.

Air pollutants commonly associated with urban activities in the past included particulate matter (PM), metals, PAHs, PCBs, and dioxins and furans. PM and PAHs were emitted from the burning of fossil fuels such as coal and oil, used in various industrial processes and for domestic purposes such as transportation and heat. Wood burning and waste incineration were other historical practices that emitted PM, PAHs, dioxins and furans, and metals (PSCAA 2003).

In the past, large quantities of lead were discharged to the atmosphere from the use of leaded gasoline, which was used primarily from the early 1920s through the 1990s, though it was largely phased out by the mid-1970s because of health concerns (Ecology 2001b). Chromium and nickel emissions are also associated with the burning of fossil fuels (PSCAA 2006a).

Other industrial processes that were common in the LDW in the past also may have contributed air pollutants. Arsenic is often emitted to the atmosphere through wood treatment processes and distillate oil combustion, and chromium is emitted by chrome electroplaters (PSCAA 2006a). Cement kiln emissions can contain chromium, mercury, lead, and zinc, as well as dioxins and dioxin-like compounds, PCBs, and PAHs (EPA 2007e). Several of the historical urban sources of atmospheric discharges are still present near the LDW today, though most emissions are now registered and regulated by PSCAA and EPA.

9.3.2.5 Waste disposal, fill areas, and abandoned landfills

Source information from upland disposal or abandoned landfills sites is generally not well characterized because information on the current conditions of these sites is either largely unavailable or has not been thoroughly investigated. This section summarizes readily available information that pertains to upland waste disposal and abandoned landfills in the vicinity of the LDW.

Waste Disposal and Fill Areas

Historical waste disposal and fill areas in the LDW are discussed in Sweet Edwards (1985) (Map 9-4). The focus of this report was to identify potential sources of contamination to groundwater through the leaching of wastes. Prior to the 1970s, it was common practice for industries to store or bury waste products on their property or on nearby vacant lots. Filling of the old Duwamish River and grading of new sites along the LDW occurred until the 1960s (Sweet Edwards 1985). Potential contamination from the fill material is uncertain, because the quality and source of fill material used in the 1920s to 1960s is undocumented and largely unknown. Fill was assumed to be primarily dredge spoils from the main channel, and fill sites were likely near dredge sites to keep transportation costs low (Sweet Edwards 1985).

Abandoned Landfills

At least four former municipal landfills, and likely numerous smaller undocumented landfills, are located in the LDW drainage area (see Map 9-4). Prior to the

establishment of municipal waste dumps, from the 1800s into the first half of the 1900s, residential and industrial solid wastes in the Seattle area were commonly dumped in low-lying and wet areas. Waste disposal in such areas served the dual purpose of providing dumping grounds for garbage and other debris while reclaiming low-lying land for future development. Unknown numbers of such waste sites underlie portions of the city historically comprising Elliott Bay and the Duwamish tidelands.

In 1984, the Seattle-King County Department of Public Health (1984) conducted a survey of some of the larger and better-documented former landfills in the City. This survey concluded that in most cases abandoned landfill sites were adequately capped and not likely to pose a direct-contact risk to people. However, the survey noted a potential for exposure to leachate that has migrated via groundwater into underground piping networks or building excavations. The survey also stated that degradation could occur at abandoned landfills through the generation of methane gas as wastes decompose.

In 1986, the department followed up with a toxicity hazard assessment of some of these landfills (Seattle-King County Department of Public Health 1986). The four landfills reviewed in the LDW drainage basin included the Judkins Park Landfill, the Sick's Stadium Landfill, the 6th Avenue S Landfill, and the South Park Landfill. No significant environmental health problems were identified at the Judkins Park and Sick's Stadium Landfills, and no further study was recommended. No conclusions or recommendations were made for the 6th Avenue S Landfill because of the limited amount of information available for this site. The assessment recommended that groundwater and stream sampling be conducted at the South Park Landfill.

The Seattle-King County Department of Health (1986) stated that for the six sites reviewed for the possible future use of groundwater as a water supply (four of which were in the LDW drainage basin), groundwater contamination at abandoned landfill sites may present public health risks in the future. The study also stated that elevated chemical concentrations in South Park groundwater samples were likely attributable to their location in an industrial area and near highways rather than from the landfill itself (Seattle-King County Department of Public Health 1986).

King County implemented an independent remedial investigation and a quarterly groundwater monitoring program at the South Park Landfill beginning in 1997. Quarterly monitoring was conducted between 1999 and 2002. Vinyl chloride was the only COC identified in downgradient wells at the site during the monitoring events. Long-term monitoring indicated that vinyl chloride concentrations rarely exceeded the MTCA Method B surface water cleanup standards at wells approximately 2,000 ft from the LDW. Vinyl chloride rarely degrades under aerobic conditions and with precipitation infiltrations and the tidal-mixing effect expected during transport through groundwater, it is highly unlikely that vinyl chloride from this site will reach

the LDW at concentrations that could influence sediment conditions (Windward 2003c).

9.4 SOURCE IDENTIFICATION AND CONTROL EFFORTS

The previous section discussed the sources and pathways of historical contamination that have the potential to affect the quality of the environment in the LDW. Identification of current and ongoing sources is equally important to promote the effective implementation of remediation strategies while ensuring that areas being remediated are not recontaminated after cleanup is complete. This section provides a detailed discussion of the present day sources of contamination that could continue to affect sediment quality in the LDW if they are not eliminated or controlled.

This section also discusses the ongoing source identification and control efforts being implemented to address these recontamination concerns. These source control programs are currently being employed by various regulatory and public entities, including the City of Seattle, City of Tukwila, the County, the Port, Ecology, and EPA. These entities use permits, regulations, pollution prevention programs, inspection and maintenance programs, environmental investigations, and pathway assessments in an effort to identify and control ongoing sources of sediment contamination in the LDW.

9.4.1 Available source information

Numerous documents and resources were reviewed and evaluated during completion of the RI to help identify potential sources of contamination in the LDW. These resources included various federal, state, and local environmental databases that document a range of information including suspected site contamination, investigative status, and past cleanup activities. Agency and SCWG representatives provided clarification and additional updated information on the status of sites along the LDW, as necessary. The following section presents an overview of these various resources and the results of the review.

9.4.1.1 Historical documents and available reports

A considerable number of regional and site-specific source investigations and associated control actions have been performed in the LDW study area under a wide range of RCRA, Clean Water Act (CWA), and MTCA actions. Chemical loadings, sources, pathways, and control efforts within the LDW were initially compiled and reviewed by Metro in 1983 (Harper-Owes 1983) based on a synthesis of numerous investigations conducted from 1960 to 1982. A more recent data compilation was prepared by King County as part of the 1999 WQA for the Duwamish River and Elliott Bay (King County 1999e). Most recently, Ecology and its agency partners have published a source control strategy for the LDW (Ecology 2004b) as discussed in Section 9.2.

9.4.1.2 Environmental databases

EDR provided documentation reporting regulator-listed facilities in the LDW area, including those in upland areas adjacent to the LDW and in the LDW drainage basin. The databases included local, state, and federal regulatory and environmental databases, along with information on spills and compliance issues (EDR 2006a).

Tables 9-2 and 9-3 list the various regulatory databases identified in the EDR report (EDR 2006a); information provided in the databases identified in Table 9-2 is summarized in Appendix G. The documentation provided by EDR included information on numerous facilities located within several miles of the LDW study area. Databases were selected for inclusion in the RI based primarily on the potential for the facilities listed on these databases to contaminate sediment within the LDW study area. Tables 9-2 and 9-3 also summarize the rationale for determining the inclusion or exclusion of database information included in Appendix G.

Table 9-2. Databases used to identify facility information on potential ongoing sources to the LDW as detailed in Appendix G

Location in Appendix G	Database	Description	Rationale for Inclusion in Report	Source
Tables G-1 through G-3	National Pollutant Discharge Elimination System Permits	Permits are issued for industrial, municipal, and large construction area discharges to surface water bodies by Ecology under the NPDES program.	Facilities discharge directly to the LDW or upriver of the LDW.	Ecology's Water Quality Permit Life Cycle database (Ecology 2007f, e; EDR 2006a)
Table G-4	King County Industrial Waste Program Permits	King County issues permits or discharge authorizations for industries discharging into its wastewater treatment system.	Facilities discharge directly to the LDW through King County and SPU CSOs.	King County (Tiffany 2006a, b)
Table G-5	Large-Quantity Generators	EPA database that tracks facilities that generate over 1,000 kg of hazardous waste or over 1 kg of acutely hazardous waste per month.	Given the volume of potential hazardous waste generated, LQGs have the potential to be a source of contamination to the LDW.	EDR (2006a) from Ecology database
Table G-6	Confirmed and Suspected Contaminated Sites List	This list is a database used by Ecology's Toxic Cleanup Program to track progress on all confirmed and suspected contaminated sites in Washington State.	Sites with toxic chemicals present or suspected to be present in various environmental media in the vicinity of the LDW.	EDR (2006a) from Ecology database
Table G-7	Leaking Underground Storage Tank; cleanup in progress	Inventory of active LUST incident reports maintained by Ecology.	LUST sites with soil and/or groundwater contamination in the vicinity of the LDW.	EDR(2006a) from Ecology database
Table G-8	Leaking Underground Storage Tank; remediation reported as complete	Inventory of inactive (remediated) LUST reports maintained by Ecology.	LUST sites may have provided a source of contamination prior to site cleanup.	EDR (2006a) from Ecology database
Table G-9	Solid Waste Facility/Landfill	Inventory of solid waste disposal facilities or landfills, both active and inactive, or open dumps that failed to meet RCRA Subtitle D Section 4004 criteria for solid waste landfills or disposal sites.	Releases of toxic waste from disposed materials at SWF/LF facilities in the vicinity of the LDW.	EDR (2006a) from Ecology SWF/LF database
Table G-10	Lists of active and inactive drycleaners	Ecology-maintained list of registered (i.e., active) and inactive drycleaners.	Drycleaners can generate hazardous waste that can contaminate groundwater and soil in the vicinity of the LDW.	EDR (2006a) from Ecology databases
Table G-11	Brownfields	Federal and state lists of Brownfield sites (i.e., abandoned, idle, or underused commercial or industrial properties, where expansion or redevelopment is hindered by real or perceived contamination). EPA and Ecology maintain separate lists of Brownfields sites.	Sites with toxic chemicals present in various environmental media in the vicinity of the LDW.	EDR (2006a) from EPA and Ecology databases

Location in Appendix G	Database	Description	Rationale for Inclusion in Report	Source
Table G-12	Toxics Release Inventory System	TRIS facilities are registered with EPA and identified as facilities that release toxic chemicals to the air, water, and land in reportable quantities under the Superfund Amendments and Reauthorization Act, Title III, Section 313.	Facilities with toxic chemical releases to various environmental media in the LDW surface drainage area.	EDR (2006a) from EPA database
Table G-12	PSCAA's Registered Air Emission Sources	A list of sites registered with the PSCAA as producing emissions of air pollutants.	Facilities with registered air emissions in the vicinity of the LDW.	PSCAA (Nehen 2007)
Table G-13	Ecology Spills Program[a]	Spills reported to Ecology's Environmental Report Tracking System.	Spills in the LDW drainage basin.	EDR (2006a) from Ecology database
Table G-14	Seattle Public Utilities Spill Response Program Records[a]	SPU maintains a record of hazardous material spills involving the City's infrastructure.	Spills in the LDW drainage basin.	Schmoyer (2007d)

[a] Databases include only a partial list of spill information maintained by the US Coast Guard.

CSCSL – Confirmed and Suspected Contaminated Sites List
CSO – combined sewer outfall
Ecology – Washington State Department of Ecology
EDR – Environmental Data Resources, Inc.
EPA – US Environmental Protection Agency
ERTS – Environmental Report Tracking System
LDW – Lower Duwamish Waterway
LUST – leaking underground storage tank
NPDES – National Pollutant Discharge Elimination System
PSCAA – Puget Sound Clean Air Agency
RCRA – Resource Conservation and Recovery Act
SARA – Superfund Amendments and Reauthorization Act
SD – storm drain
SPU – Seattle Public Utilities
SWF/LF – solid waste facility/landfill
TRIS – Toxics Release Inventory System
UST – underground storage tank

Table 9-3. Databases used to identify facility information on potential sources to the LDW but excluded from Appendix G

DATABASE	DESCRIPTION	RATIONALE FOR EXCLUSION FROM THE REPORT	SOURCE
Comprehensive Environmental Response, Compensation, and Liability Information System	EPA-maintained database that contains information on potentially hazardous waste sites as reported by states, local municipalities, and private entities under Section 103 of CERCLA.	Facilities with toxic chemical releases to various environmental media in the vicinity of the LDW. The seven facilities identified on this database are either included previously on other database listings or are outside the LDW drainage basin.	EDR (2006a) from EPA database
Resource Conservation and Recovery Act (RCRA) – treatment, storage, and disposal facilities	EPA-maintained database that provides selective information on sites that store, transport, treat, or dispose of hazardous materials as defined by RCRA.	No RCRA-TSD sites were identified within the LDW drainage basin.	EDR (2006a) from EPA database
National Priorities List	EPA's list of Superfund sites is a subset of CERCLIS that identifies priority sites for cleanup under the Superfund program.	The NPL list is a subset of CERCLIS. The three NPL listed sites included the LDW, Harbor Island, and Pacific Sound Resources (located to the west of the mouth of the West Waterway)	EDR (2006a) from EPA database
Corrective Action Sites	Database maintained by EPA that tracks hazardous waste handlers with RCRA corrective action activity.	Entries in the CORRACTS database primarily include administrative compliance issues and do not have direct implications for source control.	EDR (2006a) from EPA database
RCRA small-quantity waste generators	EPA database that tracks facilities producing between 100 and 1,000 kg of hazardous waste per month (small-quantity waste generators).	Over 1,000 facilities within the EDR search radius were identified as being RCRA SQGs. These categories often refer to generation of materials that are not associated with LDW sediment contaminants. Other databases were relied upon to identify potential sources of contamination.	EDR (2006a) from EPA database
Emergency Response Notification System	Database maintained by the National Response Center, US Coast Guard to track reported releases of oil and hazardous substances.	Ecology and SPU databases were relied upon to identify hazardous materials spills information.	EDR(2006a) from National Response Center database
Hazardous Sites List	Database maintained by Ecology listing a subset of CSCSL sites ranked by the Washington Ranking Method.	The HSL is a subset of the CSCSL; CSCSL sites within the summation area are summarized in Table G-6.	EDR (2006a) from Ecology database

DATABASE	DESCRIPTION	RATIONALE FOR EXCLUSION FROM THE REPORT	SOURCE
Voluntary Cleanup Program sites	List of sites maintained by Ecology that tracks sites that have entered either the VCP or the Independent Remedial Action Program.	Sites listed in the VCP database that are still undergoing cleanup or investigation are also listed on the CSCSL, dry cleaners, or LUST databases, all of which are summarized in Appendix G.	EDR (2006a) from Ecology database

CERCLA – Comprehensive Environmental Response, Compensation, and Liability Act
CERCLIS – Comprehensive Environmental Response, Compensation, and Liability Information System
CORRACTS – Corrective Action Sites
CSCSL – Confirmed and Suspected Contaminated Sites List
ECOLOGY – Washington State Department of Ecology
EDR – Environmental Data Resources, Inc.
EPA – US Environmental Protection Agency
ERNS – Emergency Response Notification System
HSL – Hazardous Sites List
IRA – independent remedial action
LDW – Lower Duwamish Waterway
LQG – large-quantity generator
LUST – leaking underground storage tank
NPL – National Priorities List
RCRA – Resource Conservation and Recovery Act
SPU – Seattle Public Utilities
SQG – small-quantity generator
TSD – treatment, storage, and disposal
USCG – US Coast Guard
VCP – Voluntary Cleanup Program
WARM – Washington Ranking Method

The following geographic parameters were used to identify which property reviews were most relevant to source control in the LDW:

- Facilities located within the LDW surface drainage basin, as defined by King County and the City of Seattle (King County and SPU 2005)
- Facilities within 1 mi of the shoreline of the LDW study area
- Facilities within 1 mi upriver of the Upper Turning Basin

Facilities meeting these criteria are listed in tables in Appendix G and are shown on Map 9-5. Tables 9-2 and 9-3 summarize the types of information in each environmental database, their relevance to source control in the LDW, and the rationale for their inclusion. Sites identified on Map 9-5 include those on Ecology's Confirmed and Suspected Contaminated Sites List (CSCSL), RCRA- and CERCLA-listed properties, registered Brownfield properties, and leaking underground storage tank (LUST) sites. In addition to the databases provided by EDR, business inspection programs conducted by King County and Seattle Public Utilities (SPU) also provided information about potential sources; these programs are described in Section 9.4.4.6.

9.4.1.3 Agency and Source Control Work Group information

The final source of information used to evaluate potential ongoing sources to the LDW included work products and personal communications from Ecology, the City, the County, the Port, PSCAA, USACE, and EPA. Information on upland sites available in existing SCAPs, Data Gaps Reports, and other regulatory information sources (such as the databases listed in Table 9-2), along with direct discussions with Ecology, SCWG, and EPA representatives.

Other readily available information used to summarize the status of source control in the LDW included sediment sampling in CSO/SD systems and road rights-of-way conducted by the City (in conjunction with Boeing at some locations), business inspections conducted by the City and County, a street sweeping pilot program conducted by the City, atmospheric deposition and water quality sampling conducted by the County, air monitoring conducted by PSCAA, and information on the sources, fate, and transport of phthalates gathered by several groups. Readily available information was reviewed, and agency personnel were contacted for additional information when necessary.

9.4.2 Atmospheric loading and assessment programs

Common air pollutants with the ongoing potential to affect LDW sediments through deposition include metals, PCBs, PAHs, dioxins and furans, and phthalates (EPA 2001; Floyd|Snider 2007a). PSCAA records show that over 200 businesses in the Duwamish Valley are registered as active sources of air pollution (listed in Appendix G) (PSCAA 2007a). Motor vehicle traffic on Interstate 5 (I-5) also produces nitrous oxide, black carbon, and other emissions through the burning of fossil fuels (PSCAA 2006a).

Air quality and atmospheric deposition monitoring has been conducted in the vicinity of the LDW by several groups including the PSCAA, Ecology, and the County. Results of these efforts are discussed in the sections that follow. In addition to monitoring, efforts are underway to research the sources, transport, and fate of air pollutants and to reduce emissions through regulations and voluntary programs. The Sediment Phthalates Work Group (SPWG) is one such group (see Section 9.4.2.3). Emission reduction programs and additional regulations are discussed in Section 9.4.2.4.

9.4.2.1 Particulate matter

PM refers to solid particulates and liquid droplets suspended in the air. There are two types of PM: fine particulate matter (FPM) referring to particles less than 2.5 µm in diameter ($PM_{2.5}$), and coarse particulate matter (CPM) including particles between 2.5 and 10 µm in diameter (PM_{10}) (PSCAA 2006a).

FPM is generated through the burning of wood, fossil fuels, and other materials and can also be formed when pollutant gases react in the atmosphere. In the Puget Sound region, FPM is the air pollutant of greatest concern for human health (PSCAA 2006d); it can also provide a pathway for the transport of chemicals (Floyd|Snider 2007a). Motor vehicles and wood burning are two major sources of FPM in the region (King County and SPU 2005; PSCAA 2006d).

EPA's 24-hour national ambient air quality standard for FPM is 35 µg/m^3 and the annual standard is 15 µg/m^3 (EPA 2006c). The 3-year average 98th percentile daily concentrations of FPM in the Duwamish area were approximately 34 µg/m^3 (as reported in 2001) and 29 µg/m^3 (as reported in 2005) (PSCAA 2006a). All FPM concentrations reported for the Duwamish area between 2001 and 2005 were below the federal standard (PSCAA 2006a).

Dust from roadways and construction sites, and crushing or grinding operations are sources of CPM to the atmosphere. Three-year average daily maximum CPM concentrations in the Duwamish area ranged from approximately 65 to 80 µg/m^3 between 2000 and 2005, and were below the federal standard of 150 µg/m^3 (PSCAA 2006a; EPA 2006c). CPM concentrations in the Duwamish area have decreased since the 1990s (PSCAA 2006a).

9.4.2.2 Toxic air pollutants

Toxic air pollutants are those that have the potential to adversely affect human health. The PSCAA considers over 400 chemicals to be toxic air pollutants, and the EPA regulates emissions of 188 air pollutants (EPA 2007a). Toxic air pollutants include PAHs, PCBs, phthalates, dioxins and furans, and several metals. Many of these chemicals have the potential to affect sediment quality if they migrate to the LDW through direct or indirect atmospheric deposition.

Common emission sources of air pollutants include industrial facilities, chemical manufacturing plants, solid waste incinerators, motor vehicles, marine vessels, trains,

and wood smoke (PSCAA 2006b, c). Use of solvents, degreasers, and other chemicals can also release air pollutants (EPA 2007a).

Ecology monitors several air pollutants at the Georgetown monitoring station in the Duwamish basin, and results from 2000 to 2001 were presented in PSCAA's *Final Report: Puget Sound Air Toxics Evaluation* (PSCAA 2003). During this time period, ambient concentrations in air at the Georgetown station were 0.001 $\mu g/m^3$ for arsenic, 0.003 $\mu g/m^3$ for chromium, 0.009 $\mu g/m^3$ for lead, and 0.003 $\mu g/m^3$ for nickel. The concentration for lead was below the quarterly average federal standard of 1.5 $\mu g/m^3$; standards do not currently exist for the other metals monitored. PSCAA is in the process of developing an air toxics emissions summary for data collected in 2005.

The County has collected two phases of atmospheric deposition data in the LDW basin. The first phase was collected between January and May 2005, and deposition flux values were calculated for phthalates and PAHs. The second phase was conducted between October 2005 and December 2006 and deposition flux values were calculated for PCB Aroclors in addition to phthalates and PAHs. The following sampling stations were used: Beacon Hill, Duwamish Valley (E Marginal Way S), Georgetown, South Park Community Center, and KCIA (in Phase 2 only). The samples were collected using a passive atmospheric deposition sampler that samples for total deposition (i.e., "dry" and "wet" deposition).

Atmospheric deposition flux values indicating mass deposition/area/time were provided for each chemical. Results of the Phase 1 testing indicated the following range of atmospheric flux values (King County and SPU 2005):

- BEHP: 0.26 to 3.5 $\mu g/m^2/day$
- BBP: 0.19 to 1.5 $\mu g/m^2/day$
- Individual PAHs: 0.01 to 0.3 $\mu g/m^2/day$

A comparison of results from the Phase 1 study to data from other areas (British Columbia, Denmark, the greater Puget Sound area, and the Great Lakes region) indicated atmospheric deposition flux values from the LDW were typical of other urban environments, and that there may be a regional or even global atmospheric background concentration for some PAHs and phthalates (King County and SPU 2005). More research would be necessary to investigate the background condition further.

Results of the Phase 2 testing indicated the following range of detected atmospheric deposition flux values:

- BEHP: 0.26 to 12 $\mu g/m^2/day$
- BBP: 0.19 to 7.0 $\mu g/m^2/day$
- Individual PAHs: 0.003 to 3.5 $\mu g/m^2/day$
- Total PCBs: 0.011 to 0.064 $\mu g/m^2/day$

During the Phase 2 sampling, PCBs were also detected at most of the stations; however, they were not detected for every sampling round and most of the detections were very close to the laboratory method detection limit (MDL). MDLs for PCBs ranged from 0.011 to 0.063 µg/m^2/day. The highest detected total PCB atmospheric deposition flux result was 0.064 µg/m^2/day (at the Georgetown station), and the lowest detected total PCB atmospheric deposition flux result was 0.011 µg/m^2/day (at the South Park Community Center station) (Tiffany 2007a). PCBs were not detected at the Beacon Hill station. The remaining four stations had at least one sampling round with a detected value for total PCBs (Tiffany 2007c).

Results of the atmospheric deposition monitoring for the LDW basin indicate atmospheric deposition may contribute phthalates, PAHs, and PCBs to the LDW; however, the relative contribution of atmospheric deposition compared to other sources has not been evaluated. Currently, King County has no plans for future monitoring in the Duwamish Valley (Tiffany 2007b). No dioxin and furan monitoring was conducted for this program.

9.4.2.3 Sediment Phthalates Work Group

The SPWG is a voluntary work group that includes representatives from the City of Seattle, the City of Tacoma, the County, Ecology, and EPA (Ecology 2007n). The goals of the SPWG are to better understand the route of transport of phthalates to sediments, and to understand the impact phthalates have on living things. Using existing information on phthalates, the SPWG is working to identify data gaps, potential sources of phthalates, and source control and treatment options for phthalates that are present in the environment (Ecology 2007n). The SPWG will use public outreach to communicate their findings.

The categories of information the SPWG initially set out to research were phthalate occurrence, phthalate risk and receptors, source identification, source control and treatment, and federal and state regulations relevant to phthalate contamination (proposed scope of work and responsibilities for the SPWG, in Floyd|Snider 2007a). One of the SPWG's goals was to provide recommendations for preventing phthalate recontamination of the LDW after cleanup (Floyd|Snider 2007a).

The SPWG concluded that phthalates are widespread in urban and other developed environments and that they are ubiquitous in water, soil, sediment and air (Floyd|Snider 2007a). The SPWG developed a basic working model of how phthalates from nonpoint sources may reach the LDW sediments (Floyd|Snider 2007b). This model has not yet been peer-reviewed or otherwise validated. The basic premise is that phthalates initially enter the environment primarily through off-gassing from manufactured products. Once in the atmosphere, they attach to PM and are distributed to land and water surfaces. They are often transported to water bodies like the LDW through stormwater runoff. Numerous sources of phthalates exist. An earlier phthalate source sampling project conducted in 2003 identified some common products that contain phthalates (Section 9.4.4.5). However, this was not a

comprehensive study but was aimed at identifying products that contained high concentrations of phthalates, in the hopes that the SCWG could identify which products may represent potential ongoing sources of phthalates.

Phthalates from nonpoint sources can accumulate in sediment through both direct and indirect atmospheric deposition, and atmospheric deposition has been identified as the pathway most likely contributing the majority of phthalates to sediments (Floyd|Snider 2007a). However, point sources of phthalates (i.e., direct discharges from industrial processes or wastes) may also exist. The SCWG has not yet identified any ongoing discharges, but historical discharges to the LDW have been documented. For example, a facility called Janco-United, Inc., formerly located in the 4400 block of 4th Avenue S, disposed of degreasing compounds and other wastes associated with janitorial supply operations that allegedly contained phthalates (Ecology 2004a). These wastes were initially discharged directly to the ground and then through an SD that discharged to the LDW; the discharges occurred for several years in the early 1980s.

The two phthalates most frequently detected in sediments, BEHP and BBP, are commonly attached to PM in the air. When indirect atmospheric deposition occurs, phthalates deposited on surfaces throughout an entire watershed can be carried through surface water run-off (including CSO and SD discharges) to discrete sediment areas where they then accumulate (Floyd|Snider 2007a). Although atmospheric deposition and surface water runoff are the pathways by which phthalates reach sediments, the concentrations of phthalates in these media do not generally trigger regulatory responses based on human or ecological health risks. In sediment, however, phthalates can accumulate at concentrations above sediment cleanup standards.

Animals higher up the food chain, including humans, do not appear to be at risk from phthalates. The SPWG has recommended that sediment cleanup actions not be delayed due to the potential for phthalate recontamination (Floyd|Snider 2007b). Methods for reducing and controlling sources of phthalates include using substitutes for phthalates in plasticized goods, removing particulates from the atmosphere, and reducing stormwater flow runoff by increasing pervious surfaces within watersheds.

The SPWG made several recommendations (Floyd|Snider 2007b), including:

- Phthalate recontamination at sediment cleanup sites should be managed with site-specific operation and maintenance plans.
- Other contaminants that are transferred through the air to stormwater to sediment pathway should be studied in order to further validate the SPWG's understanding of this pathway and to reduce additional contaminants (beyond phthalates) that reach sediment through this pathway.

9.4.2.4 Air Emissions Reduction Programs

EPA regulates air pollutants through the CAA (42 USC § 7401 et seq.), and local regulations include the Washington CAA (WAC 173-401) and PSCAA Regulations I, II, and III. PSCAA's regulations provide acceptable source impact levels (ASILs) for

approximately 650 chemicals, which are used to evaluate the potential for a single emission source to impact air quality. PSCAA requires dischargers of toxic air pollutants to use best available control technology (BACT), described in their regulations, to minimize emissions of these chemicals and protect human health. Currently the Puget Sound region is in compliance with federal air quality standards (PSCAA 2007b). Except for FPM and ozone, concentrations of criteria air pollutants in Puget Sound have been in compliance with regulatory standards for nearly a decade (PSCAA 2006a).

Current regulations on air quality are generally based on protecting human health and may not be adequate to address source control concerns associated with atmospheric deposition. Several local groups have initiated emissions reduction programs that go beyond regulatory requirements to further control the release of atmospheric pollutants.

The Port is helping to lead the Puget Sound Maritime Air Forum (PSMAF), an organization of private and public entities with regulatory responsibilities related to air emissions. The PSMAF is working to identify, quantify, and reduce maritime sources of air pollutants (PSMAF 2007). The primary goals of the forum are to ensure that the Georgia Basin-Puget Sound area continues to meet ambient air quality standards and to reduce port-related air quality impacts on human health and climate change (PSMAF 2007). Through the use of cleaner burning fuels, cleaner engines, and other technology, PSMAF is working to reduce FPM emitted by ships at berth by 70% by 2010. The group is also working to reduce emissions from trucks, trains, and other harbor craft used in Port activities.

The PSMAF has completed an air emissions inventory to identify air pollutants originating from diesel equipment used in maritime operations in the Puget Sound region (Starcrest Consulting Group 2007). PM and VOCs were two types of pollutants included in the inventory. The emissions inventory estimated that 1,270 tons per year (tpy) of VOCs are emitted from maritime sources, while 19,200 tpy are emitted from non-maritime sources; 220 tpy FPM are emitted through maritime emissions, while 3,600 tpy are emitted from non-maritime emissions; and 170 tpy diesel particulate matter (a type of FPM) are emitted through maritime emissions, while 260 tpy are emitted through non-maritime emissions (Starcrest Consulting Group 2007). Data provided in the inventory will be used to prioritize pollution reduction efforts.

The PSMAF is part of PSCAA's Diesel Solutions Program, which was initiated in 2001. The program works to reduce diesel emissions from numerous sources, including the construction industry, railways, and transit vehicles (PSCAA 2007b). Reducing diesel emissions will reduce FPM and can play a role in the reduction of other air pollutants that are transported through the atmosphere when they adsorb to FPM.

9.4.3 Surface water inputs

The Green/Duwamish River drainage area is approximately 1,466 km^2 (566 mi^2) and is part of WRIA 9 (Kerwin and Nelson 2000). According to information on WRIA 9, the Green/Duwamish River flows through four subwatersheds: the upper Green River, the middle Green River, the lower Green River, and the Duwamish Estuary (Map 2-2). The LDW is located within the Duwamish Estuary subwatershed. The flow of water to the LDW from upriver provides a potential source of chemicals to the study area. Upstream data is presented in Section 7.

Tributary creeks within the LDW drainage basin and direct surface water runoff are also potential sources of chemical inputs to the system. Two main creeks within the Duwamish River Estuary subwatershed, Puget Creek and Hamm Creek, drain into the LDW. The following sections discuss the potential upriver sources, tributary inputs, and direct surface water runoff to the LDW.

9.4.3.1 Upper Duwamish/Green River

Both historical and current land use practices in the Green/Duwamish River watershed provide the potential for chemicals to enter the river upstream of the LDW study area. These potential upstream sources include:

- Fecal coliform bacteria from human sewage discharges and animal wastes (both domestic and wild) (Herrera 2006)
- Pesticide and fertilizer runoff from agricultural operations
- Chemicals from various sources, including but not limited to roadways, industrial and commercial facilities, and surface water runoff from mining operations
- Chemicals deposited by atmospheric deposition
- Contaminated sediment from logging and sawmill operations and urban development throughout the watershed

Historically, the Duwamish/Green River watershed consisted almost entirely of forests, wetlands, and floodplain sloughs (Collins and Sheikh 2005). With the arrival of settlers, towns and agricultural operations were established throughout the river valley in the mid 1800s (Sato 1997; King County 2005f). Logging, sawmill, and mining operations were scattered throughout the Green River watershed, particularly in the upper and middle Green Rivers. Meat packing industries were located along the middle Green River, and manure, blood, and floor washings were often discharged directly into the river (Foster 1945). Sedimentation from logging and mining operations, as well as human sewage, dairy farm drainage, and household garbage were also sources of contamination to the Green/Duwamish River (Collins and Sheikh 2005; Foster 1945).

Mines and mining towns, including the town of Black Diamond, the Royal Reward Mine, and the St. Helens mining district, were established in the upper and middle

Green River watersheds. Though most of these mines are either no longer in use or are mined only to a limited degree, mine drainage containing copper, arsenic, mercury, and iron may enter tributaries to the Green River and still has the potential to contaminate water and sediment in the Green River (Ecology 2002b). The following section discusses the potential for these types of creeks to be an ongoing source of chemical and microbial inputs to the system.

Forestry remains the primary land use in the upper Green River watershed (King County 2004). This watershed is protected under a watershed management plan as the drinking water supply for the City of Tacoma (Tacoma Water 2007). In the middle Green River watershed, major land uses include forested land (approximately 27%), residential land (approximately 50%), and agricultural land (approximately 12%) (King County 2005f). In the lower Green River watershed, expansive urban development has been the most significant land use change from historical conditions, with approximately 50% of the land now residential, and 27% of the land developed for commercial and industrial use (King County 2005f). Agriculture, mixed land use, and park areas make up the remainder of the lower Green River watershed.

Although the upper and middle Green Rivers generally have good water quality, the lower Green River frequently exceeded Class A standards for water quality (temperature, dissolved oxygen, fecal coliform bacteria, and mercury) (Washington Environmental Hearings Office 1996). The lower Green River and the LDW have been placed on Ecology's biannual Category 5 list under Section 303(d) of the federal CWA for fecal coliform bacteria, dissolved oxygen, and pH (Ecology 2005b).

In 2003, King County conducted a WQA of the Green and Duwamish Rivers (Herrera 2005). The study spanned the area from the Howard Hanson Dam in the upper Green River watershed to the mouth of the Duwamish River at Elliott Bay, and included locations in the main river systems and in selected tributaries. One sampling station was located in the upper Green River subwatershed, nine were located in the middle Green River subwatershed, six were located in the lower Green River subwatershed, one was located at the lower Green River/Duwamish River boundary, and one was located in the Duwamish River Estuary subwatershed. This assessment included analyses of nutrients, metals, and organic analytes (Herrera 2005). Nutrients are excluded from the discussion below because they were not assessed in the ERA.

Total and dissolved concentrations of arsenic, cadmium, chromium, copper, lead, mercury, nickel, selenium, silver, and zinc were compared to Washington State surface water quality standards (WQS) (Herrera 2005). Only mercury and zinc concentrations were above state standards at some stations. Of the 86 samples collected during base flow, the concentration of dissolved mercury was above the chronic standard in one sample each from the Duwamish River subwatershed and the lower Green River subwatershed. The concentration of dissolved mercury was not above the standard during storm flow at any of the sampling locations. Of the 58 samples analyzed for dissolved zinc, the concentration of zinc was above acute WQS in one storm-flow

event sample from Mill/Springbrook Creek (tributary entering at the lower Green River/ Duwamish River boundary) (Herrera 2005). The concentration of dissolved zinc was not above the standard during base flow at any of the sampling locations.

During this study, aluminum, dissolved arsenic, cadmium, chromium, lead, mercury, dissolved nickel, and zinc concentrations did not vary significantly among upriver and downstream locations (Herrera 2005). Concentrations of total arsenic were statistically higher in downstream locations relative to upriver locations within the Green-Duwamish system. Dissolved copper and total nickel concentrations also increased significantly with distance downstream during storm flow conditions, but not during base flow conditions; the increased concentrations remained below applicable WQS (Herrera 2005). Although not reported in the 2003 WQA, a previous water quality investigation conducted in the Green/Duwamish River system between 2001 and 2002 had indicated a trend of increasing total aluminum, copper, lead, and zinc concentrations with distance downstream, and decreasing dissolved aluminum concentrations with distance downstream during storm flow conditions (Herrera 2005).

Nutrient and metal samples were collected using autosequential and autocomposite methods and occasionally as grab samples. Samples analyzed for organic compounds were only collected using grab sampling, and therefore less information is available compared to metals and nutrients (Herrera 2005). Organic compounds analyzed included PCB Aroclors, PAHs, SVOCs, phthalates, VOCs, organophosphate pesticides, and chlorinated herbicides. VOCs and PCB Aroclors were never detected; however, the detection limits for PCB Aroclors were noted to be greater than the Washington State surface water quality chronic standards for total PCBs (detection limits ranged from 0.04 to 0.06 µg/L).

Phenol was detected at low concentrations (less than 0.09 to 0.67 µg/L) at all sampling locations except for one location in the lower Green River where phenol was not detected. Phthalates and many PAHs were rarely detected; detection limits were not identified in source documentation. Multiple PAHs were detected in at least three samples from the Black River (a tributary that enters at the lower Green River/Duwamish River boundary). Organophosphate pesticides were not detected, though the detection limits were greater than the chronic Washington State surface WQS for some pesticides (detection limits for organophosphate pesticides ranged from 0.29 to 0.52 µg/L). Of the chlorinated herbicides, atrazine was detected in one sample (0.98 µg/L in a storm flow sample) from Newaukum Creek (a tributary to the middle Green River), 2,4-D was detected in two samples from Springbrook Creek, a tributary to the lower Green River (0.1 µg/L in base flow and 0.37 µg/L in a storm flow sample) and in two samples from Newaukum Creek (0.08 µg/L and 0.19 µg/L in a storm flow sample), and dichlorprop was detected in one sample from Newaukum Creek (detection limits for chlorinated herbicides ranged from 0.015 to 0.046 µg/L) (Herrera 2005).

Water quality index scores for the Green River decreased from upriver to downstream, and concentrations of most analytes increased from upriver to downriver (Herrera 2005). The water quality index is a number intended to rank the water quality in certain areas. A higher water quality index score indicates better water quality (Herrera 2005). Water quality index scores were calculated for each of the Green/Duwamish River tributaries. Many of the tributaries were ranked as being of high concern for water quality primarily because of high total nitrogen, total phosphorus, and fecal coliform bacteria concentrations, and because of low dissolved oxygen concentrations. Urban, residential, and agricultural land use in the drainage basins of these creeks and streams were directly associated with the reduced water quality index scores (Herrera 2005). The water quality index at the Duwamish River Estuary subwatershed sampling station (Hamm Creek) was ranked as being of moderate concern.

King County performed water quality statistical and pollutant loading analyses on the data collected in 2002 and 2003. The loading analyses were used to derive correlations between constituent loadings and land use patterns in the Green River drainage basin (Herrera 2007). The loading analyses showed a correlation between agricultural lands and copper loading. Forested lands were not correlated with metals loading during base flow conditions but storm events could influence metals loading within those areas. The loading analyses also indicated that there was a correlation between the presence of roads and impervious surfaces and zinc and fecal coliform bacteria loading. Commercial areas were positively correlated with metals loading. Loading from low-density residential areas was highly variable, but negatively correlated with metals loading. Within the developed basins, metals loading was associated with storm water. In general, the ratio of median storm-to-base flow concentrations of metals was greater than or equal to 1, but the ratios for dissolved metals were particularly high in Hamm Creek (4.5 for copper, 5.0 for mercury, and 4.0 for zinc) (Herrera 2007). These results indicate that chemicals entering the LDW from upriver and local sources may have long-term implications for source control.

9.4.3.2 Creeks

Hamm Creek and Puget Creek, two significant open-channel systems that discharge to the LDW, drain combined industrial and residential neighborhoods and provide potential pathways for chemicals to reach the LDW.

Puget Creek is a surface water body that includes three convergent forks. The Puget Creek natural drainage area is approximately 294 ac; however, additional stormwater conveyances (e.g., ditches) increase the drainage area to approximately 430 ac (Rheaume 2007). There is generally year-round base flow to the north of Brandon Street, but the upper reaches are typically seasonal (Rheaume 2007). The creek primarily drains runoff from the surrounding residential neighborhood before it enters the SD system at W Marginal Way SW and flows into the LDW near T-105.

Hamm Creek is a seasonal tributary to the LDW and drains approximately 1,408 ac (Kerwin and Nelson 2000). Hamm Creek flows through urban residential and portions of the West Duwamish Greenbelt before discharging to the LDW at approximately RM 4.2 (Kerwin and Nelson 2000).

The Hamm Creek drainage basin is primarily residential (850 ac), with 316 ac of open space, including both the Rainier Golf and County Club and Glen Acres Golf and Country Club. Open space is also occupied by public rights-of-way and some second-growth forest. The lower portions of the creek are heavily impacted commercial and industrial areas (242 ac). Water quality in all three forks of Hamm Creek has shown evidence of degradation (Kerwin and Nelson 2000). During base flow conditions, elevated pH values have been reported. These pH values are thought to be related to a cement kiln dust pile near S 96th Street and 10th Avenue (Kerwin and Nelson 2000).

9.4.4 Direct discharge systems

For the purposes of this report, direct discharge systems are defined as those that enter the LDW via a distinct pipe or outfall. In general, direct discharge systems include municipal or other publicly owned storm drain systems, municipal sanitary and combined sewer systems, and privately owned and managed storm drain systems. The main classifications of direct discharges include CSOs, EOFs, and public and private SDs.

In the area of the LDW, these discharge systems transport stormwater collected from industrial, commercial, and residential neighborhoods. These drainage systems also have the potential to transport treated wastewaters (from private treatment systems) and raw sewage during heavy rainfall events.

In May 2003, SPU conducted a low-tide outfall survey for the LDW study area to locate, inspect, and collect data on the horizontal and approximate vertical locations of apparent outfalls in the LDW. No attempt was made, however, to verify whether each and every pipe observed during the survey was in fact an outfall. The survey also collected information on the size and composition of the outfall piping and estimated the rate of the flow (in gpm) exiting the pipe at the time of the survey, when feasible. The survey area included the east and west shorelines of the LDW from the south end of Harbor Island to approximately RM 5.0 (near the Norfolk CSO/SD) (Herrera 2004a).

For completion of the RI, the most up-to-date and readily available outfall-specific information was synthesized, including information on discharges identified in the Phase 1 RI, the 2003 outfall survey, and updated outfall information from Ecology, EPA, City of Seattle, City of Tukwila, Boeing, Port, and County records. Through this effort, approximately 250 individual outfalls were identified within the LDW study area. To assist in this effort, Boeing also conducted an internal records review to further evaluate the current status and ownership of the outfalls on their properties along the LDW. Boeing completed an in-field inspection of the shoreline along Boeing

Plant 2 to identify and confirm outfall locations noted in the 2003 survey. During this effort, Boeing identified additional outfall locations that had not been reported in the 2003 survey and noted other locations that were either abandoned or improperly classified as outfalls by Herrera (2004a).

Based on this information, the current status, use, and ownership of the outfalls discharging to the LDW were identified. These outfalls were identified and categorized using the classification nomenclature presented in Table 9-4. Maps 9-6a through 9-6f present the identified outfall locations, their classification, and ownership status. Detailed outfall-specific information is included in Appendix H.

Table 9-4. Outfall nomenclature and descriptions

NOMENCLATURE	DESCRIPTION
CSO	Overflow point on a publicly owned and maintained combined sewer system. Combined sewers convey both sanitary wastewater and stormwater runoff. Overflows generally occur only during large storm events when the capacity of the combined sewer is exceeded and not all flow can be successfully conveyed to a treatment plant. Under these conditions, excess flow is discharged to a nearby receiving water body to prevent sewage backups.
EOF[a]	Overflow point on a combined or sanitary sewer, generally located at a pump station. Discharges are not storm related. Overflows occur as a result of mechanical failure, pipe obstruction, or power failure. Pump stations in LDW are equipped with back up generators.
CSO/SD or EOF/SD	Shared outfall that serves as both a combined sewer overflow or emergency overflow and storm drain system.
Public storm drain	Storm drains that are required to have coverage under an NPDES municipal stormwater permit (i.e., City of Seattle, King County, Port of Seattle, and WSDOT).
Private storm drain	Storm drains that collect storm and surface waters from areas in addition to those mentioned above. Discharges from these facilities may be covered under the general industrial NPDES permit, individual stormwater permits, or other site-specific requirements governing discharge to surface water bodies.
Stream, channel, or swale	Locations of open-air, convergent discharges into the waterway.
Abandoned outfall	Outfalls that are plugged or have existing documentation to show that the upland pipe has been disconnected.
Not an outfall	Location previously identified as an outfall in the 2003 survey; subsequent in-field investigation determined the location was not an outfall.
Pipe of unresolved origin and/or use	Pipe identified as an outfall by Herrera (2004a) but with an undetermined origin and/or use.

[a] This designation is also used to identify an overflow feature from a reservoir, as appropriate. Reservoir overflows are not associated with discharges from the combined or sanitary sewer system.

CSO – combined sewer overflow

EOF – emergency overflow

LDW – Lower Duwamish Waterway

NPDES – National Pollutant Discharge Elimination System

SD – storm drain

WSDOT – Washington State Department of Transportation

9.4.4.1 Storm Drain Systems

The LDW study area is served by a combination of public and private storm drain systems. Many of the privately owned storm drain systems interconnect with public systems before discharging to the LDW. In some cases, primarily for properties adjacent to the waterfront, private storm drains discharge directly to the LDW. For the purpose of this document, a public storm drain is defined as any system that is required to have coverage under a National Pollutant Discharge Elimination System (NPDES) municipal stormwater permit (i.e., systems owned and operated by the City of Seattle, Port of Seattle, King County, and WSDOT). All other drainage systems are classified as private.

Storm drains that discharge to the LDW primarily transport urban runoff generated during precipitation and storm events. A wide variety of chemicals can potentially become dissolved or suspended as water contacts and flows over land surfaces. Urban areas have the potential to accumulate particulate materials, dust, oil, asphalt, rust, rubber, metals, pesticides, detergents, and other chemicals resulting from urban activities. These materials can then be flushed into storm drains during wet weather.

An estimated 9,347 ac of land drain to the LDW. Land use in the basin is summarized in Table 9-5. Table 9-6 presents LDW basin statistics based on geographic area and system ownership. Recent analysis by the City and County estimates approximately 4,000 mgy (including volumes from Hamm and Puget Creeks) discharges in the LDW through this storm drain network (SPU 2007).

Table 9-5. Land use in the LDW drainage basin

Land Use	Area (ac)	Percentage of Total Area
Commercial	625	7
Industrial	2,591	28
Multi-family residential	303	3
Parks, open space	1,350	14
Single family residential	1,862	20
Right-of-way	2,616	28
Total	**9,347**	**100**

Source: (SPU 2007)

LDW – Lower Duwamish Waterway

Table 9-6. Relative size of sub-basin discharges to the LDW

SUB-BASIN	AREA (ac)	PERCENTAGE OF TOTAL AREA
City of Seattle municipal storm drain system	5,694	61
Public storm drains outside of City of Seattle[a]	2,220	24
Other storm drain systems and waterfront discharges within the Seattle city limits[b]	920	10
Private storm drain systems and waterfront discharges outside of Seattle city limits[c]	512	5
Totals	**9,347**	**100**

Source: SPU (2007)

[a] Includes storm drains owned and operated by the City of Tukwila and other public drainage networks in unincorporated King County.

[b] Storm drains located within Seattle that do not connect to the City's municipal storm drain network (including systems owned by the County, the Port, and other private owners along the LDW.

[c] Storm drains outside Seattle serving areas immediately adjacent to the waterfront, owned and operated by waterfront property owners.

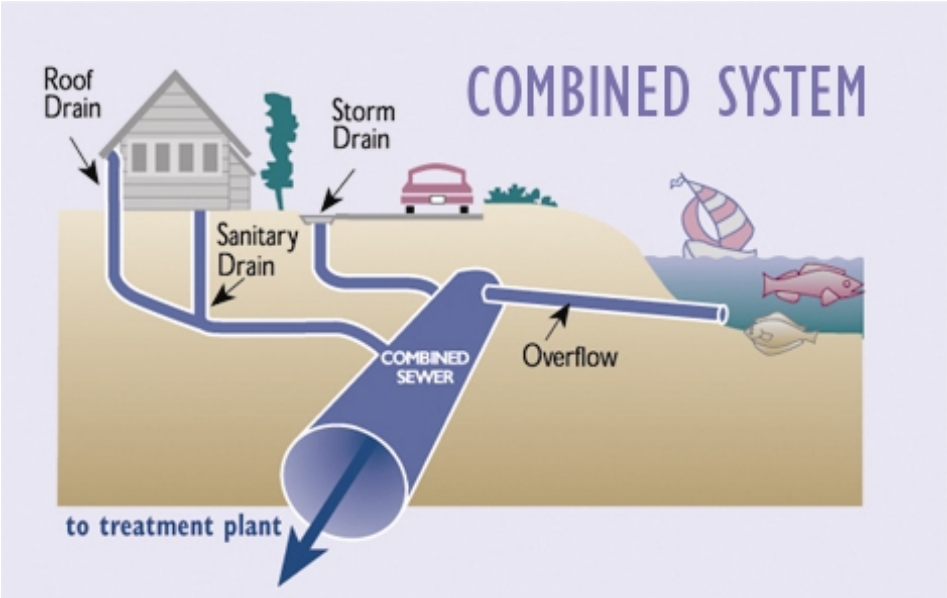
LDW – Lower Duwamish Waterway

During periods of rainfall, stormwater may enter the LDW directly from roadways or properties adjacent to the waterway or through these storm drain systems. Chemicals present on pervious and impervious surfaces (e.g., roadways, parking lots, residential yards, industrial yard areas) can be suspended or dissolved in the stormwater and transported to the LDW via direct surface runoff or via public and private storm drain systems within the study area. These drainage networks also provide a pathway for spills and leaks to reach the LDW.

9.4.4.2 Combined sewer overflows and emergency overflows

Discharges from CSOs occur during rainfall events when the CSO system capacity is insufficient to transport the volume of both sanitary wastewater and stormwater flows. When this capacity is exceeded, excess flow is discharged to the LDW through an overflow structure (Figure 9-3). This flow can consist of a combination of untreated sanitary wastewater and stormwater runoff.

The City and the County can potentially discharge combined untreated sanitary wastewater and stormwater runoff through CSOs to the LDW study area during these storm events. The City owns and operates the local sanitary sewer collectors and trunk lines, while the County owns and operates the larger interceptor lines that transport flow from the local systems to the West Point WWTP. The City's CSO network has its own NPDES permit (Appendix G); the County's CSOs are administered under the NPDES permit established for the West Point WWTP.



Source: King County (2006a)

Figure 9-3. Schematic diagram of CSO system

CSOs are a recognized source of water pollution as a result of temporary increases in bacterial counts in the LDW, aesthetic degradation of the shoreline, and public health concerns in areas where human contact with the discharged water is possible. Since the 1970s, the County and the City have been implementing CSO control projects to improve water quality.

An EOF is a discharge that can occur from either the combined or sanitary sewer systems that is not necessarily related to storm conditions and/or system capacity limitations. EOF discharges typically occur as a result of mechanical issues (e.g., pump station failures) or when transport lines are blocked. EOF events are not covered under the City's or County's existing CSO wastewater permits. Pressure relief points are provided in the network to discharge flow under emergency conditions to prevent sewer backups. Pump stations are operated by both the City and County.

CSO/EOF Inputs to the LDW

Thirteen CSO/EOF outfalls are located within the LDW study area (Maps 9-6a through 9-6f). The City owns and operates two CSOs (S Brighton Street CSO and Diagonal Avenue S CSO) and two EOFs (SPU EOF No. 117, formerly the Slip 4 EOF, and SPU EOF No. 156, formerly the Isaacson EOF). The City's two CSOs also operate as storm drains, which regularly discharge stormwater during rainfall events. The City's Diagonal Avenue S CSO outfall carries overflows from both County and City CSOs and receives overflows from seven separate overflow points, including the County's Hanford No. 1 CSO. Map 9-1 shows the basin boundaries for the area served by sanitary and combined sewers that have the potential to discharge to the LDW through a CSO, EOF, or CSO/SD outfall location.

Table 9-7 presents the 2005-2006 season annual overflow volumes and flow frequencies for the County CSO/EOFs plus the contributions of the County's Hanford No. 1 CSO to the City's Diagonal Avenue S outfall. King County uses the period between 1981 and 1983 as the baseline for measuring progress in controlling CSOs; baseline volumes were determined using computer modeling (King County 2006a).

Table 9-7. King County untreated CSO/EOF volumes, 1999-2005.

CSO/EOF	Discharge Serial Number	Annual Average Volume (mgy) June 1999 – May 2005	Baseline Annual Average Volume (mgy) 1981 – 1983	Annual Average Frequency (events/yr) June 1999 – May 2005	Baseline Annual Average Frequency (events/yr) 1981 – 1983
8th Avenue S	040	0.00	8	0	6
S Brandon Street	041	30.96	64	26.3	36
Duwamish pump station West	034	0.58	<1	1.0	<1
Duwamish pump station East	035	0.67	nr	0.17	<1
Hanford No. 1[a]	031	10.36	nr	5.50	nr
E Marginal Way S pump station[b]	043	0.00	< 1	0	< 1
E Michigan	039	19.01	190	8.14	34
W Michigan	042	0.98	2	3.57	5
Norfolk[c]	044	0.28	39	1.14	20
Terminal 115	038	3.17	2	2.00	4

Source: King County (2006b); Nairn (2007)

[a] From Hanford at Rainier which discharges at Hanford No. 1 through the City's Diagonal Avenue S SD outfall, which also discharges through the City's Diagonal Avenue S CSO.

[b] Emergency overflow.

[c] Combined CSO/SD.

CSO – combined sewer overflow

EOF – emergency overflow

mgy – million gallons per year

nr – not reported

As shown in Table 9-7, the E Michigan CSO, S Brandon Street CSO, and Hanford No. 1 (discharging via the City's Diagonal Avenue S CSO/SD) outfalls had the highest annual average overflow volumes between 1999 and 2005. In 2004, the Duwamish West CSO was reactivated under the County's NPDES permit because of concerns that it could overflow; monitoring of this CSO is now in place. The E Marginal Way S Pump Station and Duwamish East CSOs are considered to be controlled (i.e., they meet Ecology's standard of one overflow event per year based on a 5-yr average) (King County 2006b).

The County's E Marginal Way S outfall is a pump station EOF; discharge at this station occurs only under emergency conditions (i.e., power failure). The County has

no record of an overflow event ever occurring at this location. The Duwamish East CSO also functions as an emergency bypass for a pump station; there has not been an emergency overflow since 1989 (King County 2006b). However, this location has the potential to overflow if the input volume from the siphon carrying combined sewage from the western bank of the LDW and the West Seattle area is too high (King County 2006b).

The City's Diagonal Avenue S CSO is the only City CSO with significant discharges. Overflow data for 1999 to 2005 are presented in Table 9-8. The other City CSO (S Brighton Street) discharges only during extreme storm events, generally once every 5 yrs (Tetra Tech 1988a). SPU records show that the S Brighton Street CSO has not overflowed since monitoring began in March 2000 (King County and SPU 2005).

Table 9-8. Diagonal Avenue S untreated CSO discharge volumes, 1999 to 2005

Year	Volume (mgy)	Frequency (events per year)
1999	5.21	41
2000	0.58	28
2001	2.74	29
2002	0.83	22
2003	74.5[a]	11
2004	6.6	5
2005	20.4	5

Source: SPU (2007) and Schmoyer (2007a)

Note: These volumes do not include contributions from the County's Hanford No. 1 CSO outfall, which are presented in Table 9-7; volumes do include the remaining seven overflow points that discharge through the Diagonal Avenue S CSO.

[a] Fifty-nine million gal. were discharged during a single event that occurred during October 20-21, 2003.

mgy – millions of gallons per year

Whole-Water Sampling of CSOs

Whole-water effluent samples were collected from County CSOs that discharge to the Duwamish River or Elliott Bay in 1996 and 1997 (King County 1999e). Samples were collected from the Norfolk CSO and the S Brandon Street CSO, which both discharge into the LDW study area; Table 9-9 presents a summary of the effluent data collection from the Norfolk and S Brandon Street CSOs (King County 1999e).

Table 9-9. Chemical concentrations in select King County CSO effluents

Chemical[a, b]	Norfolk CSO			S Brandon St CSO		
	Detect. Freq. (%)	Median (µg/L)[c, d]	SD (µg/L)	Detect. Freq. (%)	Median (µg/L)[d, e]	SD (µg/L)
TOC	100	13,900	3,240	100	12,100	14,500
TSS	100	167,000	48,300	100	74,800	155,500
Total arsenic	100	4.43	0.49	100	2.81	1.82

Chemical[a, b]	Norfolk CSO			S Brandon St CSO		
	Detect. Freq. (%)	Median (µg/L)[c, d]	SD (µg/L)	Detect. Freq. (%)	Median (µg/L)[d, e]	SD (µg/L)
Total cadmium	100	0.30	0.021	100	8.35	9.72
Total chromium	100	11.0	0.49	100	10.2	24.3
Total copper	100	25.5	7.0	100	37.4	59.2
Total lead	100	25.8	4.53	100	39.1	52.8
Total mercury	50	0.23	0.028	24	0.20	0.32
Total nickel	100	10.9	0.92	100	12.6	173.3
Total silver	50	0.88	na	65	0.92	0.59
Total zinc	100	87.0	11.0	100	173	135
1,4-Dichlorobenzene	100	0.44	0.21	97	0.45	0.48
2-Methylnaphthalene	0	na	na	19	0.64	0.49
2-Methylphenol	0	na	na	35	0.38	6.44
4-Chloro-3-methylphenol	0	na	na	19	0.64	0.22
4-Methylphenol	50	0.60	0.49	55	0.98	6.02
4-Nitrophenol	0	na	na	61	0.96	0.74
Acenaphthene	0	na	na	6	0.16	0.05
Benzo(a)anthracene	0	na	na	13	0.17	0.05
Benzoic acid	50	1.85	0.91	90	2.5	6.1
Benzyl alcohol	50	0.36	0.035	52	0.52	0.82
BBP	100	0.30	0.48	81	0.65	4.63
BEHP	100	1.55	1.61	100	5.0	1.38
Chrysene	25	0.20	na	77	0.23	0.08
Coprostanol	100	22.8	24.55	100	22.7	22.7
Diethyl phthalate	100	1.03	0.50	87	0.51	0.39
Dimethyl phthalate	0	na	na	77	0.13	0.07
Di-n-butyl phthalate	0	na	na	77	0.57	3.6
Di-n-octyl phthalate	0	na	na	100	1.21	0.63
Fluoranthene	75	0.21	0.038	97	0.31	0.13
Fluorene	0	na	na	13	0.19	0.12
Naphthalene	0	na	na	3	0.45	na
Pentachlorophenol	0	na	na	81	0.35	0.10
Phenanthrene	75	0.18	0.090	81	0.26	0.15
Phenol	0	na	na	26	2.28	5.03
Pyrene	75	0.23	0.044	97	0.32	0.13

Source: (King County 1999e; Stern 2002)

[a] Detection limits ranged from ~0.005-1.0 µg/L for organic compounds and from ~0.001 to 1.0 µg/L for metals.

[b] Detection frequency was less than 5% for 2,4-dimethylphenol, anthracene, benzo(a)pyrene, benzo(b)fluoranthene, benzo(g,h,i)perylene, benzo(k)fluoranthene, carbazole, dibenzofuran, Lindane, indeno(1,2,3-c,d)pyrene, isophorone, n-nitrosodimethylamine, and n-nitrosodiphenylamine; all other organic compounds (including PCBs, DDTs, and other pesticides) were not detected in these CSO samples.

[c] Number of samples was fewer than five, except for TSS.

[d] Median values reflect all samples taken, regardless of type of grab sample.

[e] Number of samples was greater than 30 (40 samples for metals; 31 samples for organic compounds).

BBP – butyl benzyl phthalate
BEHP – bis(2-ethylhexyl) phthalate
CSO – combined sewer overflow
na – not applicable
nd – no data
PCB – polychlorinated biphenyl
SD – storm drain
TOC – total organic carbon
TSS – total suspended solids

In the samples collected from the Norfolk and S Brandon Street CSOs in 1996 and 1997, seven metals (i.e., arsenic, cadmium, chromium, copper, lead, nickel, and zinc) were detected in 100% of the samples. Other constituents regularly detected included other metals, phthalates, and several PAHs. Other organic compounds, including PCBs and DDTs, were not detected in any of these CSO whole-water effluent samples (detection limits for organic compounds ranged from 0.005 to 1.0 µg/L) (King County 1999e).

Sediment Chemistry Modeling Efforts

Modeling of sediment deposition patterns off of the Duwamish/Diagonal and Brandon CSOs was completed in 1999 (Anchor and Herrera 1999). The modeling effort showed that BEHP and chrysene could potentially accumulate in sediments within localized areas adjacent to the CSOs at concentrations above the CSL and SQS, respectively. Modeling results also predicted that lead could accumulate in sediments within localized sediment areas adjacent to the S Brandon St. CSO at concentrations above the SQS.

Surface sediment samples collected in the vicinity of the Duwamish/Diagonal CSO have consistently detected BEHP at concentrations above SQS (Ecology 2006b), supporting the predictions of the model. Chrysene concentrations have not exceeded the SQS as consistently as has BEHP; therefore field-collected data for this chemical do not necessarily support model predictions at the Duwamish/Diagonal CSO. At the S Brandon St CSO, BEHP, lead, and chrysene concentrations in surface sediment collected near the CSO have consistently been below the SQS and also do not support model predictions. The models did not appear to be refined enough to accurately predict sediment chemical concentrations (Anchor and Herrera 1999).

Sediment deposition was also modeled near the Norfolk and Duwamish/Diagonal CSOs (King County 2001a). This 2001 modeling effort estimated that BEHP and BBP may accumulate at in sediments at concentrations above SQS criteria at the Duwamish/Diagonal CSO location but not necessarily at the Norfolk CSO. Post-remediation sediment sampling in these two areas has generally confirmed these estimates (phthalate concentrations have exceeded SQS criteria at the Duwamish/Diagonal CSO but not at the Norfolk CSO). However, limitations in these modeling efforts may make it difficult to accurately predict chemical concentrations (King County 2001a).

9.4.4.3 King County/City of Seattle CSO management programs

An important element of source control is the use of discharge permits issued by regulatory agencies. These permits provide clear and enforceable requirements for controlling chemical concentrations in waste discharges and reducing these concentrations over time as necessary. As part of the City and County NPDES permits, Ecology requires "the greatest reasonable reduction of combined sewer overflows at the earliest possible date" (WAC 173-245-010). Ecology also requires that CSO planning documents specify the means for complying with the regulations.

The County prepares and updates its CSO control plan to reflect the current state of science and regulation and to integrate CSO control with other Wastewater Treatment Division capital improvement programs. Various strategies to monitor and control CSOs include controlling pollution at its sources, maximizing use of existing system capacity, monitoring and modeling flows in the system, and constructing new CSO control facilities.

To provide a high level of treatment, the County routes combined sewer flows that would otherwise be discharged at CSOs during storm events to regional plants for secondary treatment, to the extent possible. An automated control system manages flows through the conveyance system so that the maximum amount of flow is contained in pipelines and storage facilities until it can be conveyed to the plant. In some areas of the system, where flows cannot be conveyed to the plant, the flows are sent to CSO treatment facilities for treatment prior to discharge. CSOs discharge untreated wastewater only when flows exceed the capacity of these systems (King County 2006b).

Construction of County CSO control facilities in the region began in the late 1970s. So far, about $320 million has been spent to control CSOs and another $383 million is planned to implement the CSO control projects over the next 25 years. Early projects involved sewer separation, flow diversion, and storage tunnels. Current and future projects involve construction of conveyance improvements, storage tanks, and treatment facilities. The County has plans for reducing volumes at six CSOs discharging to the LDW (Hanford at Rainier [Hanford No. 1], E Michigan, S Brandon Street, 8th Avenue W, W Michigan, and Terminal 115). The reduction plans include conveyance expansion, increased storage capacity, and treatment (King County 2006b).

Since 1988, when monitoring and measuring of CSO flows began, these control efforts have reduced CSO volumes from an estimated 2.4 billion gal. per year to approximately 900 mgy system-wide. Examples of CSO management projects that have either decreased CSO volumes or the frequency of discharges include:
1) replacement of the pipe that leads from the Brandon Street Regulator Station to the EBI, providing some reduction in the volume of overflows, and 2) installation of portable monitoring within the Duwamish and West Seattle pump stations to better

characterize conditions at the pump stations and siphons to improve flow and system capacity management (King County 2006b).

Since about 1970, when the City CSO reduction program began, the City has reduced the annual CSO volume citywide by about 95%. In its 2001 CSO reduction plan update, SPU identified the Diagonal Avenue S CSO/SD as one of the six priority outfalls in the city; no other uncontrolled CSOs have been identified in the LDW (City of Seattle 2002; CTE 2001). In 2005, SPU replaced an undersized pipe on the largest overflow point (Diagonal Avenue S CSO/SD No.111 D) to allow more water to enter the King County system. The system is currently being monitored to determine whether additional storage capacity is needed to adequately control overflows.

9.4.4.4 Comparison of stormwater and combined sanitary sewer flows

Annual stormwater discharge volumes are usually substantially higher than annual CSO discharge volumes because storm drains discharge whenever it rains, and CSOs only discharge during larger storm events. Annual stormwater discharges to the LDW have been estimated at approximately 4,000 mgy compared to less than 65 mgy historically from the county CSOs and less than 10 mgy from the city CSOs (Huber 2002; Schmoyer 2002; SPU 2007). The original Phase 1 municipal stormwater permit requirements did not require stormwater quality monitoring; however, limited monitoring of stormwater discharges is required under the 2007 revisions to the Phase 1 permit.

Industrial stormwater permittees are currently required to collect quarterly stormwater samples from at least one outfall, but the required analyses (total copper, total lead, total zinc, oil and grease, turbidity, BOD, ammonia, nitrate-nitrite nitrogen, total phosphorus, pH) do not necessarily correspond to the chemicals of concern identified in LDW sediment.

Private storm drain systems usually support smaller drainage basins (often less than 5 ac (~2 ha). In the LDW drainage basin, these smaller areas are usually immediately adjacent to the LDW and are located primarily on commercial and industrial use properties. An estimated 1,432-ac discharge to the LDW through these private systems. Most of these small private drainage systems serve the primarily industrial areas adjacent to the LDW. In contrast, Seattle's municipal system drains about 5,694 ac, and the publicly managed system outside of Seattle's basin (including the Hamm Creek drainage and the City of Tukwila's network) drains approximately 2,200 ac (SPU 2007).

9.4.4.5 Discharge Permit Programs

Wastewater and stormwater discharges from industrial facilities, public systems, and construction sites are potential sources of contamination to the LDW. In general, discharge permit programs have been established by federal and state governments to help reduce the impact to water quality from these types of discharges. Both the NPDES program and the County's Industrial Waste Program (IWP) regulate

discharges to the LDW. Wastewater discharges are regulated by the NPDES and IWP programs; the NPDES program also regulates stormwater discharges from industrial operations and construction sites. Construction projects may also require the implementation of best management practices to manage stormwater. The following section discusses these various permit programs and how they apply to discharges to the LDW.

NPDES Programs

The NPDES program was established in 1972 and requires any facility that discharges pollutants into waters of the United States from any point source to obtain a NPDES permit. The program is administered by Ecology and regulates both wastewater and stormwater discharges to surface waters. Toxic chemicals, such as metals and man-made organics, as well as conventional parameters (5-day BOD, TSS, pH, fecal coliform bacteria, and oil and grease) are regulated under NPDES permits. Non-conventional parameters including ammonia, phosphorus, chemical oxygen demand, and whole effluent toxicity are also regulated under the NPDES program (EPA 2002b). The NPDES program was established to reduce discharges of these chemicals into surface water nationwide. Discharges into the LDW include industrial and municipal stormwater discharges, surface water discharges from construction sites, and CSOs.

The two main types of NPDES permits are general and individual permits (EPA 2002b). Facilities with similar operations, wastes, effluent limits, or monitoring requirements can be covered under general permits (EPA 2002b). Facilities with the same stormwater point source can also be covered under a general permit. The three types of general permits include sand and gravel, boatyard, and stormwater permits (Ecology 2007i). The general stormwater permit is also subdivided into three types: municipal, industrial, and construction stormwater permits (Ecology 2007f).

Sand and gravel general permits cover discharges of process water, stormwater, and mine-dewatering water (Ecology 2007p). Boatyard general permits are issued to commercial businesses that maintain, construct, or repair small vessels (Ecology 2007i). Municipal stormwater permits are divided into Phase 1 and Phase 2 permits. The permits are issued based on size designations established in *40 CFR 122.26*. Phase 1 permits cover discharges from large and medium municipal separate stormwater sewer systems (MS4s), except for municipal separate storm sewers owned or operated by WSDOT. The City, the Port, and the County are all covered under Phase 1 permits (Ecology 2007p).

Industrial general stormwater NPDES permits regulate stormwater discharges from industrial facilities. These facilities are required to monitor total copper, total lead, total zinc, oils and grease, turbidity, BOD, ammonia, nitrate-nitrite nitrogen, total phosphorous, and pH in stormwater discharges. All industrial facilities with industrial general stormwater permits are required to conduct quarterly monitoring of authorized discharges to surface water (Ecology 2007d).

Construction general stormwater NPDES permits are required for clearing, grading and excavation activities that affect 1 or more acres and that will discharge into surface water. Permits are also required if Ecology determines that an activity may potentially contaminate surface waters or if a water quality standard is expected to be violated (Ecology 2007a).

Facilities that do not qualify for a general permit must apply for an individual permit. Individual permits regulate the discharge of pollutants associated with a particular industrial or municipal facility. The individual permit is issued for no more than 5 yrs and must be renewed prior to expiration. Individual permits have been issued to the City and the County for CSO discharges (EPA 2002b).

A search was conducted to identify NPDES permitted discharges to the LDW. This search included a review of Ecology's Water Quality Permit Life Cycle System (Ecology 2007o). The system included facility addresses for each permittee but did not have information on outfall locations. A facility was assumed to potentially discharge to the LDW if the facility was within the LDW surface water drainage basin (according to the City's geographic information system [GIS] shape file). Six sand and gravel, sixty-nine industrial general, three construction general, three boatyard, and six individual permitted facilities were identified within the City's LDW surface water drainage basin (see Table G-1). In addition, 35 industrial general and 2 individual permitted facilities are located within the LDW drainage basin. It is possible that facilities discharging to the Duwamish River upstream of the LDW could also affect the water quality within the LDW. Table G-2 provides a summary of the facilities with an NPDES permit in the Black River, Duwamish River, lower Green River-East or lower Green River-West drainage basins (according to Washington State Department of Natural Resources GIS shape files). Sound Transit Light Rail also holds an individual NPDES permit for outfalls that discharge to the Duwamish River during construction (including areas upstream of and within the LDW). Only one of these outfall locations is within the LDW. A summary of the other Sound Transit Light Rail outfall locations is presented in Table G-3.

King County Industrial Waste Program

The County implemented an industrial waste pretreatment program (the IWP) in 1969. This program became an EPA-delegated program consistent with the requirements of the CWA in 1981. The IWP requires non-domestic users of the waste treatment systems within the County to meet certain standards and limits before discharging wastewater into the sewer system. Requirements may include pretreatment before discharge and/or best management practices (BMPs). The County enforces both federal standards and local limits, whichever are more stringent. Local limits were established to protect sewer facilities and treatment processes, public health and safety, and waters receiving the discharge to enable the County to comply with its NPDES permits. Regulated materials include heavy metals; flammable materials; sulfides; cyanide; pH; fats, oils, and grease; and organic compounds. Since it began,

the program has achieved significant improvements in the quality of wastewater received by the County's WWTPs and thus also in CSO discharges (King County 2006c).

The IWP issues industrial wastewater discharge permits and authorizations to companies that have industrial processes with the potential to adversely affect the County's WWTPs. Permits almost always require companies to monitor their own discharges. Waste discharge permits are issued to federally regulated industries or to facilities with discharge volumes greater than 25,000 gpd. Major discharge authorizations are issued for discharges between 5,000 gpd and 25,000 gpd. Minor discharge authorizations are issued for discharges between 1,000 and 5,000 gpd. Letters of authorization are issued for discharges less than 1,000 gpd. A list of industrial facilities that currently hold industrial waste discharge permits and authorizations governed under the IWP is provided in Table G-4 of Appendix G.

In support of this program, County staff inspect facilities with discharge permits at least once per year and sample wastewater from all permitted companies at least twice per year. Follow-up inspections and wastewater sampling are conducted to verify that identified conditions causing violations are corrected and eliminated. The County bases the enforcement response to each violation on a number of factors, including the severity and duration of the violation and the discharge and enforcement history of the facility or business (King County 2006b). Facilities demonstrating compliance with the County's permit can be issued discharge authorizations, which reduce the sampling regularity. The County inspects companies with discharge authorizations at least once every 5 yrs, but does not regularly sample them, relying instead on self-monitoring at these companies.

Construction Site Water Management

An important aspect of a construction operation is surface water management. Depending on the size and complexity of the effort, large volumes of stormwater runoff can collect at a construction site that can be very high in TSS and can potentially contain chemical contaminants. This surface water may have to be temporarily stored and potentially pretreated prior to discharge to the storm drain network or sanitary sewer.

In the City of Seattle, the Department of Planning and Development (DPD) regulates discharges from construction sites to the storm drain and combined sewer systems per the stormwater, grading, and drainage control code (Seattle Municipal Code 22.800), which requires developers to control soil erosion and runoff from all construction projects. Depending on the size of the construction site, developers are required to either implement standard erosion and control procedures or submit a temporary erosion and sediment control plan for DPD review. Projects that will conduct a significant amount of excavation dewatering are also required to submit a dewatering plan for review and approval. Construction dewatering involves the removal of groundwater encountered during excavation activities. This water can be

contaminated depending on the environmental conditions of the site. Depending on the location of the project, stormwater and dewatering water may be treated and discharged to the storm drain system or discharged directly to the combined sewer system. Discharges to the combined sewer must obtain a Side Sewer Permit from the City and comply with the County's IWP requirements.

In unincorporated King County, the King County Department of Development and Environmental Services regulates construction discharges (King County 2006b). Most construction runoff is discharged to nearby storm drain systems or directly to adjacent surface waters. Although there are separate departments responsible for regulation within the city and county limits, the City and County work together to regulate discharges during construction because the systems are interconnected.

The County's IWP regulates construction discharges when those discharges will enter the sanitary or the combined sewer system. A temporary dewatering permit is required for excavations greater than 12 ft deep, projects that disturb more than 1 ac, projects that encounter surface and/or subsurface water during construction, projects in an environmentally critical area, and disposal of contaminated surface and/or subsurface water. Water from dewatering operations can only be discharged to the public sewer system if authorized by the County. All dewatering discharges must meet County permitting requirements regardless of the discharge volume and period of discharge (King County 2006b). Stormwater discharged into the sanitary or combined sewer systems must contain less than 7 mL/L of settleable solids to prevent sewer lines from becoming restricted or blocked. The discharge of materials such as ashes, sand, grass, and gravel to the sewer system is prohibited, and entities that discharge these or other solids to the public sewer system are liable for any damages caused (King County 2006b).

Large-volume dischargers of construction-related water to the sanitary or combined sewer systems (i.e., more than 25,000 gpd) must find other means of disposal during the "wet months" (November through April) when the sewer system can be at full capacity. Discharges of construction-related water to the sewer system are not prohibited during the "dry months" (May through October).

In addition to local regulations, Ecology also requires that developers obtain a construction stormwater general NPDES permit for clearing, grading, and/or excavation projects larger than 1 ac in size that discharge stormwater to a surface water body directly or through a storm drain network.

9.4.4.6 Storm drain source identification and control programs

Various source control efforts to identify and characterize potential sources of contamination have been implemented to minimize the potential for sediment recontamination in the LDW. As part of these efforts, storm drain source identification is being conducted in the LDW drainage basin. This source identification effort includes the collection of the following types of samples:

- Sediment from in-line traps installed in the storm drain system
- Grab samples of sediment accumulated within storm drain lines
- Sediment from catch basins
- Sediment from in-line maintenance holes on the storm drain trunk lines

In-line sediment trap sampling consists of a sample bottle mounted inside the collection system pipe for a period of 4 to 6 months to passively collect suspended particulate that pass by. Grab samples of sediment that accumulated within storm drain lines (without the use of a trap) were also collected. SPU has installed sediment traps in the storm drain systems that discharge to the Duwamish/Diagonal and Slip 4 EAAs (Map 9-7).

Catch basin samples are grab samples of material that has accumulated in the catch basin sump. A catch basin sump is a feature in the storm drain that captures sediment and other debris before it can enter the drainage collection system. The sediment that accumulates in catch basins provides a qualitative measure of the quality of the stormwater runoff discharged from the drainage system, but may not reflect the chemical concentrations on particulates actually discharging to the LDW. Catch basin samples are collected either from a specific site or property (on-site) or from the streets (rights-of-way).

Since January 2002, the City of Seattle has collected sediment samples from 74 onsite catch basins and 57 right-of-way catch basins in the LDW drainage basin (not including the nearly 300 samples collected to characterize street dirt and soil in the roadways and properties adjacent to T-117). Samples from on-site catch basins have primarily been collected from sites where business inspections were conducted and at sites where sufficient sediment was available for chemical analysis (see Map 9-7 for sampling locations). Most of the catch basin samples have been collected in the Duwamish/Diagonal and Slip 4 drainage basins, but samples were also collected from the Norfolk area, the embayment at RM 1.4 to RM 1.5 W, and other areas outside the Duwamish/Diagonal and Slip 4 EAAs.

No regulatory standards exist for storm drain sediment. Chemical concentrations in storm drain sediment were compared in this section to the SMS and MTCA Method A cleanup standards for informational purposes only. This comparison provides an estimate of storm drain sediment quality. If storm drain sediment samples are below the SQS, it is likely that LDW sediment in outfall discharge areas would not be recontaminated to levels above the SQS as a result of outfall discharges. In contrast, an exceedance of the SQS in storm drain samples may indicate a potential for sediment recontamination, but additional source investigations would be necessary to determine if recontamination above the SQS might occur. This section summarizes results from the storm drain sampling efforts.

Figures 9-4 through 9-8 summarize the results of source tracing samples for arsenic, total PCBs, BEHP, LPAHs, and HPAHs, respectively. Information on other chemicals

of interest that are frequently detected in source tracing samples are summarized in Table 9-10 and Figures 9-9 through 9-15. All data present have been provided by the City of Seattle (Schmoyer 2007c).

Overall trends and conclusions interpreted from the source tracing information collected thus far are as follows:

- Although chemical concentrations in the four different types of samples were variable, the same chemicals tended to be elevated above the SQS. Chemicals that frequently exceeded the SQS included zinc, BEHP, and BBP.
- Other chemicals that exceeded the SQS in at least one sample included total PCBs, copper, lead, PAHs, di-n-octylphthalate, benzoic acid, 1,2-dichlorobenzene, 1,4-dichlorobenzene, hexachlorobenzene, dibenzofuran, 2-methylnaphthalene, n-nitrosodiphenylamine, and bis(2-chloroethoxy)methane.
- Chemical concentrations were generally higher in samples collected from onsite catch basins than in right-of-way catch basin samples likely because most onsite catch basin sampling stations were selected based on source control problems identified during business inspections.
- Total PCBs were frequently detected in all sample types (48 to 96%), but rarely exceeded the SQS (2 to 21%) and only in a few select locations in the LDW drainage basin. Elevated total PCB concentrations have generally been associated with specific sources, including PCB-contaminated exterior building paint or historical activities. The largest number of exceedances were in samples collected from the Slip 4 drainage basin. Source tracing efforts are ongoing in the Slip 4 drainage basin.
- BEHP was detected at elevated concentrations in most of the samples collected in the LDW drainage basin, and exceeded the SQS in 60 to 90% of the samples. No specific sources of BEHP have been identified.
- With the exception of the Slip 4 drainage basin, mercury was generally detected at low concentrations, exceeding the SQS in only 4 to 16% of the samples. These data suggest that there may be ongoing sources of mercury in the Slip 4 drainage basin, where nearly 40% of the samples exceeded the SQS.
- PAHs were frequently detected, but only 2 to 17% of the samples had PAH concentrations that exceeded the SQS.
- Although arsenic has been identified as a chemical of concern in LDW sediment and was detected in 29 to 56% of the source samples, the arsenic concentration in only one sample exceeded the SQS.

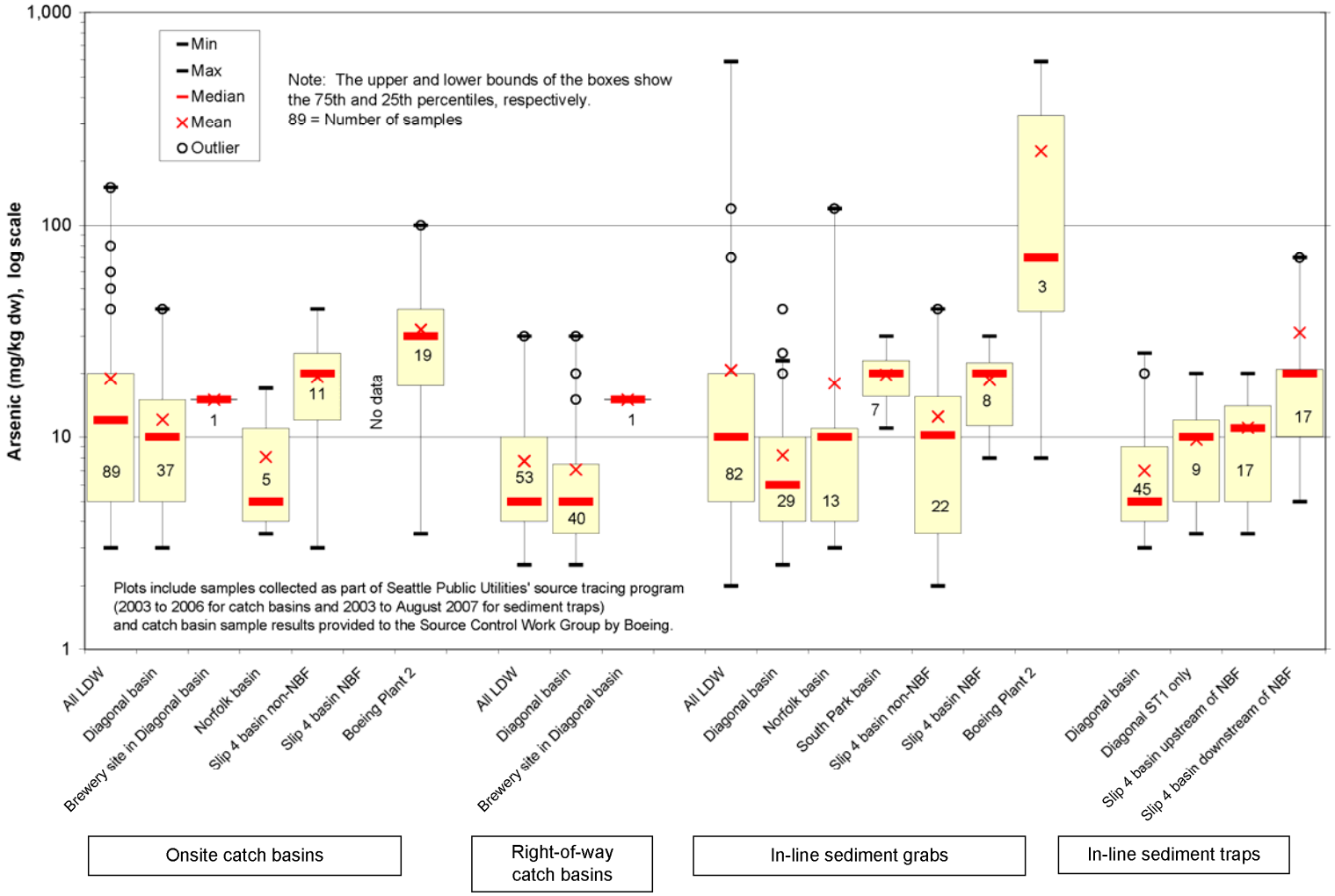


Figure 9-4. Arsenic source tracing sample results

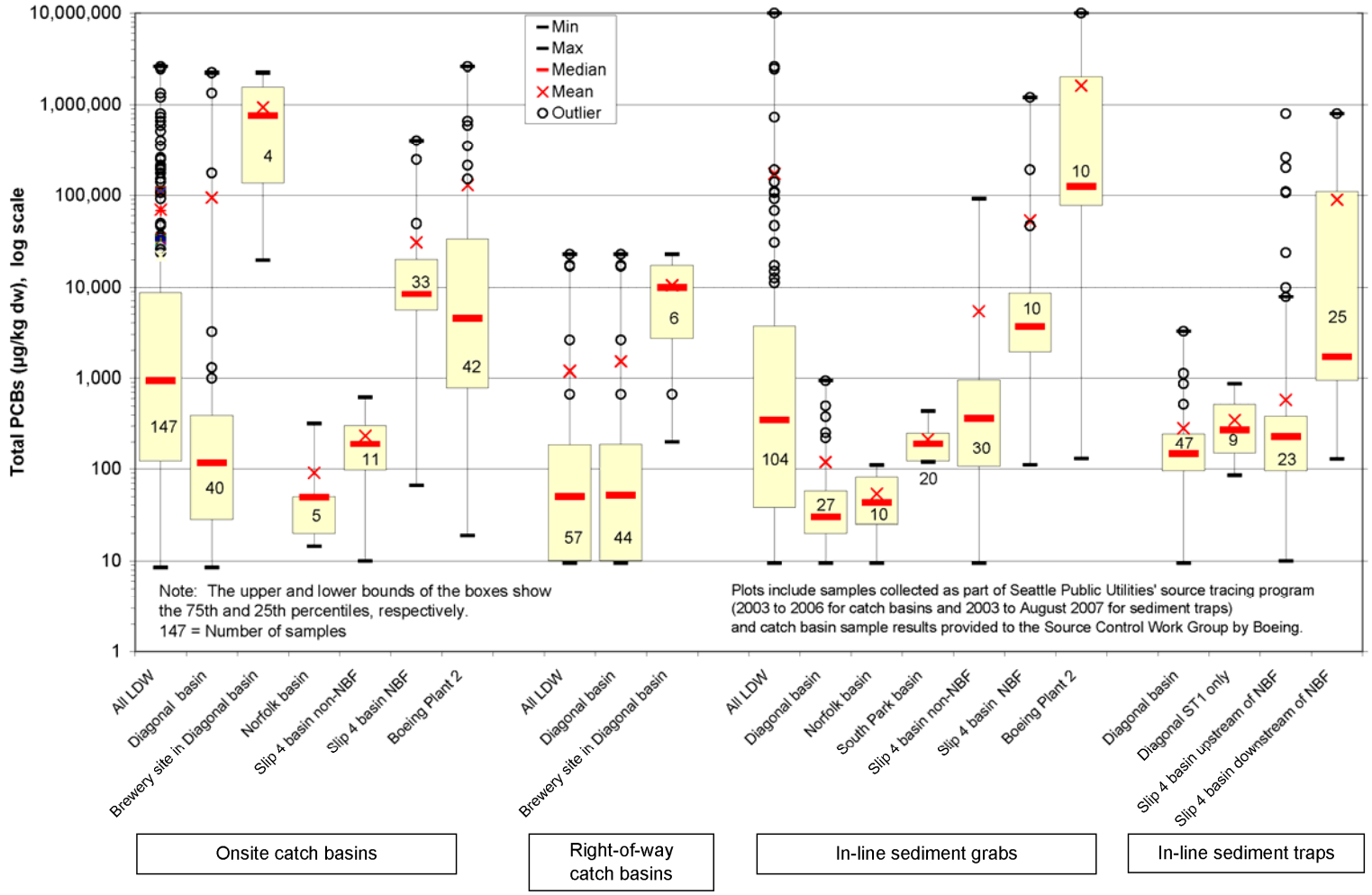


Figure 9-5. PCB source tracing sample results

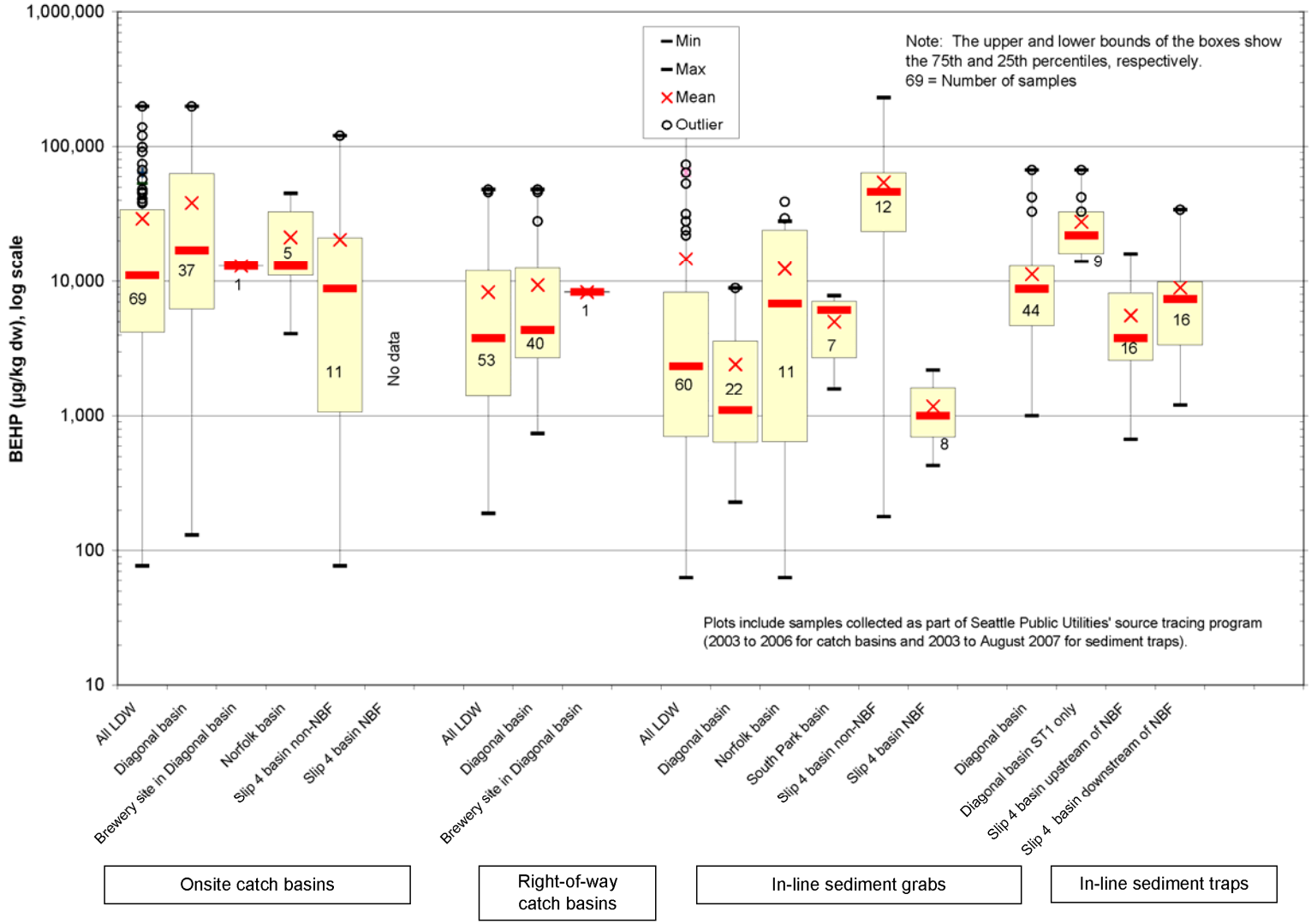


Figure 9.6. BEHP source tracing sample results

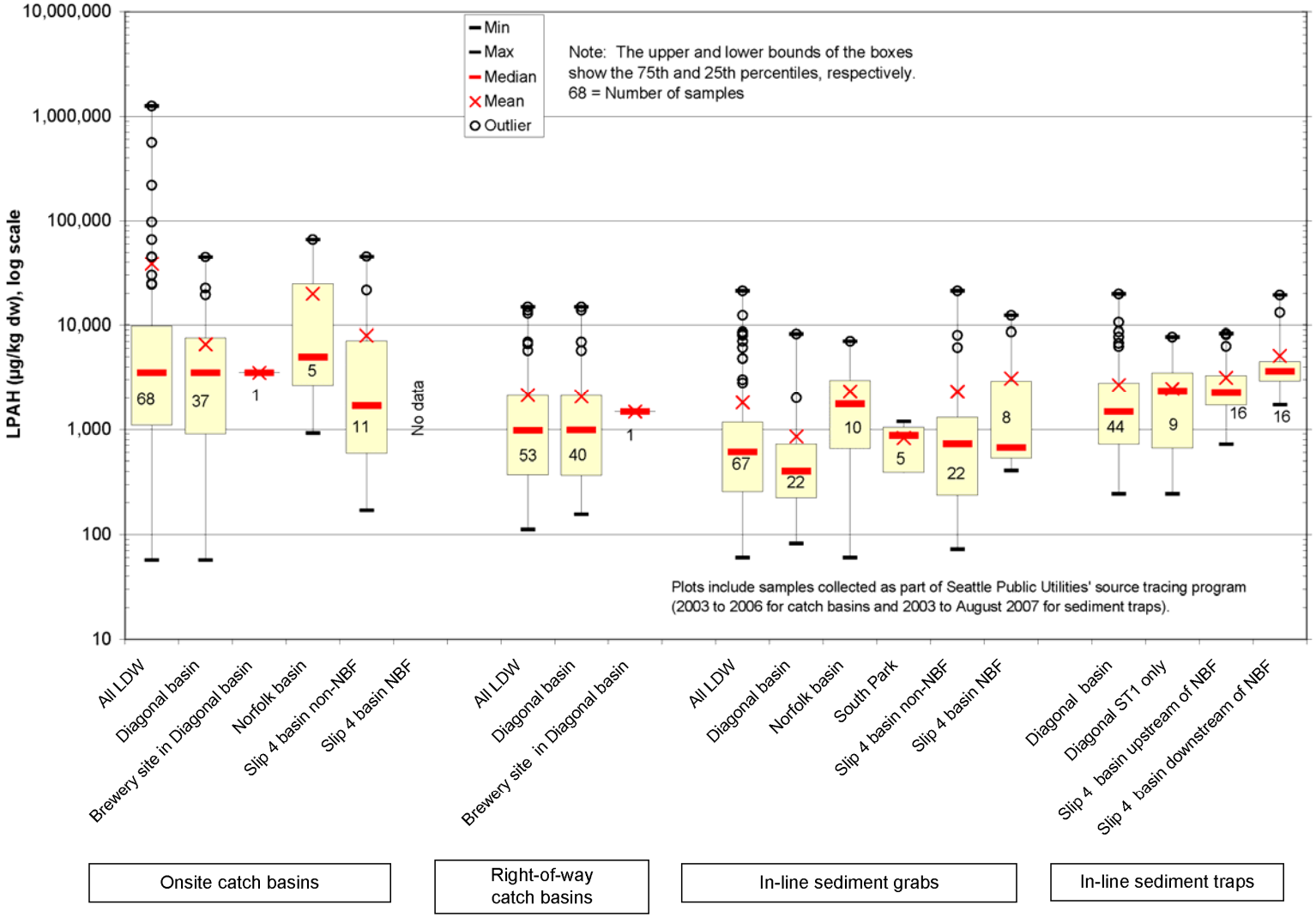


Figure 9-7. LPAH source tracing sample results

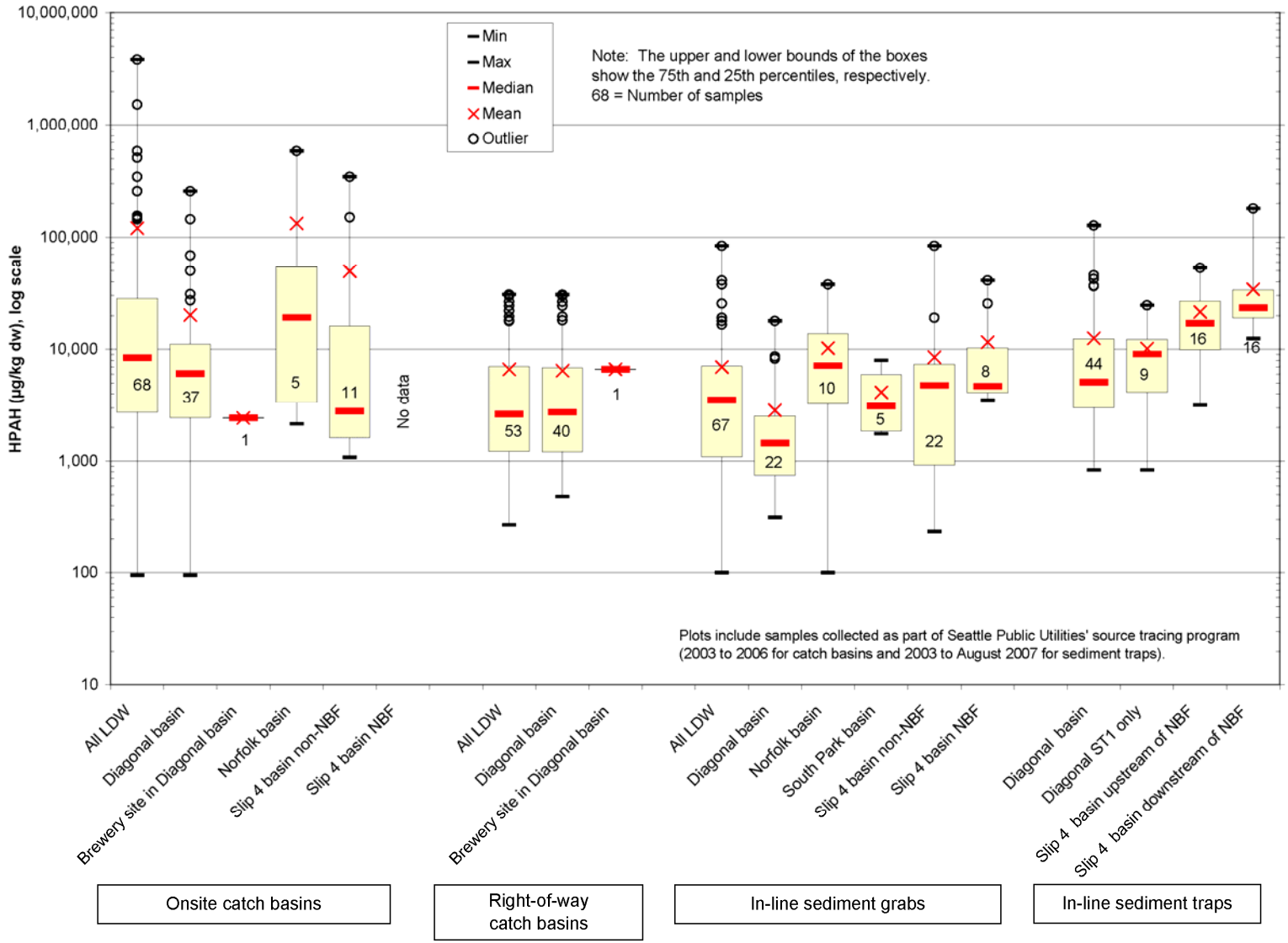


Figure 9-8. HPAH source tracing sample results

Table 9-10. Summary of COCs in source tracing samples by sample type

Sample Statistics	In-Line Sediment Traps[A]		In-Line Grab Samples[a]	Onsite Catch Basins[b]	Right-of-Way Catch Basins[b]
	Diagonal Drainage Basin	Slip 4 Drainage Basin			
Number of samples[c]	**42 – 45**	**26 – 47**	**68 – 73**	**69 – 73**	**52 – 57**
Copper					
Detection frequency (%)	100	100	100	100	100
Range (mg/kg dw)	6.6 – 597	33.9 – 3,960	13 – 3,960	29.6 – 6,320	35.9 – 751
% above SQS	4	6	1	22	6
% above CSL	4	6	1	22	6
Lead					
Detection frequency (%)	100	100	100	100	100
Range (mg/kg dw)	29 – 360	41 – 1,070	10 – 4,910	10 – 5,830	10 – 1,370
% above SQS	0	12	8	12	2
% above CSL	0	12	7	8	2
Mercury					
Detection frequency (%)	62	79	47	71	38
Range (mg/kg dw)	< 0.05 – 2.8	< 0.06 – 8.3	< 0.02 – 3.3	< 0.05 – 2.05	< 0.05 – 1.2
% above SQS	4	38	10	15	2
% above CSL	4	38	8	12	2
Zinc					
Detection frequency (%)	100	100	100	100	100
Range (mg/kg dw)	162 – 1,930	137 – 2,460	53 – 9,980	54.9 – 3,940	77.8 – 966
% above SQS	49	71	34	75	27
% above CSL	7	15	8	26	4
PCBs					
Detection frequency (%)	96	92	48	71	66
Range (mg/kg dw)	< 0.02 – 3.25	< 0.02 – 800	< 0.02 – 92	< 0.02 – 2,226	< 0.02 – 23
% above SQS	2	46	21	14	2
% above CSL	2	46	11	7	2
BEHP					
Detection frequency (%)	100	100	100	100	100
Range (mg/kg dw)	0.82 – 67	0.67 – 34	0.063 – 28	0.077 – 200	0.19 – 48
% above SQS	86	83	64	80	62
% above CSL	73	67	41	77	47
BBP					
Detection frequency (%)	77	53	36	72	79
Range (mg/kg dw)	< 0.06 – 3.4	< 0.06 – 1.6	< 0.02 – 1.9	< 0.02 – 15	< 0.04 – 37
% above SQS	48	29	19	49	42

Sample Statistics	In-Line Sediment Traps[A]		In-Line Grab Samples[a]	Onsite Catch Basins[b]	Right-of-Way Catch Basins[b]
	Diagonal Drainage Basin	Slip 4 Drainage Basin			
% above CSL	0	0	1	10	4

Note: Percent above SQS includes CSL exceedances

[a] Samples collected through August 2007.

[b] Samples collected through December 31, 2007.

[c] Some samples were not analyzed for all chemicals because of insufficient sample volume.

BEHP – bis(2-ethylhexyl) phthalate

BBP – butyl benzyl phthalate

CSL – cleanup screening level

PAH – polycyclic aromatic hydrocarbon

PCB – polychlorinated biphenyl

SQS – sediment quality standards

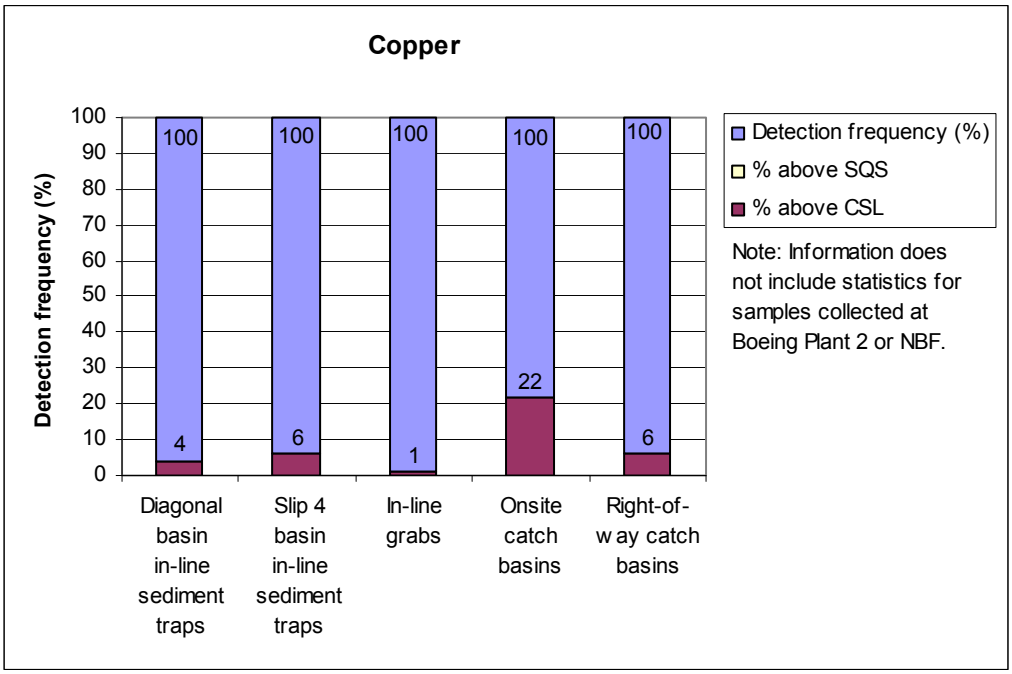


Figure 9-9. Comparison of copper concentrations in source tracing samples to SMS criteria

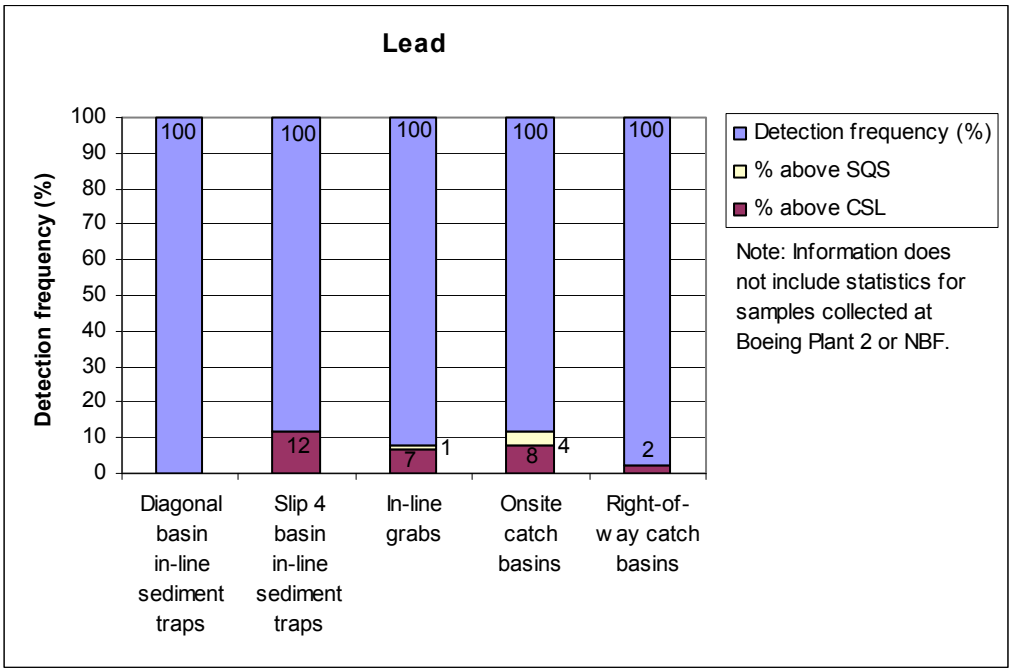


Figure 9-10. Comparison of lead concentrations in source tracing samples to SMS criteria

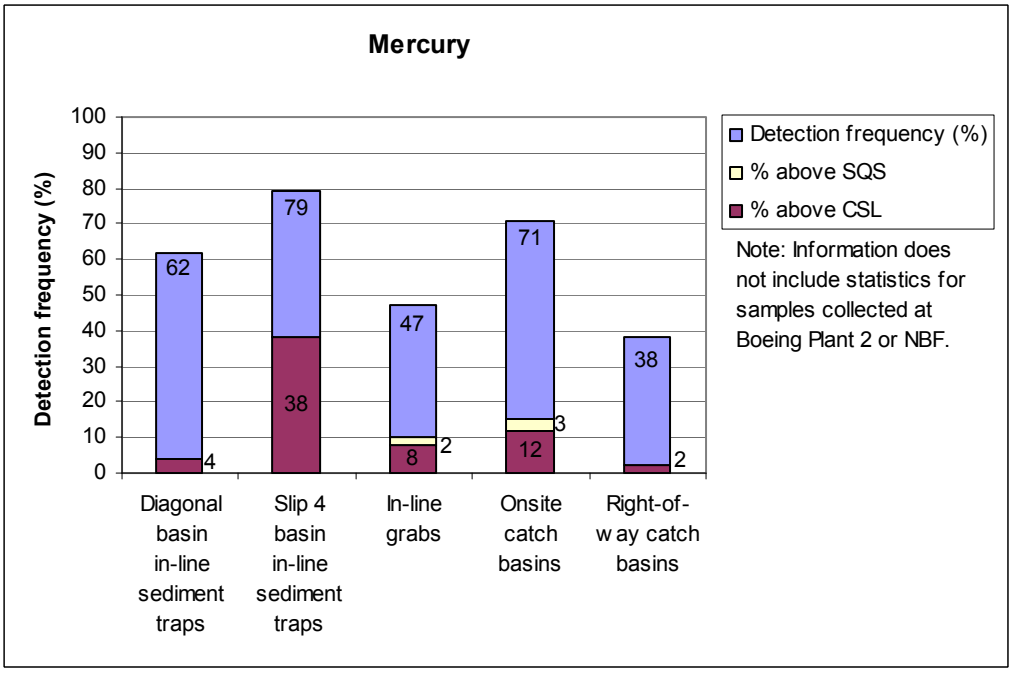


Figure 9-11. Comparison of mercury concentrations in source tracing samples to SMS criteria

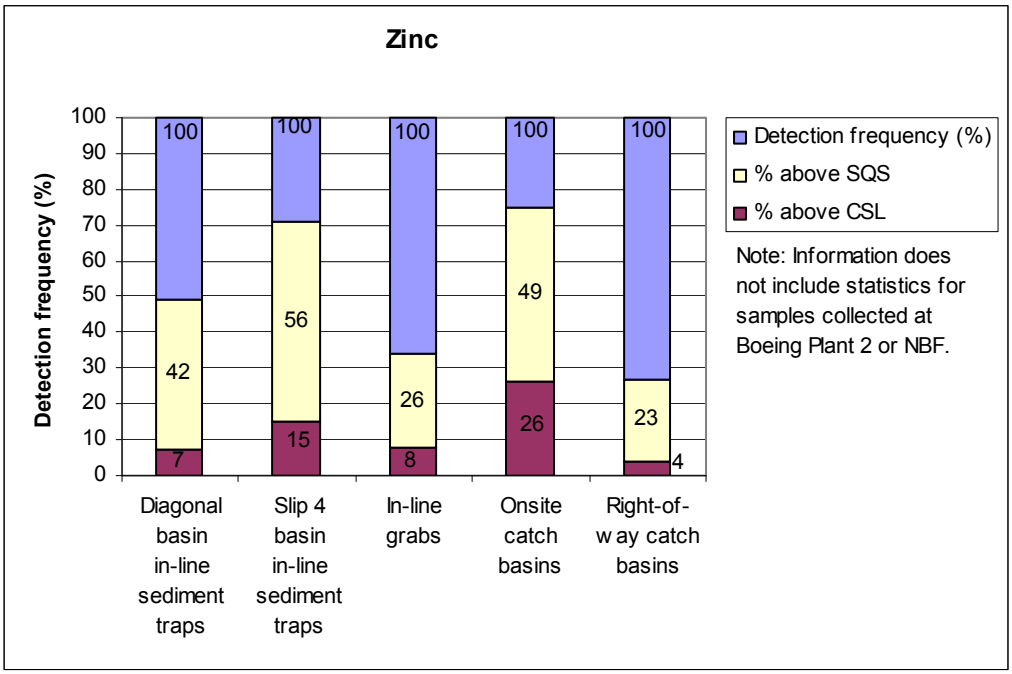


Figure 9-12. Comparison of zinc concentrations in source tracing samples to SMS criteria

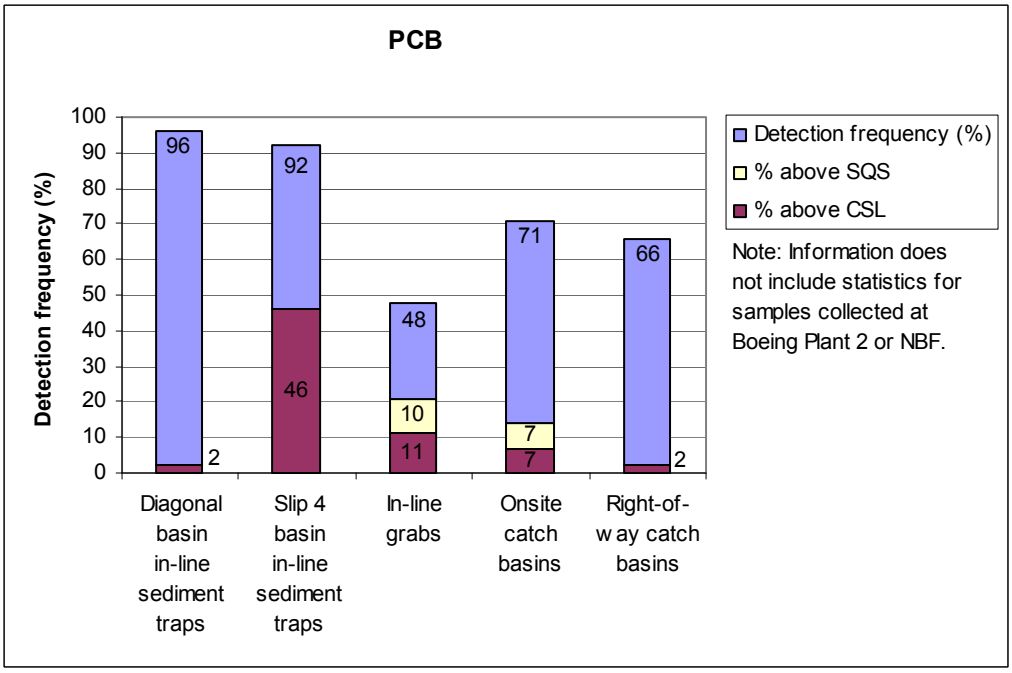


Figure 9-13. Comparison of PCB concentrations in source tracing samples to SMS criteria

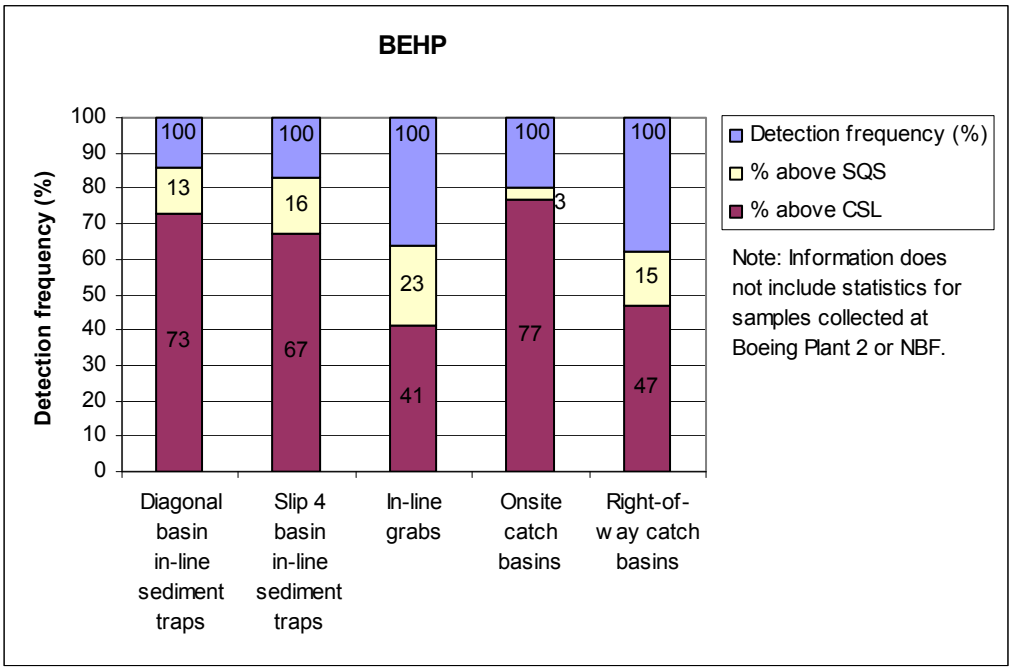


Figure 9-14. Comparison of BEHP concentrations in source tracing samples to SMS criteria

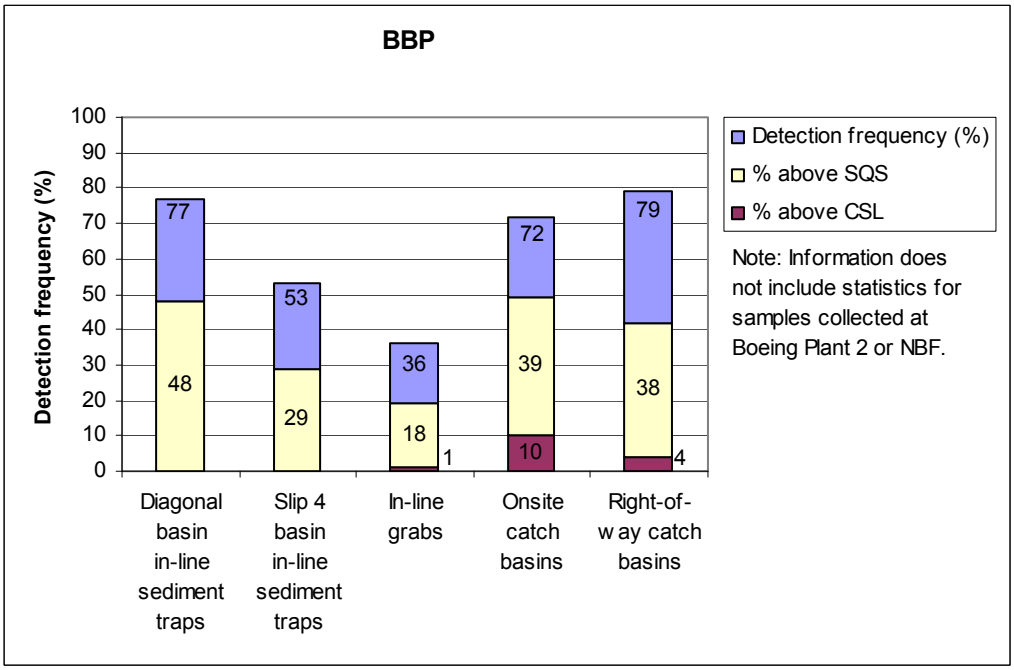


Figure 9-15. Comparison of BBP concentrations in source tracing samples to SMS criteria

Catch Basin Cleaning

In addition to sampling catch basins, the City has a maintenance program that routinely cleans catch basins in the public rights-of-way. Street dirt can contain chemicals such as metals, petroleum hydrocarbons, and other organic chemicals associated with motor vehicles. These chemicals often sorb onto particulate material and are transported along with suspended solids present in stormwater runoff. Although catch basins are primarily designed to prevent storm drains from becoming plugged by trapping large debris and solids, some of the particulate-bound chemicals can also be removed in catch basins. At the start of the LDW source control effort, SPU prioritized cleaning of catch basins in the Diagonal Avenue S CSO/SD drainage basin. In 2006, SPU cleaned catch basins and jetted the storm drain lines located in the public right-of-way in the 7th Avenue S storm drain system located in the South Park neighborhood. SPU is on schedule to re-clean all the right-of-way catch basins in the Diagonal Avenue S CSO/SD drainage basin in 2007.

Key Manhole Sampling

King County regularly samples wastewater at key locations in the sewer system that lead to the wastewater treatment plant. Week-long composite sampling is conducted twice per year (during a wet season and a dry season) at three pump stations in the LDW drainage basin: W Marginal Way SW, Duwamish, and E Marginal Way S (near Slip 4). In 2003, King County began analyzing samples for phthalates in addition to metals in wastewater samples. Data are available for samples collected in September 2003 (dry season), April 2004 (wet season), and September 2004 (dry season) for phthalates. Key conclusions for the phthalates were (King County and SPU 2005):

- Concentrations of BEHP in the wet season samples from the E Marginal Way S and Duwamish pump stations were within the range of concentrations typically detected in stormwater samples.
- In September 2003, samples collected from the W Marginal Way SW pump station contained a significantly higher BEHP concentration (148 µg/L) than the other two stations (mean ~ 5 µg/L).
- Other phthalates detected in the key manhole samples included BBP (100% of samples), diethylphthalate (97% of samples), di-n-butylphthalate (25% of samples), and dimethylphthalate (1% of samples).

Street Sweeping Pilot Program

In 2006-2007, SPU conducted a street sweeping pilot study to evaluate whether this practice reduced the chemical loadings to nearby receiving waters and/or reduced the frequency of required catch basin cleaning. New high-efficiency street sweepers were used to remove street dirt and debris from designated streets in two residential areas (i.e., West Seattle and Columbia City) and one industrial area (i.e., Duwamish/ Diagonal) in the City. At each test site, a 10-to-25-block area was divided into a control

area and a sweep area. Both control and sweep areas were cleaned at the beginning of the pilot study to establish a baseline for analyzing whether street sweeping reduced pollutants and sediment (Schmoyer 2007b).

The test area was swept every 2 weeks, but the control was not swept. Catch basin sediment, sweeper waste, and street dirt accumulation were assessed every month from the test and control areas to evaluate the quantity of material removed by the sweeping effort. Catch basin, sweeper waste, and street dirt samples were also collected every month and composited on a quarterly basis for the laboratory analysis of metals, SVOCs, PCBs, grain size, and TOC content. Field testing was completed in June 2007. SPU expects to issue a final report in fall 2007 (Schmoyer 2007b).

Phthalate Source Study

Atmospheric deposition is one likely source of phthalate compounds to the LDW (as discussed in Sections 9.4.2.2 and 9.4.2.3). To investigate the potential sources of phthalates that could be transported to the Thea Foss Waterway (in Tacoma) and the LDW through atmospheric deposition or other pathways, the County and the City of Seattle, joined by the City of Tacoma, conducted a phthalate source study in 2003. During the study period, various materials and commonly used consumer products were tested for phthalates, a literature review was conducted, and dust samples were collected from the roof of the Tacoma Dome for phthalate sampling.

King County and City of Tacoma laboratories analyzed various liquid products (e.g., soaps, inks, used and unused motor oil) and solid products (e.g., plastic bottles, automotive belts, brake pads and dust, packing peanuts, tires, cigarette butts). High concentrations of BEHP were found in some automotive belts, brake pads, and tires (King County and SPU 2005). Liquid products generally contained lower concentrations of BEHP. Analytical results from product testing were reported in the City/County source control progress report from 2004 (King County and SPU 2004). The literature review suggested that some vehicle fuel products, such as diesel, contain BEHP that may be released into the atmosphere in exhaust (King County and SPU 2005, citing California Air Resources Board, 1997).

Dust samples were collected during two sampling rounds from the roof of the Tacoma Dome; the first round was collected in May 2003 prior to roof washing, and the second round was collected in July 2003 immediately after the roof was washed. BEHP was detected at 600 $\mu g/ft^2$ in the first round sample, and at 42 $\mu g/ft^2$ in the second round sample (King County and SPU 2004).

9.4.4.6 Compliance and inspection programs

Compliance and inspection programs are conducted by various agencies and primary or secondary NPDES permit holders to identify and manage potential sources of chemicals that are typically generated from business practices within the LDW drainage basin. Inspections are used to assess compliance with source control requirements (i.e., BMPs), identify any illicit discharges, and to specify required

corrective actions. As members of the SCWG, the City of Tukwila, the City of Seattle, the Port, and the County are implementing source control programs. Most of their compliance and inspection programs pertain to discharges to the storm drainage system. The measures implemented jointly or by each member are described in greater detail below.

Business Inspection Programs

In 2001, SPU began a city-wide business inspection program to enforce the source control requirements of the City's stormwater, grading, and drainage control code (SMC 82.200). Business inspections for the LDW source control program, which began in 2003, are part of the City's overall source control program, but focus on a wider range of pollution-generating activities, including hazardous waste and industrial wastewater management. Prior to 2001, the City conducted periodic stormwater inspections for specific projects in Elliott Bay and the Duwamish River to control sources of chemicals entering city-owned storm drains. Business inspection locations are shown on Map 9-8.

One of the earliest inspection efforts in the LDW occurred in the mid 1990s under a centennial clean water fund grant from Ecology (She 1997). Approximately 100 businesses in the LDW were inspected. The study focused on outdoor activities to minimize the presence of chemicals used or generated by onsite activities that could come in contact with stormwater runoff. Inspectors worked with local businesses to improve their pollution prevention practices and developed information bulletins for businesses to promote BMPs, including disposal/storage activities, landscape maintenance, and housekeeping practices, and to increase local awareness of the importance of protecting water quality. A number of measurable actions by businesses were noted as a result of the inspections, including movement of barrels indoors, improvement of housekeeping practices, reducing or diverting contaminated water entry away from storm drains, and other similar actions.

In 2001, SPU inspected 200 businesses in the western portion of the Diagonal drainage basin (109 drive-by inspections and 91 complete onsite inspections). A total of 81 of the businesses inspected were not in compliance with City stormwater source control requirements. Most of the problems were related to inadequate maintenance of onsite storm drainage systems (33%) and inadequate spill response programs (47%). SPU inspectors worked with the business owners to improve their stormwater pollution prevention practices.

In 2003, SPU and King County initiated a joint inspection program to support LDW source control efforts. The joint program addressed various potential pollution-generating activities, including stormwater management, hazardous waste management, and industrial waste disposal. This joint program was designed to minimize the disruption to businesses by conducting a single inspection that covered all of the municipal environmental regulations rather than having representatives from each program conduct individual inspections. All inspections were conducted

under the existing code authorities of each jurisdiction. Inspectors from King County and SPU conducted joint inspections to cross-train inspectors from the various County/City programs about each other's program requirements. Later, each inspector worked independently. Since 2005, SPU has assumed all inspection duties for the LDW program, with King County providing technical assistance, as needed. King County also continues to conduct routine inspections as part of its ongoing programs (e.g., industrial waste).

As of December 31, 2006, a total of 1,009 businesses in the LDW study area have been inspected (723 full onsite inspections and 286 screening inspections). In addition, over 800 follow-up inspections have been conducted to confirm that property owners have implemented the necessary corrective actions. So far, inspections have been completed in areas draining to the Duwamish/Diagonal, Slip 4, Norfolk, and T-117 EAAs, as well as the portions of the KCIA that drain to Slip 6 and the Isaacson SD (former Slip 5).

Corrective actions were required at 476 of the 723 sites (66%) where full inspections were conducted. Inadequate maintenance of onsite drainage facilities was the most common problem (57% of the corrective actions). Approximately 45% of the corrective actions required were associated with spill prevention practices (e.g., lack of appropriate spill prevention practices, employee training, and maintenance of spill control supplies). The other problems identified during the inspections involved improper storage of hazardous products and waste materials (22%), improper hazardous waste disposal (16%), and improper outdoor storage of non-hazardous materials and products (12%).

The City will continue conducting business inspections to support the LDW source control program. In 2007, SPU will inspect businesses in EAA 2, in the vicinity of the embayment at RM 1.4 to RM 1.5 W, in areas along the east side of the LDW that drain to city-owned storm drains, and will begin re-inspecting high priority sites in the Duwamish/Diagonal drainage basin.

City of Tukwila and King County

The City of Tukwila has jurisdiction over a portion of the drainage in the southeast part of the LDW study area. The County has jurisdiction over a very small strip of land adjacent to the southwest part of the LDW study area. These entities will be brought in as appropriate to conduct any necessary stormwater source control activities consistent with the source control strategy. In 2007, the City of Seattle signed a memorandum of agreement with the City of Tukwila to allow the City of Seattle to inspect five high-priority sites in the Norfolk drainage basin that are in Tukwila's jurisdiction.

Port of Seattle

The Port is implementing a tenant audit and inspection program for its approximately 30 tenants, aimed at minimizing the potential for stormwater impacts. Tenant activities include container storage and handling, shipping, fishing vessel moorage,

barge loading, and marina operations. The Port will use inspection opportunities to observe activities that can affect stormwater discharges and oversee the implementation and operation of appropriate BMPs. These inspections will also allow Port inspectors to observe any hazardous/dangerous waste-generating activities, inspect usage and storage of hazardous materials, and investigate any other environmental concerns, including air quality, noise, and impacts on neighbors and surrounding communities.

As part of their lease, the Port requires tenants to apply for and obtain their own permits for stormwater management and air discharges and to write their own plans for industrial discharges to the County sewers and any spill prevention, control, and countermeasures. The Port reviews all the permits and plans for adequacy and implementation to protect the Port from potential liabilities and to ensure that appropriate environmental/public health concerns have been addressed. The Port also communicates with federal, state, and local agencies to make sure that any potential concerns or actions are addressed.

The Port has a multi-phased tenant compliance program that includes an environmental review of all new and renewed tenant leases, a walk-through of all new tenant facilities upon occupancy and exit, and depending on their activity, periodic inspections (weekly, monthly or annual). Most Port tenants do not have industrial activities that require a formal intensive checklist and environmental audit by the Port. The Port has a cooperative compliance program through which they communicate potential concerns with tenants, including responsiveness and technical assistance.

Hazardous Waste Management Inspection Programs

The regional Hazardous Waste Management Program (HWMP) complements the County's IWP by educating local residents and small businesses on methods to reduce hazardous waste and prevent water pollution. The program is a cooperative effort among the County's Department of Natural Resources and Parks (the Solid Waste and Water and Land Resources Divisions), Public Health, SPU, and 38 cities throughout King and Snohomish Counties. The program implements the local HWMP adopted in 1990 by the County and all of the local cities.

The HWMP targets industry groups and geographic areas to provide technical assistance. Program staff make site visits to small businesses throughout the County and all of its incorporated cities to observe operating practices. When problem materials, such as lead, mercury, and solvent-based paints, are found to be disposed of in the sanitary sewer, program staff advise the company on correct practices. When necessary, staff can refer the matter to the IWP for regulatory action. In 2000, more than 3,000 businesses were inspected. Follow-up inspections indicated that 75 to 80% of businesses made at least one positive change in hazardous waste management or environmental practices as a result of the initial visit (King County 2002b; citing Galvin 2001). The LDW area is included in the general coverage of the program;

additional efforts in the LDW targeted auto body and repair shops, machine shops, photo laboratories, and dry cleaners in the basin.

The regional HWMP will conduct follow-up activities in the Duwamish/Diagonal drainage basin to support the early action cleanup for the Duwamish/Diagonal EAA consistent with the LDW source control strategy. Past business inspections in the Diagonal CSO/SD drainage basin will be identified and compared with current business lists. This information will be used to identify the need for new site visits and to coordinate with the inspection data from both the stormwater and industrial waste inspection programs. Current efforts are focusing on visits to new (and previously visited) businesses within the Diagonal CSO/SD drainage basin or businesses in the drainage basins of other EAAs.

9.4.5 Spills and leaks

Spills and leaks may enter the LDW through various pathways. This section provides information on data resources for leaks and spills in the LDW drainage basin.

9.4.5.1 Available spill data

Ecology receives spill reports, which are tracked through the Environmental Report Tracking System (ERTS), and placed on the Spills Program database. Records of spills in the LDW drainage area currently listed on Ecology's Spills database are listed in Table G-14 of Appendix G. In addition to responding to spills, Ecology regularly refers reports of spills within the LDW area to SPU or the US Coast Guard (USCG) for response.[95] Records of SPU spill responses are including in Table G-15 (to be checked) of Appendix G.

9.4.5.2 Leaks

Leaks can occur from various sources such as pipes and storage tanks or from industrial or commercial equipment and process operations. Ecology currently regulates and monitors active and inactive LUSTs. The LUST list includes confirmed and suspected leaking tanks owned and operated by entities such as gas stations, manufacturing industries, commercial properties, and government agencies. The LUST list contains sites that have been cleaned up and sites currently in the process. Currently, about 70 of the LUST sites on Ecology's active LUST list are located in the LDW drainage area (EDR 2006a). The records for active and inactive LUST sites are summarized in Tables G-7 and G-8 of Appendix G, respectively.

9.4.5.3 Spill prevention and cleanup programs

Ecology maintains a spill prevention, preparedness, and response program whose goal is to protect public health, safety, and the environment from accidental releases of oil

[95] USCG also maintains a database of oil and chemical spills into water bodies. Spills reported to USCG are subsequently reported to Ecology; however, ERTS does not include all spills in the USCG database.

and other hazardous materials (Ecology 2006b). In addition, Ecology provides 24-hr emergency response units statewide and investigates hazardous material incidents in order to identify practices that will reduce the number of future spills. Ecology has the authority to issue fines or require changes in operations at facilities that violate spills regulations.

Ecology partnered with EPA, the USCG, and other state and local agencies from Oregon and Idaho to produce the Northwest Area Contingency Plan. The plan provides guidelines for response actions to spill incidents, and, in Washington State, it functions as the statewide master response plan for oil and hazardous substance releases (EPA and USCG 2006). Regulations calling for a regional response plan capable of responding to hazardous materials spills and other releases are contained in 40 CFR, Part 300.

SPU inspectors routinely respond to water quality complaints throughout the City. Complaints are received from a number of sources, including the City's hotline number, staff reports, and referrals from other City departments and agencies. In addition to identifying and controlling the causes of a complaint, City inspectors also provide technical assistance to local residents and businesses regarding BMPs for reducing pollution. SPU also maintains a coordinator network to respond to spills affecting City infrastructure on a 24-hour basis, seven days a week. SPU maintains an on-call contractor to provide emergency cleanup services. Records from the SPU spill coordinator program are provided in Appendix G.

9.4.6 Groundwater

This section presents the results of the groundwater assessment summary conducted as part of the Phase 1 RI, and porewater and seep studies were conducted in Phase 2 to further evaluate the potential groundwater pathway to LDW sediment. Groundwater data discussed in this section include information through 2002, as evaluated in the Phase 1 RI. Additional updated information on the potential existence and affect of the groundwater pathway to LDW sediment is discussed in Appendix I.

9.4.6.1 Phase 1 groundwater assessment summary

As part of the Phase 1 RI, an assessment was completed to evaluate the potential for groundwater contamination to reach the LDW and contaminate sediment. Groundwater information, through 2002, was summarized for 12 upland sites identified by EPA and Ecology as preliminary sites of interest for the Phase 1 RI. These sites were Advance Electroplating, Boeing Developmental Center, Boeing Isaacson, Boeing Plant 2, GWI, Long Painting, Malarkey Asphalt, PACCAR (Kenworth Truck Co.), Philip Services (Burlington Environmental), Rhône-Poulenc, South Park Landfill, and T-108/Chiyoda Property. The locations of these properties are shown on Map 9-9.

Methods

The following information was evaluated on a site-specific basis for each of the 12 sites:

- Site operations and regulatory status
- Site-specific aquifer characteristics, groundwater flow direction, and rates of flow
- Chemicals identified as COCs (different sites used different criteria to select COCs)
- COC concentrations in monitoring wells samples located nearest to the LDW and chemical contour or plume maps, when available
- COC concentrations in groundwater seeps or sediment in the LDW near potential areas of groundwater discharge
- Available fate and transport information

As part of this analysis, the LDW sediment database was reviewed to compare detections of the groundwater COCs in sediment samples located within the potential zone of groundwater discharge for each of the 12 sites (the sediment samples locations evaluated are color-coded on Map 9-9). If no exceedances of SMS sediment criteria or DMMP guidelines were identified, the analysis concluded that there was no evidence to indicate that groundwater COCs had migrated to LDW sediments at concentrations of concern. If SMS criteria or DMMP guidelines were exceeded, additional assessment would be required as part of source control activities.

For chlorinated solvents, VOCs, and petroleum compounds that do not readily accumulate in sediment, groundwater and seep data, along with fate and transport considerations, were deemed more appropriate for assessing the existence of a potential groundwater pathway.

It should be noted that this preliminary analysis of potential COC migration to LDW sediment had limitations based on the type of data used for analysis and uncertainties associated with the data. Uncertainties included: 1) all potential groundwater sources may not have been addressed, such as fill sites that have not yet been characterized, 2) few groundwater samples have been collected on the west side of the LDW (although few potential sources have been identified in that area), and 3) SMS criteria or DMMP guidelines were unavailable for some chemicals.

Results

Table 9-11 provides an overview of the conclusions reached for each of the 12 sites assessed during the Phase 1 RI (Windward 2003a).[96] The information on Table 9-11 includes evaluation of sediment data available at the time of the Phase 1 RI; sediment data collected as part of the Phase 2 program are not reflected on this table. Based on various screening criteria used at each of the sites, COCs identified in groundwater included chlorinated solvents and their breakdown products, metals, BTEX compounds, TPH, PCBs, PAHs, and a few other organic compounds at a few sites. The most common COCs were metals and chlorinated solvents, both of which were COCs at over half of the sites (Table 9-11).

[96] The agencies have also been conducting independent analyses of groundwater contamination impacts for many of these sites and for other contaminated sites as appropriate. Additional information is provided in Appendix I.

Table 9-11. Summary of groundwater sites evaluated in the Phase 1 RI

Site	COCs in Groundwater[a]	Fate and Transport Issues	Seep Data	COCs Evaluated in Sediment[b, c]
Advance Electroplating	trichloroethene, tetrachloroethene, cadmium, chromium, and nickel	Site is significant distance from LDW but may discharge to Hamm Creek first. Biodegradation and volatilization of VOCs expected between the site and LDW.	No known data.	Trichloroethene, tetrachloroethene, and metals Metals: No detects > SQS. Organic chemicals: No detects. TCE and PCE analyzed in one sample.
Boeing Developmental Center	arsenic, copper, lead, nickel, tetrachloroethene, benzene, and TPH	Plumes of chlorinated solvents are locally contained and do not extend to LDW. Natural attenuation expected for hydrocarbons and benzene (500 ft from LDW).	No known data.	Arsenic, copper, lead, nickel, trichloroethene, benzene, and TPH Metals: Only lead > SQS and CSL in 1 of 24 samples; others < SQS. Organic chemicals: No detects. TCE analyzed in 1 sample.
Boeing Isaacson	arsenic	Natural attenuation apparent from well pairs, and seep concentrations low. Sediment data do not show accumulation.	Arsenic detected at 7 µg/L in the only seep sample collected.	Arsenic, chromium, lead, and zinc Metals: Arsenic > SQS in 1 of 9 samples. Other metals < SQS.
Boeing Plant 2	vinyl chloride, cis-1,2-dichloroethene, 1,1-dichloroethene, arsenic, cadmium, copper, lead, silver, thallium, selenium, nickel, and zinc	Weston's mass loading analysis indicated no accumulation for metals following bank removal. VOCs were not included because they are non-persistent and highly volatile. PCBs were detected in seeps but not groundwater. The likely source of PCBs in seeps is sediment.	All COCs analyzed in 18 seep samples; metals and PCBs exceeded AWQC in some samples: VOCs detected at concentrations below AWQC.	Trichloroethene, vinyl chloride, 1,1-dichloroethene, PCBs, and metals Metals: > SQS in up to 20% of samples. Organic chemicals: PCBs > SQS in 89% of samples. VOCs analyzed in six samples, no detections.
GWI	chlorinated ethenes and ethanes, benzene, toluene, ethyl benzene, 1,4-dichlorobenzene, pentachlorophenol, and TPH	Of the groundwater COCs, only chlorinated ethenes and ethanes were detected at the point of discharge (seeps). These chemicals were not detected in sediment or mussels. Fate and transport analyses for VOCs indicated extensive degradation prior to discharge.	Chlorinated ethenes and ethanes detected in 5 of 12 seeps sampled in 1998 and 1999.	Tetrachloroethene, trichloroethene, 1,2-dichloroethene, vinyl chloride, 1,1-dichlororethane, and methylene chloride Organic chemicals: VOCs were not detected in all three samples.

Site	COCs in Groundwater[a]	Fate and Transport Issues	Seep Data	COCs Evaluated in Sediment[b, c]
Long Painting	arsenic, chromium, lead, and possibly VOCs	No plume identified on site, but groundwater data were insufficient.	No known data.	Arsenic, chromium, lead, 1,1,1-trichloroethane, tetrachloroethene Metals: Six samples analyzed for metals; no detects > SQS. Organic chemicals: VOCs not detected in one sample analyzed.
T-117 (formerly Malarkey Asphalt)	PCBs	PCBs not expected to migrate in groundwater because of low solubility, high affinity for organic matter (natural attenuation expected).	No known data.	PCBs Organic chemicals: PCBs > SQS in 9 of 11 samples, > CSL in 5 of 11 samples.
PACCAR	chlorinated solvents, TPH, and arsenic	Solvents likely attenuated; arsenic in range of background; likely attenuation of TPH.	No known data.	1,1-dichloroethene, trichloroethene, tetrachloroethene, vinyl chloride, arsenic, and barium Metals: No detects > SQS. Organic chemicals: Not analyzed in sediment.
Philip Services	chlorinated ethanes, chlorinated ethenes, and BTEX compounds	Phillip Services Corp. (2001) concluded no COCs from their site would reach LDW at concentrations greater than AWQC. Trichloroethene, tetrachloroethene, and vinyl chloride, attributed to other sources, may reach LDW based on plume data.	No known data.	Chlorinated ethanes and ethenes, BTEX, benzoic acid, barium, cadmium, chromium, copper, lead, manganese, nickel, and silver Metals: No detects > SQS. Organic chemicals: VOCs not detected in three samples analyzed. Benzoic acid > CSL in 1 of 27 samples.
Rhône-Poulenc	toluene and metals; also PAHs, BEHP, and nine organic compounds.	Toluene modeled to be degraded to less than AWQC by earlier analysis, but this conclusion has been contested by EPA (2000).	Seven seep samples were collected; seven metals and BEHP exceeded action levels.	Toluene, arsenic, chromium, copper, lead, mercury, vanadium, zinc, benzene, ethylbenzene, and xylene Metals: No detects > SQS. Organic chemicals: BTEX not detected in single sample available.
South Park Landfill	vinyl chloride	Vinyl chloride not expected to discharge at concentrations of concern because of relatively low concentrations and distance to LDW.	No known data	Sediment evaluation was not conducted because of distance to LDW and because vinyl chloride was not expected to accumulate in sediment.

Site	COCs in Groundwater[a]	Fate and Transport Issues	Seep Data	COCs Evaluated in Sediment[b, c]
T-108/Chiyoda	cPAHs, arsenic, cadmium, lead, chromium, copper, nickel, and zinc (PCBs detected in duplicate sample)	No PCBs and cPAHs in groundwater plumes.	No known data.	PAHs, PCBs, arsenic, cadmium, lead, chromium, copper, nickel, and zinc Metals: Of 59 to 61 detects, arsenic, cadmium, chromium, and lead > SQS or CSL in one sample; zinc > SQS in four samples. Organic chemicals: PCBs > SQS in 33 of 70 samples, > CSL in 5 samples.

Source: Windward (2003a)

[a] COCs in groundwater were identified using best professional judgment after evaluation of data presented in referenced reports based on the presence of groundwater plumes, comparisons to screening criteria, and frequency of detection. These chemicals are not necessarily considered COCs with respect to migration to the LDW or accumulation in LDW sediment.

[b] Some groundwater COCs listed in the first column for a site may not have been evaluated in sediment if not detected in downgradient wells closest to the LDW. Additional COCs may have been evaluated even if not listed as primary groundwater COCs, based on rationale presented in Appendix G of the Phase 1 RI (Windward 2003a).

[C] The information evaluated includes sediment data available at the time of the Phase 1 RI, sediment data collected as part of the Phase 2 RI program is not reflected on this table.

AWQC – ambient water quality criteria

BTEX – benzene, toluene, ethylbenzene, and xylene compounds

COC – chemical of concern

cPAH – carcinogenic polycyclic aromatic hydrocarbons

CSL – cleanup screening level

EPA – US Environmental Protection Agency

GWI – Great Western International

LDW – Lower Duwamish Waterway

PAH – polycyclic aromatic hydrocarbons

PCB – polychlorinated biphenyl

PCE – tetrachloroethene

RI – remedial investigation

SQS – sediment quality standards

T-108 – Terminal 108

T-117 – Terminal 117

TCE – trichloroethene

VOC – volatile organic compound

At 11 of the 12 sites evaluated in the Phase 1 RI, no evidence was found for metals accumulation in sediment to concentrations greater than SMS criteria or DMMP guidelines in potential discharge zones.[97] Elevated concentrations of metals in sediment adjacent to Boeing Plant 2 were likely associated with contaminated fill rather than groundwater. Remediation is proposed for this area and the fill material, and detailed mass loading analyses concluded that following remediation planned for the site, metals are not expected to accumulate in nearshore sediments through the groundwater pathway.

PCB concentrations in sediment near Boeing Plant 2,[98] Malarkey Asphalt (T-117), and T-108/Chiyoda were also elevated. PCBs near Boeing Plant 2 are known to have originated from releases via outfalls from underground transformer vaults, and PCBs have not been identified as a COC in groundwater. Pathways other than groundwater are also highly probable at Malarkey Asphalt (T-117) and the T-108/Chiyoda sites based on groundwater data and the known high retardation factors for PCBs in groundwater.

At the time of the Phase 1 RI, seep data were available at 4 of the 12 sites. In general, chemical concentrations in seep samples may be influenced by inputs from LDW water, groundwater, and sediment. Seep data presented in the Phase 1 RI indicated a few sites where chlorinated solvents have been detected (i.e., GWI and Boeing Plant 2). Porewater studies conducted in these areas indicated low risk to benthic invertebrates exposed through porewater in the high intertidal of these areas (see Section 9.4.6.2). Chlorinated solvents have not been detected in sediment at any of the potential discharge zones based on the data available.

Review of groundwater data prior to 2003 showed that select metals were identified in seeps at Boeing Isaacson, Boeing Plant 2, and Rhône-Poulenc; metals in seep samples exceeded the ambient water quality criteria (AWQC) at the latter two sites. Metals did not exceed SQS criteria in sediment adjacent to Rhône-Poulenc, whereas metals did exceed the SQS in sediment adjacent to Boeing Plant 2, likely because of fill material, as discussed above. Seep data, at Boeing Plant 2 in particular, are difficult to interpret with respect to the source of the chemicals because of additional influences (i.e., chemicals in seeps may be the result of a mix of inputs from LDW water, groundwater, and sediment).

In summary, the above preliminary review of available data conducted for the Phase 1 RI indicated that at most of the contaminated groundwater sites in the LDW basin, COCs are likely to be attenuated or diluted prior to groundwater discharge to the LDW because of source control and natural processes, resulting in concentrations

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[97] At 3 of these 11 sites, metals exceeded their SMS criteria in a small percentage of sediment samples (Table 9-11).

[98] PCBs were also detected in seep samples.

below those of concern. At the few sites where COCs have been found in both groundwater and seeps, these chemicals have not accumulated in sediment via the groundwater pathway. At the few sites with elevated PCBs in nearby sediments, a source other than groundwater has either been demonstrated or is suspected.

Overall, the groundwater pathway assessment completed for the Phase 1 RI showed that contaminated groundwater at upland properties adjacent to the LDW does not appear to be a major source of chemicals identified in surface and subsurface sediment within the LDW. COCs in upland groundwater do not correlate with those chemicals that have been identified as the risk drivers derived through completion of the human health and ecological risk assessment process. To help support and further interpret the overall conclusions of the groundwater pathways assessment, LDWG implemented a porewater and seep study in the LDW (discussed in the sections that follow).

Updated groundwater information (since 2002) for the 12 properties included in the Phase 1 RI groundwater pathways assessment and other adjacent upland properties not originally evaluated is included in Appendix I. The appendix includes information on COCs identified in groundwater for various properties and the status of their groundwater monitoring programs, if available. Raw monitoring data for these locations are not provided in this appendix, but key references are listed.

9.4.6.2 LDW porewater study

Porewater chemistry has implications for source control because chemicals, such as VOCs, can be transported through groundwater to LDW sediments; although VOCs rarely accumulate in sediment, benthic invertebrates can be exposed to VOCs in downgradient areas. Porewater samples were collected in select areas where VOCs had been historically detected in groundwater at upland properties immediately adjacent to the LDW. The GWI and Boeing Plant 2/Jorgensen Forge sites were selected for the porewater investigation because existing data indicated that these areas had the greatest potential for VOC contaminated porewater from the groundwater pathway. The results of the porewater sampling are described in detail in Section 4 and summarized below (Windward 2005f).

Porewater samples were collected at these two locations using both piezometers and peepers (Windward 2006a). VOCs were not detected in any of the 12 porewater samples collected using piezometers. In the peeper samples, 16 of the 71 VOCs analyzed were detected in at least one sample. The highest concentrations were reported for vinyl chloride and cis-1,2-dichloroethene, at 2,900 and 2,500 µg/L, respectively (both detected in sample LDW-DW-G-PE-06 at the GWI site). This peeper was located adjacent to seep S-13, where concentrations of these two VOCs were also high (1,600 µg/L for vinyl chloride and 5,400 µg/L for cis-1,2-dichlororethene (Windward 2006a)). Vinyl chloride and cis-1,2-dichloroethene were detected at all eight peeper sampling locations at the GWI site.

Four VOCs (1,1-dichloroethane, cis-1,2-dichloroethene, trichloroethene, and vinyl chloride) were detected in porewater samples collected using peepers at the Boeing Plant 2/Jorgensen Forge site. At least one VOC was detected in samples collected from five of the eight locations at this site, with concentrations ranging from 0.2 (cis-1,2-dichloroethene and trichloroethene) to 13 µg/L (vinyl chloride).

The results of the porewater investigation showed that even though VOCs were detected in porewater from these high intertidal areas, concentrations were relatively low (Windward 2006a). VOCs were only detected in peeper samples near the seeps and were not detected in piezometer samples located farther offshore. Risks to benthic invertebrates in these areas were very low (Appendix A).

Because these areas were considered to be a worst-case exposure areas with respect to the potential for adverse effects for benthic invertebrates from VOCs, and other areas with seep water data had much lower VOC concentrations, it was concluded that there was a very low likelihood of appreciable risks from VOCs in the rest of the LDW.

9.4.6.3 LDW seep study

In 2004, a seep survey was conducted to evaluate whether seeps contribute significant chemical inputs to the LDW and to determine whether additional seep water or surface sediment sampling was needed. A reconnaissance survey conducted as part of this effort involved the visual identification of 82 seep locations; 18 of these locations were selected for sample collection and chemical analysis. Seeps were selected for analysis based on the seep's proximity to potentially contaminated upland properties, flow, and any visual indications (e.g., oily, colored discharge, or sediments) that might suggest a potential for chemical input. The locations of the 82 identified seeps and the subset of 18 seeps selected for sampling are shown on Map 4-11. Two of the 18 seeps selected for chemical analysis (seep 24 near RM 4.2 E and seep 53 near RM 2.2) were not sampled because they were dry at the time of sampling. The results of the seep sampling are described in detail in Section 4 and summarized below.

Seep data from this study were compared to marine WQC to identify areas with higher concentrations (i.e., these criteria were not derived to assess risk via this exposure route). In filtered seep samples, copper, arsenic, and zinc were the only chemicals that exceeded WQC. Copper was detected in seep water from seven seeps at concentrations greater than acute WQC. Arsenic and zinc were detected in seep water at concentrations greater than acute WQC at two seeps each. In unfiltered seep samples, which may have contained suspended sediment, heptachlor epoxide, mercury, and total PCBs exceeded chronic WQC in one, three, and two samples, respectively.

Seep locations with WQC exceedances included seep 71 at RM 0.1 W (mercury), seep 64 south of Kellogg Island (PCBs), seep 76 at the head of Slip 1 (arsenic, zinc, and mercury), seep 61 at RM 1.5 W (arsenic), seep 80 at RM 1.6 E (copper), seep 82 in Slip 2 (copper and zinc), seep 54 in EAA 2 (mercury and PCBs), seeps 10 and 12 in Slip 4 and

near the mouth of Slip 4 (copper), seep 48 at RM 3.0 W (copper), seep 20 at RM 3.7 in the south end of the Boeing Plant 2/ Jorgensen Forge EAA (copper), and seep 39 at RM 4.3 W (copper and heptachlor epoxide).

Chemicals detected above WQC in the sampled seeps were compared to chemicals exceeding the SQS in adjacent sediments to assess the potential for a connection between seep discharge and sediment contamination. Fourteen of the 16 seep samples analyzed did not appear to have a connection between chemicals in seeps and sediment contamination. One seep (seep 54 in EAA 2 at RM 2.2) had a potential connection and another seep (seep 76 in Slip 1) had a more probable connection to nearby sediment quality. In seep 54, concentrations of total PCBs, mercury, and lead were greater than the chronic WQC in unfiltered seep samples and PCBs and mercury exceeded the CSL in sediment samples within this inlet. However, because the seep samples were unfiltered, the elevated concentrations could have resulted from entrained contaminated sediment during sampling. In Slip 1, concentrations of arsenic and zinc in a filtered seep sample were greater than the acute WQC and arsenic and zinc concentrations exceeded the CSL in adjacent surface sediment and were also elevated in subsurface sediments.

9.4.7 Bank erosion

Erosion of contaminated bank soils and sediments can be an ongoing source of contamination to the LDW, as discussed in Section 9.1.6. Unprotected shoreline banks can be eroded by both wind and surface water runoff. Most of the shoreline area along the LDW has been developed over the past 100 years for industrial, commercial, or manufacturing purposes. Currently, the shoreline of most of the LDW study area is armored with constructed steel and concrete bulkheads, sheet pile walls, and steep riprap banks, limiting bank erosion in many areas.

Characterization of erodable material is required at sites where bank construction or remediation is planned. Such characterization has recently been completed at two EAAs in the LDW. At Slip 4 and T-117, PCB contamination has been found in bank soils. At both of these sites, the erosion of bank material was identified as a direct source of contamination in sediment (Ecology 2005a, 2006a). Remediation activities at both of these EAAs, both planned and ongoing, will include removal or capping of contaminated bank soils to eliminate this pathway as a potential source of sediment recontamination.

9.4.8 Routine waterway activities

Potential sources of contamination from routine activities on the LDW include dredging or in-water structure maintenance, which may resuspend contaminated material. In addition, chemicals may leach from structures such as piers or pilings, or originate from recreational and commercial boats or overwater cargo transfer. Vessel movement can potentially resuspend contaminated sediment by a process known as propeller wash. Further discussion of these activities is presented below.

9.4.8.1 Sediment resuspension activities

Dredging activities in the LDW have the potential to resuspend contaminated sediments. In-water construction projects, including the removal of old structures, also have the potential to resuspend sediments. To minimize sediment resuspension during dredging, demolition, or construction activities, appropriate BMPs are required. As part of the design phase of the project, BMPs, compliance protocols, and other requirements are identified and monitored. The most common types of BMPs are operational controls or the use of special equipment.

Operational controls are used to minimize the resuspension of sediment. These controls can include slowing down operations, making sure that barges are not overfilled, and containing or filtering leaks and/or spills from barges. The use of special equipment can also minimize the resuspension of sediment. Special equipment can include specially constructed dredging buckets designed to reduce or eliminate increased turbidity of suspended solids (sometimes referred to as environmental or closed-bucket dredging), and silt curtains or Gunder booms designed to limit the distribution of suspended sediment. Water quality monitoring is also typically conducted during dredging and construction activities to ensure that potential water quality impacts are minimized.

Propwash, which generally occurs in or near berthing areas or in shallow areas LDW, may also resuspend sediment. The effect of propwash can vary depending on the sediment type, water depth, and vessel type (see Section 3). Higher bed scours may occur in the bench areas than in the navigation channel from the effects of ships maneuvering in the shallower near shore areas (Windward and QEA 2007).

9.4.8.2 Other activities

Normal day-to-day in-water operations and boat traffic in the LDW can also potentially introduce chemical contaminants to the LDW. These potential sources are generally localized. Some of these activities and potential sources are listed below.

- Direct discharge from boats such as gray, bilge, or ballast waters
- Incidental fuel releases and spills from boats
- Incidental spills from loading and offloading of materials
- Wastes and other materials from marinas and small boatyards.

9.5 SUMMARY

The LDW is a dynamic system that has been altered by over 100 yrs of industrial, commercial, and residential development. Potential sources of contamination to LDW sediment are numerous and may include both historical and ongoing operations and various pathways. Characterization of potential chemical sources to the LDW started in the late 1980s and is expected to continue for years to come.

Earlier sediment investigations have helped to identify seven EAAs that require further sediment characterization and/or remediation; remedial actions have already been implemented at two of these seven locations and are currently in the planning stages for three more.

Ecology is leading the source control evaluation and assessment of these seven EAAs and an additional eight TTAs. Ecology has or will prepare SCAPs and Data Gaps Reports to help direct future investigation and remediation efforts within these areas while assessing potential sources for the chemicals identified.

Modern regulatory requirements have established inspection and compliance programs to monitor and control the handling and disposal of manufacturing wastes produced by industrial operations along the LDW. These regulations have helped reduce sources of contamination that were common practices in the past.

Air emissions are a potential ongoing source of many chemicals to the LDW. Two hundred businesses in the greater LDW basin are registered as active sources of air pollutants. Programs are in place to help reduce these emissions over the long term. However, motor vehicles and the burning of wood and fossil fuels will remain ongoing sources of chemicals to the atmosphere until technologies evolve and these sources can be controlled.

Permit programs are in place to help monitor and control direct discharges to the LDW from CSOs, EOFs, and SDs in the basin. Since 1970, the City has reduced the annual CSO volume citywide by approximately 95%; and since 1988, the County has reduced their annual CSO volume by 63%. Direct discharges to the LDW contribute approximately 1% of the annual sediment load to the LDW compared to inputs from the Duwamish/Green River system.

In source tracing samples collected to date (including catch basin, in-line sediment, and right-of-way catch basin samples), BEHP, BBP, and zinc are the chemicals most frequently exceeding SQS criteria. Since sampling began in 2003, PCBs, BEHP, BBP, copper, lead, mercury, and zinc have generally been identified in over 50% of the samples collected.

Chemicals identified in upland groundwater or seeps along the LDW have generally not been shown to accumulate in adjacent sediments, except in a few small areas (e.g., the head of Slip 1). Most chemicals identified in upland groundwater samples are likely to be diluted or attenuated prior to discharge. Groundwater is not considered to be a major source of chemicals to the LDW.

Approximately 87% of the river bank along the LDW is covered or armored with impervious materials; however, erosion of contaminated bank material into the LDW is still possible. This potential source of chemicals to the waterway is currently not well characterized.

10 Areas of Interest for Source Control

Ecology, in consultation with EPA, has identified 15 areas of interest for source control in accordance with their source control strategy (Ecology 2007k). This section describes the origin of this list of areas (Section 10.1), discusses the surface sediment chemistry data that were the primary basis for identifying many of these areas (Section 10.2), and summarizes the key findings and ongoing source control efforts for each area (Section 10.3).

10.1 IDENTIFICATION OF AREAS

Since the initiation of the RI/FS, three analyses have been conducted to define specific areas within the LDW that require further investigation. The first analysis was conducted as part of the Phase 1 RI (Windward 2003a). One of the major objectives of the Phase 1 RI was to identify candidate areas for early action. The following types of areas were identified based on Phase 1 data:

- Areas that had at least three contiguous surface sediment samples with concentrations of any detected chemicals above the CSL
- Areas within the upper 5th percentile of total PCB concentrations to address exposure through seafood consumption
- Areas with chemical concentrations that exceeded those associated with a 1×10^{-4} cancer risk threshold for direct sediment exposure

The result of the Phase 1 analysis was the identification of seven candidate areas; (Windward 2003d) these seven areas, subsequently referred to as EAAs, are included in Table 10-1 and on Map 10-1.

Table 10-1. Early Action Areas

EARLY ACTION AREA NO.[a]	ASSOCIATED NAME	POTENTIAL PRIORITY AREA (PPA) DESIGNATION[b]	RIVER MILE	SIDE OF LDW
EAA 1	Duwamish/ Diagonal	Area 1 (RM 0.4) and Area 2 (RM 0.6)	0.4 to 0.6	east
EAA 2	Early Action Area 2 (RM 2.2)	Area 6	2.2	west (inlet area)
EAA 3	Slip 4	na	2.8 to 2.9	east
EAA 4	Boeing Plant 2/ Jorgensen Forge	Area 7	2.9 to 3.7	east
EAA 5	Terminal 117	na	3.5 to 3.7	west
EAA 6	Early Action Area 6 (RM 3.8)	Area 8	3.8	east
EAA 7	Norfolk	na	4.8 to 5.0	east

[a] Identified during the Phase 1 RI (Windward 2003d).

[b] Identified in the preliminary screening of alternatives (RETEC 2006).

EAA – Early Action Area

na – not applicable

PPA – Potential Priority Area

RI – remedial investigation

RM – river mile

The next analysis that identified specific areas within the LDW was documented in the *Technical Memorandum: Draft Preliminary Screening of Alternatives, Lower Duwamish Waterway, Seattle, Washington*, an FS document that was completed early in the process (RETEC 2006). The areas identified in the preliminary screening of alternatives were:

- Sponsored EAAs – Areas for which one or more individual LDWG members had made a commitment to conduct removal actions (i.e., Duwamish/Diagonal, Slip 4, Boeing Plant 2/Jorgensen Forge, T-117, and Norfolk) (Table 10-1)
- Potential Priority Areas (PPAs) – Areas that had been identified as EAAs but for which no entity had accepted or been assigned responsibility for sediment remediation (i.e., EAA 2 [RM 2.2] and EAA 6 [RM 3.8]), as well as other areas with characteristics similar to EAAs based on additional surface sediment data collected for the RI in 2005. Specifically, the two criteria used were: 1) at least three contiguous surface sediment samples with concentrations of any detected chemicals above the CSL, and 2) at least three contiguous surface sediment samples with concentrations of any detected chemicals above the SQS and either a single CSL toxicity test failure or two or more SQS toxicity test failures. PPA boundaries are identified in the draft preliminary screening of alternatives memo (RETEC 2006).

The most recent analysis undertaken to identify areas for further investigation was conducted by Ecology, in consultation with EPA, to prioritize areas where source identification and control efforts may be needed (Ecology 2007k). Ecology and EPA designated these areas as Tier Two Areas (TTAs)[99] (Table 10-2).

Table 10-2. Tentative Tier Two Areas

Tier Two Area[a]	TTA Letter Designation[a]	TTA No.[a]	PPA Designation[b]	River Mile	Side of LDW
Glacier Bay	TTA A	3, 4, & 5	Area 3 (RM 1.3), Area 4 (RM 1.35), and Area 5 (RM 1.4)	1.3 to 1.55	west
Ash Grove Cement[c]	TTA B	15	na	0.0 to 0.3	east
Slip 1/Manson[c]	TTA C	14	na	0.9 to 1.0	east
Glacier NW to Saint-Gobain Glass Company[c]	TTA D	13	na	1.3 to 1.7	east
Slip 3 to Slip 2	TTA E	12	na	1.8 to 2.0	east
Slip 3	TTA F	11	na	2.1	east
Slip 4 to Seattle Boiler Works[c]	TTA G	10	na	2.4 to 2.8	east
RM 3.8 (Same as EAA 6)	TTA H	na	na	3.8	east
Slip 6	TTA I	9	na	3.9 to 4.4	east

[a] Identified by Ecology and EPA (Ecology 2007k).

[b] Identified in the preliminary screening of alternatives (RETEC 2006).

[99] Tier One Areas, as identified by Ecology (2007k), are equivalent to the seven EAAs.

[c] Company names included in the TTA names designated by Ecology, in consultation with EPA, are used only to designate areas of interest for source control. The properties owned by these companies are not necessarily included in the cleanup area nor are the companies necessarily responsible for contamination in these areas.

EAA – Early Action Area
Ecology – Washington State Department of Ecology
EPA – US Environmental Protection Agency
na – not applicable
NW – northwest
PPA – Potential Priority Area
RM – river mile
TTA – Tier Two Area

The Tier Two terminology was first introduced in 2004 in the source control strategy (Ecology 2004b); however, the list of TTAs was not formally published until 2007. TTAs were identified based on sediment data, size of the upland area draining to the TTA, and general knowledge about facilities operating in the basin (Ecology 2007k). TTAs were identified only for source control evaluation and do not necessarily represent areas requiring sediment cleanup.

10.2 SURFACE SEDIMENT CHEMISTRY RELATIVE TO SMS

Surface sediment chemistry data were an important component of the process used to identify areas requiring further investigation (Section 10.1). This section briefly summarizes LDW surface sediment chemistry data, focusing on exceedances of SMS criteria. The derivation of SMS criteria is discussed in detail in the ERA (Appendix A, Section A.3.2). Concentrations of non-SMS chemicals of interest, such as TBT, dioxins and furans, and total DDT, are also noted in this section because concentrations of these chemicals in surface sediments were also considered in the identification process.

The data presentation (in relation to SMS exceedances) is divided into three parts: surface sediment chemistry data for the entire LDW, surface sediment chemistry data for areas inside the boundaries of the EAAs, and surface sediment chemistry data for areas of the LDW outside of the EAA boundaries. Subsurface sediment chemistry data that are discussed in Section 4 were not evaluated for the purposes of designating areas of interest for source control. These data are compared to SMS criteria in Appendix J.

Table 10-3 presents a summary of SMS criteria and SQS and CSL exceedances for all 47 SMS chemicals for the entire LDW.[100] Detected concentrations for 39 SMS chemicals exceeded the CSL at one or more locations. Two additional chemicals (anthracene and pentachlorophenol) had concentrations that exceeded the SQS but did not exceed the CSL. Detected concentrations of the remaining six SMS chemicals (diethyl phthalate, di-n-butyl phthalate, di-n-octyl phthalate, acenaphthylene, hexachlorobutadiene, and 2-methylphenol) did not exceed the SQS or the CSL.

The frequency at which detected chemical concentrations exceeded the SQS ranged from 1.1 to 4.9% for metals, 0 to 5.1% for PAHs, 0 to 13% for phthalates, 0 to 2.8% for

[100] The data summarized in Table 10-2 are the same data presented in Section 4 and are also known as the RI baseline surface sediment chemistry dataset.

other SVOCs, and 37% for total PCBs. The frequency at which detected chemical concentrations exceeded the CSL ranged from 0.94 to 3.1% for metals, 0 to 1.2% for PAHs, 0 to 7.1% for phthalates, 0 to 0.97% for other SVOCs, and 13% for total PCBs. Maps 10-2a through 10-2f present detailed results for each location with SQS or CSL exceedances for any detected chemical in surface sediment.

Table 10-3. Comparison of LDW surface sediment chemical data to SMS chemical criteria

Chemical	Number of Results	Detection Frequency		Sediment Chemistry								SMS Criteria		
				Detected Concentrations > SQS and ≤ CSL[a]		Detected Concentrations > CSL[a]		Locations with RLs > SMS Criteria[b]		Maximum Ratio of Detected Conc. to SQS[a]	Maximum Ratio of Detected Conc. to CSL[a]			
		No. of Detects	%	No.	%[c]	No.	%[c]	> SQS and ≤ CSL[b]	> CSL			SQS	CSL	Unit
Metals														
Arsenic	852	794	93	5	0.59	9	1.1	0	0	19	12	57	93	mg/kg dw
Cadmium	838	584	70	2	0.24	11	1.3	0	0	24	18	5.1	6.7	mg/kg dw
Chromium	850	850	100	1	0.12	8	0.94	0	0	4.2	4.1	260	270	mg/kg dw
Copper	852	852	100	0	0	12	1.4	0	0	31	31	390	390	mg/kg dw
Lead	852	852	100	2	0.23	19	2.2	0	0	51	43	450	530	mg/kg dw
Mercury	868	746	86	16	1.8	27	3.1	0	0	11	7.8	0.41	0.59	mg/kg dw
Silver	823	499	61	0	0	10	1.2	0	0	44	44	6.1	6.1	mg/kg dw
Zinc	849	849	100	26	3.1	16	1.9	0	0	24	10	410	960	mg/kg dw
PAHs														
2-Methylnaphthalene	818	139	17	0	0	3	0.37	6	1	4.2	2.5	38	64	mg/kg OC
Acenaphthene	828	304	37	15	1.8	3	0.36	9	2	16	4.6	16	57	mg/kg OC
Acenaphthylene	818	128	16	0	0	0	0	0	1	0.38	0.38	66	66	mg/kg OC
Anthracene	828	576	70	2	0.24	0	0	0	0	1.7	0.32	220	1,200	mg/kg OC
Benzo(a)anthracene	828	748	90	9	1.1	4	0.48	0	0	4.0	1.6	110	270	mg/kg OC
Benzo(a)pyrene	822	747	91	5	0.61	4	0.49	0	0	4.2	2.0	99	210	mg/kg OC
Benzo(g,h,i)perylene	823	682	83	10	1.2	8	0.97	4	1	5.8	2.3	31	78	mg/kg OC
Total benzofluoranthenes	822	757	92	5	0.61	5	0.61	0	0	3.9	2.0	230	450	mg/kg OC
Chrysene	828	773	93	25	3.0	2	0.24	0	0	3.7	1.3	110	460	mg/kg OC
Dibenzo(a,h) anthracene	828	436	53	16	1.9	4	0.48	10	6	5.9	2.2	12	33	mg/kg OC
Dibenzofuran	827	248	30	6	0.73	3	0.36	9	2	15	3.8	15	58	mg/kg OC
Fluoranthene	828	797	96	32	3.9	10	1.2	0	0	8.1	2.7	160	1,200	mg/kg OC

Chemical	Number of Results	Detection Frequency		Sediment Chemistry								SMS Criteria		
				Detected Concentrations > SQS and ≤ CSL[a]		Detected Concentrations > CSL[a]		Locations with RLs > SMS Criteria[b]		Maximum Ratio of Detected Conc. to SQS[a]	Maximum Ratio of Detected Conc. to CSL[a]			
		No. of Detects	%	No.	%[c]	No.	%[c]	> SQS and ≤ CSL[b]	> CSL			SQS	CSL	Unit
Fluorene	828	382	46	10	1.2	3	0.36	8	1	13	3.7	23	79	mg/kg OC
Indeno(1,2,3-cd)pyrene	823	726	88	16	1.9	9	1.1	1	0	5.9	2.3	34	88	mg/kg OC
Naphthalene	818	152	19	0	0	2	0.24	0	0	2.6	1.7	99	170	mg/kg OC
Phenanthrene	828	759	92	24	2.9	3	0.36	0	0	15	3.1	100	480	mg/kg OC
Pyrene	828	788	95	2	0.24	4	0.48	0	0	1.8	1.5	1,000	1,400	mg/kg OC
Total HPAH	828	803	97	24	2.9	4	0.48	0	0	4.7	1.5	960	5,300	mg/kg OC
Total LPAH	828	763	92	3	0.36	3	0.36	0	0	6.2	2.9	370	780	mg/kg OC
Phthalates														
BEHP	832	674	81	48	5.8	59	7.1	3	2	11	7.4	47	78	mg/kg OC
BBP	822	425	52	71	8.6	8	0.97	75	1	110	8.3	4.9	64	mg/kg OC
Diethyl phthalate	832	41	4.9	0	0	0	0	4	0	0.33	0.18	61	110	mg/kg OC
Dimethyl phthalate	822	156	19	0	0	1	0.12	8	5	2.5	1.1	53	53	mg/kg OC
Di-n-butyl phthalate	822	189	23	0	0	0	0	0	0	0.64	0.082	220	1,700	mg/kg OC
Di-n-octyl phthalate	832	49	5.9	0	0	0	0	5	0	0.60	0.0078	58	4,500	mg/kg OC
Other SVOCs														
1,2,4-Trichlorobenzene	816	5	0.61	0	0	1	0.12	231	128	2.3	1.4	0.81	1.8	mg/kg OC
1,2-Dichlorobenzene	816	17	2.1	0	0	3	0.37	0	110	15	10	2.3	2.3	mg/kg OC
1,4-Dichlorobenzene	816	39	4.8	0	0	3	0.37	77	18	21	13	3.1	9.0	mg/kg OC
2,4-Dimethylphenol	813	5	0.62	0	0	1	0.12	0	222	10	10	29	29	µg/kg dw
2-Methylphenol	821	7	0.85	0	0	0	0	0	114	0.92	0.92	63	63	µg/kg dw
4-Methylphenol	831	82	9.9	0	0	4	0.48	0	9	6.9	6.9	670	670	µg/kg dw
Benzoic acid	822	70	8.5	0	0	8	0.97	0	107	6.9	6.9	650	650	µg/kg dw
Benzyl alcohol	812	15	1.8	2	0.25	3	0.37	7	106	12	9.2	57	73	µg/kg dw
Hexachlorobenzene	819	46	5.6	4	0.49	2	0.24	283	105	10	1.7	0.38	2.3	mg/kg OC

Chemical	Number of Results	Detection Frequency		Sediment Chemistry								SMS Criteria		
				Detected Concentrations > SQS and ≤ CSL[a]		Detected Concentrations > CSL[a]		Locations with RLs > SMS Criteria[b]		Maximum Ratio of Detected Conc. to SQS[a]	Maximum Ratio of Detected Conc. to CSL[a]			
		No. of Detects	%	No.	%[c]	No.	%[c]	> SQS and ≤ CSL[b]	> CSL			SQS	CSL	Unit
Hexachlorobutadiene	818	0	0	0	0	0	0	58	88	na	na	3.9	6.2	mg/kg OC
n-Nitrosodiphenylamine	818	23	2.8	0	0	2	0.24	34	33	3.9	2.8	11	11	mg/kg OC
Pentachlorophenol	785	12	1.5	1	0.13	0	0	88	29	1.1	0.59	360	690	µg/kg dw
Phenol	831	257	31	17	2.0	6	0.72	5	0	6.7	2.3	420	1,200	µg/kg dw
PCBs								0						
Total PCBs	1,327	1,243	94	314	24	174	13	0	0	830	150	12	65	mg/kg OC

[a] For individual locations with TOC ≥ 4% or ≤0.5%, locations were counted as greater than the SQS or CSL if the dry-weight concentration of chemicals with OC-normalized SMS criteria was greater than the LAET or 2LAET, respectively.

[b] Number of locations with RLs greater than the SQS or CSL.

[c] Percentage of total number of locations with detected concentrations that were greater than the SQS and less than or equal to the CSL, or greater than the CSL.

2LAET – second lowest apparent effects threshold
BEHP – bis(2-ethylhexyl) phthalate
BBP – butyl benzyl phthalate
CSL – cleanup screening level
dw – dry weight
HPAH – high-molecular-weight polycyclic aromatic hydrocarbon
LAET – lowest apparent effects threshold
LPAH – low-molecular-weight polycyclic aromatic hydrocarbon
na – not applicable
OC – organic carbon
PCB – polychlorinated biphenyl
RL – reporting limit
SMS – Washington State Sediment Management Standards
SQS – sediment quality standard
SVOC – semivolatile organic compound

Total PCB concentrations in surface sediment exceeded the SQS at more locations than any other chemical (488 locations exceeded the SQS, and, of those, 174 also exceeded the CSL). Most of the total PCB concentrations that exceeded the CSL were in samples collected in or near EAAs between RM 0.0 and RM 1.0, between RM 3.2 and RM 4.0, and in Slip 4 (Map 10-1). The chemical with the next highest number of locations with detected concentrations in surface sediment exceeding the SQS was BEHP (107 locations exceeded the SQS, and, of those, 59 also exceeded the CSL), followed by BBP (79 locations exceeded the SQS, and, of those, 8 also exceeded the CSL) and mercury (43 locations exceeded the SQS, and, of those, 27 also exceeded the CSL) (Table 10-2). Approximately 50% of the phthalate exceedances were located between RM 0.4 and RM 0.6, in and around the Duwamish/Diagonal EAA. Mercury exceedances were located in discrete locations throughout the LDW, although approximately 60% of the exceedances were also between RM 0.4 and RM 0.6 (in the Duwamish/Diagonal EAA) and between RM 3.5 and RM 3.6 (in the Boeing Plant 2/Jorgensen Forge EAA).

Maps 10-2a to 10-2f also present the results of toxicity tests conducted with split sediment samples of some of the surface sediment samples collected in 2005. As described in the SMS regulation (WAC 173-204) and the ERA (Appendix A), toxicity test results, relative to SQS and CSL biological criteria, overrule SQS or CSL designations based on chemistry because the toxicity test results provide a more direct assessment of sediment toxicity at the location tested.

The SMS designation based on toxicity test results agreed with the SMS designation based on chemistry for 17 of the 46 samples (Map 10-1). The toxicity designation represented a higher quality condition (i.e., lower toxicity than estimated based on chemistry alone) at 20 of the 29 locations where the toxicity and chemistry designations did not agree and a lower quality condition at the remaining 9 locations (i.e., higher toxicity than estimated based on chemistry alone). See Appendix A, Section A.3.2.2, for a comprehensive discussion of the toxicity test results.

A total of 574 locations (42% of all baseline surface sediment locations) had detected concentrations of at least one chemical that exceeded the SQS. Many of these locations were located in EAAs. The EAAs displayed on Map 10-1 represent 33 ac (7.2%) of the 460 ac LDW study area; more than half of the locations (331 or 58%) with SQS exceedances in the RI baseline dataset were within the EAA boundaries (Table 10-3). More than half of the 331 locations within the EAAs (193 or 58%) also had one or more chemical concentrations above the CSL (Table 10-4). Almost all of the 331 locations with SMS exceedances (305 locations of 92%) were within the four largest sponsored EAAs (Duwamish/Diagonal, Slip 4, T-117, and Boeing Plant 2/Jorgensen Forge), totaling 30 ac in size; these areas have been sampled extensively. Thus, the most highly contaminated areas of the LDW are within a relatively small portion of the total study area.

Table 10-4. SMS exceedances in surface sediments within and outside of the EAAs

Parameter	Number of Locations Within EAAs	Number of Locations Outside of EAAs	Total Number of Locations with SMS Exceedances
Locations with CSL exceedances	193	68	261 (45%)
Locations with SQS exceedances[a]	138	175	313 (55%)
Total number of locations with SMS exceedances	**331 (58%)**	**243 (42%)**	**574**

[a] Includes only those sampling locations that had detected concentrations above the SQS and less than or equal to the CSL.

CSL – cleanup screening level

EAA – Early Action Area

SMS – Washington State Sediment Management Standards

SQS – sediment quality standard

Total PCB concentrations exceeded the CSL at six of the seven EAAs (at EAA 6 [RM 3.8], PCB concentrations only exceeded the SQS). Each of the EAAs also had SQS or CSL exceedances for other chemicals.

Outside of the EAAs, 243 sampling locations had surface sediments that exceeded SMS criteria (42% of the total number of SMS exceedances in the LDW) (Table 10-4). Concentrations of 34 chemicals exceeded the SQS and concentrations of 30 chemicals exceeded the CSL outside the EAAs (Table 10-5).[101] As with the LDW-wide data (Section 10.2.1), total PCBs were found at the highest number of locations in areas outside of the EAAs with concentrations above the SQS (174). BEHP, fluoranthene, and BBP had the next highest numbers of locations with concentrations above the SQS (41, 30, and 29 locations, respectively) (Table 10-5). Total PCBs also had the highest number of locations with CSL exceedances (18), followed by BEHP (17) and mercury (13). Twenty-six of the other chemicals listed in Table 10-5 had concentrations that exceeded the CSL at three or fewer locations. Approximately 30 of the 243 detected exceedances were just outside the margins of the Duwamish/Diagonal EAA. Other exceedances in other areas with multiple exceedances were within and just downstream of the embayment on the west side of the LDW between RM 1.4 and RM 1.5. There were also many locations with isolated SMS exceedances.

[101] Detected concentrations of chromium; di-methyl phthalate; 1,2,4-trichlorobenzene; 1,2-dichlorobenzene; 1,4-dichlorobenzene; 2,4-dimethyl phenol; and n-nitrosodiphenylamine had SMS exceedances only within the EAAs.

Table 10-5. Summary of SMS exceedances in surface sediments outside of the EAAs

Chemical	No. of Locations with Detected Concentrations > SQS and ≤ CSL[a]	Maximum Ratio of Detected Conc. to SQS	No. of Locations with Detected Concentrations > CSL[a]	Maximum Ratio of Detected Conc. to CSL
Metals				
Arsenic	4	19	9	12
Cadmium	1	1.4	1	1.1
Copper	0	3.6	5	3.6
Lead	0	1.7	4	1.5
Mercury	5	4.4	13	3.1
Silver	0	1.0	1	1.0
Zinc	10	6.9	4	2.9
PAHs				
2-Methylnaphthalene	0	4.2	1	2.5
Acenaphthene	8	16	2	4.6
Anthracene	1	1.7	0	0.32
Benzo(a)anthracene	5	2.0	2	1.6
Benzo(a)pyrene	2	2.0	1	1.1
Benzo(g,h,i)perylene	7	2.4	2	2.2
Total benzofluoranthenes	2	1.5	2	1.3
Chrysene	16	2.7	2	1.3
Dibenzo(a,h)anthracene	11	2.2	0	0.65
Dibenzofuran	2	11	2	2.9
Fluoranthene	24	6.9	6	2.7
Fluorene	5	11	1	3.3
Indeno(1,2,3-cd)pyrene	11	2.7	2	2.3
Naphthalene	0	2.6	1	1.5
Phenanthrene	13	8.3	2	1.7
Pyrene	1	1.8	3	1.5
Total HPAHs	14	2.6	3	1.5
Total LPAHs	0	4.6	2	2.2
Phthalates				
BEHP	24	4.5	17	2.7
BBP	29	4.9	0	0.20
Other SVOCs	0			
4-Methylphenol	0	2.1	3	2.1
Benzoic acid	0	6.9	7	6.9
Benzyl alcohol	1	12	3	9.2
Hexachlorobenzene	4	10	2	1.7
Pentachlorophenol	1	1.1	0	0.57
Phenol	10	3.3	2	1.2

Chemical	No. of Locations with Detected Concentrations > SQS and ≤ CSL[a]	Maximum Ratio of Detected Conc. to SQS	No. of Locations with Detected Concentrations > CSL[a]	Maximum Ratio of Detected Conc. to CSL
PCBs				
Total PCBs	156	830	18	150

[a] For individual locations with TOC ≥ 4% or ≤ 0.5%, locations were counted as greater than the SQS or CSL if the dry-weight concentration of chemicals with OC-normalized SMS criteria was greater than the LAET or 2LAET, respectively.

2LAET – second lowest apparent effects threshold

BEHP – bis(2-ethylhexyl) phthalate

BBP – butyl benzyl phthalate

CSL – cleanup screening level

EAA – Early Action Area

HPAH – high-molecular-weight polycyclic aromatic hydrocarbon

LAET – lowest apparent effects threshold

LPAH – low-molecular-weight polycyclic aromatic hydrocarbon

PCB – polychlorinated biphenyl

SMS – Washington State Sediment Management Standards

SQS – sediment quality standard

SVOC – semi-volatile organic compound

Outside of EAAs, PCB concentrations above the 95th percentile are typically single locations surrounded by much lower concentrations. The highest total PCB concentration in LDW surface sediment (223 mg/kg dw, or 10,000 mg/kg OC) in the RI baseline dataset was 150 times higher than the CSL. This sample was collected at RM 4.9, just north of the Norfolk EAA in December 1995. Other nearby samples collected at the same time had much lower concentrations: 1.37 mg/kg dw, or 45 mg/kg OC (17 ft away), 0.45 mg/kg dw, or 50 mg/kg OC (46 ft away), and 0.615 mg/kg dw, or 26 mg/kg OC (50 ft away). The nearby samples exceeded the SQS but not the CSL. An attempt was also made in 2005 to reoccupy the location with the highest PCB concentration. The resampled location was 11 ft away from the original coordinates; the total PCB concentration in the 2005 sample was 2.7 mg/kg dw, or 390 mg/kg OC (a CSL exceedance). Based on these data, it appears the location with this high PCB concentration is limited to a very small area.

Elevated (relative to the rest of the LDW) concentrations of non-SMS chemicals of interest have also been detected in surface sediment outside of the EAAs. As summarized in Section 4.2.3, there are 29 locations outside of the EAAs with TBT concentrations ranging between 110 and 3,000 µg/kg dw. The two highest concentrations were located within the Glacier Bay TTA (3,000 µg/kg dw, sampled in 2005) and at RM 0.1, just west of the navigation channel (2,300 µg/kg dw, sampled in 2004) (Map 4-47).

One location outside of the EAAs was identified with a total DDT concentration greater than 69 µg/kg dw (the DMMP maximum level) (Map 4-58). This location, with a total DDT concentration of 75 µg/kg dw, was at RM 2.1, just outside of Slip 3. Total PCBs at this location were 310 µg/kg dw, or 14 mg/kg OC (an SQS exceedance). The total DDT concentration was qualified as an estimated value (J-qualified), although it was not JN-qualified (estimated value, tentative identification), as were 35 other total

DDT results in the surface sediment dataset because of presumed interference from PCB congeners.

Seven locations outside of EAAs have also been identified with dioxin and furan TEQs in surface sediments greater than 20 ng/kg dw. The three highest concentrations (2,100, 565, and 463 ng/kg dw), plus a fourth dioxin and furan TEQ of 49 ng/kg dw, were all detected in surface sediment samples collected in 2005 within the embayment between RM 1.4 and 1.5. The three other locations with concentrations greater than 20 ng/kg dw had dioxin and furan TEQs of 27.1 ng/kg dw (RM 1.0, just south of Kellogg Island), 124 ng/kg dw (RM 1.0 in the navigation channel), and 33.3 ng/kg dw (RM 2.4 E) (Map 4-37).

For certain chemicals, analytical laboratory RLs exceeded the SQS (and in fewer cases, the CSL) (Table 10-2). These elevated RLs, which generally reflected dilutions or analytical challenges, create some uncertainty in interpreting the data relative to potential effects on benthic invertebrates. Appendix A, Section A.6.1.1, presents a detailed discussion of RL exceedances at all locations, including those with detected chemical exceedances.

The RL for at least one chemical was greater than the SQS at 294 locations with no detected exceedances of the SQS, including only two locations from sampling events conducted for the RI. The increased sensitivity of the analytical methods used during the RI greatly reduced the frequency of RL values above the SQS. There was no increase in the detection frequency of these compounds, which suggests that these chemicals are likely not present in the LDW at concentrations above the RL.

Of the 294 locations with RLs greater than the SQS and no detected exceedances, 94% of the locations had RLs for 1,2,4-trichlorobenzene, hexachlorobenzene, or 2,4-dimethylphenol that were greater than the SQS. As discussed in Section A.6.1.1 of the ERA, 1,2,4-trichlorobenzene, hexachlorobenzene, and 2,4-dimethylphenol are difficult to analyze at low concentrations using the standard SVOC analytical method and were rarely detected in LDW sediments (Table 10-3). Overall, approximately 44, 50, and 27% of all the surface sediment samples in which 1,2,4-trichlorobenzene, hexachlorobenzene, and 2,4-dimethylphenol, respectively, were not detected had RLs greater than the SQS. Detected concentrations of these compounds were greater than the SQS in one sample for both 1,2,4-trichlorobenzene and 2,4-dimethylphenol (out of five detected concentrations for each) and in six samples (out of 46 detected concentrations) for hexachlorobenzene.

There were 18 locations with no detected exceedances of the SQS but with RLs greater than the SQS for one or more chemicals other than 1,2,4-trichlorobenzene, hexachlorobenzene, and 2,4-dimethylphenol. These other chemicals included: benzoic acid (1 location), benzyl alcohol (1 location), hexachlorobutadiene (9 locations; never detected in LDW), n-nitrosodiphenylamine (14 locations; 2 of 23 detected concentrations were greater than the SQS), pentachlorophenol (1 location), phenol (1 location), one individual PAH (1 location), and two phthalate compounds

(1 location each). Benzoic acid, benzyl alcohol, hexachlorobutadiene, n-nitrosodiphenylamine, pentachlorophenol, and phenol are also difficult to analyze at low concentrations. Thus, the available data suggest that locations with RLs greater than the SQS but no detected exceedances are likely the result of analytical difficulties; the likelihood of actual chemical concentrations greater than the SQS at these locations is low.

10.3 SUMMARY OF FINDINGS FOR EACH AREA

As described in Section 10.1, 15 areas of interest for source control were identified in the LDW (Tables 10-1 and 10-2). A review of readily available source control information for these areas was performed to provide an overview of the environmental conditions of each identified area (Section 10.3.1). The summaries of findings from this review and associated maps is provided in Section 10.3.2 for each area of interest for source control.

Information in this section was based on readily available resources, most notably Ecology's Summary of Existing Information and Data Gaps Reports and Source Control Action Plans, other past summaries of source information, and an Environmental Data Resources, Inc. document that was prepared in 2006 for the LDW. Some of these documents are known to contain errors and to have omitted data and other information. None of the resource information has been independently verified beyond the citations noted. Furthermore, the primary data themselves have had varying levels of QC and data validation, which have not generally been documented in the literature reviewed. Therefore, information included in this section should not be relied upon as complete and factual. Additional evaluations of the data would be required to use the data quantitatively or as an inference to identify actual sources. It is expected that more thorough evaluations of the data will be conducted in the course of further source control investigations.

10.3.1 Methodologies used for information summation

Summaries of relevant source control information focused on chemicals determined to be risk drivers in either the HHRA or the ERA that exceeded SMS in the surface sediment within the in-water sediment boundaries of each area. Characterization and source control documentation was then reviewed to establish the environmental conditions of each area, to provide an initial review of the history of commercial and industrial operations on adjacent properties, and to obtain other historical information for the areas that may affect source identification and control efforts.

Depending on the status of source control investigations, upland properties and drainage basins associated with these 15 areas, as applicable, were identified by Ecology. In areas where relevant agency information was not available, upland properties in the vicinity of these areas (usually only those directly adjacent to the LDW shoreline) were selected for further evaluation based only on their proximity to the area's in-water boundary.

At the time of the review, Ecology had identified relevant upland properties for the seven EAAs and the Glacier Bay TTA. The associated upland properties and historical operations for the remaining TTAs were identified using the methodology previously described to establish a datum for source control evaluation. Table 10-6 summarizes the upland properties and historical upland facilities reviewed for this evaluation. The inclusion of these properties in the review effort does not necessarily infer a connection to any identified sediment contamination in the areas. Likewise, the properties included in the discussions do not encompass all potential sources of contamination for the areas.

Table 10-6. Associated properties reviewed for the areas of interest for source control

AREA OF INTEREST (EAA/TTA)	PRIMARY UPLAND PROPERTIES	HISTORICAL UPLAND FACILITIES
Early Action Areas		
Duwamish/Diagonal EAA	Container Care International (T-106 W)	Chiyoda Property (former T-108)
	T-106 Northeast and southeast/Warehouse W-4 and Warehouse W-6	
	Federal Center South	
	Union Pacific Railroad Argo Yard	
	T-108/Former Diagonal Avenue WWTP	
EAA 2 (RM 2.2)	Douglas Management Company (Alaska Marine Lines)	Pacific Metal and Salvage Co.
	Trotsky/Former Northwest Cooperage Property (present-day Industrial Container Services LLC)	Seabell Shipbuilding Co.
	Boyer Towing	Knik Construction Company
		Geo. E Mitzel & Company/Pacific Drum Company
		Consolidated Drum Reconditioning Company
		Palex
Slip 4 EAA	KCIA – northern drainage basin	North Boeing Field Fire Training Center
	Georgetown Steam Plant	Boeing Fire Training Pit
	North Boeing Field	Hydraulic Supply Manufacturing Company
	Crowley Marine Services	Pankrantz Lumber Company
	Emerald Services	Washington Excelsior and Manufacturing Company
	Boeing Plant 2 (northwestern portion)	Washington Machinery and Storage Company
		J.A. Jack & Son Lime Plant
		Northwest Precote asphalt plant
		Ellis Substation
		KCIA Maintenance Shop
Boeing Plant 2/ Jorgensen Forge EAA	Boeing Plant 2	Seattle City Light Substation
	Jorgensen Forge	Isaacson Iron Works
	KCIA (drainage to EAA 4)	Bethlehem Steel
	Portion of E Marginal Way S arterial	
Terminal 117 EAA	South Park Marina	A & B Barrel
	Terminal 117	North Star Trading Company
	Former Basin Oil Company	Evergreen Boat Transport

AREA OF INTEREST (EAA/TTA)	PRIMARY UPLAND PROPERTIES	HISTORICAL UPLAND FACILITIES
	Nearby street rights-of-way	R.P. Boatbuilding
		Dekker Engineering
		Duwamish Manufacturing Co.
		Malarkey Asphalt Co.
		Evergreen Lumber
		Basin Tank and Environmental Services
		Northwest Antifreeze Service
EAA 6 (RM 3.8)	Boeing Isaacson and Thompson	Mineralized Cell Preserving Co.
	KCIA – north-central drainage basin	Bissell Lumber Co.
		Isaacson Steel
Norfolk EAA	Boeing Developmental Center	
	Boeing Military Flight Center	
	KCIA – southern drainage basin	
Tier Two Areas		
Glacier Bay TTA (A)	Alaska Marine Lines	Carlisle Lumber Co.
	Duwamish Shipyard	U.S. Army
	Glacier Northwest	Reichold Chemical, Inc.
	Former MRI Corporation (historical T-115 tenant)	Port of Seattle
		Kaiser Cement Company
		Lone Star Northwest, Inc.
		Ash Grove Cement West, Inc.
Ash Grove Cement TTA (B)	Harbor Real Estate	Pacific Coast Cement Company
	Harbor Island Marina/Terminal 102	Superior Portland Cement Company
	Terminal 104	
	Ash Grove Cement Company	
	Terminal 106 West	
Slip 1/Manson TTA (C)	Federal Center South (U.S. property)	Tilbury Cement Company
	SnoPac Products, Inc.	Ocean Cement and Supplies Ltd.
	Manson Construction (tenant on King County property)	Cadman Gravel Company
	Lehigh Northwest Cement Company (tenant on King County property)	
Glacier NW to Saint-Gobain Glass Company TTA (D)	Saint-Gobain Containers, LLC	Ball-Foster Glass Container Co., LLC
	Longview Fibre Paper and Packaging, Inc.	Ball-InCon
	BPB Gypsum, Inc.	James Hardie
	Glacier Northwest, Inc.	Lone Star Northwest, Inc.
		Ash Grove Cement West, Inc.
Slip 3 to Slip 2 TTA (E)	Duwamish Marine Center (northern parcel)	
	Hale's Construction	
	Duwamish Marine Center (southern parcels)	
	WSDOT parcel	
	Frank's Used Cars	

AREA OF INTEREST (EAA/TTA)	PRIMARY UPLAND PROPERTIES	HISTORICAL UPLAND FACILITIES
Slip 3 TTA (F)	Silver Bay Logging	Northland Services, Inc.
	Rainier Petroleum Corporation	Glacier Marine Company
	Seattle Cold Storage Company	
	Seattle Distribution Center	
	Glacier Marine Services	
	SeaTac Marine Services LLC	
	Big John's Truck Repair	
Slip 4 to Seattle Boiler Works TTA (G)	Seattle Boiler Works	Crowley Marine Services
	Seattle Iron & Metals Corporation	
	Puget Sound Truck Lines, Inc.	
	Seattle City Light Pumping Station	
Slip 6 TTA (I)	Former Kenworth Truck Company/PACCAR, Inc.	Monsanto facility (formerly on Rhône-Poulenc site)
	Former Rhône-Poulenc	
	Boeing Developmental Center	
	KCIA – south-central drainage basin	

EAA – Early Action Area
KCIA – King County International Airport
NW – northwest
RM – river mile
T-106 – Terminal 106
T-108 – Terminal 108
TTA – Tier Two Area
WSDOT – Washington State Department of Transportation

For each area of interest for source control, the chemicals exceeding SMS (in both surface and subsurface sediment within the area) were evaluated and documentation was reviewed to identify the potential occurrence of those same chemicals in upland media (i.e., soil, seeps, groundwater, stormwater, and source tracing samples). The chemicals identified as risk-driver chemicals included 41 SMS chemicals and dioxins and furans (Sections 5 and 6).[102]

A variety of information was reviewed for each area and their associated upland properties and historical facilities as outlined on Table 10-6. This information was summarized to support future source identification efforts to be conducted by Ecology for each area discussed in Section 10.3.2. In addition to the risk driver chemicals identified in surface and subsurface sediment samples, the following types of information were reviewed:

- Anthropogenic markers in subsurface sediment
- Seep and/or porewater sampling information

[102] cPAHs were also identified as a risk driver in the HHRA and they were included in the evaluation of the areas of interest. Although Ecology and EPA also included other chemicals, such as TBT and DDT in their assessments, these chemicals are not further discussed in this section because they are not risk driver chemicals. These chemicals are discussed in Section 4 (nature and extent of contamination).

- Hydrogeologic information (i.e., former river meander locations, nature and depth of fill, surface topography, shoreline/bank conditions, groundwater hydrology)
- Historical dredging information
- Available monitoring and source tracing data relevant to the basins discharging into the area
- Public access issues (i.e., beach access, clamming)
- Investigation and remedial action information for areas and their associated upland properties
- Outfall information (i.e., types, sizes, ownership, permit status)
- Other pertinent information depending on the unique aspects of each area

Upon review of this information, area-specific summaries of the findings were prepared. To support each summary, maps were prepared to provide a visual summary of pertinent information for each area. These maps include surface sediment data for chemicals exceeding SMS in surface sediment, an outline of the in-water boundary[103] and upland properties associated with each area, locations of subsurface, seep, and porewater samples, outfall locations and classifications, and drainage basins, as applicable. In addition, the Duwamish/Diagonal and Norfolk EAAs include maps that summarize recent monitoring data following remedial actions in these areas and locations of source tracing sampling efforts.

It was not the intention of this source document review and evaluation to provide definitive conclusions regarding the origin and responsibility for chemicals identified in surface sediment within the LDW study area. The area-specific information in Section 10.3.2 has been prepared to provide an initial overview of the environmental conditions for each area of interest. Definitive conclusions regarding the origin of contamination would require detailed source evaluations, which were outside the scope of this effort. Further information on source identification and control issues is presented in Ecology's Data Gaps Reports and SCAPs documentation.

10.3.2 Area-specific summaries

Separate sections that summarize the results of the evaluations prepared for each area of interest for source control are included in this section. The availability of potential source information and the complexity of the various areas from a source perspective varied widely. Depending on the availability of area-related information, area-specific summary tables are included in this section to indicate whether chemicals that

[103] These boundaries were identified by Ecology as an organizational tool to identify upland areas of interest for source control.

exceeded the CSL[104] in surface sediment were also identified in upland media during site characterization or source control assessments.

These tables are intended to indicate the potential for a localized source to exist (indicated on the table by an X); however, detection of a chemical in upland media or in source tracing samples does not necessarily infer a connection to surface sediment contamination. Instead, they indicate whether further evaluation may be appropriate. To the extent possible, as provided by the information available, the X on the table indicates that the chemical has been detected in upland media at concentrations above a regulatory threshold, such as MTCA Method A cleanup levels, ambient water quality standards, NPDES permit limits, or SMS criteria (in the case of source tracing samples). In other cases, particularly for organic chemicals, the X may merely indicate that the chemical has been detected in an upland media sample but no concentration information was readily available in the documentation that was reviewed. Conversely, the absence of an X in any cell in these tables does not necessarily imply that that chemical is absent in the upland media or in source tracing samples; in many cases, that chemical may not have been analyzed.

10.3.2.1 Duwamish/Diagonal EAA

The Duwamish/Diagonal EAA was originally identified as a priority cleanup area by the Elliott Bay/Duwamish Restoration Program in the mid-1990s because of sediment contamination associated with the Duwamish EOF and Diagonal Avenue S CSO/SD outfalls. The area was subsequently one of the seven EAAs recommended to EPA and Ecology (Windward 2003d). As part of the ongoing source control efforts for the LDW, Ecology prepared a SCAP for the Duwamish/Diagonal EAA in 2004; a Data Gaps Report is currently not planned for this EAA (Cargill 2007d).

In pre-dredge surface sediment data, CSL exceedances in surface sediment were identified for 23 individual chemicals, including phthalates, PAHs, total PCBs, and four metals (cadmium, chromium, mercury, and silver). Pre-dredge surface sediment sample concentrations were greater than the SQS for 31 individual chemicals, including phthalates, PAHs, total PCBs, and five metals (cadmium, chromium, mercury, silver, and zinc). Historical and current operations within the vicinity of the EAA have included a former WWTP, a processing and disposal site for PCB contaminated dredged material, shipping container refurbishing, railroad refueling, federal government offices and warehouses, and cement manufacturing. Several outfalls (including one historical outfall), associated with a large drainage basin that includes industrial, commercial, and residential properties, discharge to this EAA. The combined service area for the Diagonal Avenue S CSO/SD covers 4,900 ac and the storm drain basin encompasses about 2,600 ac (King County and SPU 2005). The basin includes a 3.6-mi portion of I-5, parts of the Central District, the Duwamish industrial

[104] Information was limited to chemicals with CSL exceedances in these tables to focus the summary discussion.

area, Rainier Valley, and Beacon Hill. Relevant information on the Duwamish/ Diagonal EAA is provided on Map 10-3a.

Various investigations at upland properties associated with this EAA have identified chemicals in soil, groundwater, stormwater effluent, and source tracing samples (Table 10-7). Onsite catch basin, right-of-way catch basin, and in-line sediment samples have been collected from within the Duwamish/Diagonal EAA service area since 2002; these source tracing sampling locations are provided on Maps 10-3b and 10-3c. Between 2002 and 2006, portions of the Diagonal Avenue S CSO/SD main line, the S Dakota Street lateral, and the downstream sections of the 1st Avenue S lateral and the Denver Avenue S lateral were cleaned.

Table 10-7. Chemicals identified in upland media and source tracing samples potentially associated with the Duwamish/Diagonal EAA

Chemicals Exceeding CSL in Surface Sediment	Soil	Ground-water	Storm-water	Source Tracing Samples[a]	Source
BEHP[b, c]			X	X	Ecology (2004a); Schmoyer (2007a)
BBP[b]				X	Ecology (2004b); Schmoyer (2007a)
Mercury[b]		X	X	X	Ecology (2004b); King County (1997); King County and SPU (2005); Schmoyer (2007a)
Total PCBs[b]		X		X	Ecology (2004a); King County and SPU (2005); Pacific Groundwater Group (2006); Schmoyer (2007a)
Benzoic acid[c]					
Benzyl alcohol[c]					

Note: The absence of an X in any cell in this table does not necessarily imply that that chemical is absent in the upland media or in source tracing samples; in many cases, that chemical may not have been analyzed or, if analyzed, could have been present but at concentrations below the RLs.

[a] Source tracing samples for the Duwamish/Diagonal include sediment traps, in-line sediment samples, onsite catch basin samples, and samples collected from catch basins located in public rights-of-way.

[b] Chemicals identified as chemicals of primary concern in sediment prior to sediment remediation in the 1990s.

[c] Detected concentrations of BEHP, benzoic acid, and benzyl alcohol exceeded the CSL in post-dredge monitoring samples in the June 2004, April 2005, March 2006, and April 2007 sampling events on the cap.

BBP – butyl benzyl phthalate

BEHP – bis(2-ethylhexyl) phthalate

CSL – cleanup screening level

EAA – Early Action Area

SPU – Seattle Public Utilities

Remedial actions were implemented in this EAA between November 2003 and March 2004 including dredging of contaminated sediment and the placement of clean cap material over a 5-ac rectangle (Area A) and a 2-ac rectangle (Area B) as presented on Maps 10-3a and 10-3d; a sediment remediation project closure report was prepared in 2005 (King County et al. 2005). A thin-layer placement of sand around the dredged

area in February 2005 was placed in response to elevated chemical concentrations that are thought to have resulted from the dredging activity in 2003 (King County et al. 2005; Ecology 2004a). Long-term sediment monitoring began in the summer of 2004 and will continue until 2014.

SMS exceedances, including exceedances of total PCBs and BEHP, were detected on the cap in 2004 through 2007 (King County et al. 2005). The monitoring locations on cap Areas A and B are summarized in Table 10-8; analytical results are presented on Map 10-3d. Additional information on the perimeter monitoring program is included in Section 4-2; the thin-layer placement is also being monitored.

Table 10-8. Monitoring locations in the vicinity of Duwamish/Diagonal EAA

AREA	LOCATIONS	DATES SAMPLED
Cap A	1A, 2A, 3A, 4A, 5A	June 2004,[a] April 2005, March 2006, and April 2007
Cap B	1B, 2B, 3B	June 2004, April 2005,[b] March 2006, and April 2007

[a] Location 3A was not sampled in June 2004 because the substrate was cobble.

[b] Location 2B was sampled in August 2005 instead of in April 2005. Location 3B was not sampled in 2005.

In Area A, concentrations of total PCBs and BEHP were relatively low in June 2004, the first monitoring event conducted after the area was capped in February/March 2004. At the four locations closest to the outfalls (1A, 2A, 3A, and 4A) (Map 10-3d), total PCB and BEHP concentrations were higher in 2005, 2006, and 2007 than in 2004. However, with the exception of BEHP at location 4A, these same four locations all had lower concentrations in 2007 than in 2006.

Surface sediment samples from four of the locations (1A, 2A, 3A, and 4A)[105] had SQS exceedances in samples collected in 2004 through 2007, as follows:

- **Location 1A –** In both 2005 and 2006, total PCB exceeded the SQS, and BEHP exceeded the CSL in 2005, 2006, and 2007.
- **Location 2A –** BEHP exceeded the SQS in 2004, the CSL in 2005, and the SQS in 2006 and 2007.
- **Location 3A –** In 2005, BEHP exceeded the SQS.
- **Location 4A –** In 2006, total PCBs exceeded the SQS. BEHP exceeded the CSL in 2006 and the SQS in 2007.

Total PCB and BEHP concentrations were much lower at location 5A; all sample results for location 5A were below SQS criteria.

In Area B, total PCB concentrations were relatively similar in 2004, 2005, 2006, and 2007 (Map 10-3d), and there was one SQS exceedance in 2005 at location 1B. At all three locations on Cap B, concentrations of BEHP were lowest in 2004 and highest in

[105] Concentrations of total PCBs and BEHP were low in all three years at location 5A, the farthest downstream location.

2006 or 2007. No BEHP SQS exceedances were identified in any samples from Area B. Thus, no SMS exceedances were detected on Cap B for total PCBs or BEHP since 2005. On Cap A, total PCBs did not exceed SMS in 2007 although BEHP still exceeded either the SQS or CSL at the three locations closest to the shoreline.

In addition to exceedances discussed for total PCBs and BEHP, five other chemicals were detected at concentrations exceeding SMS criteria. Benzoic acid and benzyl alcohol each exceeded the CSL at one location on Cap A. Fluoranthene and dimethyl phthalate each exceeded the SQS at one location on Cap A. BBP exceeded the SQS at three locations on Cap A.

10.3.2.2 EAA 2 (RM 2.2)

EAA 2 (RM 2.2) was one of the seven EAAs recommended to EPA and Ecology (Windward 2003d). This EAA is considered to be unsponsored because no party has accepted responsibility for further study and cleanup at this site. A Data Gaps Report was completed for this EAA in February 2007 (SAIC 2007d) and was used as the basis for development of a SCAP, which was finalized in June 2007 (Ecology 2007i; SAIC 2007d). Ecology also completed an additional site characterization activities data report for this EAA in July 2007 (SAIC 2007c).

CSL exceedances in surface sediment at this EAA included total PCBs, BEHP, mercury, and lead. SQS exceedances in surface sediment included zinc and BBP. In addition, the dioxin and furan TEQ was elevated in surface sediment in this area. Historical and current industrial operations at upland sites near this EAA have included cargo handling and storage, marine vessel salvage and construction, barrel reconditioning, warehousing, and vehicle repair and maintenance (Ecology 2007i). Relevant information on EAA 2 (RM 2.2) is provided on Map 10-4.

Various investigations at upland properties in this EAA have identified chemicals in soil, groundwater, seeps, and source tracing samples (including in-line sediment samples and drainage ditch grab samples) (Table 10-9). Sampling completed by Ecology in 2007 indicated that PCB contamination in this EAA is highest along the southern shoreline of the inlet.

Table 10-9. Chemicals identified in upland media and source tracing samples potentially associated with EAA 2 (RM 2.2)

Chemicals Exceeding CSL in Surface Sediment	Soil	Ground-water	Seeps[b]	Source Tracing Samples[c]	Source
Total PCBs	X	X	X	X	Ecology (2007i); SAIC (2007c); Windward (2004b)
Mercury	X		X		SAIC (2007c); Windward (2004b)
BEHP				X	SAIC (2007c; 2007d)
Lead	X	X	X		Ecology (2007i); SAIC (2007c); Windward (2004)
Dioxins and furans[a]					

Note: The absence of an X in any cell in this table does not necessarily imply that that chemical is absent in the upland media or in source tracing samples; in many cases, that chemical may not have been analyzed or, if analyzed, could have been present but at concentrations below the RLs.

[a] A CSL does not exist for dioxins and furans. They were included in this table because they are a risk driver chemical with a highly elevated concentration in surface sediment in this area. Dioxins and furans were not analyzed in the upland media samples in this table.

[b] Available documentation did not indicate whether all seeps samples were filtered or unfiltered; entrained solids could affect sampling results if samples were unfiltered. PCBs, mercury, and lead were detected in both filtered and unfiltered seep samples collected as part of the RI (Windward 2004).

[c] In-line sediment and whole water samples were collected from the drainage ditch along 2nd Avenue S and an in-line sediment sample was collected approximately one foot from the mouth of a SD pipe discharging from 2nd Avenue S.

BEHP – bis(2-ethylhexyl) phthalate

CSL – cleanup screening level

EAA – Early Action Area

PCB – polychlorinated biphenyl

RM – river mile

SAIC – Science Applications International Corporation

TEQ – toxic equivalent

10.3.2.3 Slip 4 EAA

The Slip 4 EAA was one of the seven EAAs recommended to EPA and Ecology (Windward 2003d), and is a candidate for a non-time-critical removal action (NTCRA). Several documents have been prepared by Ecology, the SCWG, the City, and the County in preparation of the NTCRA and to assess source control in the Slip 4 EAA drainage basin. These documents include summary of existing information and analysis of data gaps reports (SAIC 2007g; SEA 2004), a draft engineering evaluation/cost analysis (EE/CA) (Integral 2006), a 100% design analysis report (Integral 2007), a SCAP (Ecology 2006a), several smaller property-specific reviews (SAIC 2006a, 2007h, 2006b, c), and a technical memorandum on the status of source control in the Slip 4 EAA (SAIC 2007a).

CSL exceedances in the surface sediment at this EAA included total PCBs, BEHP, mercury, and four individual PAHs. SQS exceedances included total PCBs, BEHP, BBP, mercury, total HPAHs, and nine individual PAHs. Total PCB concentrations in

Slip 4 surface sediment have consistently been highest at the head of the slip, but have generally decreased with time based on available data (see Section 4.2.1.1).

Historical and current commercial and industrial operations in the vicinity of the Slip 4 EAA include airplane manufacturing, airport operations, steam power generation, cargo shipping, manufacturing of various metal products, lumber milling and processing, and lime and asphalt production. Five public outfalls discharge at the head of the Slip 4 EAA, including the I-5 SD, the Georgetown flume SD, the North Boeing Field SD, the KCIA SD No. 3/Pump Station 44 EOF, and the E Marginal Way S EOF. In total, approximately 470 ac of land drain to Slip 4. Relevant information on the Slip 4 EAA is provided on Map 10-5a; the Slip 4 drainage basin is provided on Map 10-5b.

Various investigations at upland properties included in this EAA have identified chemicals in soil, groundwater, seeps, stormwater, and source tracing samples (Table 10-10).

Table 10-10. Chemicals identified in upland media and source tracing efforts potentially associated with Slip 4 EAA

Chemicals Exceeding CSL in Surface Sediment	Soil	Ground -water	Seep[b]	Storm -water	Source Tracing Samples[a]	Source
Total PCBs	X	X	X		X	Ecology (2006a); King County and SPU (2005); SAIC (2007a; 2007g); Schmoyer (2007a); (Integral 2007)
Mercury	X		X		X	Ecology (2006a); King County and SPU (2005); SAIC (2007g); Schmoyer (2007a); Windward (2004)
BEHP	X	X			X	Ecology (2006a); King County and SPU (2005); SAIC (2007g); Schmoyer (2007a)
Benzo(a)pyrene	X	X			X	Ecology (2006a); King County and SPU (2005); SAIC (2006c; 2007g); Schmoyer (2007a)
Benzo(g,h,i)perylene[c]	X				X	Ecology (2006a); King County and SPU (2005); SAIC (2007g); Schmoyer (2007a)
Dibenzo(a,h)anthracene[c]	X				X	Ecology (2006a); King County and SPU (2005); SAIC (2007g); Schmoyer (2007a)
Indeno(1,2,3-cd)pyrene[c]	X				X	Ecology (2006a); King County and SPU (2005); SAIC (2007g); Schmoyer (2007a)

Note: The absence of an X in any cell in this table does not necessarily imply that that chemical is absent in the upland media or in source tracing samples; in many cases, that chemical may not have been analyzed or, if analyzed, could have been present but at concentrations below the RLs.

[a] Source tracing efforts included collection of onsite catch basin, in-line sediment trap, in-line sediment grab samples.

[b] Six seep samples were collected from five locations during the pre-design investigation for the Slip 4 NTCRA. PCB Aroclors were detected in two of the samples. The resource documentation did not specify whether the

seep samples were filtered or unfiltered; entrained solids could potentially affect sampling results if samples were unfiltered (Integral 2007). Mercury was detected in the filtered sample collected as part of the RI (Windward 2004).

[c] PAHs have been detected in soil at several properties within the Slip 4 EAA; specific information regarding which individual PAHs detected was often not available. Therefore, it is not known if these three specific PAHs were detected.

BEHP – bis(2-ethylhexyl) phthalate

CSL – cleanup screening level

EAA – Early Action Area

PAH – polycyclic aromatic hydrocarbon

PCB – polychlorinated biphenyl

SAIC – Science Applications International Corporation

SPU – Seattle Public Utilities

10.3.2.4 Boeing Plant 2/Jorgensen Forge EAA

The Boeing Plant 2/Jorgensen Forge EAA was one of the seven EAAs recommended to EPA and Ecology (Windward 2003d). Since the early 1990s, various soil, groundwater, and sediment investigations have been conducted within the Boeing Plant 2/Jorgensen Forge EAA under both RCRA and MTCA. As part of the source control efforts for this EAA, a Data Gaps Report and a SCAP were completed in August 2007 (Ecology 2007j; SAIC 2007e); annual updates to the SCAP are expected.

CSL exceedances in surface sediment in this EAA included total PCBs, BEHP, BBP, cadmium, chromium, copper, lead, mercury, phenol, silver, and zinc. SQS exceedances were reported for individual PAHs, total HPAHs, and total LPAHs. Historical and current commercial and industrial operations in the vicinity of this EAA have included airplane manufacturing, cargo handling and storage, and metals fabrication (SAIC 2007e). Raw materials including metals (such as chrome, zinc, copper, cadmium, and silver), lead-based paints, solvents, petroleum products, and acids have also been used at nearby properties. Relevant information on the Boeing Plant 2/Jorgensen Forge EAA is provided on Map 10-6a through 10-6c.

Various investigations at upland properties in this EAA have identified chemicals in upland soil, groundwater, seeps, and source tracing samples (Table 10-11). To date, several potential sources identified during upland investigations have been controlled or removed. Stormwater lines were removed and/or cleaned, catch basins connected to the storm drain conveyance system are routinely sampled and cleaned as needed, and soils in areas with elevated chemical concentrations have been removed.

Table 10-11. Chemicals identified in upland media and source tracing samples potentially associated with the Boeing Plant 2/Jorgensen Forge EAA

Chemicals Exceeding CSL in Surface Sediment	Soil	Ground-water	Seeps	Storm-water	Source Tracing Samples[a]	Source
BBP	X				X	SAIC (2007e)
BEHP	X				X	SAIC (2007e)

Chemicals Exceeding CSL in Surface Sediment	Soil	Ground-water	Seeps	Storm-water	Source Tracing Samples[a]	Source
Cadmium	X	X				SAIC (2007e)
Chromium					X	SAIC (2007e)
Copper	X	X	X[b]		X	SAIC (2007e)
Lead	X	X			X	SAIC (2007e)
Mercury	X				X	SAIC (2007e)
Total PCBs	X		X[b]		X	SAIC (2007e)
Phenol						
Silver		X				SAIC (2007e)
Zinc	X	X	X[b]		X	SAIC (2007e)

Note: The absence of an X in any cell in this table does not necessarily imply that that chemical is absent in the upland media or in source tracing samples; in many cases, that chemical may not have been analyzed or, if analyzed, could have been present but at concentrations below the RLs.

[a] Catch basin and in-line storm drain samples have been collected. Catch basins and storm drain lines were cleaned at the Boeing Plant 2 and Jorgensen Forge properties subsequent to sampling.

[b] Metals samples were filtered; PCB samples not specific but likely unfiltered; entrained solids can affect sampling results if unfiltered.

BBP – butyl benzyl phthalate

BEHP –bis(2-ethylhexyl) phthalate

CSL – cleanup screening level

EAA –Early Action Area

PCB – polychlorinated biphenyl

SAIC – Science Applications International Corporation

10.3.2.5 T-117 EAA

The T-117 EAA was one of the seven EAAs recommended to EPA and Ecology (Windward 2003d). As part of the initial investigations for the T-117 EAA, a Data Gaps Report (Windward et al. 2003), and an EE/CA (Windward et al. 2004d) have been prepared for the NTCRA. In 2005, Ecology prepared a SCAP (Ecology 2005a); annual updates to the SCAP are expected.

CSL exceedances in surface sediment in this EAA included primarily total PCBs; however, individual PAHs and phenol also exceeded the CSL. SQS exceedances were noted for several individual PAHs and benzyl alcohol. Total PCBs concentrations exceeding the CSL were detected predominantly along the shoreline, with concentrations decreasing with distance from the shoreline. The in-water sediment NTCRA boundary defined in the sediment/bank EE/CA encompasses the sampling locations with the CSL and higher SQS exceedances. Historical and current commercial and industrial operations within this EAA have included asphalt manufacturing, boat marina operations (including boat building, maintenance, storage), barrel reconditioning, lumber storage, cargo handling and storage, metals fabrication, food processing, and oil reclamation. Relevant information on the T-117 EAA is provided on Map 10-7.

Various investigations at upland properties in this EAA have identified chemicals in soil, groundwater, seep, and source tracing samples (including catch basin solids and street dirt samples) (Table 10-12). Several potential sources of PCBs and PAHs identified during upland investigations have been controlled or removed during completion of two separate time-critical removal actions (TCRAs) and PCBs have been cleaned up in the right-of-way areas in the vicinity of T-117. Soils on the upland portion of T-117 with elevated concentrations of PCBs have been removed, PCB-contaminated areas in the right-of-way areas have been paved, and a new stormwater collection system has been installed, which conveys most runoff from the roadways adjacent to T-117 to the combined sewer system. Industrial activities in the adjacent upland area have ceased (except for the marina) and a new sediment, upland soils, and adjacent street rights of way NTCRA is currently being implemented, starting with an updated EE/CA.

Table 10-12. Chemicals identified in upland media and source tracing samples potentially associated with the T-117 EAA

Chemicals Exceeding CSL in Surface Sediment	Soil	Ground-water	Seeps	Source Tracing Samples[a]	Source
2-methylnaphthalene	X			X	Windward et al. (2004d)
Acenaphthene	X			X	Windward et al. (2004d)
Benzo(a)anthracene	X			X	Windward et al. (2004d)
Benzo(a)pyrene	X			X	Windward et al. (2004d)
Dibenzo(a,h)anthracene	X			X	Windward et al. (2004d)
Fluoranthene	X			X	Windward et al. (2004d)
Fluorene	X			X	Windward et al. (2004d)
Indeno(1,2,3-cd)pyrene	X			X	Windward et al. (2004d)
Phenanthrene	X			X	Windward et al. (2004d)
Total LPAHs	X			X	Windward et al. (2004d)
Total PCBs	X	X		X	Windward et al. (2004d)
Phenol					

Note: The absence of an X in any cell in this table does not necessarily imply that that chemical is absent in the upland media or in source tracing samples; in many cases, that chemical may not have been analyzed or, if analyzed, could have been present but at concentrations below the RLs.

[a] Catch basin and street dirt sampling have been collected in this EAA; Catch basins were cleaned after sampling.

CSL – cleanup screening level

EAA – Early Action Area

PAH – polycyclic aromatic hydrocarbons

PCB – polychlorinated biphenyl

10.3.2.6 EAA 6 (RM 3.8)

EAA 6 (RM 3.8) was one of the seven EAAs recommended to EPA and Ecology (Windward 2003d). This EAA is considered to be unsponsored because no party has accepted responsibility for further study and cleanup at this site. To date, no source control efforts have been conducted by Ecology in this EAA. Completion of a Data Gaps Report and a SCAP for the EAA 6 is scheduled for 2008 (SAIC 2007b).

CSL exceedances in surface sediment in this EAA included five individual PAHs, benzoic acid, and BEHP. SQS exceedances in surface sediment included six individual PAHs, total HPAH, total LPAH, arsenic, total PCBs, BBP, and dibenzofuran. Historical and current operations in the vicinity of the EAA 6 have included the storage and treatment of wood, and steel smelting, forging, and fabrication. A City of Seattle/King County EOF/SD that drains the north central portion of the KCIA discharges into this EAA. Relevant information on EAA 6 is provided on Map 10-8.

Data collected in 2005 for the RI indicated that surface sediment contamination in this EAA is lower than in samples collected in 1997 and 1998. No CSL exceedances for PAHs and BEHP were detected in 2005 samples suggesting that conditions in this area are improving.

Various investigations at upland properties associated with this EAA have included soil, groundwater, seep, and source tracing sampling (catch basin sample) (Table 10-13).

Table 10-13. Chemicals identified in upland media and source tracing samples potentially associated with EAA 6 (RM 3.8)

Chemicals Exceeding CSL in Surface Sediment	Soil	Ground-water	Seeps	Source Tracing Samples[a]	Source
Benzo(g,h,i)perylene				X[b]	King County and SPU (2005)
Indeno(1,2,3-cd)pyrene				X[b]	King County and SPU (2005)
Dibenzo(a,h)anthracene				X[b]	King County and SPU (2005)
Benzo(a)pyrene				X[b]	King County and SPU (2005)
Total benzofluoranthenes				X[b]	King County and SPU (2005)
Benzoic acid					
BEHP				X	King County and SPU (2005)

Note: The absence of an X in any cell in this table does not necessarily imply that that chemical is absent in the upland media or in source tracing samples; in many cases, that chemical may not have been analyzed or, if analyzed, could have been present but at concentrations below the RLs.

[a] Onsite catch basin samples were collected from KCIA.

[b] TPH-heavy oil was detected in source tracing samples (specific PAHs were not identified).

BEHP – bis(2-ethylhexyl) phthalate

CSL – cleanup screening level

EAA – Early Action Area

SPU – Seattle Public Utilities

10.3.2.7 Norfolk EAA

The Norfolk EAA was one of the seven EAAs recommended to EPA and Ecology (Windward 2003d). A sediment removal action under the EB/DRRP, which included dredging followed by backfilling with clean fill material, was completed in front of the Norfolk CSO/SD outfall in 1999. A sediment removal action (excavation followed by capping) was also conducted in this EAA in 2003 near the Boeing Developmental Center south SD. As part of the ongoing source control efforts for the LDW, Ecology is preparing a Data Gaps Report and a SCAP for the Norfolk EAA, which are scheduled for completion in 2007 (SAIC 2007b).

CSL exceedances in surface sediment in this EAA[106] included 1,4-dichlorobenzene, total PCBs, and BEHP. SQS exceedances included BBP and fluoranthene. Historical and current operations in the vicinity of this EAA have included aircraft and aerospace research, development, and manufacturing. The Norfolk CSO/SD drains approximately 769 ac of industrial and residential land and road rights-of-way; discharge from a total area of approximately 4,900 ac is possible during a major storm event. Relevant information on the Norfolk EAA is provided on Map 10-9a.

Various investigations at upland properties in this EAA have identified chemicals in soil, groundwater, stormwater, and source tracing samples (Table 10-14). Source tracing sampling has been conducted within the Norfolk CSO/SD drainage basin since 2003; these sampling locations are provided on Map 10-9b.

Table 10-14. Chemicals identified in upland media and source tracing samples potentially associated with the Norfolk EAA

Chemicals Exceeding CSL in Surface Sediment	Soil	Ground-water	Storm-water	Source Tracing Samples[a]	Source
1,4-dichlorobenzene	X[b]	X[b]		X[b]	EDR (2006a); King County and SPU (2005); Schmoyer (2007a)
BEHP				X	King County and SPU (2005); Schmoyer (2007a)
Total PCBs	X			X	Ecology (2007h); EDR (2006b)

Note: The absence of an X in any cell in this table does not necessarily imply that that chemical is absent in the upland media or in source tracing samples; in many cases, that chemical may not have been analyzed or, if analyzed, could have been present but at concentrations below the RLs.

[a] Source tracing samples included manhole sediment, sediment/sludge and water from oil/water separators, catch basins, and in-line sediment.

[b] Unspecified halogenated organic compounds have been identified in soil and groundwater.

BEHP – bis(2-ethylhexyl) phthalate

CSL – cleanup screening level

[106] Surface sediment data for this area included post-remediation samples collected within the 1999 Norfolk CSO/SD removal area, pre-removal samples collected within the 2003 Boeing removal area, samples collected prior to the 1999 action from locations outside of the dredged area, and samples collected after the 2003 action from locations outside of the capped area.

EAA – Early Action Area
EDR – Environmental Data Resources, Inc.
PCB – polychlorinated biphenyl
SPU – Seattle Public Utilities
TPH – total petroleum hydrocarbon

The sediment removal action associated with the Norfolk CSO/SD was conducted in February and March 1999. After remediation, four locations (NFK501, NFK502, NFK503, and NFK504) on the clean cap were monitored on an annual basis by King County until 2004 (Map 10-9c). These locations were also sampled in October 2006 as part of the RI (Windward 2007c). Sediment samples collected by King County at the four monitoring locations were collected from both the 0-to-2-cm and 0-to-10-cm intervals. Sediment sampling results for total PCBs, cPAHs, and BEHP in the 0-to-10-cm interval are provided on Figure 10-1.

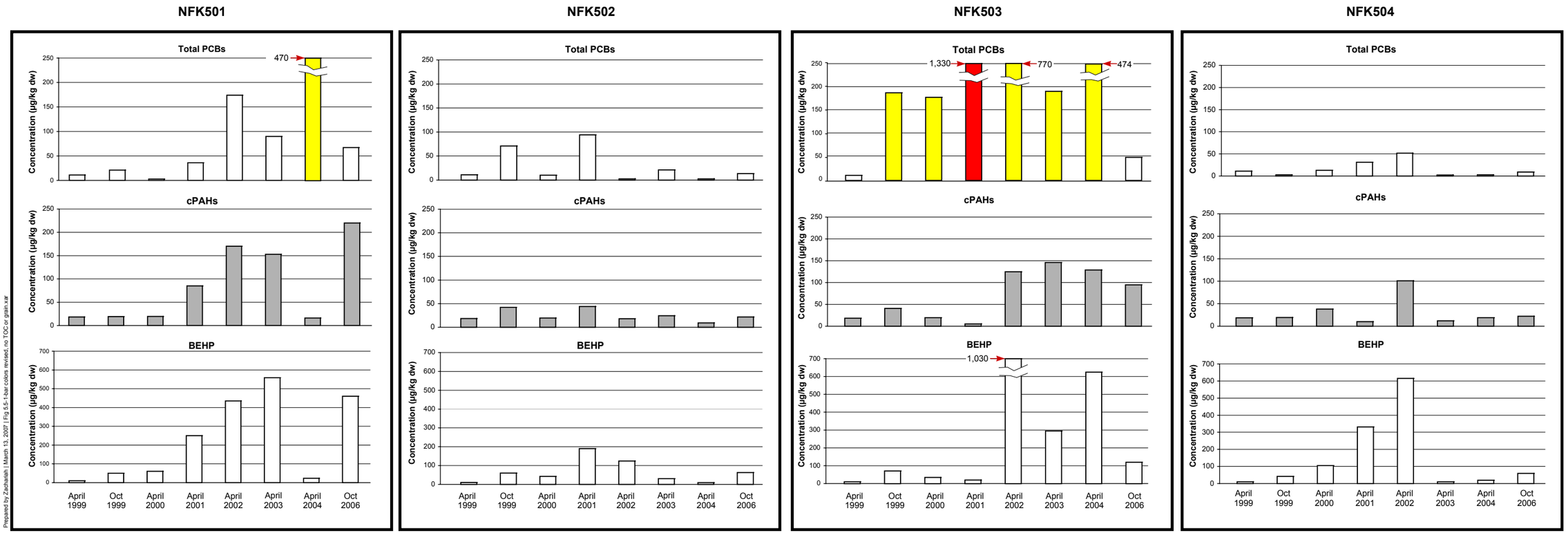
The highest total PCB concentrations in surface sediment were detected at NFK503 (Map 10-9c). Total PCBs exceeded the SQS on five of the eight sampling dates at this location (Figure 10-1). The CSL was also exceeded at this location in 2001. At NFK501 (the location closest to the Norfolk CSO/SD), total PCBs exceeded the SQS in 2004. There were no total PCB SQS exceedances at locations NFK502 or NFK504. The highest BEHP concentrations were also detected at NFK503, although there were no BEHP SQS exceedances at any locations (Figure 10-1).

Other chemicals with detected SQS exceedances in 0-to-10-cm samples from at least one of the four monitoring locations were as follows: benzo(g,h,i)perylene (in 2000), butyl benzyl phthalate (in 2001), hexachlorobenzene (in 1999 and 2000), 2-methylnaphthalene (in 1999), fluoranthene (in 2004), and phenanthrene (in 2004).

A cleanup action was also conducted at the Boeing Developmental Center in 2003 to remove sediment containing PCBs in the vicinity of the south SD outfall (2093) (Map 10-9c). Three locations on the cap were monitored for PCBs in surface sediment in 2004 and 2005, including S01, the location that had the highest PCB concentrations (46,000 µg/kg dw) prior to the action. PCBs were detected only at location S01 in 2004 and 2005 but at concentrations lower than those detected before the remedial action. In 2005, total PCBs exceeded the SQS at location S01 but were non detect in the field duplicate sample.

10.3.2.8 Glacier Bay TTA

The Glacier Bay TTA was identified as by Ecology, in consultation with EPA, because of surface sediment chemistry and potential source concerns (Cargill et al. 2007). A Data Gaps Report was prepared for this TTA in April 2007; the SCAP is scheduled for completion in late 2007 (SAIC 2007b).



Legend

- No criteria
- > CSL
- > SQS, ≤ CSL
- ≤ SQS

Figure 10-1. Concentrations of total PCBs, BEHP, and cPAHs in surface sediment at four monitoring locations on the Norfolk cap

DRAFT

Wind Ward environmental LLC

Note: SQS and CSL exceedances are based on OC-normalized concen-trations if TOC is ≥ 0.5% and ≤ 4.0%. For samples with TOC < 0.5% or > 4.0%, the dry weight concentration is compared to AETs to determine the exceedance status. No criteria are available for carcinogenic PAHs.

TEQs were calculated with mammalian PEFs for seven individual PAH compounds (California EPA 1994), using one-half the RL for undetected compounds.

CSL exceedances in surface sediment in this area included arsenic, copper, lead, mercury, zinc, and total PCBs. SQS exceedances were reported for BEHP, BBP, and six individual PAHs. In addition, dioxins and furans were elevated in surface sediment samples.[107] Core logs collected in this TTA identified sand blast grit, asphalt-like conglomerate materials, and foliated chips (possibly paint) in the 5.5-to-7.5-ft sampling interval (Windward and RETEC 2007).

Historical and current commercial and industrial operations within this area have included cargo handling and storage, marine vessel/shipyard repair and maintenance, concrete manufacturing, lumber milling, charcoal production, glue and resin manufacturing (including pentachlorophenol), and tin reclamation. Several outfalls discharge into this TTA, including a City main line that collects stormwater from areas along W Marginal Way SW.

Various investigations at upland properties in the Glacier Bay TTA have identified chemicals in soil, groundwater, seeps, stormwater effluent, and source tracing samples (including catch basin samples) (Table 10-15). PCBs and dioxins and furans detected in surface sediment within this TTA were located predominantly along the northern shoreline of the embayment between RM 1.4 and RM 1.5; metals contamination was identified in the northern portion of the TTA. Maintenance dredging was completed adjacent to the floating docks near the navigation channel in 1992 and 2005. The 2005 dredge event did not remove the surface sediment with the identified CSL exceedances identified in that area. Relevant information on the Glacier Bay TTA is provided on Map 10-10.

Table 10-15. Chemicals identified in upland media and source tracing samples potentially associated with the Glacier Bay TTA

Chemicals Exceeding CSL in Surface Sediment	Soil	Ground -water	Seeps[b]	Storm-water[c]	Source Tracing Samples[d]	Source
Arsenic	X		X			Anchor (2006, as cited in SAIC 2007f); SAIC (2007f); Windward (2004b)
Copper			X	X	X	Anchor (2006, as cited in SAIC 2007f); SAIC (2007f); Windward (2004b)
Lead	X	X	X			SAIC (2007f); Windward (2004)
Mercury			X		X	Anchor (2006, as cited in SAIC 2007f); SAIC (2007f); Windward (2004)
Zinc			X	X	X	Anchor (2006, as cited in SAIC 2007f); SAIC (2007f); Windward (2004)

[107] TBT was also identified in sediment in this TTA. TBT and dioxin and furans detected in the Glacier Bay TTA were each the highest reported for the LDW study area.

Chemicals Exceeding CSL in Surface Sediment	Soil	Ground -water	Seeps[b]	Storm-water[c]	Source Tracing Samples[d]	Source
Total PCBs	X					SAIC (2007f)
Dioxins and furans[a]						

Note: The absence of an X in any cell in this table does not necessarily imply that that chemical is absent in the upland media or in source tracing samples; in many cases, that chemical may not have been analyzed or, if analyzed, could have been present but at concentrations below the RLs.

[a] A CSL does not exist for dioxins and furans. They were included in this table because they are a risk driver chemical with highly elevated concentrations in surface sediment in this area.

[b] Arsenic, lead, mercury, and zinc were detected in both filtered and unfiltered seep samples collected as part of the RI (Windward 2004); copper was not detected in either filtered or unfiltered seep samples collected as part of the RI (Windward 2004).

[c] Stormwater effluent samples were collected at the Alaskan Marine Line and Duwamish Shipyard properties in support of NPDES compliance monitoring.

[d] Onsite catch basin samples were collected.

CSL – cleanup screening level

PCB – polychlorinated biphenyl

TEQ – toxic equivalents

TTA – tentative two area

10.3.2.9 Ash Grove Cement TTA

The Ash Grove Cement TTA was identified by Ecology, in consultation with EPA, because of surface sediment chemistry and potential source concerns (Cargill et al. 2007). A SCAP and Data Gaps Report for this TTA are scheduled for completion in 2010 (SAIC 2007b).

CSL exceedances in surface sediment in this TTA included total PCBs, arsenic, lead, and BEHP. SQS exceedances included total PCBs, arsenic, and zinc. A subsurface core sample collected in this TTA had sub-rounded gravel, silt and rock flour, balls of hard silt attributed to anthropogenic sources, and wood fragments (Windward and RETEC 2007). Current and historical commercial and industrial activities in the vicinity of this TTA have included cement and brick manufacturing, marina operations, cargo storage and shipping, vehicle storage, and office uses. Relevant information for the TTA is presented on Map 10-11.

Little information on investigations at upland properties was available for this TTA. Readily available information reviewed during the assessment of the area was insufficient to create a tabular summary of chemical occurrences in upland media (i.e., chemicals with CSL exceedances were not identified in upland media or source investigations).

10.3.2.10 Slip 1/Manson TTA

The Slip 1/Manson TTA was identified by Ecology, in consultation with EPA, because of surface sediment chemistry and potential source concerns (Cargill et al. 2007). A Data Gaps Report and a SCAP for this TTA are scheduled for completion in 2010 (SAIC 2007b).

CSL exceedances in surface sediment in this TTA included arsenic, copper, mercury, zinc, dibenzofuran, five individual PAHs, and total LPAHs. SQS exceedances in surface sediment in this TTA include eight individual PAHs, total HPAH, and total PCBs. Historical and current commercial and industrial operations in the vicinity of the Slip 1/Manson TTA have included warehousing, auto manufacturing, waterway construction, boat mooring, fish processing, and cement manufacturing.

Two separate dredge events were completed in Slip 1 in an attempt to clean up a PCB spill that occurred in 1974 at the US General Services Administration (GSA) dock when a transformer was dropped and cracked while being loaded onto a barge (King County 2002b; EPA 1975 as cited in Windward 2003a). Approximately 980 L (260 gal.) of near-pure PCB (Aroclor 1242) were spilled into the slip (King County 2002b; EPA 1975 as cited in Windward 2003a). EPA conducted the first dredge event in 1974 using hand dredges. The second dredging event was conducted by EPA and the USACE in 1976 (King County 2002b; EPA 1975 as cited in Windward 2003a). Additional information on this spill is provided in Section 9.3.2.3. Relevant information for this TTA is present on Map 10-12. An area at the head of the slip had the highest metals contamination.

Various investigations at upland properties in this TTA have identified chemicals in soil, groundwater, seep, and porewater samples (Table 10-16).

Table 10-16. Chemicals identified in upland media potentially associated with the Slip 1/Manson TTA

Chemicals Exceeding CSL in Surface Sediment	Soil[a]	Groundwater[b]	Seeps	Source
Acenaphthene	X	X		Ecology (2004a; 2007k)
Arsenic			X[c]	Windward (2004b); Weston (1999a)
Copper			X[d]	Windward (2004b)
Dibenzofuran	X	X		Ecology (2004a; 2007k)
Fluorene	X	X		Ecology (2004a; 2007k)
Mercury			X[c]	Windward (2004b)
2-Methylnaphthalene	X	X		Ecology (2004a; 2007k)
Naphthalene	X	X		Ecology (2004a; 2007k)
Phenanthrene	X	X		Ecology (2004a; 2007k)
Total LPAHs	X	X		Ecology (2004a; 2007k)
Zinc			X[c]	Windward (2004b)

Note: The absence of an X in any cell in this table does not necessarily imply that that chemical is absent in the upland media or in source tracing samples; in many cases, that chemical may not have been analyzed or, if analyzed, could have been present but at concentrations below the RLs.

[a] Petroleum was detected in soil on an upland property.

[b] A gasoline plume was identified in groundwater on an upland property. Diesel-, gasoline-, and heavy-oil-range hydrocarbons were detected in soil and groundwater; however none of these individual PAHs were specifically identified.

[c] Detected in both filtered and unfiltered seep samples.

[d] Detected in unfiltered seep sample only.

CSL – cleanup screening level

LPAH – low-molecular-weight polycyclic aromatic hydrocarbon

TTA – tentative two area

10.3.2.11 Glacier NW to Saint-Gobain Glass Company TTA

The Glacier NW to Saint-Gobain Glass Company TTA was identified by Ecology, in consultation with EPA, because of surface sediment chemistry and potential source concerns (Cargill et al. 2007). No Ecology-led source control efforts have been conducted at the Glacier NW to Saint-Gobain Glass Company TTA. A Data Gaps Report and a SCAP for this TTA are scheduled for completion in 2010 (SAIC 2007b).

CSL exceedances in surface sediment in this TTA included zinc and total PCBs. SQS exceedances in surface sediment in this TTA included benzyl alcohol, chrysene, and phenol. Commercial and industrial operations in the vicinity of this TTA have included glass manufacturing and storage, cardboard manufacturing, printing, gypsum wall board manufacturing, and cement manufacturing. Relevant information for this TTA is provided on Map 10-13. Maintenance dredging has occurred within this TTA between RM 1.55 W and RM 1.75 W during two separate events in 1995 and 1999.

Readily available information reviewed during the assessment of this TTA was insufficient to create a tabular summary of chemical occurrences in upland media (i.e., chemicals with CSL exceedances in surface sediment were not identified in upland media characterization and source investigation documentation available for review).

10.3.2.12 Slip 3 to Slip 2 TTA

The Slip 3 to Slip 2 TTA was identified by Ecology, in consultation with EPA, because of surface sediment chemistry and potential source concerns (Cargill et al. 2007). A Data Gaps Report and a SCAP for this TTA are scheduled for completion in 2010 (SAIC 2007b).

No CSL exceedances were identified for any chemicals in surface sediments within the Slip 3 to Slip 2 TTA. Total PCBs exceeded the SQS in four surface sediment samples. Current and historical operations in this TTA have included construction, auto wrecking, shipyard and marina activities, and commercial warehousing. The E Michigan CSO (ID 2502) discharges to the Slip 3 to Slip 2 TTA and is discussed further in Section 9.4.4.2. In addition, a WSDOT SD that drains the 1st Avenue S Bridge (ID 2503) also discharges to this TTA. Relevant information for this TTA is provided on Map 10-14.

Because no CSL exceedances were identified in surface sediment within this TTA, a table summarizing the occurrence of chemicals in upland media was not prepared. Information regarding potential sources in this TTA was limited.

10.3.2.13 Slip 3 TTA

The Slip 3 TTA was identified by Ecology, in consultation with EPA, because of surface sediment chemistry and potential source concerns (Cargill et al. 2007). A Data Gaps Report and a SCAP for this TTA are scheduled for completion in 2010 (SAIC 2007b).

Benzyl alcohol exceeded the CSL in surface sediment within this TTA. The only SQS exceedance in surface sediment within the Slip 3 TTA was a single exceedance for arsenic. These chemistry exceedances were limited to the head of the slip. Current and historical operations in the TTA have included cargo transport, barge berthing, general warehousing, cold storage, shipbuilding, auto repair, and boat storage. Relevant information for this TTA is provided on Map 10-15.

Little information on investigations at upland properties was available for this TTA. Readily available information reviewed during the assessment of the area was insufficient to create a tabular summary of chemical occurrences in upland media (i.e., chemicals with CSL exceedances were not identified in upland media or source investigations).

10.3.2.14 Slip 4 to Seattle Boiler Works TTA

The Slip 4 to Seattle Boiler Works TTA was identified by Ecology, in consultation with EPA, because of surface sediment chemistry and potential source concerns (Cargill et al. 2007). No Ecology-led source identification or control activities have been conducted, and neither a Data Gaps Report nor a SCAP has been prepared (Cargill et al. 2007). Completion of these two reports for this TTA is scheduled for 2008 (SAIC 2007b).

CSL exceedances in surface sediment in this TTA included total PCBs and mercury. Fluoranthene exceeded the SQS in surface sediment within this TTA. Commercial and industrial operations in the vicinity of this TTA include the storage of boilers, heaters, and vehicles, as well as metal recycling. Historical operations in the area have included metalworking, foundry work, painting activities, steel can manufacturing equipment construction, fuel storage, warehousing, and terminal operations (Terra Vac and Floyd & Snider 2000). Relevant information for this TTA is provided on Map 10-16.

Readily available information reviewed during the assessment of this TTA was insufficient to create a tabular summary of chemical occurrences in upland media.

10.3.2.15 Slip 6 TTA

The Slip 6 TTA was identified by Ecology, in consultation with the EPA, because of surface sediment chemistry and potential source concerns (Cargill et al. 2007). Although source control activities have been conducted at some of the properties in this TTA, neither a Data Gaps Report nor a SCAP has been prepared. These documents are scheduled for completion in 2008 (SAIC 2007b).

CSL exceedances in surface sediment within this TTA were reported for benzoic acid, phenol, total PCBs, lead, and mercury. SQS exceedances were reported for BBP, BEHP, individual PAHs, dibenzofuran, and total HPAH. Most exceedances were located in the slip. The one identified lead exceedance was located in the northern portion of the TTA (outside of the slip), and the mercury exceedance was centrally located within the slip. Total PCB exceedances were identified in the northern portion of the TTA (outside of the slip).

Historical and current commercial and industrial operations in the vicinity of this TTA have included cargo handling and storage; vehicle manufacturing and storage; airplane manufacturing, research, testing and development; lumber milling; and resin, glue, and chemical manufacturing. Several private outfalls discharge from adjacent upland properties into Slip 6. One public outfall (2080) at the head of the slip drains approximately 70 ac of the south-central portion of the KCIA and an unknown acreage of E Marginal Way S. Relevant information for this TTA is provided on Map 10-17.

Various investigations at upland properties in this TTA have identified chemicals in soil, groundwater, seep, stormwater, and source tracing samples (Table 10-17).

Table 10-17. Chemicals identified in upland media and source tracing samples potentially associated with the Slip 6 TTA

Chemicals Exceeding CSL in Surface Sediment	Soil	Ground-water	Seeps[a]	Storm-water	Source Tracing Samples[b]	Source
Benzoic acid						
Total PCBs	X	X	X		X	EPA (2005d)
Phenol	X	X			X	Rhône-Poulenc (1995)
Mercury	X	X	X	X		Anchor (2006); EPA (2005d)
Lead	X	X	X	X	X	Anchor (2006); EPA (2005d)

Note: The absence of an X in any cell in this table does not necessarily imply that that chemical is absent in the upland media or in source tracing samples; in many cases, that chemical may not have been analyzed or, if analyzed, could have been present but at concentrations below the RLs.

[a] Metals seep samples were filtered; PCB seep samples were unfiltered. Entrained solids can affect sampling results if unfiltered.

[b] Source tracing samples included in-line sediment grab samples.

CSL – cleanup screening level

PCB – polychlorinated biphenyl

TTA – tentative two area

10.4 Summary

Since the initiation of the LDW RI/FS, three analyses have been conducted to define specific areas within the LDW that require further investigation. The first analysis was conducted as part of the Phase 1 RI (Windward 2003d) and the second analysis was presented in an early FS deliverable (RETEC 2006). Both analyses were focused primarily on SMS exceedances in surface sediment.

A third analysis, conducted by Ecology (2007k), in consultation with EPA, also utilized surface sediment chemistry data but was focused on areas where source control investigations were likely to be needed. The result of Ecology's recent efforts was the designation of 15 areas of interest for source control, including the seven EAAs originally identified in the Phase 1 RI (Windward 2003d). The other areas of interest for source control included eight TTAs identified for the purpose of source control evaluation. Only one of these TTAs (Glacier Bay TTA) had previously been identified as an area of concern, encompassing three PPAs identified in RETEC (2006).

Given the importance of surface sediment chemistry data relative to SMS in the designation of EAAs, and their significance with respect to ongoing removal actions, these data were summarized for the entire LDW site, then for areas both inside and outside the EAA boundaries. Detected concentrations for 41 of the 47 SMS chemicals exceeded the SQS at one or more locations. Total PCB concentrations exceeded SMS criteria at more locations than any other chemical. Surface sediment locations inside the seven EAAs, which make up 7.2% of the total LDW study area, account for 58% of the locations (331 of 574) with SMS exceedances. Almost all of the 331 locations with SMS exceedances within EAAs (92% or 305 locations) were within the four largest sponsored EAAs (Duwamish/Diagonal, Slip 4, T-117, and Boeing Plant 2/Jorgensen Forge), which have been sampled extensively.

The 15 areas of interest for source control identified through the processes mentioned above are the focus of Ecology's ongoing source identification and control efforts in the LDW. The size, composition, and number of chemical exceedances in each of these areas varies greatly; many of the TTAs have little identified surface sediment contamination. Most importantly, the in-water sediment boundaries of these areas are being used for source evaluation purposes only and do not necessarily constitute areas that will require sediment remediation. These areas will likely evolve as source identification and evaluation efforts continue into the future.

Ecology is continuing with their source evaluation and control efforts for the areas of sediment contamination in the LDW. Data Gaps Reports and SCAPs have been developed by Ecology for many of the areas; completion of the remainder of these documents, especially for the TTAs, is scheduled through 2011. Supplemental characterization and sampling investigations may be completed at these areas as needed and existing reports will be updated as necessary as new information on potential sources and recontamination issues arise. Source tracing sample collection will also continue into the future.

11 Key Findings

This section highlights the important findings of the RI and synthesizes these findings into a unified CSM. The physical, chemical, and ecological processes that have been examined in the RI are interrelated and must be considered in combination when assembling and analyzing alternatives in the FS, and when making remedy and risk management decisions for the LDW.

11.1 GEOGRAPHIC LANDSCAPE OF THE LDW

Prior to the significant alterations that began in the early 1900s, the lower sections of the Duwamish River were characteristic of the many estuarine rivers in the Pacific Northwest. The river exhibited high sinuosity as it meandered through the valley floor and emptied into Elliott Bay as a broad delta. Given the nature of the rivers feeding the lower Duwamish and the flat topography of the valley floor, the terminal section of the river was highly depositional, giving rise to the broad expanse of mudflats depicted in early maps of the area.

As a result of its development, the Green River/Duwamish River system has undergone significant hydrological changes, which have reduced flow by an estimated 78% from historical levels. These changes have included the natural diversion of the White River following the flood of 1906, the loss of flow from the Black River when the water level in Lake Washington dropped in response to the opening of the Ship Canal and Hiram M. Chittenden locks, and the diversion of the Cedar River to Lake Washington. Even with the loss of these water sources, the Green River/Duwamish River system continued to flood low-lying areas. The Howard Hansen Dam, built in 1961 for flood control, effectively decreased peak flows in the river system. Currently, peak flows rarely exceed 12,000 cfs. Despite these changes, the Green River still delivers an estimated 221,000[108] metric tons of sediment load to the Duwamish River each year (QEA 2007).

A portion of the LDW is maintained as a federal navigation channel by USACE. Elevations within the navigation channel are maintained between -30 ft MLLW and -15 ft MLLW, with the shallower elevations maintained in the upper reaches of the LDW. King County Waterway Commercial Waterway District No. 1 constructed an Upper Turning Basin at the head of the navigation channel (RM 4.6) prior to 1920, which serves as a catchment for sediment from the Green/Duwamish River system. The need for this engineered catchment, along with the periodic dredging required to maintain the navigation channel, is evidence that the LDW continues to be a depositional environment.

[108] Estimated sediment load from the Green River to the LDW from 1960 to 1980 (QEA 2007). This value remains relevant because no significant changes have occurred to the morphology of the river that would be expected to change the sediment load.

The lower 5 mi of the Duwamish River have been highly modified. In the early 1900s, issues with flooding in the area precipitated the installation of levees and dams and the subsequent channelization of the river. Most of the tidal marsh, mudflats and shallows, and tidal wetlands historically present in the lower Duwamish River have either been filled or dredged. Today, the majority of the LDW shoreline consists of riprap, pier aprons, or sheet pilings, especially in the downstream portion of the site.

Intermittent, shallow benches exist in the nearshore, intertidal, and shallow subtidal zones of the LDW outside the navigation channel. These benches are of various dimensions and elevations, with minimum elevations of less than 3 ft MLLW. Intertidal habitats are dispersed in relatively small patches (i.e., generally less than 1 ac in size) downstream of RM 3.0, with the exception of Kellogg Island, which represents the largest contiguous area of intertidal habitat remaining in the LDW. Low intertidal mudflats are present below upper bank riprap and miscellaneous armoring in the reaches upstream of RM 3.0.

The LDW provides a navigational corridor for the movement of raw materials and products used by local facilities, and serves as a major base for marine transport to Alaska. Current land use, zoning requirements, and land ownership within most of the LDW corridor are consistent with the characteristics of an active industrial waterway. Non-industrial uses also occur on and along the waterway. Two mixed residential/commercial/industrial neighborhoods, South Park and Georgetown, are located west and east, respectively, of the LDW. The presence of these neighborhoods and various public access points along the LDW results in some recreational uses of the waterway. Recreational users can collect seafood from the various public access areas, can walk or play on the beaches, or can access the river by boat (most of the LDW is publicly owned and accessible by boat). In addition, the Muckleshoot Tribe has an active fishery on the LDW for returning salmon.

Despite the highly modified river habitat, the LDW is home to a diverse assemblage of aquatic and wildlife species and a robust food web that includes top predators. More than 53 resident and anadromous fish species are found in the LDW (Windward 2004d, 2005b, 2006c). A total of 61 invertebrate taxa were identified in a survey of the intertidal and subtidal benthic invertebrate community, including crabs, clams, mussels, crustaceans, marine worms, and gastropod species (Windward 2005e). In addition, surveys have indicated that up to 87 species of birds and 6 species of mammals use the LDW during part of the year to feed, rest, or reproduce. Avian species include passerines, raptors, shorebirds, waterfowl, and seabirds (Canning et al. 1979); mammals include river otters, which may be permanent residents, and harbor seals and sea lions, which may visit the area. Fifteen species identified in the LDW study area are listed under the ESA or by WDFW as candidate species, threatened species, or species of concern.

While the LDW will continue to be used as an industrial corridor, there is a high interest by the resource trustees and the public to create higher quality habitat where

opportunities exist. The focus of these efforts thus far has been in small areas of marsh and unvegetated intertidal marsh habitat, including Terminal 105 (RM 0.1); Terminal 107 and Herring's House Park near Kellogg Island (RM 0.3 to RM 0.5); T-108 (RM 0.6); GSA marsh restoration (RM 0.8); Hamm Creek (RM 4.3); Muckleshoot Kenco Marine, Coastal America, and Port of Seattle mitigation sites near the Upper Turning Basin (RM 4.7); and Cecil B. Moses Park (North Winds Weir, RM 5.7).

11.2 CONCEPTUAL SITE MODEL

A CSM for the LDW that describes key features and processes of the waterway was developed as part of the RI/FS process. The CSM will aid in the assembly and evaluation of remedial action alternatives in the FS and in future decision-making for the site. The CSM is considered a working model throughout the RI/FS process, and may continue to be refined as new information and analyses are developed.

The LDW CSM provides a synthesis of the major processes governing the movement and distribution of chemicals at the site as well as potential exposure pathways for the people and animals that use the site (Figure 11-1). Three key components of the LDW CSM are worthy of discussion because of their importance:

- **Physical CSM –** The influence of hydrodynamic and sediment transport processes on the physical distribution of sediment within the LDW
- **Chemical CSM –** Patterns in the distribution of chemicals in the LDW
- **Exposure CSM –** Exposure of human and ecological receptors to chemicals within the LDW

11.2.1 Physical CSM

The LDW can be characterized as a well-stratified salt wedge estuary. The LDW receives inflows of marine (or salt) water from Elliott Bay and fresh water from the Green/Duwamish River system. Fresh water from the Green/Duwamish River system moving downstream overlies the tidally influenced salt water entering from Elliott Bay. Typical of tidally influenced estuaries, the LDW has a relatively sharp interface between the freshwater outflow at the surface and saltwater inflow at depth.

During low-flow conditions in the Green River, the saltwater wedge extends, on average, up to approximately RM 4.5 to RM 4.8, depending on the tidal cycle and river flows (Figure 11-1). Under extreme conditions, the saltwater wedge can extend as far upstream as RM 8.7. Thus, much of the LDW is dominated by two-layer estuarine circulation, in which the denser salt water is transported upstream in the lower layer of the water column and the fresh water is transported downstream in the upper layer. Above RM 4.8, the LDW is typically a tidally influenced river in which the entire water column is generally composed of fresh water.

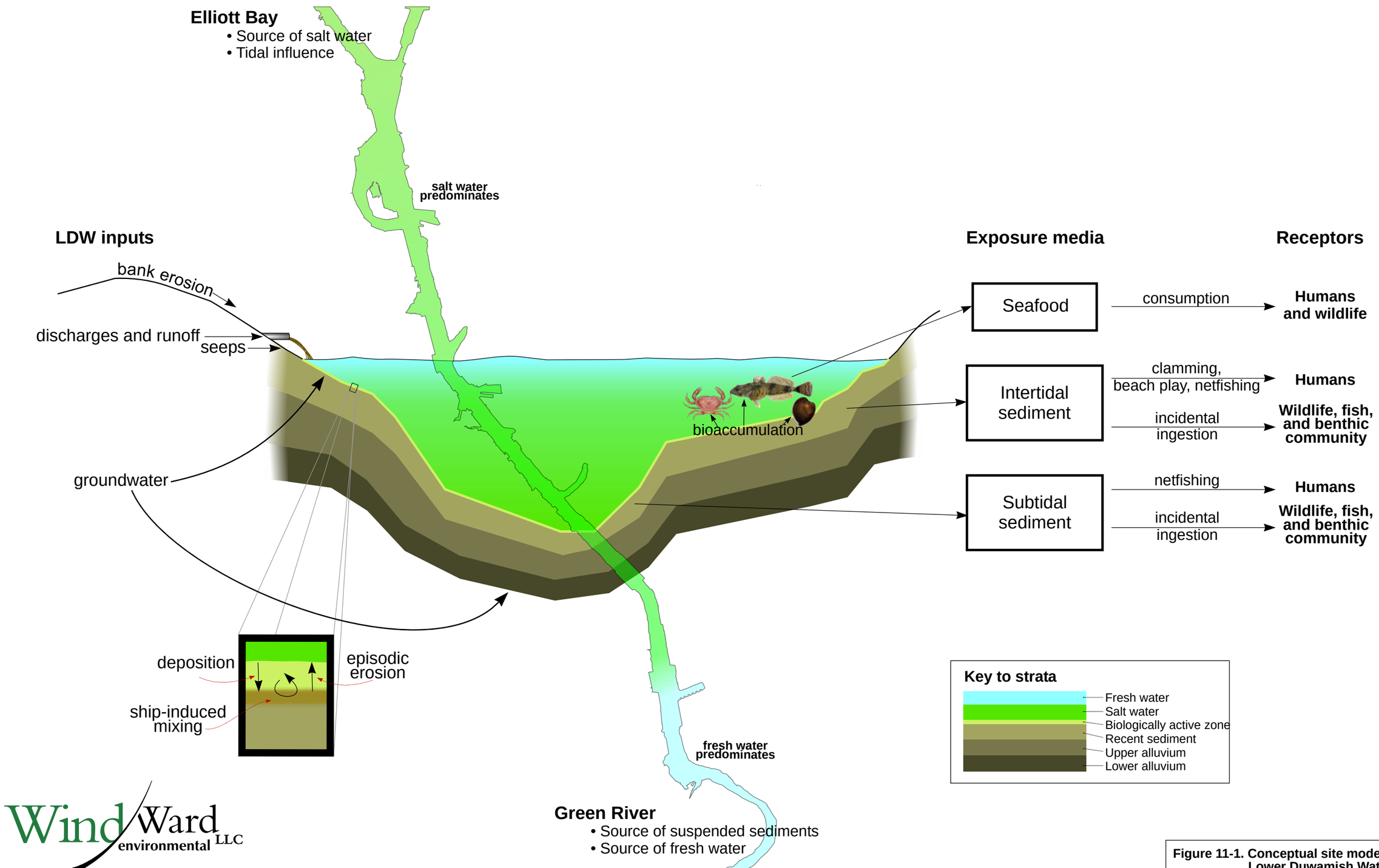


Figure 11-1. Conceptual site model for the Lower Duwamish Waterway

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The two-layer estuarine circulation plays a key role in patterns of sediment deposition and scour potential. Reaches of the LDW that are occupied by the saltwater wedge experience minimal bed scour from fresh water coming from upstream because much of the sediment bed is covered by tidally driven salt water. The tidal velocities within the saltwater wedge are much lower than the current (river) velocities. Reaches of the LDW that are entirely fresh water during high flow events are more susceptible to bed scour because the near-bed velocities are driven by river flow and not tidal velocities.

The physical CSM, which is based on both empirical data collected from the LDW and an STM, identified three reaches of the LDW that have distinct sediment transport characteristics: RM 0.0 to RM 2.2 (Reach 1), RM 2.2 to RM 4.0 (Reach 2), and RM 4.0 to RM 4.8 (Reach 3). Differences among these reaches are defined by the hydrodynamic characteristics of each reach, particularly during high-flow events. The sediment dynamics for Reaches 1, 2, and 3 are illustrated in Figures 11-2, 11-3, and 11-4, respectively, and summarized below.

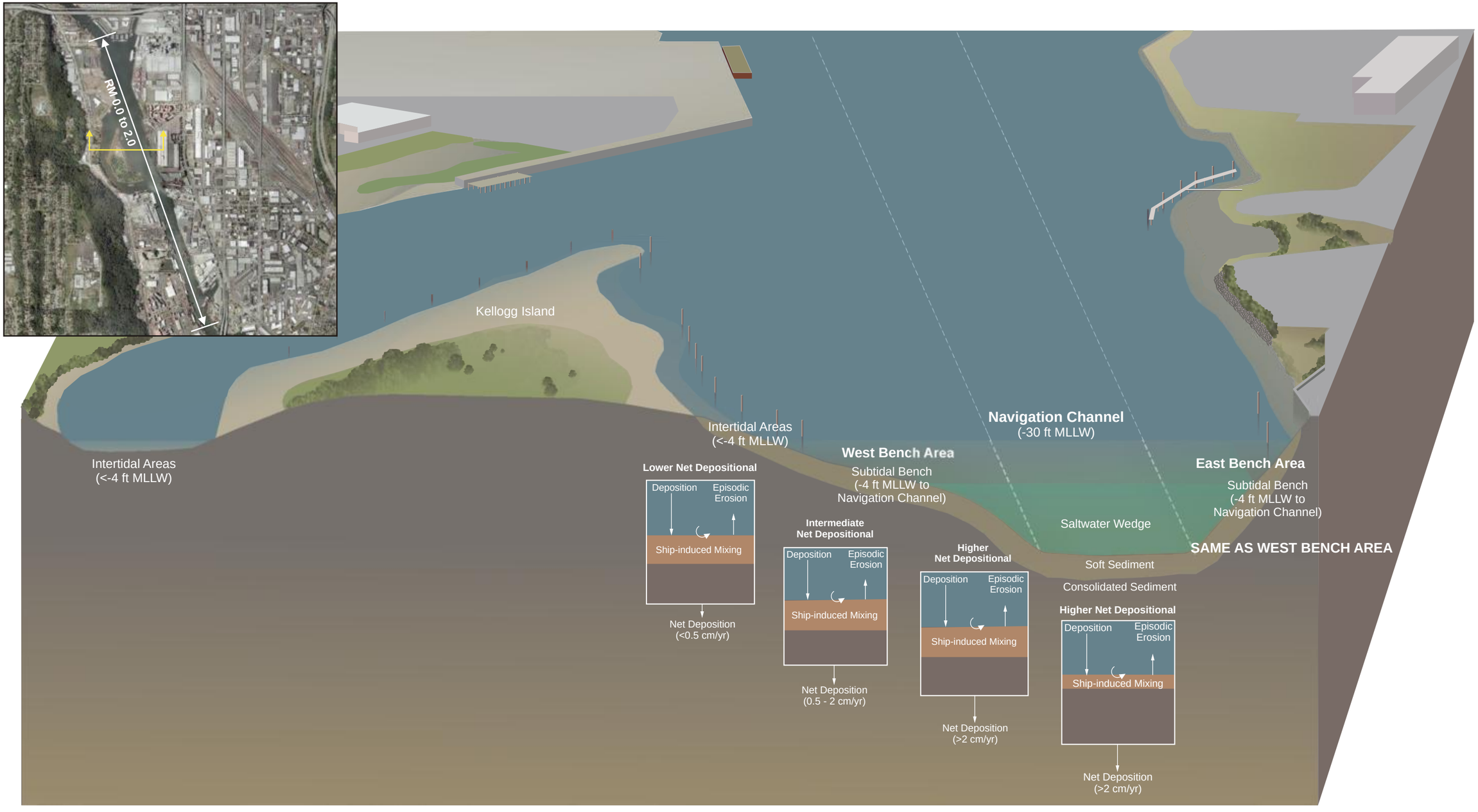
- **Reach 1: RM 0.0 to RM 2.2 —** The permanent presence of the saltwater wedge in this reach serves as a protective barrier for the river bed. Consequently, the potential for bed scour, even during high-flow events, is relatively low, as is the potential for buried sediments to be re-exposed. This reach is net depositional on an annual time scale. Approximately 60% of the area within and outside the navigation channel has sedimentation rates between 1 and 2 cm/yr; an additional 30% of the area has rates greater than 2 cm/yr.
- **Reach 2: RM 2.2 to RM 4.0 —** Reach 2 is occupied by the saltwater wedge during most flow conditions, but during high-flow events the toe of the saltwater wedge may only reach the downstream portion of this reach. As a result, Reach 2 could experience net erosion during high-flow events with 2-yr or greater return intervals. Even during 100-yr storms, net erosion is generally limited to the upper 10 cm of the sediment bed (i.e., within the biologically active zone). Reach 2 is also net depositional on an annual time scale. Net sedimentation rates are estimated to be spatially variable. Between RM 2.2 and RM 2.6, approximately 90% of the area outside the navigation channel and 70% of the area within the navigation channel has sedimentation rates greater than 2 cm/yr. Between RM 2.6 and RM 4.0, approximately 50% of the area both within and outside of the navigation channel has sedimentation rates less than 1 cm/yr; less than 35% of this reach has sedimentation rates greater than 2 cm/yr.

Reach 3: RM 4.0 to RM 4.8 — During most flow conditions, the saltwater wedge extends into this reach, with the location of the toe of the wedge dependent on a combination of river-flow and tidal influences. During high-flow events, the saltwater wedge is not present in this reach, and freshwater flows dominate. This reach is wider than the downstream reaches, which creates lower current velocities that are less likely to erode the sediment bed. The upstream portion of the reach serves as a catchment for

sediments transported into the LDW from upstream; by dredging this area every several years, the USACE is able to keep a large portion of the sediments (especially bedload) entering the LDW from upstream from moving into the lower reaches of the LDW. Approximately 90% of the area outside of the navigation channel and all of the area within the navigation channel has net sedimentation rates greater than 2 cm/yr. Over shorter time scales associated with high-flow events (i.e., storms that may last for days or weeks), over 80% of the LDW area is still expected to be net depositional. Slightly more erosion is predicted for the 100-yr high-flow event (12,000 cfs) compared to the 10-yr (10,800 cfs) and 2-yr (8,400 cfs) high-flow events, but the areas experiencing erosion as well as the net erosion depths within those areas are similar for all high-flow events (Maps 3-6, 3,-7, and 3-8). Approximately 0.4% of the LDW is predicted to experience greater than 10 cm net erosion during a 100-yr high-flow event, with a maximum predicted net erosion depth of approximately 14 cm. Based on these model results, high-flow events are not expected to expose subsurface sediment at greater depths.

On a smaller scale, there are a number of processes that can affect localized sediment deposition and erosion. Net sedimentation rates may be higher than estimated by the STM in the vicinity of specific outfalls if the actual particulate load from those outfalls is greater than assumed or not specifically included in the model. Net sedimentation rates in specific areas may also be higher than estimated by the model if there have been significant inputs of anthropogenic materials (e.g., sandblast grit in the vicinity of shipyards or bulk material spilled during offloading) that are not addressed by the STM. Prop wash associated with ship and tugboat operations in the LDW can also have localized effects and has the potential to erode deeper holes in small areas than the net erosion depths estimated by the STM. Ship-induced bed scour from passing vessels is estimated to result in a continual reworking of the uppermost 1 to 2 cm of the sediment bed within the navigation channel. In a typical event, sediment is suspended for a brief period of time, but is then re-deposited shortly thereafter once the propeller-driven forces are no longer present.

- Based on a mass balance of sediment loads entering and leaving the LDW system, it is estimated that less than 1% of the sediment load enters the LDW from lateral inputs such as storm drains, CSOs, and surface water runoff. Over 99% of the sediment entering the system is from upstream. Approximately 50% of the incoming sediment exits the system as suspended sediment; the remaining material settles and becomes incorporated into the sediment bed. The sediment transported from upstream of the LDW is deposited onto the sediment bed and is mixed with the surface sediment layer (i.e., top 10 to 15 cm) by both biological and physical processes. Over time, the bed-source content of the surface layer decreases at an approximately exponential rate, primarily because of the deposition of upstream source sediment (Map 3-13).



Notes: 1. Approximate net depositional rates from Sediment Transport Analysis Report, Windward and QEA 2006.
2. Inserts are qualitative illustrations and are not to scale.

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NOTE: Figure prepared January 11, 2006, by Brick Tudor Studios for RETEC, Inc. (RETEC | ENSR). Figure footer updated November 5, 2007, by Windward Environmental LLC.

Figure 11-2. LDW sediment transport CSM for Reach 1 (RM 0.0 to RM 2.0)

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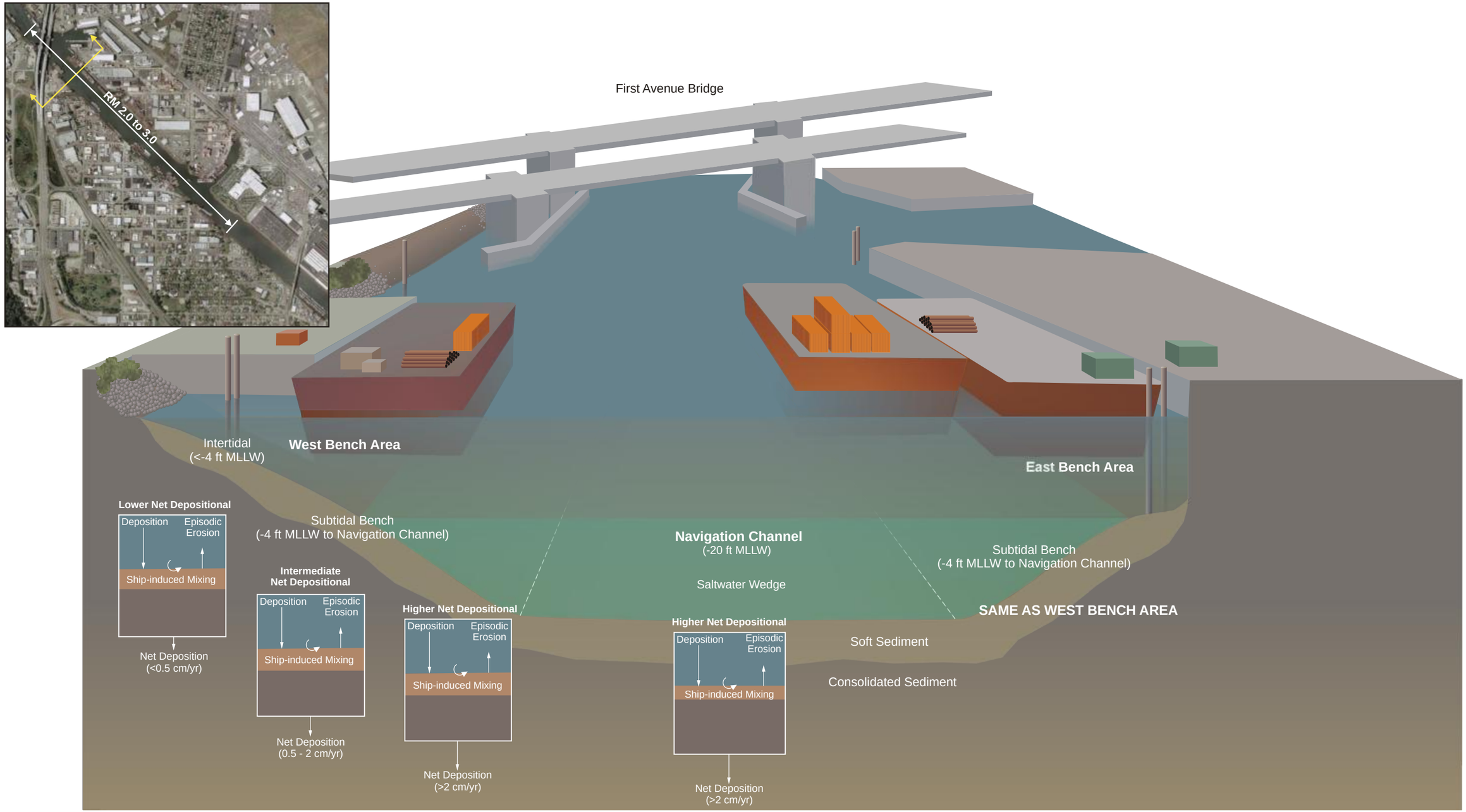

Notes: 1. Approximate net depositional rates from Sediment Transport Analysis Report, Windward and QEA 2006.
2. Inserts are qualitative illustrations and are not to scale.

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NOTE: Figure prepared January 11, 2006, by Brick Tudor Studios for RETEC, Inc. (RETEC | ENSR). Figure footer updated November 5, 2007, by Windward Environmental LLC.

Figure 11-3. LDW sediment transport CSM for Reach 1 (RM 2.0 to RM 3.0)

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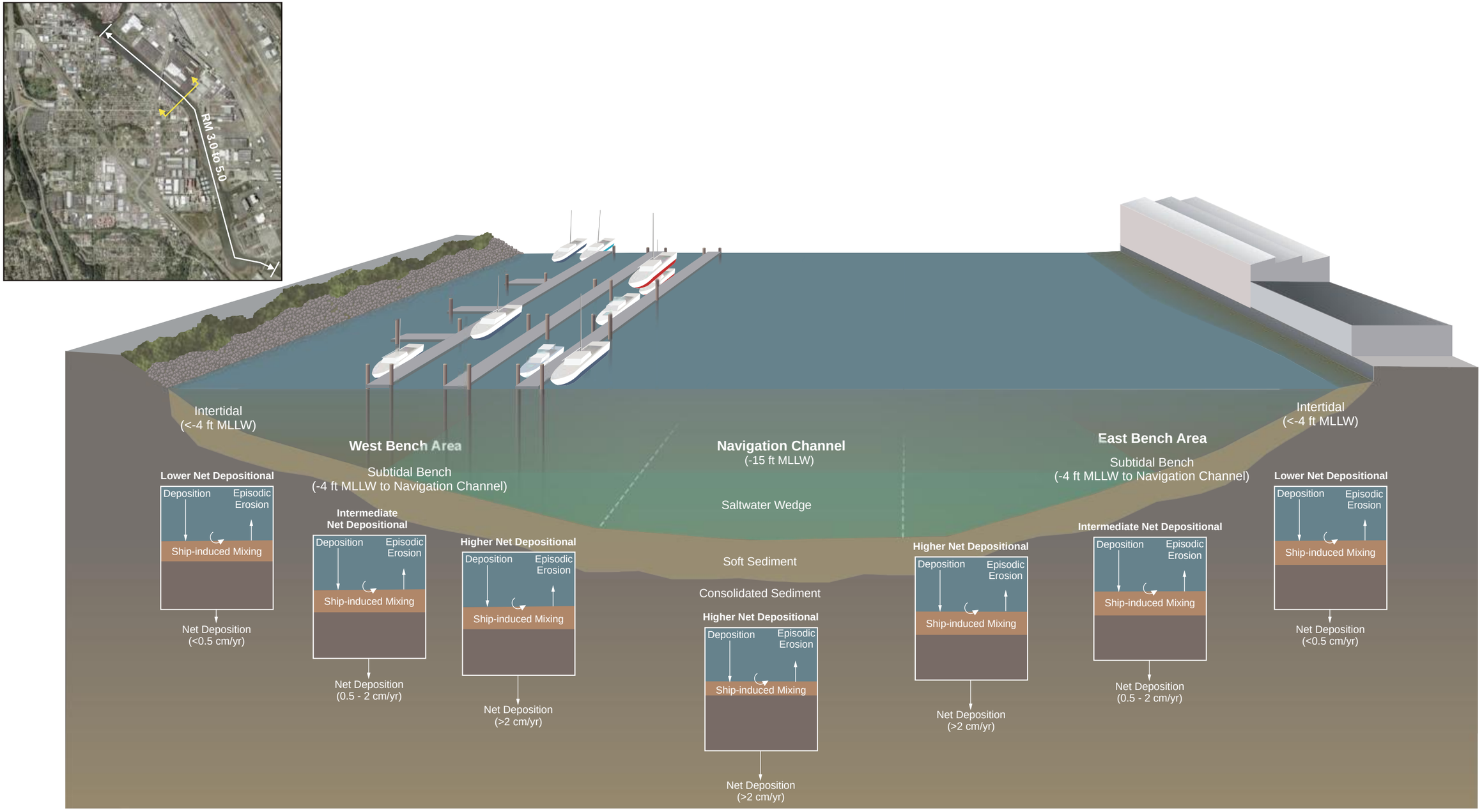

Notes: 1. Approximate net depositional rates from Sediment Transport Analysis Report, Windward and QEA 2006.
2. Inserts are qualitative illustrations and are not to scale.

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Figure 11-4. LDW sediment transport CSM for Reach 3 (RM 3.0 to RM 5.0)

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Approximately 90% of the total bed area in the LDW has an estimated half-time[109] of 10 yrs or less (Map 3-15).

The sediment transport model developed for the LDW to estimate scour and deposition patterns was collaboratively developed and calibrated by a working group that included representatives from EPA, Ecology, USACE, and LDWG. EPA considers the model to be functioning acceptably for site-wide applications.

11.2.2 Chemical CSM for sediment

The extensive RI dataset for both surface sediment (e.g., 0 to 15 cm in depth) and subsurface sediment provides a thorough characterization of sediment contamination patterns within the LDW. The observed patterns in the spatial and vertical distributions of chemicals in surface and subsurface sediment, respectively, are the result of interactions among several factors, including the proximity and status of chemical sources (present-day and historical), mixing of the surface layer, transport and deposition of sediment within the LDW over time, and localized conditions that affect sediment mixing (e.g., scour from outfalls and tugboats, dredging).

Numerous chemicals are present in surface sediment in the LDW. Risk driver chemicals (referred to as indicator hazardous substances in the MTCA program [WAC 173-340-703]) were selected as part of the baseline risk assessments, in consultation with EPA and Ecology, to focus the remedial analyses in the FS.[110] The nature and extent of all chemical contamination are discussed in Section 4.

Four risk drivers were identified in the HHRA based on risks associated with seafood consumption and direct sediment contact: total PCBs, arsenic, cPAHs, and dioxins and furans. Total PCBs were also identified in the ERA as a risk driver for river otter. In addition, 41 chemicals (including total PCBs, arsenic, several phthalates, and various PAHs) were identified as risk drivers for benthic invertebrates because detected concentrations of these 41 chemicals exceeded the SQS of the Washington State SMS at one or more locations (Table 10-2). Total PCB concentrations in surface sediment exceeded the SQS at more locations than any other chemical. The chemical with the next highest number of locations with detected concentrations in surface sediment greater than the SQS was BEHP. The distributions of these risk driver chemicals in sediment are discussed below.

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[109] Half-time is defined as the time needed for a 50% reduction of a particular sediment component (e.g., bed sediment composition change by 50%).

[110] Other chemicals that exceeded risk thresholds but were not selected as risk drivers for a particular exposure pathway may be addressed through focused evaluation in the FS, as part of the 5-yr review, or included in the post-remedial monitoring program, as appropriate.

Surface Sediment Patterns

In general, high concentrations of chemicals were detected in localized areas separated by larger areas of the LDW with relatively low concentrations. Contamination was generally more widespread in the lower reaches of the LDW, although areas with high concentrations also exist in more upstream locations (Maps 4-60a, 4-60b, and 4-60c).

PCBs are widely distributed throughout the LDW and concentrations are highly variable. Total PCBs were detected in 94% of the 1,365 surface sediment samples in which they were analyzed, at concentrations ranging from 1.6 to 220,000 µg/kg dw, with a median concentration of 115 µg/kg dw (Map 4-14). Arsenic from both natural and anthropogenic sources was detected throughout the surface sediments in the LDW, with concentrations ranging from 1.2 to 1,100 mg/kg dw, with a median concentration of 11 mg/kg dw (Map 4-24). The highest arsenic concentrations were also highly localized.

Despite the widespread distribution of both of these chemicals in the LDW, the locations of the highest total PCB and arsenic concentrations were generally not in the same areas, indicating that sources likely differ for these two chemicals. The majority of the high total PCB concentrations were located within fairly well defined areas (Map 4-14), suggesting the nearby presence of one or more ongoing or historical sources. Areas with the highest total PCB concentrations in the baseline surface sediment dataset included the Duwamish/Diagonal EAA, EAA 2 (RM 2.2 on the west shore), the Slip 4 EAA, the T-117 EAA, the Boeing Plant 2/Jorgensen Forge EAA, and the Norfolk EAA (Map 4-14). Removal actions within some of the EAAs have already substantially reduced surface sediment PCB concentrations in those areas. Likely sources of PCBs have been identified in most of these EAAs and source identification efforts are ongoing. However, specific sources have not been identified for all areas with higher total PCB concentrations (e.g., RM 1.0 in the navigation channel).

The majority of the higher arsenic concentrations were also located within fairly well defined areas (Map 4-24), suggesting the presence of one or more nearby ongoing or historical sources. Areas with the highest arsenic concentrations in the baseline surface sediment dataset included the Duwamish/Diagonal EAA, Slip 1, RM 1.1 to RM 1.5, RM 2.0, RM 2.8, and RM 3.7 to RM 3.9 (Map 4-24). Otherwise, most of the arsenic concentrations were similar to urban background concentrations (Section 7.2).

Areas with the highest cPAH concentrations in the baseline surface sediment dataset were located in many of the same areas identified for arsenic and total PCBs, including the Duwamish/Diagonal EAA, RM 1.3 to RM 1.4, Slip 4, and RM 3.7 to RM 3.8 (Map 4-31). Other areas with high cPAH concentrations were generally downstream of RM 1.1, at RM 2.1, RM 2.7, and in Slip 6. Sources of cPAHs for some of these areas have been identified. In general, high cPAH concentrations were more dispersed than PCBs and arsenic, suggesting more ubiquitous sources.

Dioxin and furan TEQs in the LDW were generally uniformly distributed and similar to urban background concentrations detected in the greater Seattle area (Section 7.2), except for a few areas with substantially higher TEQs. Dioxin and furan TEQs were highest in the embayment on the west side of the LDW between RM 1.4 and RM 1.5 (Map 4-37). Other high concentrations were generally located in EAAs, such as the Duwamish/Diagonal EAA, EAA 2 at RM 2.2, and the Boeing Plant 2/Jorgensen Forge EAA. A few other locations also exist, such as the navigation channel at RM 1.0, where both the dioxin and furan TEQ and the total PCB concentrations were high.

Higher concentrations of numerous chemicals, including total PCBs, PAHs, and arsenic, are represented on Maps 10-2a through 10-2f as SMS exceedances. Most samples with high metals concentrations were located either downstream of RM 2.2 or generally in areas identified for other risk drivers above, such as Slip 1, Slip 4, RM 1.3 to RM 1.5 on the west side, RM 2.2 on the west side, and the Boeing Plant 2/Jorgensen Forge EAA. The majority of locations with high BEHP concentrations in surface sediment were in many of these same locations (most notably in and near the Duwamish/Diagonal EAA, near RM 2.2 on the west side of the LDW, Slip 4, and in the Boeing Plant 2/Jorgensen Forge EAA). Thus, to a large extent many of the highest chemical concentrations were co-located. However, there is also a broad scattering of isolated SMS exceedances from different chemicals for which sources have generally not been identified.

Subsurface Sediment Patterns

Data from subsurface sediments collected from cores also provide important information and offer insight into processes that contributed to present-day distributions. Physical events, such as dredging, river channelization, and debris disposal are recorded in the geological structure of subsurface sediments. Physical and chemical markers within sediment cores can be used to reconstruct historical records of contamination and estimate the age of subsurface sediments.

Subsurface sediment chemistry data, radioisotope data, and physical characteristics of the material retained in the cores, when combined with the history of industrial development in the waterway, including chemical spills, channel diversions, and construction of the Howard Hansen Dam, allow for interpretation of the history of sedimentation in the LDW. Based on subsurface sediment stratigraphy in sediment cores, LDW sediments can be divided into the following three relatively distinct zones:

- **Recent sediment –** This upper zone represents the sediments deposited in the LDW over approximately the last 50 yrs and generally provides a record of the more recent activities that have occurred within the LDW since construction of the Howard Hansen Dam.
- **Upper alluvium –** The second zone, located beneath the recent sediment, represents sediments deposited in the LDW following the construction of the

waterway in the early 20th century but before construction of the Howard Hansen Dam. This layer contains the record of activities that occurred within the LDW through much of the early commercial and industrial development along the LDW.

- **Lower alluvium –** The deepest zone, located beneath the upper alluvium, represents historical sediment deposits that predate construction of the LDW.

The upper alluvium and the recent sediment zone consist of material that has been deposited in the LDW since the onset of urbanization and industrial activity in the uplands surrounding the LDW.

The distribution of chemicals within cores illustrates the sedimentation history. The trend of total PCB concentrations in LDW sediment cores generally reflects the introduction of PCBs (circa 1935) and the ban on the use of PCBs (1978). Peak PCB use has been recorded in Puget Sound sediment cores between 1960 and 1970 (Lefkovitz et al. 1997; Van Metre and Mahler 2004); peak total PCB concentrations are often preserved in subsurface intervals of LDW cores as well. An analysis was conducted to compare the depths of peak total PCB concentrations in subsurface sediment to expected depths based on the net sedimentation rates estimated using the STM and assumptions regarding the period of peak PCB usage in the LDW and minimal sediment disturbance. Most of the cores had peak concentrations as deep or deeper than expected, supporting the CSM that historical PCB contamination is being buried in most areas by sediment that is less contaminated. Some cores had peak total PCB concentrations that were shallower than expected; most, if not all, of these cores were either at locations with low net sedimentation rates or in areas with potential localized ongoing sources or localized disturbances, such as dredging (Map 4-59).

Detected total PCB concentrations ranged from 0.52 to 890,000 µg/kg dw in subsurface sediments, with the highest concentrations generally detected in deeper core intervals, except in a few areas with low net sedimentation or potential ongoing sources, as discussed above (Maps 4-20a through 4-20f). Arsenic concentrations in subsurface sediment, which ranged from 3.9 to 2,000 mg/kg dw, also followed this pattern, with the highest concentrations generally detected in deeper core intervals, with a few exceptions (Maps 4-27a through 4-27e).

Profiles of cPAH concentrations in subsurface cores were less consistent. The onset of PAH contamination, typically associated with industrialization, and the decrease of PAH contamination, associated with implementation of source controls, were more gradual. Nevertheless, peak concentrations of cPAHs within cores were often found in deeper core intervals in areas with the highest cPAH concentrations, suggesting a reduction in inputs of cPAHs over time. However, moderately elevated concentrations of cPAHs were also often found in uppermost core intervals, suggesting continued low-level contributions of PAHs to the LDW, typical of urbanized environments.

Dioxins and furans were generally analyzed in subsurface cores in areas with higher concentrations in surface sediment as part of the RI. In the core near RM 1.0 in the navigation channel, the highest TEQ occurred in the 4-to-6-ft interval. This TEQ was higher than the TEQ in the surface sediment sample from that area. In contrast, within the embayment between RM 1.4 and RM 1.5, which had the highest dioxin and furan TEQs in the LDW, the highest TEQ was in surface sediment and the uppermost core interval. In a subsurface core collected just outside the inlet, the highest TEQ was detected in the 6-to-8-ft interval, but TEQs were lower. The estimated net sedimentation rate in this embayment was 2 to 3 cm/yr, although some of the embayment was not included in the model and is isolated by a pier across its mouth. The higher TEQs in surface sediment in this area indicate that burial by cleaner material is not occurring, potentially because of lower-than-estimated sedimentation rates in this embayment or continuing sources in this area.

Overall, 77% of the cores collected from the LDW during RI sampling events had chemical concentrations that were less than the SQS in the lowest interval of the core that was analyzed. This depth was typically reached at about 4 to 6 ft. In the cores where the lower alluvium was analyzed (50% of the cores), only one SQS exceedance was detected.

In summary, the lateral and vertical distributions of chemicals in the LDW show a general trend of decreasing concentrations with time. These observations complement and validate the physical CSM, which predicts site-wide stability and recovery of most bedded sediments with time. The LDW is much cleaner than it was in the past and will continue to improve in the future where cleaner sediments are accumulating.

11.2.3 Exposure CSM

One of the key components of the RI is the assessment of risk for people and ecological receptors that may be exposed to site-related contamination. This section discusses the pathways of exposure that may lead to unacceptable risks. The risk assessments are summarized in Section 11.3.

The physical and chemical CSMs discussed above provide an overview of the distribution and movement of chemicals in the LDW, which influence the exposure of receptors to these chemicals. The exposure CSM includes chemical sources, transport mechanisms, exposure pathways, exposure routes, and potentially exposed receptors. Specific CSMs representing exposure pathways were presented for ecological receptors in Figures 5-1 and 5-2 and for people in Figure 6-1. Source information and transport mechanisms were discussed in Section 9.

The importance of various exposure pathways is dependent on the chemical and the receptor of interest (e.g., people, fish), but the principal pathways for exposure to contaminated sediment in the LDW are: 1) diet, primarily through ingestion of contaminated seafood; and 2) direct contact. Other pathways of exposure were also

included in the risk assessments, but were generally less important or had low risks associated with them. For example, porewater and seep data were collected in areas along the banks of the LDW where EPA and Ecology considered the likelihood of contamination from upland groundwater to be the greatest. Risks to aquatic life associated with exposure to water-borne chemicals were found to be low in these areas.

Dietary Exposure

People and wildlife can be exposed to chemicals associated with sediment contamination in the LDW through ingestion of contaminated seafood, such as fish, crabs, and clams. Most resident fish, which integrate their exposure throughout their foraging ranges in the LDW, are primarily exposed through the ingestion of benthic invertebrates, which have very localized exposures, or the ingestion of other fish that have a broader exposure.

Aquatic species that serve as food for other receptors are exposed to sediment-associated chemicals through various pathways (Figures 5-1 and 5-2). Chemical concentrations in the tissues of some species (e.g., benthic invertebrates) were strongly correlated with concentrations of total PCBs and arsenic in sediment. Weak tissue/sediment relationships were observed for clams and arsenic and cPAHs, suggesting that concentrations in clam tissues are complex and involve important variables that may never be well understood.

For fish and crabs, the relationship between total PCB concentrations in sediment and tissues was modeled using a steady-state bioaccumulation model (Appendix D). This model showed that total PCB concentrations in tissues could be related to concentrations in surface sediment within acceptable uncertainty bounds. The model also showed that as the total PCB concentrations in sediment decrease in the future, concentrations in water will exert a relatively greater effect on total PCB concentrations in tissues.

Dietary exposures in the risk assessments were estimated based primarily on consumption rates and chemical concentrations in tissues, although other factors were also included. Consumption rates for people were estimated for various scenarios. EPA and Ecology required the application of seafood consumption rates that have been developed for adult and child consumers based on seafood harvest from King County or Puget Sound. There is considerable uncertainty about the applicability of some of the seafood consumption rates to this HHRA, particularly for clams, given the size of the LDW and the availability of suitable habitat.

WSDOH has issued health advisories for LDW fish and crabs, warning the public that any seafood consumption from the LDW is unsafe because of elevated total PCB concentrations in these species (WSDOH 2005). WSDOH (Office of Food Safety and Shellfish) and Public Health—Seattle and King County also advise against

consumption of clams from all of King County, including the LDW, because of pathogens from urban sewage discharges (WSDOH 2005). A creel survey conducted by King County as part of their water quality assessment indicated that very few people fish or crab in the LDW (Simmonds et al. 1998). This intensive survey was conducted for 2 months in the summer of 1997, the period of highest use, and showed that use in the LDW was significantly lower than in other Puget Sound sites. However, people fish seasonally for salmon in the LDW and a relatively high number of individuals fished/crabbed from Spokane Street Bridge, just downstream of the LDW. Future harvest rates are uncertain and are anticipated to be higher following remediation, although the availability and size of seafood available for consumption will still be dictated largely by the availability of suitable habitat within the LDW.

To estimate exposure, chemical concentrations in tissues have been analyzed in various fish and invertebrate species that have been collected from throughout the LDW. Most of the tissue data are for English sole, shiner surfperch, Pacific staghorn sculpin, juvenile chinook salmon, Dungeness and slender crabs, soft-shell clams, and small invertebrates that live in the sediment, such as amphipods and marine worms. These species were selected because they were assumed to be representative of species that could be consumed by people, fish, or wildlife within the LDW. Their tissues were analyzed for a wide variety of chemicals.

Mean total PCB concentrations in the four sampling areas were highest for English sole, shiner surfperch, and crab hepatopancreas tissue (concentrations in edible crab meat were lower). Mean total PCB concentrations were lowest in juvenile chinook salmon. Mean inorganic arsenic concentrations were highest in soft-shell clams. The highest cPAH concentrations in tissue were detected in clams, mussels, and benthic invertebrates; concentrations were not as high in fish because fish metabolize PAHs. Other SVOCs were infrequently detected in tissue. DDT and chlordane compounds were detected in tissue samples, but there is high uncertainty in these data because of analytical interference from PCBs. The highest mean concentrations of metals (including TBT) were detected in invertebrates.

The most comprehensive tissue sampling event was conducted in 2004, several months after three dredging events were completed in the lower reach of the LDW and near Harbor Island (Duwamish/Diagonal in the LDW, Todd Shipyard in the West Waterway, and Port of Seattle in the East Waterway). Total PCB concentrations were much higher in some species in 2004 than in older (1995 to 1998) and more recent (2005 and 2006) samples, suggesting that exposure to total PCBs may have been higher immediately following the dredging events than is typical for the LDW. Temporary increases in total PCB concentrations in tissue have been documented elsewhere in the country following remedial dredging of PCB-contaminated sediments, such as in the Grasse River in New York (Alcoa 2001).

Direct contact

People are also exposed to chemicals in contaminated sediment through direct contact. The HHRA evaluated several scenarios for this pathway, including beach play, clamming, and netfishing, where people would come into direct contact with sediment. While many ecological species can be exposed through direct contact, the benthic invertebrate community is most closely related to direct sediment exposure pathways because they are in direct contact with sediment year-round.

11.3 RISK ASSESSMENT RESULTS

Baseline risk assessments were conducted to evaluate the potential for adverse effects for people and ecological species that may be exposed directly or indirectly to contaminated sediment in the LDW, as discussed above. Key findings from these assessments are presented in this section. The risk assessments are presented in full in Appendices A and B and summarized in Sections 5 and 6.

11.3.1 Ecological Health

The ERA assessed risks to a number of receptor groups that were selected to be both representative and protective of the range of species found in the LDW, either as resident species or transient/migratory species. Receptor groups included the benthic invertebrate community (including gastropods), fish, crabs, birds, and mammals. A summary of COCs based on the ERA is presented in Table 11-1.

Table 11-1. Summary of COCs based on the baseline ERA

COC	BENTHIC INVERTEBRATE COMMUNITY[a]	FISH	SPOTTED SANDPIPER	GREAT BLUE HERON	OSPREY	RIVER OTTER	HARBOR SEAL
PCBs	X	X	X			X	
Cadmium	X	X					
Chromium	X		X				
Copper	X		X				
Lead	X		X				
Mercury	X		X				
Vanadium		X	X				
BEHP	X						
Zinc	X						
Silver	X						
BBP	X						
Fluoranthene	X						
Indeno(1,2,3-cd)pyrene	X						
Arsenic	X						

[a] Forty-one chemicals were considered to be COCs for the benthic invertebrate community, based on exceedances of the chemical criteria of the SMS in one or more surface sediment samples within the LDW. The full list is presented in Table 5-6. COCs with eight or more CSL exceedances are included in this table.

BBP -- butyl benzyl phthalate

BEHP – bis(2-ethylhexyl) phthalate

COC – chemical of concern

ERA – ecological risk assessment

PCB – polychlorinated biphenyl

Underlined Xs signify risk drivers

Overall, seven chemicals were identified as COCs (total PCBs, cadmium, chromium, copper, lead, mercury, and vanadium) for at least one fish or wildlife species (Table 11-1). COCs were defined as chemicals with exposure levels that exceeded lowest effects thresholds. For most ecological species, risks were estimated to be low. Somewhat higher risks were estimated for river otter based on PCB exposures and for spotted sandpiper based on exposure to lead in one EAA (Table 5-16). No quantitative ecological risk estimates were calculated for dioxins and furans.

For the benthic invertebrate community, 41 COCs were identified based on at least one sediment sample exceeding its SQS; three additional COCs were also identified with concentrations higher than TRVs. Single locations with COC concentrations above the SQS for one or more chemicals were widespread. However, because many of these locations were isolated, site-wide sediment chemistry and toxicity test results indicated that adverse effects are not likely for 75% of the overall LDW area (i.e., the area in which chemical concentrations were less than or equal to chemical SQS criteria and where sediments were non-toxic[111] according to SQS biological criteria). Thus, the benthic invertebrate community, which serves as the base of the LDW food web, is likely to be healthy in most areas of the LDW.

There is a higher likelihood for adverse effects in approximately 7% of the LDW area, which was designated as having chemical concentrations or biological effects in excess of the CSL. The remaining 18% of the LDW area had chemical concentrations or biological effects between the SQS and CSL criteria, indicating that risks to the benthic invertebrate community are less certain than in areas with concentrations greater than one or more CSL. Some areas had multiple locations with chemical concentrations greater than the CSL criteria. Most of these areas are located within or near the EAAs.

Total PCBs and BEHP were the chemicals that most frequently exceeded the SQS and the CSL in surface sediment samples. Over 60% of the SQS exceedances for both total PCBs and BEHP and 71% and 90% of BEHP and total PCB CSL exceedances, respectively, were in EAAs. Mercury was the only other chemical with more than

[111] Sediment samples from 46 locations with chemical SQS exceedances were tested for toxicity. Sixteen of the samples were not toxic relative to SQS biological effects criteria. Toxicity tests were generally not conducted in areas with the highest contamination (e.g., EAAs), which were assumed to be toxic.

20 locations with CSL exceedances. Other chemicals with more than 20 locations with SQS exceedances were butyl benzyl phthalate, lead, mercury, zinc, and a few PAHs. Risks to the benthic invertebrate community from TBT were low.

11.3.2 Human Health

For the HHRA, risks were evaluated for several different seafood consumption scenarios and also for direct sediment exposure scenarios associated with beach play, clamming, or netfishing. There is limited data available on the amount of resident seafood organisms currently being harvested and consumed from the LDW. In the absence of site-specific human seafood consumption rates, EPA and Ecology required the application of seafood consumption rates that have been developed for tribal consumers based on seafood harvest from other areas of Puget Sound. The tribal seafood consumption rates represent relatively high rates that might occur for harvest from the LDW at some time in the future, although they likely overestimate the current consumption of resident seafood organisms from the LDW. In addition, API consumption rates were based on rates derived from self-harvested seafood from throughout King County (not limited to the LDW). It is not known if the seafood resources within the LDW at some future time would be sufficient to support these consumption rates, but these rates were assumed to represent hypothetical consumption rates for the estimation of risks.

Similarly, there are no LDW-specific data to estimate the degree to which humans may currently be directly exposed to sediments via beach play or clamming activities. The exposure scenarios assumed for these activities were intended to represent exposures for a health-protective estimation of risks. The tribal netfishing scenario, on the other hand, reflects exposure conditions that could occur under current tribal fishing activities within the LDW.

Using health-protective exposure assumptions, most of the risk posed by chemical contamination is associated with only a few chemicals (i.e., PCBs, arsenic, cPAHs, and dioxins and furans), which were designated as risk drivers (Table 11-2). Estimated excess cancer risks in the LDW were found to be highest for the seafood consumption scenarios. There were distinct differences in the contributions from each seafood type to the risk estimate for each chemical. For total PCBs, the excess cancer risks for the tribal adult RME scenario were attributed to several seafood types, including clams (35%), crabs (29%), and pelagic fish (24%). For arsenic and cPAHs, approximately 95% of the risk was attributed to consumption of clams.

Table 11-2. Summary of COCs based on the baseline HHRA

COC	Seafood Consumption	Clamming	Beach Play	Netfishing
Arsenic (inorganic)	X	X	X	X
cPAHs	X	X	X	X

COC	SEAFOOD CONSUMPTION	CLAMMING	BEACH PLAY	NETFISHING
Total PCBs	X	X	X	X
Dioxins and furans	X[a]	X	X	X
Aldrin	X			
BEHP	X			
alpha-BHC	X			
beta-BHC	X			
Carbazole	X			
Total chlordane	X			
Total DDTs	X			
Dieldrin	X			
gamma-BHC	X			
Heptachlor	X			
Heptachlor epoxide	X			
Hexachlorobenzene	X			
Pentachlorophenol	X			
Toxaphene		X		X
TBT	X			
Vanadium	X			

[a] Dioxins and furans were assumed to be a risk driver in the absence of any tissue chemistry data, based on an expectation that if such data would have been collected, the risks would have been unacceptable.

BEHP – bis(2-ethylhexyl) phthalate

BHC – benzene hexachloride

COC – chemical of concern

cPAH – carcinogenic polycyclic aromatic hydrocarbon

HHRA – human health risk assessment

PCB – polychlorinated biphenyl

TBT – tributyltin

Underlined Xs signify risk drivers

Excess cancer risks for direct sediment contact RME scenarios for netfishing, clamming, and beach play were much lower than risk estimates for the seafood consumption scenarios. Most risks for beach play and clamming were associated with arsenic and cPAHs; for netfishing, most risks were associated with dioxins and furans. None of the direct sediment contact RME scenarios were estimated to result in non-cancer health risks.

11.4 RISK MANAGEMENT CONSIDERATIONS

This section summarizes key findings of the RI that are related to ongoing risk management decisions that will be made for the site. Key findings are presented for risk-based threshold concentrations and background concentrations of the risk drivers,

and the relationship between the spatial distribution of risk driver chemicals in sediment and LDW sediment dynamics.

11.4.1 RBTCs and background concentrations

Where appropriate, conclusions presented in the baseline risk assessments have been translated into sediment RBTCs that will be considered in the FS in combination with background concentrations and other factors. Sediment RBTCs were estimated for the four risk driver chemicals (total PCBs, arsenic, cPAHs, and dioxins and furans) that were identified for the direct sediment contact scenarios evaluated in the HHRA (Table 11-3). Sediment RBTCs for benthic invertebrates were set equal to SMS criteria.

Table 11-3. Comparison of sediment RBTCs and background concentrations for selected risk driver chemicals

Risk Driver Chemical	Unit	Risk Level	Sediment RBTC	Natural and Urban Background Sediment Concentration	
				Natural Background[a]	Urban Background[b]
Arsenic	mg/kg dw	1×10^{-6}	1.3 – 3.7[c]	12	**Upstream suspended solids:** 37 **Upstream sediment:** 6 – 9 **Urban bays:** 11
		1×10^{-5}	13 – 37[c]		
		1×10^{-4}	130 – 370[c]		
		HQ = 1	na		
		SQS	57		
		CSL	93		
cPAHs	µg/kg dw	1×10^{-6}	90 – 380[c]	77	**Upstream suspended solids:** nd **Upstream sediment:** 101 – 106 **Urban bays:** na
		1×10^{-5}	900 – 3,800[c]		
		1×10^{-4}	9,000 – 38,000[c]		
		HQ = 1	na		
Dioxin and furan TEQ	ng/kg dw	1×10^{-6}	13 – 37[c]	0.52	**Upstream suspended solids:** na **Upstream sediment:** 2.41 **Urban bays:** 14.9[d]
		1×10^{-5}	130 – 370[c]		
		1×10^{-4}	1,300 – 3,700[c]		
		HQ = 1	na		
Total PCBs	µg/kg dw	1×10^{-6}	500 – 1,700[c]	31	**Upstream suspended solids:** 60 **Upstream sediment:** 21 – 61 **Urban areas:** 135
		1×10^{-5}	5,000 – 17,000[c]		
		1×10^{-4}	50,000 – 170,000[c] 19 – 320[e]		
		HQ = 1	218 – 250[f]		
		SQS	130[g]		
		CSL	1,000[h]		

[a] The derivation of these preliminary natural background values is presented in Section 7.1. Final values, if any, needed for the FS will be developed in consultation with EPA and Ecology.

[b] The derivation of these urban background values is presented in Section 7.2. This table presents the range of mean concentrations (where more than one mean is available) from the studies examined in Section 7.2. The

upstream suspended solids concentrations were based on particulate concentrations in water normalized to TSS. The upstream sediment values include upstream surface sediment samples and Upper Turning Basin area sediment core samples, as described in Section 7.2.

[c] RBTCs developed from the human health direct sediment contact RME scenarios.

[d] Mean dioxin and furan TEQ concentrations from five locations (Elliott Bay [Terminal 91], Lake Union [I-5 bridge], Lake Washington [Bothell], Lake Washington [Bellevue], and Lake Washington [Renton]) sampled for the RI in 2005 (see Appendix B, Section B.5.5.2.1).

[e] RBTCs developed from the human health seafood consumption RME scenarios.

[f] RBTCs developed from the otter prey ingestion scenario, estimated through the FWM.

[g] Reported value is lowest AET, which is functionally equivalent to the SQS, but in dry weight units.

[h] Report value is second lowest AET, which is functionally equivalent to the CSL, but in dry weight units.

AET – apparent effects threshold
cPAH – carcinogenic polycyclic aromatic hydrocarbon
CSL – cleanup screening level
FWM – food web model
na – not available
nd – not detected
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
RBTC – risk-based threshold concentration
RME – reasonable maximum exposure
SQS – sediment quality standard
TEQ – toxicity equivalent quotient
TSS – total suspended solids

Information was compiled on the concentrations of four risk driver chemicals (total PCBs, arsenic, cPAHs, and dioxins and furans) in background areas. Background concentrations provide important context for RBTCs because EPA generally does not establish cleanup levels below background concentrations in consideration of the potential for recontamination from sources unrelated to the site, cost effectiveness, and technical practicability (EPA 2002d). Under Ecology's regulations, when area background concentrations would result in recontamination of the site to levels that exceed cleanup levels, that portion of the cleanup action that addresses cleanup below area background concentrations may be delayed until the offsite sources of hazardous substances are controlled. Also, Ecology will not set cleanup levels at concentrations below natural background concentrations.

Both natural and urban background concentrations will be considered in the FS and their role in determining cleanup levels will be determined by EPA and Ecology in the ROD. Preliminary natural background sediment concentrations for risk driver chemicals are presented in Section 7.1 and summarized in Table 11-3. Relevant data from urban environments that may be used to assess urban background conditions are presented in Section 7.2 and Table 11-3.

Sediment RBTCs associated with the 1×10^{-6} risk threshold for the direct sediment contact scenarios are above the preliminary natural background concentrations presented in Table 11-3 for total PCBs, cPAHs, and dioxins and furans. For cPAHs and dioxins and furans, these direct contact sediment RBTCs are also similar to urban background concentrations, while the direct contact sediment RBTC for total PCBs is above urban background concentrations. For arsenic, the sediment RBTCs associated with the 1×10^{-6} risk threshold for the direct sediment contact scenarios are well below both preliminary natural background and urban background concentrations.

RBTCs derived for the direct sediment contact scenarios should be applied to sediment concentrations averaged over the exposure areas identified in the HHRA, not on a point-by-point basis. The exposure area for the netfishing RME scenario included all of the intertidal and subtidal areas of the LDW. For the beach play RME scenario, eight separate intertidal areas were selected as exposure areas where children would be most likely to contact sediment (Map 6-1). The exposure area for the RME clamming scenario included all intertidal areas where clamming could occur because such areas are potentially accessible either from a boat or from shore (Map 6-2).

Sediment RBTCs for total PCBs were estimated for the RME seafood consumption scenarios and for river otter using a FWM calibrated to LDW environmental conditions and species. Sediment RBTCs estimated for seafood ingestion scenarios are also intended to be average concentrations over the relevant exposure area, and not applied on a point-by-point basis. Sediment RBTCs for cPAHs and arsenic associated with the human consumption of clams could not be derived because a clear relationship has not been established between concentrations of these chemicals in clams and in sediment. Sediment RBTCs for dioxins and furans were not derived because insufficient data were available.

A wide range of sediment RBTCs for total PCBs was derived for the seafood consumption scenarios (Table 8-6). Very few sediment RBTCs (and none of the sediment RBTCs for the RME scenarios) could be derived for the 1×10^{-6} risk threshold because the contribution from water alone led to estimated total PCB concentrations in tissue greater than this risk threshold, even in the absence of any contribution from sediment. The sediment RBTCs that could be derived for the 1×10^{-6} risk threshold were lower than the preliminary natural background concentration for total PCBs. At the 1×10^{-5} risk threshold, sediment RBTCs could not be derived for the RME seafood consumption scenarios, but sediment RBTCs for the CT scenarios and some of the one-meal-per-month scenarios could be calculated. Some of these sediment RBTCs (e.g., one meal per month for crabs and clams) were greater than the preliminary natural and urban background concentrations for total PCBs, while others (e.g., adult tribal CT, one meal per month of benthic fish) were within the range of urban background concentrations. At the 1×10^{-4} risk threshold, sediment RBTCs for total PCBs could be calculated for all of the seafood consumption scenarios. Except for the adult tribal RME scenario based on Tulalip data, all of the sediment RBTCs were greater than the preliminary natural and urban background concentrations, including the sediment RBTC for the API scenario at the 1×10^{-4} risk level.

Tissue RBTCs for total PCBs were also calculated for a given risk threshold and seafood consumption rate. For example, the total PCB concentration in seafood that would be necessary to achieve an excess cancer risk of 1×10^{-5} would be 4.2 µg/kg ww for the adult tribal RME scenario and 14 µg/kg ww for the API RME scenario. Just as it is important to put sediment RBTCs in the context of sediment background levels, it is important to put tissue RBTCs in the context of the PCB concentrations in other

common dietary food items. As described in Section 7.3.1 and in WSDOH (2006), PCBs are frequently detected in many different seafood species throughout Puget Sound at concentrations greater than these tissue RBTCs. These tissue RBTCs are also lower than the PCB concentrations that have been detected in some common food items from a typical diet (WSDOH 2006). Consumption of many of these common food items at the rates assumed under these RME scenarios would subject the consumer to risks that would be deemed unacceptable in the MTCA and CERLCA context. Tissue RBTCs for lower consumption rates, such as one meal per month, are much lower than tissue RBTCs for the RME scenarios. However, even a 1×10^{-6} tissue RBTC for the one-meal-per-month scenario (11 µg/kg ww) is low.

11.4.2 Implications of chemical patterns and trends

As presented in Section 4 and further discussed in Section 11.2.2, chemicals in LDW sediments exhibit distinct patterns and trends that are instructive in understanding future conditions. Chemical patterns that are of interest include spatial distributions in surface sediments, where exposure to ecological receptors and humans is most likely, and vertical distributions, which provides a record of the past and an indication of the rate of net sedimentation that has resulted in the burial of historical surface sediments. The presence of peak total PCB concentrations below the sediment surface at many different locations throughout the entire site is consistent with the independent lines of evidence that indicate that the LDW is largely a net depositional environment and that contamination associated with historical releases is being buried in most locations, with a few notable exceptions. Buried chemicals remain largely sequestered because the LDW is net depositional; areas in which net erosion can occur during high-flow events are limited to a few locations and to depths within the current biologically active zone (e.g., < 10 to 15 cm).

Despite the prevalence of peak total PCB concentrations at depth, elevated concentrations were also detected in surface sediments in distinct areas. The locations of these elevated concentrations were predominantly in areas where elevated concentrations were also detected at depth and historical contamination remains at the surface because of low net sedimentation rates in these areas. In other areas, the presence of elevated concentrations in surface sediments may be evidence of ongoing localized sources or localized disturbances. Focused sampling and source control evaluations at some of the EAAs confirm that ongoing sources are the most likely explanation for the presence of elevated concentrations at some of the EAAs.

Because multiple risk driver chemicals have been identified in sediments, each with its own set of sediment RBTCs, the spatial relationship among the chemicals will be an important element in the FS (Maps 4-60a, 4-60b, and 4-60c). Cleanup alternatives may be identified for individual chemicals or for multiple chemicals with overlapping areas of elevated concentrations. Such overlap occurs in most EAAs, but there are locations outside of EAAs with elevated concentrations of one or more risk drivers as well.

The STM and the physical CSM are useful for evaluating future exposure conditions, especially in areas outside the EAAs. Sediment particles will be continually transported into the LDW, predominantly from the Green River, which is the source of approximately 99% of the sediments to the LDW. Consequently, surface sediment quality in the LDW will resemble the quality of the particles originating from the Green River, assuming that local sources are sufficiently controlled and areas with elevated concentrations and low sediment deposition rates are remediated. Results from the STM model indicate that in 90% of the LDW, the existing surface sediment would be diluted by one-half within 10 yrs or less from burial by material flowing into the LDW from the Green River (Map 11-1). These results suggest that surface sediment quality will improve over much of the LDW. The significance of these results, and the scale at which the predictive model is valid for estimating recovery, will be evaluated in the FS.

The Slip 4 area is instructive when considering the roles both net deposition and local sources can have on the distribution of risk drivers. This area has been intensively sampled recently in preparation for a non-time critical removal action (Section 10.3.2.3). Slip 4 is within an area that has estimated net sedimentation rates between 1 and 3 cm/yr. Map 4-17 presents a time series of total PCB concentrations in surface sediment within Slip 4. There has been a marked decrease in surface sediment concentrations between 1990 and 2004 for much of the slip. This decrease suggests that chemical concentrations in depositing sediment (which includes sediment from upstream as well as from storm drains and CSOs) have decreased sufficiently to improve surface sediment quality during this period. The area of Slip 4 where elevated concentrations of total PCBs persist, even though at relatively lower concentrations, is in the head of the slip where there are active storm drain outfalls. Source identification studies in the storm drain systems have indicated ongoing sources of PCBs, which (until addressed) have the potential for limiting further improvements in sediment quality or recontaminating the slip after the removal action. This example illustrates the integrated temporal and spatial patterns that can be expected in depositional areas of the larger LDW.

The overall implication is that both active source control and remedial actions are likely to be required to reduce overall risks in the LDW. Focused investigations in the EAAs have highlighted the need to implement control measures for ongoing sources prior to remediation, while time trend data from Slip 4, for example, suggest that continuing inputs of sediment from the Green River will result in improvements of surface sediment quality over time, except where additional efforts at source control are required.

11.5 ONGOING SOURCE CONTROL EFFORTS

Controlling sources is critical to the long-term success of any remedial action (EPA 2002c). Consistent with this concept, Ecology (2007k) has developed a source control

strategy for the LDW, which it is currently implementing. This program is being conducted in parallel with the RI/FS and is expected to continue into the foreseeable future as remedial actions are undertaken, where needed, to reduce human health and environmental risks. Source identification and source control actions will need to precede remediation or these areas may become recontaminated.

Initial source control efforts have been focused on source identification at those EAAs that are being remediated as non-time critical removal actions. Ecology, along with the SCWG, is leading the source control evaluation and assessment of the EAAs and additional Tier Two Areas. As part of their source control evaluation process, summary of existing information and identification of data gaps reports and SCAPs are being prepared by Ecology to help direct future investigation and remediation efforts while assessing potential sources for the chemicals identified. These efforts will continue to evaluate potential sources to the LDW and implement corrective actions to reduce or eliminate identified sources.

In addition, current regulatory requirements have established inspection and compliance programs to monitor and control handling and disposal of manufacturing wastes produced by industrial operations along the LDW. These regulations have helped reduced sources of contamination that, in past practices, might have resulted in releases to the LDW. Permit programs are in place to help monitor and manage direct discharges to the LDW from CSOs, EOFs, and SDs in the basin. For example, since 1970, the City has reduced the annual CSO volume citywide by approximately 95%, and since 1988, the County has reduced their annual CSO volume by 63%.

Continued coordination of the CERCLA/MTCA cleanup actions and source control programs (both SCWG and other regulatory programs) will ensure that future remedial actions will not be unduly impacted by local sources.

11.5 SUMMARY OF KEY FINDINGS

Key findings for the RI are summarized below.

- Over the past 100 years, the LDW has been highly modified from its natural configuration to support urban and industrial development. Changes have included reductions and control of water flow, significant shoreline modifications, loss of intertidal habitat, and installation of riprap, pier aprons and sheet pile walls. Some limited areas of natural shoreline still exist within the LDW.
- Industrial and commercial facilities occupy most of the shoreline; one residential community is also located along the shoreline with another community nearby.
- The LDW is currently used primarily as an industrial navigational corridor. Secondarily, it supports recreational uses such as boating, kayaking, fishing and

beach play. It also supports the Muckleshoot Tribe's commercial fishery for salmon.

- Despite significant alterations in habitat and areas with elevated chemical concentrations, the LDW contains a diverse assemblage of aquatic and wildlife species and a robust food web that includes top predators.
- Much of the sediment contamination in the LDW comes from historical releases that are now generally buried under cleaner, more recently deposited sediment. The highest levels of chemical contamination are generally detected at depth.
- In general, high concentrations of chemicals, including PCBs, were detected in surface sediment in localized areas separated by larger areas of the LDW with relatively lower concentrations. Relatively high surface sediment contamination is present in some areas as a result of a number of processes, including low net sedimentation rates in a few areas with primarily historical contamination or because of the presence of ongoing, localized sources.
- The highest risks to people are associated with consumption of fish, crabs, and clams, with lower risks associated with activities that involve direct contact with sediment, such as clamming, beach play, and netfishing.
- Most of the human health risk is from PCBs, arsenic, cPAHs, and dioxins and furans.
- Ecological risks to fish and wildlife were relatively low, with the exception of risks to river otter from PCBs.
- Sediment contamination in approximately 75% of the LDW is estimated to have no effect on the benthic invertebrate community; approximately 7% of the surface sediment has chemical concentrations exceeding the higher of the two state standards, associated with potential adverse effects to the benthic invertebrate community. The potential for effects in the remaining 18% of the LDW is more uncertain. Most of the state sediment standard exceedances were for PCBs and phthalates, although 41 different chemicals had at least one exceedance.
- Sediment is depositing within the LDW, resulting in the burial of historical contamination over much of the area. Almost all new sediment (99%) that enters the LDW originates in the Green River.
- A few areas in the LDW may be scoured during high flow events, such as storms. Based on the sediment transport model, the scour depth is relatively shallow and is generally limited to sediment in the biologically active zone; thus, deeper sediments would not be exposed from high flow events. Tug maneuvering is a potential source of localized erosion that could disturb sediment at greater depths in relatively small areas. Routine boat traffic is expected to only mix the

top few cm of sediment, which is part of the biologically active zone mixed by benthic invertebrates.

- The site conceptual model has been confirmed by several lines of evidence, including chemistry profiles in sediment. The depths of most peak PCB concentrations were consistent with the expected rate of burial, with a few exceptions that may be related to localized influences.
- Based on this understanding of sediment transport in the LDW, LDW surface sediment is gradually becoming more similar in character to the sediment being transported by the Green River and will thus become cleaner with time.
- Assuming local sources are controlled and hotspots are remediated, chemical concentrations in surface sediment are estimated to be reduced by up to 50% every 5 to 10 years from transport of cleaner sediment from the Green River.
- A comparison of risk-based sediment concentrations with background concentrations will be important in risk management decisions by EPA and Ecology because cleanup levels cannot be established or maintained below background concentrations because of the potential for recontamination from sources not related to the site.
- Most of the key risk-based goals for total PCB concentrations in sediment associated with the human consumption of seafood are below background concentrations in reference areas in Puget Sound. The only risk-based goal for an RME seafood ingestion scenario that is higher than urban or natural background concentrations is the 1×10^{-4} risk level for the API RME scenario. The risk-based goal for otter is approximately the same as the 1×10^{-4} API RME risk-based goal.
- Risk-based goals for seafood consumption for arsenic, cPAHs, and dioxins and furans could not be calculated because the relationships between concentrations in clam tissue and sediment are complex and involve important variables that may never be well understood.
- The risk-based goals for direct sediment contact for arsenic at the 1×10^{-6} risk level were lower than the natural background concentration.
- The risk-based goals at the 1×10^{-6} risk level for direct sediment contact via netfishing and clamming for cPAHs and dioxins and furans were higher than the natural background concentrations and were similar to the ranges of urban background concentrations.
- The risk-based goals at the 1×10^{-6} risk for direct sediment contact related to beach play for cPAHs were similar to natural background concentrations and lower than typical urban background concentrations.

- The risk-based goals for direct sediment contact for total PCBs were higher than both the natural background concentration and the range of urban background concentrations.
- The control of local sources is critical to the long-term success of specific remedial actions in the LDW. Ecology, as lead agency of a consortium of regulatory authorities, has begun implementing a source control program in parallel to the RI/FS process.
- Continued coordination of cleanup actions and source control will be necessary to ensure any actions taken are not unduly impacted by local sources.

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