Lower Duwamish Waterway Group

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

Lower Duwamish Waterway Remedial Investigation

PHASE 2 REMEDIAL INVESTIGATION REPORT

APPENDIX A:

BASELINE ECOLOGICAL RISK ASSESSMENT FINAL

For submittal to:

The US Environmental Protection Agency
Region 10
Seattle, WA

The Washington State Department of Ecology
Northwest Regional Office
Bellevue, WA

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Acronyms

Acronym	Definition
AET	apparent effects threshold
Ah	aryl hydrocarbon
ANOVA	analysis of variance
AOC	Administrative Order on Consent
ATSDR	Agency for Toxic Substance and Disease Registry
AWQC	ambient water quality criteria
BCA	benthic community analysis
BEHP	bis(2-ethylhexyl) phthalate
BMF	biomagnification factor
bw	body weight
CERCLA	Comprehensive Environmental Response, Compensation, and Liability Act

Acronym	Definition
COC	chemical of concern
COI	chemical of interest
COPC	chemical of potential concern
CSL	cleanup screening level of SMS
CSM	conceptual site model
CSO	combined sewer overflow
CSO/SD	combined sewer overflow/storm drain
DDTs	DDT and its metabolites
DMMP	Dredged Material Management Program
DNA	deoxyribonucleic acid
dw	dry weight
EC50	concentration that causes a non-lethal effect in 50% of an exposed population
Ecology	Washington State Department of Ecology
EPA	US Environmental Protection Agency
ERA	ecological risk assessment
ERED	Environmental Residue Effects Database
EROD	ethoxyresorufin-O-deethylase
ESA	Endangered Species Act
FAV	final acute value
FCV	final chronic value
FIR	food ingestion rate
FMR	free-living metabolic rate
FS	feasibility study
GWI	Great Western International
HHRA	human health risk assessment
HPAH	high-molecular-weight polycyclic aromatic hydrocarbon
HQ	hazard quotient
IDW	inverse distance weighting
IP	intraperitoneal
IR	ingestion rate
kcal	kilocalories
LAET	lowest apparent effects threshold
2LAET	second lowest apparent effects threshold
LC50	concentration that causes the death of 50% of a group of test animals

Acronym	Definition
LC100	concentration that causes the death of 100% of a group of test animals
LD50	dose that causes the death of 50% of a group of test animals
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
MHHW	mean higher high water
ML	maximum level of DMMP
MLLW	mean lower low water
MTCA	Model Toxics Control Act
NMFS	National Marine Fisheries Service
NOAEL	no-observed-adverse-effect level
NOEC	no-observed-effect concentration
NPL	National Priorities List
OC	organic carbon
PAH	polycyclic aromatic hydrocarbon
PCB	polychlorinated biphenyl
ppt	parts per thousand
PSAMP	Puget Sound Ambient Monitoring Program
PSDDA	Puget Sound Dredged Disposal Analysis
PSEP	Puget Sound Estuary Program
QAPP	quality assurance project plan
QA/QC	quality assurance/quality control
QSAR	quantitative structure-activity relationship
RCRA	Resource Conservation and Recovery Act
RGS	reporter gene system
RFI	RCRA facility investigation
RI	remedial investigation
RI/FS	remedial investigation/feasibility study
RL	reporting limit
RM	river mile
ROC	receptor of concern
RPF	relative potency factor

Acronym	Definition
RPS	relative penis size
SD	standard deviation
SL	screening level of DMMP
SMS	Washington State Sediment Management Standards
SQG	sediment quality guideline
SQS	sediment quality standards of SMS
SUF	site use factor
SVOC	semivolatile organic compound
SWAC	spatially weighted average concentration
T&E	threatened and endangered
TBT	tributyltin
TEC	toxic equivalence concentration
TEF	toxic equivalency factor
TCDD	tetrachlorodibenzo-*p*-dioxin
TEQ	toxic equivalent
TOC	total organic carbon
TRV	toxicity reference value
UCL	upper confidence limit
USACE	US Army Corps of Engineers
USGS	US Geological Survey
VOC	volatile organic compound
WAC	Washington Administrative Code
WDFW	Washington State Department of Fish and Wildlife
WERF	Water Environment Research Foundation
WHO	World Health Organization
WIR	water ingestion rate
WQA	water quality assessment
ww	wet weight

Executive Summary

This document presents the Phase 2 (baseline) Ecological Risk Assessment (ERA) for the Lower Duwamish Waterway (LDW), as outlined in the work plan for the Phase 2 Remedial Investigation (RI) (Windward 2004e). Baseline risk assessments, as defined in EPA (1988) guidance, "provide an evaluation of the potential threat to human health and the environment in the absence of any remedial action. They provide the basis for determining whether or not remedial action is necessary and the justification for performing remedial actions." The Phase 2 RI is being conducted by the Lower Duwamish Waterway Group (LDWG) under the oversight of the US Environmental Protection Agency (EPA) and the Washington State Department of Ecology (Ecology). These parties agreed in an Administrative Order on Consent (AOC) to conduct the RI in accordance with applicable Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) and Model Toxics Control Act (MTCA) guidance, policies, and procedures. Compliance with MTCA includes compliance with the Washington State Sediment Management Standards (SMS) (Washington Administrative Code [WAC] 173-204).

The baseline ERA presents risk estimates for benthic invertebrate, fish, and wildlife species that may be exposed to chemicals of potential concern (COPCs) found in sediment, water, and aquatic biota from the LDW. The dataset used in the baseline ERA consists of historical data and sediment and tissue chemistry data collected from the LDW during Phase 2 to supplement the historical data that were used in the Phase 1 ERA (Windward 2003a). The baseline ERA consists of separate sections on problem formulation, exposure assessment, effects assessment, risk characterization, and uncertainty analysis, each of which is briefly summarized below.

ES.1 Problem Formulation

The problem formulation of the ERA establishes the overall scope of the assessment. Because it is impractical to evaluate risks to 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 than other wading birds. In addition, risks to some species were analyzed because those species are highly valued by society, such as endangered or threatened species.

Representative receptors of concern (ROCs) selected for this Phase 2 ERA were the benthic invertebrate community, crabs, English sole, Pacific staghorn sculpin, great blue heron, spotted sandpiper, osprey, river otter, and harbor seal. In addition, juvenile chinook salmon was selected as an ROC because they are a federally protected species that use the LDW during outmigration to Puget Sound.

The problem formulation includes 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 allows the risk assessment to focus on COPCs and eliminate chemicals that do not pose risks to ecological receptors.

Data used in the ERA consisted largely of:

- Surface sediment (uppermost 0 to 15 cm) chemistry data
- Site-specific sediment toxicity test results
- Sediment porewater 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

For each ROC selected, COPCs were identified through a conservative risk-based screening process. COPCs identified included: 45 chemicals (including tributyltin [TBT], metals, polychlorinated biphenyls [PCBs] and other organic compounds) for benthic invertebrate communities; 2 chemicals (total PCBs and zinc) for crabs; 6 chemicals (arsenic, cadmium, copper, total PCBs, TBT, and vanadium) for at least one fish ROC, and 12 chemicals (arsenic, cadmium, chromium, cobalt, copper, lead, mercury, nickel, selenium, total PCBs, zinc, and vanadium) for at least one wildlife ROC. COPCs that were never detected in tissue because reporting limits (RLs) were above the screening criteria and COPCs without effect-level toxicity information were evaluated in the uncertainty analysis. In addition, organochlorine pesticides in some sediment samples and 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 2005b, c). Therefore, risks from organochlorine pesticides were also assessed in the uncertainty analysis.

The problem formulation also presents the conceptual site models for the ROCs. 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 sediment exposure and indirect exposure through the ingestion of prey or water from the LDW. Exposures associated with direct contact with water in the LDW were based on water data and risk estimates from the King County water quality assessment ERA (King County 1999b) and more recent water sampling for PCBs.

Finally, assessment and measurement endpoints were identified in the problem formulation. Survival, growth, and reproduction were the key endpoints under review for most ROCs in this assessment. 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 representative ROCs, COPCs, pathways, and endpoints formed the scope for the Phase 2 ERA. Uncertainties associated with these analyses are discussed in the uncertainty analysis.

ES.2 EXPOSURE ASSESSMENT

The exposure assessment estimates the potential exposure of each ROC to the sediment-associated COPCs identified in the problem formulation. Exposure of benthic invertebrates to COPCs was assessed primarily by evaluating the distribution, concentration, and co-occurrence of COPCs in surface sediment, with the exception of risks to crabs and risks from sediment-associated TBT, which were both assessed using a critical tissue-residue approach. Risks to gastropods from exposure to TBT were also evaluated in two field studies of the most sensitive endpoint for these benthic invertebrates. In addition, risks to benthic invertebrates from exposure to volatile organic compounds (VOCs) were assessed using sediment porewater data.

Exposure of fish to COPCs was characterized based either on COPC concentrations in fish tissue or on COPC concentrations in likely fish prey. For wildlife ROCs, the exposure assessment identified equations and parameters to quantify the ingested dose of COPCs. Dietary doses for wildlife were estimated using available information on ROC biology and life histories, including body weight, feeding behavior, site usage, and diet.

ES.3 EFFECTS ASSESSMENT

Potential adverse effects (i.e., reduced survival, reduced growth, or impaired reproduction) were identified in the effects assessment. For the benthic invertebrate community, direct measures of sediment toxicity provided by site-specific sediment toxicity tests were given primary consideration over comparisons of sediment chemistry to chemical criteria. For locations without site-specific toxicity information, chemical criteria provided by the SMS were used to set benthic invertebrate effects levels for most of the COPCs. For COPCs without chemical-specific sediment criteria, toxicologically based guidelines from the Dredged Material Management Program (DMMP) or toxicity information from the literature was used. For gastropods, an invertebrate group highly sensitive to TBT, a direct, site-specific assessment of effects was also given primary consideration. For assessment of the effects on benthic invertebrates of VOCs in porewater, toxicity data from the literature were used.

For crabs, fish, and wildlife, a detailed evaluation was conducted of studies in the scientific literature documenting effects of COPCs on the ROCs or similar species. This literature review identified COPC concentrations in receptor tissue or doses associated with no effects (i.e., safe concentrations or doses), in addition to concentrations or doses indicating a threshold of adverse effects. Both sets (i.e., no-observed-adverse-effect level [NOAEL] and lowest-observed-adverse-effect level [LOAEL]) of toxicity reference values (TRVs) are summarized in tables, and the rationale for TRV selection is provided.

ES.4 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 the ROCs. This analysis identified the following conclusions.

- **Benthic Invertebrate Community —** The goal of the SMS is to reduce and ultimately eliminate adverse effects on biological resources (WAC 173-204-100). Sediment chemistry and site-specific toxicity test results indicate that no adverse effects to benthic invertebrates living in intertidal and subtidal sediments are predicted for 75% of the LDW area (i.e., the area in which chemical concentrations were less than or equal to chemical sediment quality standard [SQS] criteria and where sediments were nontoxic according to biological SQS criteria). 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. 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 are less certain in these areas than in areas with concentrations greater than one or more CSL values. Some uncertainty is associated with these area estimates because areas were estimated by interpolating from individual points at which sediments were sampled. The SQS and CSL criteria were exceeded by 39 chemicals; 2 additional chemicals exceeded only the SQS criteria.[1] Risks to the benthic invertebrate community from all VOCs detected in sediment porewater were very low, except for cis-1,2-dichloroethene, which had concentrations 21 times the no-effects concentration in a small area at RM 2.4; all concentrations of cis-1,2-dichloroethene were less than the concentration associated with adverse effects. Therefore, there is uncertainty whether exposure to cis-1,2-dichloroethene within the LDW is sufficiently high to result in adverse effects in this small area. Risks to benthic invertebrates from TBT were very low based on NOAEL-based hazard quotients (HQs) less than 1.0 and the absence

[1] Total DDTs, nickel, and total chlordane also exceeded their DMMP guidelines or literature-based TRVs at one or more locations.

of imposex in all gastropods, except one neogastropod with imposex characterized as Stage 2, a stage that is not expected to impact reproduction.

- **Crabs —** Exposure concentrations of total PCBs in tissue were equal to concentrations associated with adverse effects in crabs, indicating the potential for adverse effects. Exposure concentrations of zinc in tissue were greater than concentrations associated with no effects but less than those associated with adverse effects, indicating there is uncertainty whether exposure within the LDW is sufficiently high to result in adverse effects.
- **Fish —** Exposure concentrations for three of the six COPCs for fish (PCBs, cadmium, and vanadium) were greater than concentrations associated with adverse effects for English sole. LOAEL-based hazard quotients (HQs[2]) for cadmium and vanadium were both 1.2, and LOAEL-based HQs for PCBs ranged from 0.98 to 5.0 based on effects concentrations in the study reporting the lowest TRVs. Therefore, there is a potential for adverse effects from PCBs, but risk estimates are uncertain because the exposure concentration was in between the concentrations selected as LOAELs. Estimated exposures of English sole to two additional COPCs (arsenic and copper) were greater than their respective no-effects levels but were lower than the adverse effect levels associated with survival, growth, or reproduction, indicating that there is uncertainty whether exposure within the LDW is sufficiently high to result in adverse effects. Site-specific studies of English sole indicate the potential for reproductive effects that correlate with exposure to chemical mixtures in the field. However, the relationship of such effects to specific chemicals has not been established. Exposure concentrations of PCBs, cadmium, and vanadium for Pacific staghorn sculpin were equal to or greater than the concentrations associated with adverse effects in at least one area within the LDW. LOAEL-based HQs up to 1.0 and 1.2 were estimated for cadmium and vanadium, respectively, indicating a potential for adverse effects. LOAEL-based HQs for PCBs ranged from 0.30 to 3.8 based on effects concentrations in the study reporting the lowest TRVs. Therefore, there is a potential for adverse effects from PCBs, but risk estimates are uncertain because the exposure concentrations were in between the concentrations selected as the LOAEL range. The exposure concentrations of TBT and copper were greater than their respective no-effects concentrations for Pacific staghorn sculpin in at least one area within the LDW but less than those associated with adverse effects. Thus, the potential for adverse effects is uncertain. For juvenile chinook salmon, exposure concentrations of cadmium were greater than concentrations associated with adverse effects in any fish species but lower than concentrations

[2] The HQ is the ratio of the exposure concentration (or dose) to a concentration (or dose) associated with adverse effects.

associated with adverse effects in salmonids. Exposure concentrations of arsenic, copper, and vanadium in the diet of juvenile chinook salmon were greater than their respective no-effect concentrations but less than concentrations associated with adverse effects.

- **Birds** – Estimated exposures of spotted sandpiper to six of the 12 COPCs (copper, chromium, lead, mercury, PCB toxic equivalent (TEQ),[3] and vanadium) for spotted sandpiper were greater than the dietary doses associated with adverse effects on survival, growth, or reproduction in at least one area within the LDW (LOAEL-based HQs of up to 1.1, 1.8, 5.5, 1.0, 1.5, and 1.4, respectively). Therefore, there is a potential for adverse effects from these COPCs. Estimated doses to great blue heron of all four COPCs (chromium, lead, mercury, and total PCBs) were less than no-effects levels, indicating very low risk. For osprey, estimated doses of PCBs were greater than no-effect levels for osprey using a TEQ approach but less than those levels using a total PCBs approach; the latter risk estimate is less uncertain. Therefore, the potential for adverse effects from PCBs is uncertain for osprey. Estimated doses of the remaining three COPCs to osprey (chromium, lead, and mercury) were less than the doses associated with no-effects, indicating very low risk.
- **Mammals –** Estimated dietary doses of total PCBs were greater than doses associated with adverse effects for river otters, with a LOAEL-based HQ of 2.9. Estimated exposure of river otters to mercury was greater than a no-effects level but was less than adverse effects levels associated with survival, growth, or reproduction, indicating that the potential for effects is uncertain. Exposures of otter to the remaining three COPCs (arsenic, cobalt, and selenium) and exposures of harbor seals to both COPCs (mercury and total PCBs) were less than their respective no-effects levels, indicating very low risk.

Table ES-1 provides a summary of COPCs for crabs, fish, or wildlife for which either the NOAEL-based HQ was greater than 1.0 or the LOAEL-based HQ was greater than or equal to 1.0. Table ES-2 lists the COPCs for benthic invertebrates that exceeded SMS criteria, DMMP guidelines, or TRVs. In summary, risk estimates for PCB exposure indicated a potential for adverse effects to the benthic invertebrate community, crabs, English sole, Pacific staghorn sculpin, spotted sandpiper, and river otter. There is also a potential for adverse effects to osprey from PCB exposure, but risk estimates for this ROC are more uncertain because exposures were greater than no-effect levels but less than levels associated with adverse effects. Other COPCs with exposures greater than levels associated with adverse effects were cadmium, chromium, copper, lead, mercury, and vanadium. Numerous additional chemicals may pose a risk to the benthic invertebrate community as shown in Table ES-2.

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[3] EPA refers to this as the toxic equivalence concentration or TEC.

Table ES-1. COPCs and ROCs with HQs ≥ 1.0

COPC	ROC	NOAEL HQ	LOAEL HQ
COPCs 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**
	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.

[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 section. These calculations were presented only in the uncertainty section because of analytical interferences from PCB Aroclors in the pesticide analyses of LDW tissue samples, resulting in uncertainties in pesticide identification and a high bias in pesticide concentrations.

[b] The NOAEL-based HQs were ≥1 for the following COPC/ROC pairs based on risk calculations discussed in the uncertainty section: 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 crab (3.6). These calculations were presented in the uncertainty section because of analytical interferences from PCB Aroclors in the 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.

Table ES-2. 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
Bis(2-ethylhexyl) phthalate	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
Butyl benzyl phthalate	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

COPC	Number of Detected Concentrations > SQS and ≤ CSL	Number of Detected Concentrations > CSL
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
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; 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.

COPC – chemical of potential concern

CSL – cleanup screening level

HPAH – high-molecular-weight polycyclic aromatic hydrocarbon

LPAH – low-molecular-weight polycyclic aromatic hydrocarbon

PCB – polychlorinated biphenyl

SQS – sediment quality standards

Based on the risk estimates, uncertainties discussed in this ERA, natural background concentrations, and residual risks following early actions in the LDW, chemicals were identified as risk drivers for ecological receptors in accordance with EPA (1998) and MTCA (WAC 173-340-703) guidance. The risk drivers from both this ERA and the human health risk assessment (HHRA) will be the focus of remedial analyses in the feasibility study (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 Washington SMS in one or more locations.

Other COCs, 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.

No quantitative ecological risk estimates were calculated for dioxins and furans and thus the level of ecological risk from dioxins and furans is unknown. Ecological risks associated with exposure to dioxins and furans within the LDW were not assessed for several reasons. Primarily, human health risks from dioxins and furans through seafood consumption were assumed to be unacceptable, and therefore, neither tissues from the LDW nor from background areas were analyzed for dioxins and furans. Dioxins and furans were determined to be a risk driver based on human health risks from both seafood consumption and direct contact pathways. Risk management decisions to address dioxin and furan contamination in LDW sediment will be based on MTCA and CERCLA regulations and guidance. Remedial decisions to address dioxin and furan contamination in sediment will be made by EPA and Ecology as part of the FS process and will be documented in the Record of Decision. Additional detail on dioxins and furans is provided in Section B.5.5.2 of the HHRA.

This ERA is based on the baseline surface sediment dataset, which includes 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 storm drain in the Norfolk area). Therefore, the risks discussed in this ERA may represent an overestimate of current risks for areas where remediation has already occurred.

A.1.0 Introduction

This document presents the baseline ecological risk assessment (ERA) as part of the remedial investigation and feasibility study (RI/FS) for the Lower Duwamish Waterway (LDW). The LDW was added to the US Environmental Protection Agency's (EPA's) National Priorities List (NPL) under the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA), also known as Superfund, on September 13, 2001. The LDW was added to Ecology's Hazardous Sites List on February 26, 2002.

Under Superfund regulations, EPA requires that an RI/FS be conducted for all listed sites. An RI evaluates the nature and extent of chemical contamination, estimates baseline human health and ecological risks, and is used by risk managers to identify areas that should be remediated because they pose an unacceptable risk to human health or the environment. An FS proposes a number of alternative approaches to remediating the areas with unacceptable risk, and analyzes and compares these alternatives. The study area is shown in Map A.1-1.

The key parties involved in the LDW RI/FS are the City of Seattle, King County, the Port of Seattle, and The Boeing Company, working together for this project as the Lower Duwamish Waterway Group (LDWG), under the oversight of EPA and the Washington State Department of Ecology (Ecology). These parties agreed (in an Administrative Order on Consent, or AOC) to conduct the RI for the LDW in two phases. The first phase is complete, including a Phase 1 ERA based on data that existed at that time (Windward 2003c). The AOC requires that the RI be conducted in accordance with applicable CERCLA and MTCA guidance, policies, and procedures. Compliance with MTCA includes compliance with the Washington State Sediment Management Standards (SMS).

This document presents the Phase 2 (baseline) ERA as outlined in the Phase 2 RI work plan (Windward 2004e). Baseline risk assessments, as defined by EPA (1988) guidance for conducting an RI/FS, "provide an evaluation of the potential threat to human health and the environment in the absence of any remedial action. They provide the basis for determining whether or not remedial action is necessary and the justification for performing remedial actions." This baseline ERA presents risk estimates for ecological receptors that may come in contact with sediment–associated chemicals of potential concern (COPCs) through exposure to or ingestion of sediment, water, fish, and invertebrates (e.g., polychaete worms, clams, crabs) in the LDW.

This baseline ERA is based on data previously summarized in the Phase 1 ERA (Windward 2003b) and data collected since the Phase 1 RI was completed. It was developed in accordance with both national and regional EPA guidance (EPA 1992, 1997a, b, 1998). This baseline ERA includes the following sections:

- Section A.2.0 – Problem Formulation

- Section A.3.0 – Exposure and Effects Assessment: Benthic Invertebrates
- Section A.4.0 – Exposure and Effects Assessment: Fish
- Section A.5.0 – Exposure and Effects Assessment: Wildlife
- Section A.6.0 – Risk Characterization and Uncertainty Analysis
- Section A.7.0 – Selection of Primary Ecological Risk Drivers/Indicator Hazardous Substances
- Section A.8.0 - Conclusions
- Section A.9.0 – References

Details on site background, previous investigations, and environmental setting are provided in the Phase 2 RI and referenced accordingly. Table A.1-1 presents the sections in which the primary components of the risk assessment process are discussed for each receptor type.

Table A.1-1. Document organization for primary ERA components

ERA Component	ERA Section Number		
	Benthic Invertebrates	Fish	Wildlife
ROC selection	A.2.3.1	A.2.3.2	A.2.3.3
COPC selection	A.2.5.1	A.2.5.2	A.2.5.3
Conceptual site model	A.2.6	A.2.6	A.2.6
Exposure assessment	A.3.1 (benthic invertebrate community) A.3.3 (crabs)	A.4.1	A.5.1
Effects assessment	A.3.2 (benthic invertebrate community) A.3.4 (crabs)	A.4.2	A.5.2
Risk characterization and uncertainty analysis	A.6.1	A.6.2	A.6.3

[a] Available data used in the ERA are discussed in Section A.2.4.1, and exposure concentration calculations are presented in Attachment 11.

A.2.0 Problem Formulation

This section presents the problem formulation for this baseline ERA. Through the use of a risk-based screening approach, the problem formulation establishes which receptor of concern (ROC) and COPC pairs are further evaluated in the exposure and effects assessment, the risk characterization, and the uncertainty analysis. This section includes information regarding the environmental setting, ecological resources that use the site, selection of ROCs, a summary of relevant available data collected from the LDW, a COPC screen for ROCs, and the conceptual site model (CSM) for the LDW. Together, these elements establish the scope for this ERA.

A.2.1 Environmental Setting

This section presents an overview of the site's setting, hydrology, and habitat. Additional information regarding the setting, history, hydrology, and sediment regime of the LDW is presented in the Phase 2 RI.

A.2.1.1 Site description

The confluence of the Black and Green Rivers forms the Duwamish River 10.5 miles upstream from the southern end of Harbor Island. The LDW consists of the downstream portion of the Duwamish River, excluding the East and West Waterways around Harbor Island, and 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. The LDW is tidally influenced over its entire length, with the degree of tidal influence varying depending on stream flow and on tide stage at the mouth of the LDW. The mean tide range is 7.5 ft, and the mean diurnal range is 11 ft. Recorded tides have ranged from -4.6 to 14.7 ft mean lower low water (MLLW). Annual river discharge ranges from 43 to 51 m^3/sec (2,300 to 2,350 cfs) (NOAA 1998).

The US Army Corps of Engineers (USACE) maintains the LDW as a navigable waterway through dredging (Dexter et al. 1981). The typical cross section of the LDW includes a deeper, maintained navigation channel at the middle of the waterway, with intermittent shallow benches along the margins of the channel. The navigation channel is maintained throughout the study area, with typical depths ranging from greater than -30 ft MLLW downstream of RM 2.0 to less than -15 ft MLLW near the Upper Turning Basin. Shallower bench areas exist in the nearshore intertidal zones outside of the navigation channel, with variable dimensions and elevations. The width of the LDW is relatively uniform, ranging between about 500 and 700 ft. The navigation channel is approximately 200 ft wide.

The LDW is a highly modified estuary that has been straightened and dredged in the lower 4.8 miles to facilitate navigation and industrial development. The only remnant of the river's natural meanders exists west of Kellogg Island. The banks of the LDW are dominated by structures, including piers and buildings. Where they are not occupied by structures, the banks are typically armored with a combination of riprap, concrete debris, and other structures for stabilization of banks, especially in mid to upper tidal elevations of the bank. Industrial land use dominates on the east bank in the immediate vicinity of the LDW. The west bank includes industrial, commercial, and mixed residential land uses in the vicinity of the LDW.

Exceptions to the industrialized condition of the LDW include the area around Kellogg Island, which is partially formed by a remnant meander of the natural Duwamish River channel, and some areas of mixed commercial, recreational, and residential uses within both the upstream and downstream areas of the LDW.

The LDW is a stratified saltwater wedge estuary (Stoner 1972). Circulation within a stratified estuary results from a net upstream movement of water within a bottom-layer saltwater wedge and a net downstream movement of fresher water in the layer that overrides the wedge. Recent average annual discharges from the Duwamish River have ranged between 43 and 51 m^3/s (2,300 to 2,350 cfs), as measured at the US Geological Survey (USGS) Tukwila gaging station, located at USGS-designated RM 12.4. Flow rates at the Auburn gaging station, located at USGS-designated RM 32.0, ranged from 4.3 to 329 m^3/s (200 to 15,200 cfs, the record high) between 1962 and 1994 (NOAA 1998). The saltwater wedge, which has its source in Elliott Bay, oscillates upstream and downstream with the tide and stream flow. At freshwater inflow greater than 1,000 cfs, the saltwater wedge does not extend upstream beyond the East Marginal Way Bridge (RM 6.3), regardless of the tide height (Stoner 1967). During periods of low freshwater inflow and high-tide stage, the saltwater wedge has extended as far upstream as the Foster Bridge at RM 8.7 (Stoner 1967).

Because of their circulation, estuaries naturally act as sediment traps for incoming sediment. Sediment from freshwater sources is transported into the estuary at the upstream end while sediment from coastal waters is transported into the estuary via the saltwater wedge. The presence of the man-made channel increases the cross-sectional area of the estuary relative to natural conditions, which results in a decrease in current velocities and an increase in deposition. Additional information related to LDW sediment stability and erosion potential is presented in the Phase 2 RI.

A.2.1.2 Habitat features

Benthic habitats within the LDW include intertidal habitat (exposed by low tides) and subtidal habitat (never exposed by low tides) (Table A.2-1). A typical cross section of the LDW includes intertidal habitats, subtidal transition areas (often with steep slopes), and a deeper navigation channel in the center of the LDW. Sediment characteristics (i.e., grain size and organic carbon [OC] content) vary throughout the LDW, depending on several physical parameters such as currents, quiescent areas (e.g., slips), and sediment sources (i.e., creeks and outfalls).

Table A.2-1. Habitat types represented in the LDW

Habitat Type	Description	Influential Physical Parameters	Extent and Condition in LDW
Upland	area outside the immediate influence of wetted area of the LDW (below +14 ft MLLW), many uplands consist of fill material	temperature, precipitation, soil type, groundwater elevation	dominated by industrial uses; some mixed-use residential, commercial, and recreational areas
Intertidal marshes	Intertidal area 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 acres in LDW), high and low marsh on Kellogg Island (Blomberg et al. 1988)
Intertidal mudflats	Intertidal area 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 acres 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)	salinity gradients, wave action, water temperature, oxygen content	riprap-covered areas comprise approximately 17% of intertidal areas in LDW (USFWS 2000b)
Subtidal	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, and OC content, water depth, temperature	throughout LDW, including navigation channel

LDW – Lower Duwamish Waterway

MLLW – mean lower low water

OC – organic carbon

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; but 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). However, because of the shoreline armoring, these intertidal flats 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 flats (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).

Sections of natural shoreline in the LDW occur only upstream of the Upper Turning Basin (Tanner 1991). Most (98%) of approximately 510 hectares (ha) (1,270 ac) of tidal marsh and 590 ha (1,450 ac) of flats and shallows and all of about 500 ha (1,230 ac) of tidal wetland have been either filled or dredged (Blomberg et al. 1988) or altered by the hydrologic changes resulting from the channelization of the estuary and maintenance of the navigation channel. Remnant tidal marshes account for only 2 ha (5 ac), and mudflats account for 12 to 22 ha (30 to 54 ac) (Leon 1980).

Remnants of natural intertidal habitat are present in occasional patches throughout the LDW; the largest remnant of intertidal habitat surrounds Kellogg Island, located south of Harbor Island. Kellogg Island has been highly altered from its historical shape and function. In the 1950s and 1960s, it was filled with hydraulic dredge spoils by USACE. Later, in 1974, an upland component of Kellogg Island was created when the Port of Seattle deposited 1,700 m^3 (2,200 yd^3) of dredged materials on the south end of the island (Sato 1997). A mixture of introduced and native plant and tree species rapidly colonized the 7-ha island. Current habitat associated with the island includes high and low marsh, intertidal mudflats, and filled uplands (Canning et al. 1979). The island, including intertidal area near Kellogg Island, has been designated as a wildlife refuge (Hotchkiss 2006).

Subtidal habitat is variable throughout the LDW. Subtidal sediment composition ranges from sand to mud, depending on the sediment source and current speed (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; whereas 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.

Few surveys have investigated the vascular plant communities present in the LDW (Tanner 1991; Canning et al. 1979; USFWS 2000a; Cordell et al. 2001). The methods used to assess plant communities have ranged from analysis of aerial photos to field surveys. Many of these surveys were conducted to investigate habitat availability in the LDW and mainly addressed the plant communities of tidal marsh areas.

In estuaries in general, tidal elevation and salinity gradients determine the potential distribution of estuarine plants. Intertidal elevation gradients between MLLW and mean higher high water (MHHW) create habitats such as low-, mid-, and high-elevation tidal marshes. Salinity gradients range from saline to brackish to fresh tidal waters. The most productive areas for estuarine plant communities are found in tidal marshes. Marsh soils are generally fine-textured and nutrient-rich and support grasses, sedges, rushes, and various other types of plants associated with marine and estuarine habitats. In the LDW, there are 1.75 ha of habitat for vascular plants, primarily limited to portions of Kellogg Island and other small areas (USFWS 2000a).

Carex sp. and *Scirpus* sp. are the predominant type of vegetation between the Upper Turning Basin and Kellogg Island. Downstream from Kellogg Island are more marine plants, such as *Salicornia* sp., *Distichli* sp., and *Atriplex* sp. 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). No eelgrass is found in the LDW, and habitats with aquatic vegetation are rare (Battelle et al. 2001) Thus, plants present in the LDW are seldom present under water.

In recent years, there have been several restoration efforts in the LDW, including the Terminal 105 (T-105) channel along the western shoreline at RM 0.1, Herring's House off-channel marsh along the western shoreline at RM 0.5, Duwamish Diagonal south along the eastern shoreline at RM 0.65, Terminal 107 (T-107) at RM 0.6, the General Services Administration marsh along the eastern shoreline at RM 1.2, the Hamm Creek off-channel marsh on the western shoreline at RM 4.3, and two separate restoration sites in the Upper Turning Basin at RM 4.7. Restoration activities include removal of riprap and overwater structures, placement of log booms to decrease debris deposition and boat wakes, creation of intertidal benches to promote use by juvenile salmon, creation of new off-channel habitat (including emergent salt marshes), and creation of freshwater wetlands (including excavation and planting with native species) (Cordell et al. 2001).

A.2.2 RESOURCES POTENTIALLY AT RISK

This section provides an overview of the ecological resources that use the LDW, including threatened, endangered, and sensitive species. These resources are considered in three groups, which include species that could be directly or indirectly exposed to contaminated sediments: benthic invertebrates, fish, birds, and mammals. Representative species from these groups were selected as ROCs (Section A.2.3) and further evaluated to determine whether they may be at risk from contaminated sediments. Reptiles and amphibians are not likely to be exposed to sediment contamination in the LDW because habitat for these species is limited, and their presence has not been reported in any wildlife surveys conducted in the area[4] (Canning et al. 1979; Cordell et al. 1996; 1997; 1999). Therefore, they are not evaluated further in this ERA. In addition, risks to vascular plants and algae will not be evaluated in the Phase 2 ERA because they were evaluated as part of the Phase 1 ERA (Windward 2003b), and risk estimates were highly uncertain. No new information is available to resolve the uncertainties.

A.2.2.1 State and federal threatened, endangered, and sensitive species in the LDW

Fifteen species that are reported to occur either as residents or during migration in the LDW are listed under either the federal Endangered Species Act (ESA) or by the Washington State Department of Fish and Wildlife (WDFW) as candidate species, or threatened species (Table A.2-2). No species reported to occur in the LDW are listed under the ESA or by WDFW as endangered.

[4] A large tadpole was observed once in Slip 4.

Table A.2-2. Threatened and candidate species listed under ESA or by 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 species of concern
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 species of concern, state candidate species
Pacific cod	*Gadus macrocephalus*	federal species of concern, state candidate species
Walleye pollock	*Theragra chalcogrammus*	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)

a Listing currently under review for removal.

b Downlisted from state endangered to state sensitive April 2002 (WDFW 2002)

Nine of these fifteen listed species are fish and six are birds. With the exception of chinook salmon, coho salmon, Puget Sound steelhead, bald eagle, western grebe, peregrine falcon, and Pacific herring, use of the LDW by these species is rare or incidental, and thus they are not likely to have frequent exposure to sediment-associated chemicals from the LDW. Reports of these rare or incidental species in the LDW include: loons (Canning et al. 1979, rare), merlin (Cordell et al. 1997, rare), common murre (believed to be rare), rockfish (Matsuda et al. 1968, rare; Malins et al. 1980, present; Windward 2005c, 2006b), river lamprey (Warner and Fritz 1995, rare; Matsuda et al. 1968, rare), walleye pollock (Matsuda et al. 1968, rare; Miller et al. 1975, rare), bull trout (Shannon 2006, rare; Warner and Fritz 1995, rare), and Pacific cod (Miller et al. 1975; 1977b; Weitkamp and Campbell 1980). Use of the LDW by chinook salmon, coho salmon, Puget Sound steelhead, bull trout, and herring is described in Section A.2.2.3. Use of the LDW by bald eagle, western grebe, and peregrine falcon is described in Section A.2.2.4.

The Puget Sound Southern Resident Orca whale distinct population segment was added to the federal endangered species list in February 2006 (50 CFR 224) and Washington State's endangered species list in 2004 (WDFW 2006). Orca whales do not use the LDW but are occasionally present in Elliott Bay and the Seattle area in the fall and early winter when chum salmon are running (Nelson 2006). Records from the Whale Museum's sightings database indicate that Orca whales may be seen in Elliott

Bay two or three times during the chum salmon season, although this may be an underestimate because it does not account for unreported sightings (Traxler 2006). Orca whales may be exposed to chemicals from the LDW through the consumption of prey that spend part of their time in the LDW.

A.2.2.2 Benthic invertebrate community

Benthic invertebrate communities (including species such as polychaetes and amphipods) are an important component of the LDW ecosystem because they serve as a major food resource for fish and wildlife, and because they are active in detrital processing and nutrient cycling.

In general, key physical factors that may influence the distribution and abundance of benthic invertebrates in the LDW are salinity, water depth (i.e., intertidal and subtidal), sediment grain size, and OC content. The LDW is a stratified salt-wedge estuary influenced by river flow and tidal effects (Section A.2.1.1). The daily salinity fluctuations select for species that are tolerant of such variability. A salinity range of approximately 5 to 8 parts per thousand (ppt) has been identified as a critical transition range that corresponds to a pronounced minimum of benthic invertebrate species richness (Levinton 1982). Benthic species richness, in general, diminishes steadily in an estuary until it reaches a minimum at the critical salinity (Levinton 1982). In general, a more diverse benthic invertebrate community exists in the downstream, more saline part of the waterway (RM 0.0 to RM 2.0) (Windward 2005f). Similarly, the benthic invertebrate community in the subtidal habitat is, in general, more diverse than the community in the intertidal habitat (Windward 2005f). The benthic invertebrate communities in the intertidal and subtidal habitats of the LDW are discussed in more detail in Section A.2.2.2.1, while larger epibenthic invertebrates are discussed in Section A.2.2.2.2. A table of benthic taxa identified in the LDW is presented in Attachment 1.

A.2.2.2.1 Epibenthos and infauna

Benthic invertebrates present in the LDW are characterized as either infaunal or epibenthic. The infaunal community is typified by burrowing polychaetes and bivalves. This community is dominated by deposit- and filter-feeding organisms. Many of the polychaetes, such as *Hobsonia florida* and *Capitella capitata* complex, are surface detrital- and deposit-feeding organisms. Bivalves obtain their food either from the water column (filter feeders) or the sediment surface (surface deposit feeders). Other infauna, such as oligochaetes, feed on bacteria, diatoms, and other microorganisms (King County 1999b). Small crustaceans (including copepods) feed on diatoms, detritus, and larvae. Epibenthic invertebrate communities include larger crustaceans, mussels, anemones, and echinoderms. Many of these invertebrates, such as echinoderms and anemones, are surface detrital- or filter-feeding organisms. The larger crustaceans are both carnivores and scavengers.

LDWG conducted a survey of the benthic communities (including infauna and smaller epibenthic species) at 12 intertidal locations throughout the LDW in 2004 (Map A.2-1) (Windward 2005f). Descriptions of the benthic communities were based primarily on samples collected with a 0.0024-m^2 core sampler to a depth of 10 cm. Five replicate core samples from each location were composited into one sample for community analysis. To augment the core data by enumerating larger benthic invertebrates from a larger area, samples were also collected within a 0.1-m^2 transect frame and sieved on a 2-mm sieve. Five replicate frame samples from each location were composited into one sample for these additional community analyses.

The numerical abundance of organisms in core samples ranged from 500 to 16,233/0.1-m^2 (Table A.2-3). A total of 61 invertebrate taxa were identified in the core and transect frame samples. The most abundant organisms at the 12 intertidal locations were annelids and crustaceans. The most abundant annelids were *Capitella capitata* complex, *Hobsonia florida, Manayunkia aestuarina*, and *Pygospio elegans*. The crustaceans were dominated by two amphipod species of the genus *Americorophium.* Entognathous hexapods (a subphylum of insects that includes springtails, *Collembola*) and mollusks, including *Macoma baltica,* were also identified at some locations. Echinoderms were not observed at any of the locations (Windward 2005f).

Swartz's dominance index was calculated for each intertidal location. Because replicate samples were not analyzed at each location, these indices should be viewed as qualitative indicators of community structure. Swartz's dominance index is defined as the minimum number of taxa that comprises 75% of the total abundance (Swartz et al. 1985; as cited in PTI 1993). As indicated by the index (Table A.2-3), the intertidal benthic invertebrate community consisted of relatively few species. The index ranged from 2 at location BCA-3 to 8 at location B2a. Numerous physical factors such as salinity, temperature fluctuations, desiccation, and wave action present physiological challenges to benthic invertebrates (Levinton 1982), and are therefore contributing factors to the relatively low number of dominant species in the intertidal areas. For example, salinity in the intertidal areas frequently reaches the critical range of 5 to 8 ppt from approximately Slip 1 (RM 1.0) to upstream sections of the LDW (King County 1999b). In these areas, the salinity is below 5 ppt more than 30% of the time (King County 1999a). Prior to the Phase 2 survey, Cordell et al. (2001) conducted epibenthic and infaunal surveys at seven restoration and reference sites throughout the LDW from 1993 through 1999. They found diversity and abundance of intertidal organisms varied seasonally and among locations in the LDW. The greatest diversity of organisms (i.e., species richness) occurred in the lower (higher salinity) LDW; diversity was comparatively lower in the Upper Turning Basin. Seasonally, species diversity and abundance increased from winter through summer as primary productivity increased. In spring, community composition was generally dominated by two to three species. By summer, the species composition was generally more evenly distributed among a greater number of species. At all sites sampled, the macrofauna were generally numerically dominated by oligochaetes, polychaete

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

River Mile	Location ID[a]	Total Abundance[b]	Taxa Richness[c]	Swartz's Dominance Index	Taxa Richness by Major Taxonomic Group[d]					Abundance by Major Taxonomic Group[f]				
					Annelida	Crustacea	Insecta	Mollusca	Misc. Taxa[e]	Annelida	Crustacea	Insecta	Mollusca	Misc. Taxa[e]
0.2	B1a	4,842	13	3	4	6	1	0	2	3,950	667	8	0	217
0.6	BCA-1	11,667	22	6	7	9	2	1	3	6,083	5,417	17	8	142
0.8	B3a	11,958	25	4	9	13	0	2	1	9,175	2,767	0	17	0
0.9	B2a	16,233	30	8	8	12	2	3	5	7,725	8,183	25	42	258
1.4	B4a	8,858	23	4	6	13	2	0	2	6,933	1,833	25	0	67
2.1	B6a	1,475	19	5	8	5	0	2	4	1,217	100	0	17	142
2.3	B5a-2	500	10	3	6	2	0	0	2	475	17	0	0	8
2.9	BCA-3	9,050	16	2	8	6	0	0	2	7,658	1,267	0	0	125
3.1	B7a	8,600	20	4	6	11	1	0	2	5,983	2,592	8	0	17
3.5	B8a	14,100	21	6	7	11	0	0	3	7,383	6,583	0	0	133
4.5	B9a	5,875	24	5	8	13	1	0	2	3,992	1,867	8	0	8
4.8	B10a	6,600	20	6	7	10	0	0	3	3,292	2,767	0	0	542

a Sampling locations are shown on Map A.2-1.

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 Total number of taxa in a composite of five core samples, representing a total area of 0.012 m^2, at each location.

d Total number of taxa in each major taxonomic group in a composite of five core samples, representing a total area of 0.012 m^2, at each location.

e Miscellaneous taxa include Nemertea, Nematoda, Cnidaria, and Platyhelminthes.

f Number of individual organisms in each major taxonomic group at each location in a standard 0.1-m^2 area determined by extrapolating what was enumerated in the composite sample representing a total area of 0.012 m^2 to the larger area (0.1 m^2) by multiplying by 8.33.

ID – identification

worms of the genus *Manayunkia*, and gammarid amphipods of the genus *Corophium*. The meiofauna, defined as organisms passing through a 0.5-mm sieve but retained on a 0.153-mm sieve, at all sites sampled were generally dominated by nematodes and harpacticoid copepods.

Williams (1990) and Leon (1980) also conducted benthic invertebrate surveys in the LDW (Map A.2-1). 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 organisms 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 Shipyards, and there was a greater degree of seasonal variability in the benthic invertebrate community at a mudflat site in the marina near Kellogg Island (Leon 1980).

In 2003, LDWG conducted a clam survey at 11 intertidal locations between RM 0.0 and RM 4.0 (Map A.2-1) (Windward 2004b). A random sampling design, based on WDFW guidance (Campbell 1996), was employed to survey each of the 11 locations for clam abundance. The sediment was excavated to a depth of 30 cm and all clams were sorted from the substrate. The mean number of clams per 1 ft^2 ranged from 0.18 to 1.0 (Table A.2-4). The majority of clams collected were identified as *Macoma baltica* (60%), followed by *Mya arenaria* (20%) and *Macoma nasuta* (18%). Other less common *Macoma* species included *Macoma inquinata* and *Macoma secta* (both < 1%). The potential catch rates were also assessed at four areas with the highest abundances of clams found during the intertidal survey. *Mya arenaria* was the most common clam collected during the assessment of potential catch rates comprising more than 98% of the total biomass.

Table A.2-4. 2003 intertidal clam survey results

Beach	River Mile	Mean Number of Clams per 1-ft^2 Sample	Total Number of Individuals per Species					
			Macoma baltica	*Macoma inquinata*	*Macoma nasuta*	*Macoma secta*	*Macoma* spp.	*Mya arenaria*
1a	0.1	0.28	6	0	0	0	1	4
2a-island	0.5	1.0	75	0	12	1	1	14
2a-mainland	0.5	0.67	17	1	32	0	1	12
2b-mainland	0.5	0.70	2	0	4	0	0	1
2c-mainland	0.5	0.17	1	0	0	0	0	1
7	1.8	0.46	4	0	0	0	0	9
8	2.1	0.94	39	0	0	1	0	6
11	2.6	0.30	2	0	0	0	0	1

Beach	River Mile	Mean Number of Clams per 1-ft² Sample	Total Number of Individuals per Species					
			Macoma baltica	*Macoma inquinata*	*Macoma nasuta*	*Macoma secta*	*Macoma* spp.	*Mya arenaria*
1a	0.1	0.28	6	0	0	0	1	4
12	2.8	0.71	6	0	0	0	0	4
13a	2.9	0.47	2	0	3	0	0	3
16	3.5	0.18	19	0	0	0	1	2
All beaches	na	0.53	173	1	51	2	4	57

na – not applicable

In 2004, LDWG collected clams for chemical analysis at 10 intertidal locations throughout the LDW (Windward 2005b) (Map A.2-2). Two composite clam tissue samples were collected at 4 of these 10 locations. The collection method involved three field crew members actively searching and collecting clams from locations within the intertidal area with the highest clam abundance, as determined by evidence of siphon holes. The majority of clams collected for chemical analysis were *Mya arenaria,* with only a few *Macoma nasuta* collected at two of the locations (Table A.2-5).

Table A.2-5. Numbers of clams collected for chemical analysis in 2004

Beach[a]	Total Number of Individuals per Species	
	Mya arenaria	*Macoma nasuta*
C1-T	25	0
C2-T1	32	0
C2-T2	52	0
C3-T1	26	0
C3-T2	22	0
C4-T	22	0
C5-T	28	0
C6-T	22	0
C7-T1	17	3
C7-T2	22	0
C8-T	23	0
C9-T	22	0
C10-T1	19	2
C10-T2	17	1

[a] Replicate samples were collected at three beaches along two separate transects (T1 and T2).

C – clam sampling location

T – sampling transect

LDWG conducted a qualitative survey of the benthic communities (including infauna and smaller epibenthic species) at 14 subtidal locations throughout the LDW in 2004 (Map A.2-1) (Windward 2005f). Subtidal samples were collected with a 0.1-m² van

Veen grab sampler and organisms were retained using a 1.0-mm mesh sieve. Three replicate samples were composited into one sample at each location. The total number of organisms ranged from 72 to 2,300 per 0.1 m^2. 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. The most abundant annelids were *Aphelochaeta* cf *glandaria, Capitella capitata* complex, *Hobsonia florida, Polydora cornuta*, and *Scoletoma luti*. The crustaceans were dominated by two amphipods of the genus *Americorophium,* especially at locations between RM 3.9 and RM 5.0. *Eogammarus confervicolus* and *Grandidierella japonica* were also common crustaceans. The most abundant mollusks were the bivalves *Axinopsida serricata, Parvilucina tenuisculpta*, and *Macoma* sp., and the most common gastropod was *Alvania compacta*. Echinoderms were also present at six locations with abundances ranging from 0.1 to 1.3% of all identified organisms.

As a way to summarize the subtidal benthic invertebrate community data, Swartz's dominance index was calculated for each subtidal location. Because replicate samples were not collected at each location, these indices should be viewed as qualitative indicators of community structure. The highest values of Swartz's dominance index were, in general, calculated for the more saline locations in the lower part of the waterway (RM 0 to RM 1.5), and the lowest value was calculated for a mostly brackish location in the Upper Turning Basin at RM 4.3. In the Upper Turning Basin, the salinity is less than 5 ppt 70 to 84% of the time (King County 1999a). This pattern is in accordance with general descriptions of benthic communities in estuaries, in which diversity diminishes steadily up an estuary along a salinity gradient (Levinton 1982). Table A.2-6 summarizes the results of the subtidal benthic invertebrate community survey.

Prior to the Phase 2 survey, Ecology (2000) evaluated the benthic invertebrate community at three subtidal locations in the LDW as part of the sediment quality reconnaissance study for central Puget Sound. The benthic invertebrate community at the three locations was dominated by annelids. Mollusks were also common, whereas crustaceans and echinoderms were present in low abundances.

King County (1999b) evaluated risks to benthic infauna and epibenthos as a component of their water quality assessment (WQA) of combined sewer overflow (CSO) discharges to the Duwamish River and Elliott Bay. Subtidal samples were collected with a 0.1-m^2 van Veen grab sampler and organisms were retained using a 1.0-mm mesh sieve. Sampling sites included transects located at Kellogg Island and downgradient from the Duwamish/Diagonal CSO (Map A.2-1). Polychaetes were abundant and were the dominant organisms at all subtidal locations, except at two stations downstream of the Duwamish/Diagonal CSO, where oligochaetes and mollusks were dominant. A Kellogg Island station also had relatively high abundance of mollusks. Arthropods were more abundant in deeper waters.

Table A.2-6. Phase 2 subtidal benthic invertebrate community survey results

River Mile	Location ID[a]	Total Abundance[b]	Taxa Richness[c]	Swartz's Dominance Index	Taxa Richness by Major Taxonomic Group[d]					Major Taxonomic Group Abundance[f]				
					Annelida	Crustacea	Echino-dermata	Mollusca	Misc. Taxa[e]	Annelida	Crustacea	Echino-dermata	Mollusca	Misc. Taxa[e]
0.1	B1b	326	107	15	43	22	4	28	10	176	77	3	66	3
0.6	BCA-4	937	107	13	62	12	2	25	6	499	60	<1	367	11
0.9	B2b	537	92	7	43	17	1	23	8	124	85	<1	323	4
1.0	B3b	559	93	12	54	14	2	22	1	397	42	<1	85	35
1.4	B4b	521	78	8	39	11	3	19	6	208	28	7	276	2
1.5	B5b	643	60	8	34	11	0	11	4	468	105	0	44	27
1.5	BCA-5	72	50	15	27	7 (1)[g]	0	12	3	52	6	0	13	2
1.7	BCA-2	328	54	6	29	12	1	11	1	282	17	<1	27	<1
2.2	B6b	1,137	83	3	41	17	0	21	4	1,041	17	0	77	2
2.7	B7b	497	75	5	38	16	1	18	2	391	11	<1	94	<1
3.9	B9b	935	36	6	17	12	0	6	1	435	493	0	8	<1
4.2	B8b	2,300	27	4	14	12	0	1	0	502	1,793	0	5	0
4.3	B10b	1,541	16	2	9	7	0	0	0	195	1,347	0	0	0
4.6	BCA-6	1,689	14	3	7	7	0	0	0	405	1,284	0	0	0

a Sampling locations are shown on Map A.2-1.

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.

c Total number of taxa in a composite of three van Veen grab samples, representing a total area of 0.3 m^2, at each location.

d Total number of taxa in each major taxonomic group in a composite of three van Veen grab samples, representing a total area of 0.3 m^2, at each location.

e Miscellaneous taxa include Nemertea, Nematoda, Cnidaria, and Platyhelminthes.

f Number of individual organisms in each major taxonomic group 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.

g One insect specimen was collected at this location.

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

ID – identification

Williams (1990) sampled epibenthic sediment biota near Kellogg Island and found 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.

Leon (1980) used van Veen grab samplers to characterize the epibenthic and infaunal sediment biota from subtidal locations near Kellogg Island. More than 60 different taxa were identified, which was greater than the number found in the intertidal habitat from the same survey. The most abundant taxon was deposit-feeding cirratulid polychaete worms. While some of these invertebrates were also found in intertidal habitats (oligochaetes, *Capitella* sp., *Pygospio* sp., ostracods), most subtidal species were deposit-feeding polychaete worms that are characteristic of the 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.

A.2.2.2.2 Larger epibenthic invertebrates

Larger epibenthic invertebrates identified in the LDW include crabs, shrimp, sea stars, anemones, and mussels. Mussels, anemones, and echinoderms are surface detrital- or filter-feeding organisms, whereas crabs and shrimps couple predaceous feeding with scavenging. Numerous larger epibenthic invertebrate species were caught during two fish and crab surveys conducted in late summer (August and September 2004 and 2005) throughout the LDW by LDWG (Windward 2005c, 2006b) (Maps A.2-1 and A.2-2; Table A.2-7). The invertebrates were collected in high-rise otter trawls, beach seines, crab traps, and shrimp traps. The most abundant epibenthic invertebrates were slender crabs, crangon shrimps, and coonstripe shrimps. Dungeness crabs were also caught in both surveys. The distribution of Dungeness crabs and other crabs was generally limited to the downstream portion of the LDW where the salinity is greater; however, a few adult Dungeness and slender crabs were caught between RM 4.2 and 4.5. A pilot survey and three quarterly surveys were performed in 2003 and 2004 to estimate the abundance of crabs and shrimps in the LDW (Windward 2004a). Three crab species and one shrimp species were caught in the surveys. Slender crabs and Dungeness crabs were the most abundant species. The majority of these crabs were caught in the downstream, more saline part of the LDW, with a few adults caught between RM 4.2 and 4.6. Red rock crabs and dock shrimps were less abundant and they were also primarily caught in the downstream portion of the LDW, with a few adults caught between RM 1.6 and 2.2. In October 1998, adult Dungeness and red rock crabs were collected at multiple locations near Kellogg Island but were not caught farther upstream (ESG 1999).

Table A.2-7. Numbers of individual invertebrate species caught using trawls and traps in the LDW during the 2004 and 2005 surveys

Common Name	Scientific Name	Number of Specimens Caught	
		2004	2005
Anemone, plumose	*Metridium senile*	nc	49
Anemone	unknown	81	nc
Ascidian[a]	unknown	1	nc
Crab, black-clawed	*Lophopanopeus bellus*	4	nc
Crab, decorator	*Loxorhynchus crispatus*	32	19
Crab, Dungeness	*Cancer magister*	62	33
Crab, hermit	*Pagurus* sp.	11	3
Crab, kelp	*Pugettia producta*	8	11
Crab, red rock	*Cancer productus*	16	19
Crab, slender	*Cancer gracilis*	942	483
Frilled dogwinkle	*Nucella lamellosa*	nc	2
Mussel, blue	*Mytilus edulis*	6	nc
Moon snail	*Polinices lewisii*	nc	5
Nudibranch, striped	*Armina californica*	39	118
Nudibranch	unknown	41	nc
Sea star, mottled	*Evasterias troschelii*	11	7
Sea star, sunflower	*Pycnopodia helianthoides*	23	18
Sea star, sand	*Luidia*	nc	2
Sea star	*Pisaster* sp.	50	27
Sea star	unknown	11	nc
Sea pen	unknown	38	8
Shrimp, coonstripe	*Pandalus danae*	314	231
Shrimp, crangon	*Crangon* sp.	538	172
Shrimp	unknown	8	nc
Solaster	*Solaster stimpsoni*	nc	1
Tunicate[a]	unknown	nc	2
Urchin	unknown	4	12

[a] Ascidian is a class of tunicates, and tunicate (or Urochordata) is the subphylum. However, because the two other classes of tunicates are pelagic, the two common names refer to the same group of invertebrates.

nc – not collected

During the 1989-1999 Puget Sound Ambient Monitoring Program (PSAMP), invertebrates were collected throughout the Puget Sound, including locations in the LDW (West et al. 2001). Epibenthic invertebrate species, similar to those caught in the LDWG surveys, were collected with otter trawls. The most common invertebrates were slender crabs and crangon shrimps. Other species caught during the PSAMP survey, but not in the LDWG surveys, included porcelain crabs (family Porcellanidae), chitons (class Polyplacophora), and several sea stars.

Commonly observed mollusks include epibenthic mussels which have been observed in large numbers on pilings and other structures in the downstream, more saline portion of the LDW with fewer mussels reported up to and slightly above the Upper Turning Basin (Windward 2000).

A.2.2.2.3 Summary

In summary, benthic invertebrates in the LDW consist of infauna and epibenthic organisms in both intertidal and subtidal habitats. Invertebrates surveyed in the LDW include more than 670 taxa, representing 178 families in 13 phyla (Attachment 1). Typical of most estuaries, the invertebrate community is dominated by annelids, mollusks, and crustaceans. Crustaceans are the most diverse of these three groups in the LDW, representing more than 250 taxa. These taxa included numerous macrofauna species from the orders Amphipoda, Isopoda, Cumacea, Tanaidacea, and Decapoda and numerous meiofauna species from the orders of Harpacticoida and Calanoida. The mollusks are represented by various bivalves and to a lesser extent by gastropods. The most abundant large epibenthic invertebrates include slender crabs, crangon, and coonstripe shrimps.

The taxonomic survey performed by LDWG, which was designed as a qualitative study of the benthic invertebrate community throughout the LDW, did not evaluate the distribution of benthic invertebrates in the LDW or their use of the LDW. Because of the qualitative nature of the study, no conclusions can be drawn about potential adverse effects of sediment chemicals on the benthic invertebrate community or about benthic invertebrate distribution among habitat types.

A.2.2.3 Fish

Diverse fish communities inhabit the LDW. Data are available from 14 studies conducted in the LDW investigating site usage by fish (Table A.2-8). Fifty-three resident and non-resident fish species were captured in the LDW during Phase 2 sampling (Windward 2004c, 2005c, 2006b). During historical sampling, Warner and Fritz (1995) recorded 33 resident and seasonal fish species, Miller et al. (1975; 1977b) observed a total of 29 species, and Matsuda et al. (1968) recorded a total of 28 species. In these studies, shiner surfperch, snake prickleback, Pacific sandlance, Pacific staghorn sculpin, longfin smelt, English sole, and starry flounder were particularly abundant, as were juvenile chinook, chum, and coho salmon. Fish numerical abundance reaches its maximum in late summer to early fall and is generally lowest in winter (Miller et al. 1977b; Dexter et al. 1981). Based on otter trawl data, species richness was shown to follow a similar trend but did not vary greatly with season (Miller et al. 1977b). Fish species reported to occur in LDW studies are listed in Attachment 2 with habitat, diet, and abundance information for each.

This section presents a summary of studies of LDW fish as well as a brief summary of the life history characteristics and dietary preferences of some fish species found in the LDW.

Table A.2-8. Summary of studies assessing the fish community in the LDW

Study	Year Completed	Location	Sampling Period	Equipment Type	No. of Locations Sampled
Phase 2 fish and crab tissue collection and chemical analyses (Windward 2005c, 2006b)	2005	four areas throughout the LDW	August and September, 2004 August and September, 2005	otter trawl, beach seine, shrimp traps, crab traps	24
Habitat utilization, migration timing, growth, and diet of juvenile chinook salmon in the Duwamish River and Estuary (Ruggerone et al. 2006)	2005	throughout LDW	February to July, 2005	beach seine	14
Fish assemblages and patterns of juvenile chinook salmon abundance, diet, and growth at restored sites in the Duwamish River (Cordell J et al. 2006)	2005	restoration and reference sites throughout LDW	February to July 2005	enclosure net	6
Phase 2 juvenile chinook salmon collection and chemical analyses (Windward 2004c)	2003	lower waterway (RM 0.1 to RM 0.9), and mid waterway (RM 1.4 to RM 2.9)	May (2 days) and June (3 days) 2003	beach seine	8
East Waterway channel deepening project, juvenile salmonid and epibenthic prey assessment (Shannon 2006)	2003	Kellogg Island and Harbor Island area	biweekly April – August 1998, 2000, 2003	beach seine	7 (2 in LDW)
East Waterway juvenile chinook salmon (Windward 2002)	2002	Kellogg Island	June 2002	beach seine	1
Waterway sediment operable unit, Harbor Island Superfund site. Assessing human health risks from ingestion of seafood (Robertson 2004)	1998	Harbor Island to south side of 1st Ave S. Bridge	single visit to each site	SCUBA	8 (6 in LDW)
PSAMP (West 2001)	1997	Kellogg Island	May 1992 – 1997	otter trawl	1
Distribution and growth of Green River chinook salmon and chum salmon outmigrants in the Duwamish estuary (Warner and Fritz 1995)	1994	Kellogg Island to above rapids	February – April (biweekly); April – May (weekly); May – September (biweekly) 1994	beach seine	9
Distribution and food habits of juvenile salmonids in the Duwamish Estuary (Meyer et al. 1981)	1980	Kellogg Island and at S Kenyon Street (RM 3.0)	April to June (weekly); July (biweekly) 1980	purse seine	2
				beach seine	2

Study	Year Completed	Location	Sampling Period	Equipment Type	No. of Locations Sampled
Port of Seattle Terminal 107 fisheries study (Weitkamp and Campbell 1980)	1978	Kellogg Island and adjacent channel	monthly October 1977 to February, July, and August 1978; more frequently from March to June 1978	purse seine	5
				beach seine	5
		South end of Kellogg Island	October 1977 – August 1978 (quarterly)	gill net (surface and bottom)	1
Chemical contaminants and biological abnormalities in central and southern Puget Sound (Malins et al. 1980)	1979	South end of Harbor Island	quarterly	7.5-m otter trawl	1 in LDW
Ecological survey of demersal fishes in the Duwamish River and at West Point or near Metro sewage treatment plants (Miller et al. 1975; 1977a; 1977b)	1974, 1975, and 1976	West Waterway to the Upper Turning Basin	1974 and 1975 (monthly) and January 1976	5-m otter trawl	8 (7 in LDW)
Fishes of the Green-Duwamish River (Matsuda et al. 1968)	1966	upper and lower LDW (exact locations unknown)	1964 – 1966 (weekly)	beach seine	2

Note: The majority of these studies used active capture techniques such as beach seining and otter trawls. These techniques preferentially capture less mobile species and are not effective for rough substrates or near structures. However, passive techniques employed in LDW sampling that included gill nets (Weitkamp and Campbell 1980), shrimp traps, and crab traps (Windward 2005c, 2006b) yielded no additional fish species beyond those observed using beach seines or otter trawls, indicating that the trawl data are generally reflective of the LDW fish community. Five of the 11 studies were conducted prior to 1986 when the Renton Wastewater Treatment Plant outfall was diverted from the Green River to Central Puget Sound. Because the diversion of the wastewater treatment plant effluent decreased summer flows by as much as 25% (~1.6 m3/s [56 cfs]), the diversity and abundance of fish in the LDW may have changed since these studies were conducted.

LDW – Lower Duwamish Waterway

PSAMP – Puget Sound Ambient Monitoring Program

RM – river mile

A.2.2.3.1 Pacific salmon

Five species of Pacific salmon (coho, chinook, chum, sockeye, and pink) occur in the LDW (Kerwin and Nelson 2000). These anadromous fish use the estuary for rearing and as a migration corridor for adults and juveniles. Among numerous beneficial uses identified for the LDW, habitat for outmigrating juvenile salmonids is one of the most important (Harper-Owes 1983).

The time spent by salmonids in the LDW is limited for all species. The amount of time spent each year depends on the specific life history of the species. Salmon found in the LDW spawn mainly in the middle reaches of the Green River and its tributaries (Grette and Salo 1986).

Adult salmon generally do not feed to any significant extent once they enter the estuary on their upstream spawning migrations. The peak timing of outmigration for juveniles of all species generally corresponds with March-to-June high flows. Outmigration usually lasts through mid-July to early August for most species (Warner and Fritz 1995; Nelson et al. 2004). During this time, juveniles use the estuary to feed and begin their physiological adaptation to higher salinity.

Chinook Salmon

Historically, the Green/Duwamish River supported spring and fall runs of chinook salmon. Fall-run chinook, a sub-population of the Puget Sound chinook population, are the only naturally sustaining run that still uses the Green/Duwamish River corridor. This run was among those listed as threatened under ESA in March 1999. These fish use the LDW for migration to and from spawning grounds in the mainstem Green River and larger tributary streams. Production is from hatcheries, naturally spawning hatchery-reared fish, and naturally spawning native fish (Grette and Salo 1986; WDFW 1993).

Returning fall chinook salmon enter the LDW from late June through mid November, with peak upstream migration in mid August (Grette and Salo 1986). In the mid-1970s, WDFW established an escapement goal of 5,800 naturally produced fall chinook using average escapement of natural origin fish and hatchery strays from 1965-1976 (Ames and Phinney 1977; as cited in Weitkamp and Ruggerone 2000). For the period from 1989-99, spawning escapements have been relatively high, averaging 8,578 fish, exceeding the WDFW goal for eight of the 10 years (WDFW unpublished data, as cited in King County 1999c). The contribution of Green River chinook salmon to the total chinook run entering Puget Sound and the Strait of Juan de Fuca ranged from 1.9 to 7.0% for the period 1979 to 1984 (Grette and Salo 1986).

In a 2003 tagging study of natural origin fry released approximately 30 miles upstream in February and March, fry were recovered at RM 5.0 to 6.0 of the LDW within one to 31 days after release (53% of the fry were found within 1 to 4 days) (Nelson et al. 2004). In 2002 and 2003, several year classes of outmigrant chinook (fry, yearlings, and possibly 2-year-old fish) were found in the LDW between January and September (Warner and Fritz 1995; Nelson et al. 2004). Two peaks in abundance occurred: the first peak (of fry) was observed from late February to early March, and the second peak (of fingerlings) occurred between mid-May and mid-June. Peak catch data in 2002 and 2003 suggest that most naturally spawned fingerlings arrive in the estuary in May and reside in the LDW for approximately 2 weeks before departing to marine waters (Nelson et al. 2004). However, some fry may stay in the LDW from January until outmigration in June. Subyearlings have been consistently captured at RM 5.0 and 6.0,[5] suggesting that this is an important zone where juvenile chinook salmon transition between fresh and salt water (Ecology 2000; Nelson et al. 2004; Warner and Fritz 1995).

Ruggerone and Volk (2004) estimated residence time in the LDW prior to capture using incremental uptake of strontium in otoliths. Residence time of naturally spawned[6] chinook salmon collected from throughout the LDW averaged from a low of 16 ± 4 days in early July to a high of 58 ± 13 days in early September. The average residence time of individual hatchery chinook salmon increased from 16.6 days during late May through June (ranging from 6 to 25 days) to 45.6 days in mid-September.

Gut content analyses have shown that juvenile chinook in the LDW prey on a wide variety of benthic organisms such as *Corophium* spp. (amphipods) and *Cumella vulgaris*, drifting organisms such as adult dipterans, and zooplankton such as barnacle nauplius larvae (Cordell et al. 1997; 1999). Seasonal sampling suggests juvenile chinook shift their diet as different prey become available (Cordell et al. 1997; 1999).

Coho Salmon

Green River coho constituted from 0.9 to 1.4% of the total coho run entering Puget Sound and the Strait of Juan de Fuca for the period 1979-1984 (Grette and Salo 1986). Production was from hatcheries, naturally spawning hatchery-reared fish, and naturally spawning native fish (Grette and Salo 1986; WDFW 1993).

Adult coho return to the LDW between August and January, move through the LDW in a few days, and spawn and rear in all accessible reaches of the Green River drainage (Williams et al. 1975; as cited in Grette and Salo 1986). Juvenile coho rear in the Green River and move quickly through the LDW estuary as smolts (Warner and Fritz 1995; Weitkamp and Schadt 1982). The timing of outmigration is dependent on releases from Green River hatcheries (Warner and Fritz 1995; Weitkamp and Schadt 1982).

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[5] Reported as RM 5.5 and 6.5 in original report; mapped locations were consistent with RM 5.0 and 6.0 as defined in this document (relative to the south end of Harbor Island).

[6] Fish that have not been fin-clipped and may be native fish or progeny of hatchery adults spawning in the wild.

Chum Salmon

The current status of the native chum population of the Green River watershed is unknown, but this population is suspected to have declined dramatically (Grette and Salo 1986; WDFW 1993). WDFW (1993) reported the state of the Green/Duwamish chum stock as unknown, whereas Nehlsen et al. (1991) reported the stock as at risk of extinction. Chum from the Green River constitute an insignificant portion of the total south Puget Sound run, and are not specifically addressed with harvest strategies for south Puget Sound stocks (Grette and Salo 1986).

Outmigrating chum salmon spend from several days to two months rearing in the Duwamish River estuary prior to moving offshore (Grette and Salo 1986). Warner and Fritz (1995) captured juvenile chum in beach seines throughout the LDW from February through September 1994 and showed a continuous increase in size of fish captured, suggesting a relatively extended residence time in the LDW. However, in the same study, catch rates declined rapidly following peak catches, suggesting that fish were likely moving through the estuary within a few days. Adult chum salmon return to the LDW between September and December.

Gut content analyses showed that in the LDW, juvenile chum preyed on both epibenthic species and drift insects during outmigration, with a large temporal variation in prey composition (Cordell et al. 1997; 1999).

Pink Salmon

Pink salmon appear infrequently and in low numbers. A run of odd-year pink salmon existed in the Green River in the 1930s (Williams et al. 1975; as cited in Grette and Salo 1986), though this run is believed to be currently extinct (Grette and Salo 1986). Warner and Fritz (1995) captured a total of 14 juvenile pink salmon in beach seines from nine stations throughout the LDW sampled approximately every two weeks from February through September 1994. Adult pink salmon have also been observed spawning in the mainstem Green River (Kerwin and Nelson 2000). Grette and Salo (1986) suggest that pink salmon have a high incidence of straying and that the few pink salmon captured in Green/Duwamish River are likely strays from other systems.

Sockeye Salmon

There is limited evidence that sockeye salmon spawn and rear in the Green River watershed (Jeanes and Hilgert 2000). Juvenile sockeye appear to have the shortest residence time in the nearshore of all salmon species (Kerwin and Nelson 2000).

A.2.2.3.2 Other salmonids

The Coastal-Puget Sound population of bull trout was proposed for listing under the federal ESA in June 1998 and was formally listed as threatened on November 1, 1999. The decline of bull trout has been primarily attributed to habitat degradation and fragmentation, blockage of migratory corridors, poor water quality, past fisheries management practices, and the introduction of non-native species (64FR 210: 58910-

58933). Bull trout were historically found in the LDW, but current stock status is unknown (WDFW 2000).

Muckleshoot tribal biologists captured one adult bull trout during beach seining in the LDW on May 24, 1994 during the period of peak juvenile salmon outmigration. However, it is unknown whether the fish reared in the Green River or was an opportunistic resident (Warner and Fritz 1995). Eight subadult bull trout ranging in length from 271 to 373 mm were captured in beach seines in the Upper Turning Basin during two sampling events in August and September 2000 (Shannon 2001). Peak numbers of juvenile shiner surfperch were captured at the same site the previous week, and near-peak numbers of shiner surfperch were captured in the same sampling in which the bull trout were caught (Shannon 2001). The co-occurrence of bull trout with high abundance of potential prey suggest that they may be opportunistically occupying the LDW to prey on these small fish. There is no evidence that bull trout are spawned or reared within the LDW. Bull trout juveniles typically remain in the upper tributaries for a period of two to three years prior to migrating to saltwater during spring. Adults typically return to their native streams in summer and fall (Grette and Salo 1986).

Summer steelhead (*Oncorhynchus mykiss*) is a non-native stock sustained by wild spawning of hatchery-reared fish (WDFW 1993). The run size is unknown, but approximated at a few hundred fish (WDFW 1993). The winter steelhead run consists of wild and hatchery fish with annual returns of 944 to 2,378 fish (WDFW 1993). Spawning of winter steelhead, which return to the Green River from December through May, generally begins about mid-March and continues to early June, with a peak in mid-May (Cropp 1985, as cited in Grette and Salo [1986]). Grette and Salo (1986) report that repeat spawners make up approximately 19% or less of returning wild adult steelhead in the Green River (1976/77 to 1983/84). Summer steelhead outmigrate from the Green River after rearing for two years as smolts, and do not have an extensive residence time in the LDW. Winter steelhead outmigrate from the Green River as subyearling adults and also do not rear extensively in the LDW.

Sea-run cutthroat trout may occur in the LDW, but little is known about this population. A total of 11 adult 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). In Washington, adult cutthroat return to their home stream from July to January, with the peak occurring in October and November (Wydoski and Whitney 2003). Smolt outmigration occurs from April through May (Wydoski and Whitney 2003).

A.2.2.3.3 Non-salmonid fishes

The most abundant non-salmonid fishes in the LDW are shiner surfperch, starry flounder, three spine stickleback, English sole, Pacific staghorn sculpin, Pacific herring, Pacific sandlance, surf smelt, and longfin smelt (Warner and Fritz 1995; Windward 2004c, 2005c, 2006b; Shannon 2006; West 2001).

Of these fish, shiner surfperch, longfin smelt, and Pacific herring are seasonally abundant in the LDW. Pacific herring, Pacific sandlance, surf smelt, and longfin smelt were encountered infrequently in recent beach seine and trawling attempts, but occasionally occurred in large numbers (Shannon 2006; Windward 2005c, 2006b). Three spine stickleback were abundant in monthly beach seine samples at both the Upper Turning Basin and Kellogg Island sampling locations June through September but were uncommon in February through May samples (Shannon 2006). Historical otter trawl data show peaks in longfin smelt abundance in summer, fall, and early winter (Miller et al. 1977b). Miller et al. (1977b) suggest that the fall-winter peak (80- to 115-mm fish) may represent part of a spawning run and that the late summer peak (30- to 50-mm fish) may represent downstream migrant young of the year. Pacific herring were reported in purse seine samples throughout the year (Weitkamp and Campbell 1980), were present in trawl samples in August and September (Windward 2005c, 2006b), and were reported in beach seine samples in May, June, July, November, and December (Weitkamp and Campbell 1980; Shannon 2006). In Puget Sound, threespine stickleback and surf smelt feed on both epibenthic and pelagic invertebrates. Epibenthic invertebrates constitute a slight majority of their diet (Miller et al. 1977c; Fresh et al. 1979). Pacific herring and longfin smelt generally feed on pelagic invertebrates but also ingest epibenthic invertebrates to a lesser extent (Miller et al. 1977c; Fresh et al. 1979).

In all studies, Pacific staghorn sculpin was consistently one of the most abundant fish captured in the LDW (Attachment 2). Taylor and Associates (Shannon 2006) captured Pacific staghorn sculpin during all months from February through October. Their abundance was highest from May through July. Miller et al. (1977b) reported that Pacific staghorn sculpin were abundant in otter trawls during all seasons, but were particularly abundant in the fall. Few Pacific staghorn sculpin larger than 150 mm were collected in beach seines by Weitcamp and Campbell (1980). During otter trawl sampling conducted by LDWG in August and September 2004 and 2005, juvenile (< 120 mm) and adult Pacific staghorn sculpin were commonly encountered throughout the LDW (Windward 2005c, 2006b).

Pacific staghorn sculpin are opportunistic feeders that feed at a higher trophic level than other common resident fish species. As discussed in a meeting of local fish experts (Windward 2004j), other upper-trophic-level fish that occur in the LDW, such as brown rockfish and sand sole, occur primarily as juveniles that feed at a similar trophic level as Pacific staghorn sculpin (e.g., brown rockfish) or have foraging ranges that likely extend well beyond the LDW (e.g., sand sole). Pacific staghorn sculpin feed mostly on crabs, shrimps, and benthic invertebrates, but also ingest larval, juvenile, and adult fish (Fresh et al. 1979; Wingert et al. 1979; Miller et al. 1977c). Larger sculpin are more likely to eat at a higher trophic level than smaller sculpin.

Shiner surfperch abundance peaks in summer during the bearing of young (Miller et al. 1975). Taylor and Associates recorded abundant shiner surfperch May through October with peak abundance in July (Shannon 2006). Shiner surfperch are opportunistic omnivores, feeding on zooplankton, small crustaceans, algae, and detritus (Gordon 1965; Bane and Robinson 1970), as well as polychaetes, mollusks, and benthic organisms (Fresh et al. 1979; Wingert et al. 1979; Miller et al. 1977c).

English sole are common in the LDW over all seasons, with peak abundance in spring (Miller et al. 1977b). English sole were abundant in recent trawl samples but were absent from beach seine samples (Windward 2006b; Shannon 2006; West 2001; Windward 2005c). In Puget Sound, English sole are typically found on soft sand or mud bottoms at depths of 25 to 50 m (Smith 1936). Juvenile English sole (those less than 110 mm) ingest annelids (Smith 1936), copepods, amphipods, and mollusks (Holland 1954). Adult English sole studied in Puget Sound ingest clams, clam siphons, small mollusks, marine worms, small crabs, and small shrimps (Wingert et al. 1979; Fresh et al. 1979). It has been suggested that English sole exist in discrete populations with some site fidelity (Day 1976). Day (1976) conducted a tagging study in Puget Sound that suggested that fish captured and released at the same location remained within an area approximately equal to 5 to 10 km^2. In addition, catch rates for fish captured and released dozens of miles from their original capture site were higher at their original capture site than at the release site or other sites sampled.

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 after spawning (Pallson 2001). Angell et al. (1975; as cited in King County 1999c) reported off-shore migration in winter and spring of all age groups of central Puget Sound English sole from Meadow Point to Carkeek Park (northwest Seattle) at depths of 3 to 30 m. Juveniles (10 to 25 mm standard length), not all completely metamorphosed, migrated from spawning areas to nursery grounds as pelagic fish and moved to benthic habitats in December or May and June (King County 1999b). Data from Malins et al. (1982) show that during the winter and spring, greater than 50% of the English sole in the LDW are juveniles (< 150 mm).

Starry flounder are also noted to migrate seasonally between very shallow water and in estuaries during the summer, moving into deeper water in the winter (Morrow 1980). Young and adult starry flounder are tolerant of fresh water and move up rivers as much as 120 km (Morrow 1980). Because they have a larger mouth, starry flounder are capable of consuming somewhat larger organisms than English sole ingest, although their diets greatly overlap. Starry flounder in Puget Sound were found to ingest primarily benthic invertebrates, with bivalves, amphipods, and shrimp serving as important prey items (Fresh et al. 1979).

A.2.2.3.4 Summary

In summary, the LDW provides habitat for many migrating and resident fish. Anadromous salmon are present during rearing and migration to and from spawning sites in the upper watershed. In the LDW, juvenile salmonids are an important part of the food web, preying on various epibenthic, water column, and drift organisms and serving as prey for larger fish and wildlife. Of the non-salmonid fish, shiner surfperch, English sole, Pacific staghorn sculpin, snake prickleback, longfin smelt, and starry flounder are among the most abundant species in the LDW. Seasonal abundance of fish in the LDW varies, peaking in the summer and early fall. Fish in the LDW are primarily carnivorous and are likely to rely extensively on epibenthic invertebrates as prey.

A.2.2.4 Wildlife

The intertidal and subtidal 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 utilize the LDW during at least part of the year to feed, rest, or reproduce. This section provides an overview of these bird and mammal species.

A.2.2.4.1 Birds

The bird species associated with the LDW are presented in Table A.2-9. The birds using the site can be grouped as follows:

- Passerine/upland species
- Raptors
- Shorebirds/waders
- Waterfowl
- Seabirds

Table A.2-9. Bird species using the LDW

Common Name	Scientific Name	Common Name	Scientific Name
Passerine/Upland Species			
Blackbird, red-winged	*Agelaius phoeniceus*	Sparrow, English (house)	*Passer domesticus*
Bushtit, common	*Psaltriparus minimus*	Sparrow, fox	*Passerella iliaca*
Chickadee, black-capped	*Poecile atricapillus*	Sparrow, golden-crowned	*Zonotrichia atricapilla*
Cowbird, brown-headed	*Molothrus ater*	Sparrow, savannah	*Passerculus sandwichensis*
Crow, northwestern	*Corvus corrinus*	Sparrow, song	*Melospiza melodia*
Dove, rock	*Columba livia*	Sparrow, white-crowned	*Zonotrichia leucophrys*
Finch, house	*Carpodacus mexicanus*	Starling, European	*Sturnus vulgaris*
Flicker, northern	*Colaptes auratus*	Swallow, barn	*Hirundo rustica*
Goldfinch, American	*Spinus tristis*	Swallow, cliff	*Petrochelidon pyrronota*
Hummingbird, Anna's	*Calypte anna*	Swallow, tree	*Iridoprocne bicolor*
Junco, dark-eyed	*Junco hyemalis.*	Swallow, violet-green	*Tachycineta thalassina*
Kingfisher, belted	*Ceryle alcyon*	Thrush, Swainson's	*Hylocichla ustulata*
Kinglet, ruby-crowned	*Regulus calendula*	Towhee, rufous-sided	*Pipilo erythrophthlamus*
Purple martin	*Progne subis*	Warbler, orange-crowned	*Vermivora celata*

Common Name	Scientific Name	Common Name	Scientific Name
Quail, California	*Lophortyx californicus*	Wren, Bewick's	*Thryomanes bewickii*
Robin, American	*Turdus migratorius*	Wren, house	*Troglodytes aedon*
Siskin, pine	*Carduelis pinus*		
Raptors			
Eagle, bald	*Haliaeetus leucocephalus*	Hawk, sharp-shinned	*Accipiter striatus*
Falcon, peregrine	*Falco peregrinus*	Hawk, Swainson's	*Buteo swainsoni*
Hawk, Cooper's	*Accipter cooperii*	Merlin	*Falco columbarius*
Hawk, red-tailed	*Buteo jamaicensis*	Osprey	*Pandion haliaetus*
Shorebirds/Waders			
Dowitcher	*Limnodromus sp.*	Sanderling	*Crocethia alba*
Dunlin	*Erolia alpina*	Sandpiper, least	*Calidris minutilla*
Heron, great blue	*Ardea herodias*	Sandpiper, spotted	*Actitis macularia*
Heron, green	*Butorides virescens*	Sandpiper, western	*Calidris mauri*
Killdeer	*Charadrius vociferus*	Yellowlegs, lesser	*Totanus flavipes*
Waterfowl			
Bufflehead	*Bucephala albeola*	Goose, domestic	*Branta domesticus*
Canvasback	*Aythya valisineria*	Mallard	*Anas platyrhynchos*
Coot, American	*Fulica americana*	Merganser, common	*Mergus merganser*
Duck, domestic	*Anas domesticus*	Merganser, hooded	*Lophodytes cucullatus*
Gadwall	*Anas strepera*	Merganser, red-breasted	*Mergus serrator*
Goldeneye, Barrow's	*Bucephala islandica*	Scoter, surf	*Melanitta perspicillata*
Goldeneye, common	*Bucephala clangula*	Teal, greenwinged	*Anas carolinensis*
Goose, cackling Canada	*Branta canadensis minima*	Wigeon, American	*Mareca americana*
Goose, Aleutian	*Branta canadensis*		
Seabirds			
Cormorant, double-crested	*Phalacrocorax auritus*	Gull, glaucous-winged	*Larus glaucescens*
Cormorant, pelagic	*Phalacrocorax pelagicus*	Gull, mew	*Larus canus*
Grebe, eared	*Podiceps capsicus*	Gull, ring-billed	*Larus delawarensis*
Grebe, horned	*Podiceps auritus*	Loon, common	Gavia immer
Grebe, pied-billed	*Podilymbus podiceps*	Loon, Pacific	Gavia Pacifica
Grebe, red-necked	*Podiceps grisegena*	Loon, red-throated	Gavia stellata
Grebe, western	*Aechmophorus occidentalis*	Murre, common	*Uria aalge*
Guillemot, pigeon	*Cepphus columba*	Tern, Caspian	*Hydroprogne caspia*

Canning et al. (1979) conducted extensive surveys of the avifauna 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: 26 passerines/upland birds, 3 raptors, 11 shorebirds/waders, 17 waterfowl, and 13 seabirds. Kellogg Island had a much higher diversity of birds than the rest of the LDW because of its seclusion and greater variety of habitats.

Cordell et al. (1999) monitored bird populations monthly from 1995 to 1997 at four sites: two sites in the Upper Turning Basin, one on Kellogg Island, and one at Terminal 105. They recorded 75 species of birds: 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.

LDWG conducted a 4-day survey in June 2004 to identify the presence and quality of spotted sandpiper habitat along the LDW (Windward 2004h), as discussed below in Section A.2.2.4.4. This survey noted the presence of both spotted sandpiper and killdeer in specific locations along the LDW shoreline. Great blue herons, ospreys, and bald eagles were also observed along the LDW during this survey.

The following sections provide a brief summary of site usage by the various types of bird species in the LDW.

A.2.2.4.2 Passerines/upland birds

Thirty-two species of passerine/upland birds have been documented along the LDW (Canning et al. 1979; Cordell et al. 1999). Though generally associated with upland habitats, these birds occasionally forage in the exposed mudflats or use freshwater habitats along the river for bathing (Canning et al. 1979). Because they primarily use upland habitat, passerine birds likely experience very limited exposure to sediments in the LDW.

A.2.2.4.3 Raptors

Eight raptor species have been reported to use the LDW (Cordell et al. 1999). The bald eagle is listed under ESA as a threatened species but is currently under review for delisting. In Washington, it is also listed as a state-threatened species (WDFW 2006). Five nests were occupied within 5 miles of the LDW in 1999 (King County 1999c). The closest known nest is located in West Seattle within 1.6 km (1 mi) of the LDW. One or two pairs of resident eagles may be found in the LDW vicinity during the summer (King County 1999b). Overwintering migrant eagles are routinely observed in the vicinity of the LDW from the beginning of October through late March.

The bald eagle is an opportunistic forager with site-specific food habits based on available prey species (Buehler 2000). Bald eagles ingest dead and live fish, birds, and mammals. In most regions, bald eagles seek out aquatic habitats for foraging and prefer to ingest fish (Buehler 2000). Spawned-out salmon are a particularly important food item for eagles in the Pacific Northwest, though not in the LDW because salmon spawn farther upstream. Of the 45 individual fish identified in a study of prey remains at the base of eagle nest trees throughout Puget Sound, 8 were rockfish, 10 were starry flounder, and the remainder included cod, pollock, hake, cabezon, red Irish lord, sculpin, surfperch, salmon, plainfin midshipman, and channel catfish (Knight et al. 1990). Although eagles feed primarily on fish, birds, such as grebes, gulls, and waterfowl, make up a portion of their diet during winter months. Eagles have been reported to kill western grebe in the Duwamish River during winter (Strand 1999; as cited in King County 1999a). Eagles also have been reported to prey on great blue heron chicks (Norman 1999; as cited in King County 1999c).

Cooper's and sharp-shinned hawks have been observed to overwinter in the LDW. These relatively small raptors generally feed on birds up to the size of quail. They rarely feed on aquatic birds (Canning et al. 1979; Cordell et al. 1999). Red-tailed hawks, a resident species commonly observed along grassland/woodland margins along the LDW, feed primarily on rodents but have been observed pursuing ducklings in the study area. Swainson's hawks and merlin are rare in the LDW and not likely to prey on aquatic associated species (Canning et al. 1979; Cordell et al. 1999).

The US Geological Survey (USGS) observed five osprey nests along the LDW between RM 0.0 and RM 5.0 in 2003, as well as one nest on Harbor Island and three nests along the Duwamish and Green Rivers within about 4 miles of the Upper Turning Basin. Ospreys feed opportunistically and almost exclusively on live fish from fresh or salt water. Ospreys can penetrate about 1 m below the water surface. Therefore, they generally catch pelagic fish or those that frequent shallow flats and shorelines. A west-central Idaho osprey study reported 89% of fish ingested by osprey were 11 to 30 cm long, suggesting a preference for medium-sized fish (Van Daele and Van Daele 1982). A USGS study conducted in the LDW noted that approximately 73% of the prey delivered by osprey to their nests along the LDW were LDW fish species; salmon, perch, and sole/flounder accounted for approximately 33, 25, and 15% of total prey remains, respectively (Henny 2005). Twenty-five percent of the total prey remains were peamouth, a freshwater fish, and two percent were not reported. This dietary information was collected during only a portion of the nesting season. Additional dietary data have been collected and are currently being analyzed, but those data are not yet available (Kaiser 2006).

There are currently two known nesting pairs of peregrine falcons along the LDW; one in a nest box on the West Seattle Bridge, and one in a natural nest on the First Avenue South Bridge (Anderson 2006). Peregrine falcons in western Washington feed primarily on rock pigeons and European starlings, although they may also ingest some waterfowl (Anderson 2006).The peregrine falcon is listed as a federal species of concern and was downlisted from a state endangered species to a state sensitive species in April 2002 (WDFW 2006).

A.2.2.4.4 Shorebirds/waders

Eight species of shorebirds and wading birds have been documented in the LDW (Cordell et al. 1999), including green heron and great blue heron. Of these species, great blue heron make up the only sizeable or consistent population.

The great blue heron is a wading bird that has a range from the coasts of southeast Alaska and Northern British Columbia, through Canada and the US, and south to Belize and Guatemala. The great blue heron is found primarily in natural wetlands and along riverbanks, but can also be found in brackish marshes, lagoons, lakes, and along ocean shores. They were the most abundant shore/wading bird recorded by Cordell et al. (1996) in the LDW, and are a year-round resident. Great blue herons nest in colonies of up to several hundred pairs, preferably on islands or wooded swamps

(Butler 1992). A heron colony of up to 37 active nests was located in West Seattle a few hundred meters from Kellogg Island until 1999, but no successful nesting has occurred there since 2000 (Norman 2006). This colony was likely abandoned because of either eagle predation or disturbance from construction at the nearby Herring's House Park (Norman 2006). Great blue herons are sensitive to interferences during their nesting period, and disturbance by eagles and humans is a common reason for nesting failures (Heron Working Group 2006a). Other heron colonies in the vicinity of the LDW are located about 12 km (7.5 mi) south in Renton and 11 km (6.8 mi) northwest near Salmon Bay.

Great blue herons feed in shallow water primarily on small fish, such as juvenile salmonids, but they also take crustaceans, insects, amphibians, reptiles, and occasionally small mammals (Butler 1992; Kushlan 1978). Great blue herons hunt by sight and stalk or ambush their prey. They also feed by probing, quickly moving their bills in and out of the water and substrate. Great blue herons feed on small fish that range in size from 8 to 33 cm (Kirkpatrick 1940; Alexander 1977; Hoffman 1978). Butler (1992) reports that shiner surfperch is a major food source for female and hatchling great blue herons and may be important for juvenile survival.

The two most common shorebirds observed in the LDW are sandpipers and killdeer. The spotted, least, and western sandpipers are commonly observed in the LDW. These birds feed primarily on insects, small crustaceans and mollusks, worms, and other invertebrates, and rarely on seeds and berries. 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 (Paulson 1993), but are reported to frequent Kellogg Island from September through May (Canning et al. 1979). Most are thought to be migrants, though some may reside in the LDW throughout the winter.

Spotted sandpipers are a common bird in western Washington, and are known to nest along the LDW. They have been observed in the LDW from late June through September (Cordell et al. 1996), but have also been known to overwinter locally (Paulson 1993). Spotted sandpipers breed in open habitats along the margins of water bodies (Oring and Lank 1986) and mostly forage in open areas on the ground within 200 m of the shoreline (Oring et al. 1997). Canning et al. (1979) recorded seven spotted sandpiper nests located on Kellogg Island, and at least three additional nest sites were suspected based on the behavior of adult or juvenile birds. LDWG conducted a survey in June 2004 to identify the presence and quality of spotted sandpiper habitat along the LDW (Windward 2004h). The results of this survey indicate that sandpipers may be nesting along Kellogg Island, at the Herring's House restored marsh area (western shoreline RM 0.5), near the channel beneath the 1st Avenue South Bridge (western shoreline RM 2.1), and along the western shoreline of the Upper Turning Basin (RM 4.6 to RM 4.9).

Killdeer, another common shorebird, feed in intertidal mudflats. Their diet includes small invertebrates, insects, and some vegetative matter. Killdeer are a common bird that uses the LDW year-round. In a year-long survey conducted in 1977-78, 20 to 60 birds were reported to use the Kellogg Island area in the winter and approximately 10 birds in the fall and spring (Canning et al. 1979).[7] Two killdeer nests were observed on Kellogg Island (Canning et al. 1979). During the spotted sandpiper survey conducted in June 2004, killdeer were most frequently observed in the vicinity of Kellogg Island from RM 0.0 to RM 1.0, but were also observed at RM 2.1 to 2.2, RM 4.0 to 4.2, and RM 4.7 to 4.9 (the Upper Turning Basin) (Windward 2004h). Killdeer were confirmed to nest on Kellogg Island based on an observation of a killdeer leaving its nest during the sandpiper survey (Windward 2004h).

A.2.2.4.5 Waterfowl

Cordell et al. (1999) reported 16 species of waterfowl utilizing the LDW, including nine species of dabbling ducks. All species are generally migratory, though some non-migratory populations exist. In general, these birds overwinter in the Puget Sound area (and farther south) and migrate north in the summer. The dabbling ducks feed on aquatic plants, seeds, and grasses and to some extent small aquatic animals and insects. Feeding occurs primarily in shallow water and over intertidal mudflats. A resident population of approximately 25 mallards lives year-round in the LDW, and an additional population of approximately 15 mallards overwinters in the LDW. As many as 290 migratory mallards have been reported to move through the LDW (Canning et al. 1979). The other dabbling duck species use the LDW seasonally for nesting and forage during migration. The most significant of these are gadwalls. Approximately 10 gadwall nests were observed along the LDW in the vicinity of Kellogg Island during a survey conducted in 1977-78 (Canning et al. 1979).

Canvasback, greater scaup, bufflehead, and common and Barrow's goldeneye are reported to use the LDW. These birds dive for small aquatic animals and plants. Canvasback feed primarily on plants, scaup on equal portions of plants and animals, and bufflehead and goldeneyes exclusively on aquatic animals and insects. During a survey conducted in 1977-78, approximately 60 canvasbacks arrived in the LDW in November and departed in late February, using Kellogg Island as their primary feeding area (Canning et al. 1979). During the same survey, greater scaup and common and Barrow's goldeneyes arrived in the LDW in late November and departed by early May, and a small population of approximately eight buffleheads overwintered in the LDW from December to May (Canning et al. 1979). In general, Canning et al. (1979) found that feeding by all diving duck species occurred primarily in the vicinity of Kellogg Island in 1977-78.

All three species of North American mergansers have been recorded to use the LDW, two substantively. Common mergansers were reported to use the LDW from September to March in 1977-78; these birds did not winter on the LDW but were likely

[7] Killdeer were present in the summer, but the number observed was not reported.

migrating through the area (Canning et al. 1979). Approximately 30 red-breasted mergansers were reported to overwinter in the LDW from December 1977 to March 1978 (Canning et al. 1979). These birds feed primarily on small fish and are reported to feed in the deeper water of the channel (Canning et al. 1979).

A resident population of approximately 1,000 Canada geese resides in the vicinity of Lake Washington. The geese foraging in the LDW are thought to be a part of the Lake Washington population. Migratory Canada geese arrive in the LDW in January and February and remain until the end of July as a spring nesting population. Canada geese swim in the LDW and feed in intertidal habitats. They feed primarily on grass and terrestrial vegetation (Canning et al. 1979). In the LDW, 40 to 50 birds overwinter from September to April along Kellogg Island and the west bank of the waterway along the South Park district and in the Upper Turning Basin (Canning et al. 1979).

A.2.2.4.6 Seabirds

Thirteen species of seabird have been recorded in the LDW (Canning et al. 1979; Cordell et al. 1999), including two species of cormorants (pelagic and double-crested). Cormorants feed primarily on small fish and occasionally crustaceans. Wintering cormorants use the LDW from November to May, with large numbers present from December to April (Canning et al. 1979; Cordell et al. 1996).

Several species of gulls are reported to use the LDW. Gulls feed on fish and shellfish and are omnivorous scavengers. Glaucous-winged gulls and mew gulls are the only species reported to use the area in large numbers. Glaucous-winged gulls are reported to use the area throughout the year. Mew gulls frequent the area, occasionally in large numbers, from September through May (Canning et al. 1979).

Caspian terns have been observed near Kellogg Island (Luxon 2004). Pigeon guillemots and common murres have been reported in the LDW; however, their use of the LDW is infrequent. These birds feed primarily on pelagic fish, though bottomfish and crustaceans may also be taken.

Common loons are listed as a state sensitive species (WDFW 2006). They are present in Puget Sound in the winter and use local waters for resting during migrations to and from wintering areas farther south. Their diet consists primarily of small fish and other aquatic animals. Ten to 30 birds have been observed in the Seattle area during annual winter counts, although they are reported to be rare visitors to the LDW (Canning et al. 1979).

Three species of grebe are reported in the LDW. Of these, only western grebes are found in substantial numbers. Grebes and other marine bird species have been declining in recent years (Nysewander et al. 2001). Feeding behavior varies with species. In marine waters, the eared grebe primarily ingests crustaceans while the western grebe favors fish. The most common fish species ingested by western grebes are Pacific herring, pilchard, stickleback, sculpin, sea perch, and smelt. Western grebes occasionally feed on juvenile salmonids. The LDW population was estimated to include about 90 birds in the 1970s (Canning et al. 1979). Grebes arrive in the LDW from October to November and depart by early May.

In summary, the LDW is a corridor frequented by a diverse avian group. It is utilized mostly by shore birds, waders, seabirds, and waterfowl, which feed in areas of mudflats and other shallow-water habitats. Raptors also use the LDW for foraging.

A.2.2.4.7 Mammals

Three marine mammal species may occasionally enter the LDW: harbor seal, California sea lion, and harbor porpoise (Dexter et al. 1981). Harbor seals and California sea lions have been documented in the LDW (WDFW 1999), but recent information on harbor porpoise usage was not available. The harbor seal can be found along both North American coasts (Hoover 1988; Payne and Selzer 1989). Along the Pacific coast they can be found from Alaska to Baja California and mainland Mexico, and are the most commonly observed pinniped species (Hoover 1988). They can be seen in protected harbors year-round (Boulva and McLaren 1979). Harbor seals are commonly seen in Elliott Bay and occasionally enter the LDW (Kenney 1982).

Harbor seals are opportunistic feeders, selecting prey based on availability and ease of capture (Pitcher 1980; Pitcher and Calkins 1979; Schaffer 1989). Their diet can vary seasonally and includes bottom-dwelling fishes, invertebrates, and species that congregate for spawning (Pitcher and Calkins 1979; Everitt et al. 1981; Lowry and Frost 1981; Roffe and Mate 1984). In Washington, the most important prey include Pacific whiting, tomcod, walleye pollock, flatfishes, Pacific herring, shiner surfperch, plainfin midshipman, and sculpins (NMFS 1997). Fish ingested are generally between 40 and 280 mm (Brown and Mate 1983). Harbor seals have been shown to forage over large areas ranging from 5 km (Stewart et al. 1989) to 55 km (Beach et al. 1985).

California sea lions and harbor porpoises are also opportunistic feeders, consuming various fish species depending on availability (Marine Mammal Center 2006). California sea lions and harbor porpoises will, like harbor seals, also feed on non-fish species such as squid and octopus (Yates 1998).

A survey of sea lions and harbor seals was conducted in the LDW from December 1998 to June 1999 (WDFW 1999). This survey monitored the presence of sea lions and harbor seals in the East and West Waterways and in the LDW up to the 16th Avenue South Bridge for a total of 307 hours on 52 days. In the LDW, sea lions were observed on 16 occasions and seals on 17 occasions, with most observations for both species occurring below the 1st Avenue South Bridge. In the East and West Waterways, sea lions were observed 69 times and seals 6 times; both species used the West Waterway most frequently.

Three species of semi-aquatic terrestrial mammals use the LDW: river otters, raccoons, and muskrats. Anecdotal information indicates that a river otter family lives year-round on Kellogg Island in the LDW, although otters were not observed by Cordell during wildlife surveys (Cordell 2001). River otters are almost exclusively aquatic and prefer food-rich habitats such as the lower portions of streams and rivers, estuaries, and lakes and tributaries that feed rivers (Tabor and Wight 1977; Mowbray et al. 1979). Local river otters feed primarily on fish but will also feed on crabs and sometimes mussels and clams (Strand 1999; as cited in King County 1999a). River otters range over an area sufficiently large enough for foraging and reproduction (Melquist and Dronkert 1987); however, they are typically found in a limited number of activity centers within their overall range. In streams, the river otter's home range can average 30 km (Melquist and Hornocker 1983).

Raccoons are reported to be common along the forested ridge slopes to the west of the LDW. Raccoons are scavengers that feed on carrion and occasionally on fish. Muskrat populations are reported to exist at Terminal 107 and at the Upper Turning Basin (Canning et al. 1979). Muskrats are herbivores, feeding primarily on aquatic and semi-aquatic plants.

In summary, the LDW corridor provides habitat for a limited number of mammal species. It may serve as a significant part of the home range of a river otter family, but is used only occasionally as a foraging site by marine mammals.

A.2.3 RECEPTOR OF CONCERN SELECTION

This section presents the ROCs selected to represent benthic invertebrate, fish, and wildlife species based on a set of ROC selection criteria. Inherent to the ROC selection process is the realization that not all species in the LDW can be evaluated individually because of the large number and variety of species present. Instead, representative species are chosen to include species that are most likely to be exposed to contaminated sediment. In this way, species not selected should also be protected. Sensitive species are also preferred, although the relative sensitivity of most species is not known.

A systematic process was followed to select representative species as ROCs based on the available information for the resources presented in Section A.2.2. This process is consistent with SMS, available EPA guidance, and the process commonly used in Superfund risk assessments.

Key considerations in the selection of ROCs included:

- 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)
- Data availability

To ensure that ROCs were selected to represent all potential exposure pathways for sediment-associated COPCs, key direct and indirect exposure routes from sediment were identified (e.g., direct exposure to sediment or ingestion of prey associated with sediment either directly or through prey). Groups of organisms that may be exposed via these pathways were then identified, and representative species that are thought to be most exposed were selected from these groups representing the greatest potential for exposure. Next, human or ecological significance was considered (i.e., species valued by society, have special regulatory status [i.e., threatened or endangered], or serve a unique ecological function).

Site usage, sensitivity to COPCs at the site, and data availability were also evaluated to determine the final list of ROCs. Site usage is an important criterion because it determines the exposure of a species; species that occupy the LDW during a significant part of the year or during sensitive periods, such as gestation and rearing of young, were preferred. Sensitivity to COPCs was evaluated based on available toxicological data, although in many cases the availability of data specific to LDW resident species is low. Therefore, where necessary, toxicological information from surrogate species, or a wide range of species, was used because species-specific data were not available. Finally, data availability regarding both site-specific exposure and effects was assessed, and species for which there are related site-specific data (such as COPC concentrations in food, site usage, and feeding) and toxicological data (such as sediment toxicity tests) were preferred. The following sections provide additional rationale for each of the ROCs selected; a summary of the selected ROCs is provided in Section A.2.3.4.

A.2.3.1 Benthic invertebrate community

The benthic invertebrate community as a whole is evaluated in this Phase 2 ERA as an ROC. As discussed in Section A.2.2.2, a wide variety of benthic invertebrates inhabit the LDW; most of these species are in direct contact with sediment year round and have a limited home range. Benthic invertebrates use various techniques to nourish themselves, and thus may be exposed to sediment through several different pathways (e.g., filter feeder vs. detritus feeder). Benthic invertebrates include sediment dwellers (benthic infauna) and organisms closely associated with the sediment surface (epibenthos).

Benthic invertebrates are an important food source for other invertebrates, fish, birds, and mammals, and perform essential functions such as nutrient cycling. Thus, the diversity and abundance of benthic invertebrates is an important component of the ecosystem. In addition, benthic organisms have been shown to be sensitive to sediment-associated chemicals, and data are available to assess their exposure and predict or measure effects, including sediment criteria from the SMS.

Among the benthic invertebrates, gastropods were considered by EPA to be potentially at risk from tributyltin (TBT).[8] LDWG performed two focused investigations to address this risk (Windward 2004g; 2005j). The scope and findings of these investigations are summarized in Sections A.3.1.3, A.3.2.4, A.6.1.1.3, and A 6.1.3.

SMS, which were used to evaluate risks to the benthic invertebrate community ROC, were not developed to explicitly address issues associated with bioaccumulation of COPCs by benthic invertebrates. Although this issue is implicitly addressed through the incorporation of benthic invertebrate community structure into the overall development of the SMS, the sediment quality standards (SQS) and cleanup screening levels (CSLs) do not specifically address risks to higher-trophic-level benthic invertebrate species, such as crabs. Therefore, crabs were selected as an ROC to represent higher-trophic-level benthic invertebrate species present in the LDW.

Crabs have a relatively larger home range than most of the benthic invertebrate species covered by the SMS. Dungeness crabs are the largest crab species observed in the LDW. Mating usually occurs in offshore locations, but may occasionally occur in estuaries (Pauley et al. 1988). Spawning takes place offshore. Juvenile crabs are found in shallow coastal waters and estuaries, like the LDW, and large numbers can live among eelgrass or other aquatic vegetation.[9]

The diet of crabs is dependent on their life stage (Pauley et al. 1986). Larvae ingest both zooplankton and phytoplankton. The diet of juvenile crabs consists largely of fish, mollusks, and crustaceans. Adult crabs prey on clams, crustaceans, and fish. Crabs eat bivalves their first year, shrimp their second year, and teleost fish their third year. Megalopae[10] are preyed upon by many fish, including juvenile salmon. Juvenile crabs are preyed upon by various demersal fish in the nearshore area. Flatfish, such as starry flounder and English sole, are the most important predators. Adult and juvenile crabs are preyed upon by sea otters, fish, and octopuses. Cannibalism is also common among crabs.

A.2.3.2 Fish

The potential fish ROCs were grouped into the following three broad categories based on potential sediment exposure at the site:

- **Anadromous juvenile salmonids**—representing species such as juvenile chinook, chum, and coho salmon

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[8] Risks to gastropods from exposure to all other chemicals evaluated in this baseline ERA were addressed by the assessment of risks to the benthic invertebrate community.

[9] No eelgrass is found in the LDW, and habitats with aquatic vegetation are rare (Battelle et al. 2001).

[10] Crab larvae progress through five zoeal states before molting into megalopae. Megalopae first appear in April in Washington waters, with abundance peaking in May through June, after which they molt into juveniles.

- **Benthivorous fish—** representing species such as English sole, rock sole, and starry flounder. This category is also protective of fish, such as Pacific herring and pile perch, which prey on pelagic and encrusting organisms.
- **Upper-trophic-level fish —**representing species such as bull trout, sand sole, and Pacific staghorn sculpin

Using the criteria discussed in Section A.2.3, the following fish species were selected as ROCs in the LDW:

- Wild juvenile chinook salmon – anadromous juvenile salmonids
- English sole – benthivorous fish
- Pacific staghorn sculpin – upper-trophic-level fish

The remainder of this section discusses the rationale for selecting each fish ROC and how these species serve as surrogates for protection of other similar and important species within the LDW.

A.2.3.2.1 Wild juvenile chinook salmon

Wild juvenile chinook salmon (*Oncorhynchus tshawytscha*) were selected primarily because the Puget Sound evolutionary significant unit of chinook salmon (to which the Green River belongs) is a federally threatened species under ESA. In addition, they serve as a surrogate for other juvenile anadromous salmon.

Juvenile chinook salmon are representative of other salmonids because they have estuarine residence times similar to or greater than those of other juvenile salmonids in the LDW. During their spring outmigration, juvenile chinook salmon are among the most abundant fish in the LDW and are an important prey item for birds and piscivorous fish in the LDW (Warner and Fritz 1995). Juvenile chum salmon are also present in large numbers in the LDW from April through June and may rear extensively in the LDW (Warner and Fritz 1995). Residence times of all species of juvenile salmonids in the LDW are uncertain; however, juvenile chinook salmon are generally regarded as the most estuarine-dependent juvenile salmonid and their exposure to sediment-associated chemicals is likely equal to or greater than that of other juvenile salmonids.

Juvenile chinook are exposed to sediment-associated chemicals, primarily through their ingestion of benthic invertebrates, which are an important prey item in their early estuarine residence (Cordell et al. 1999). Juvenile chinook salmon have been studied in the LDW, and data are available on their exposure within the LDW, as well as potential effects associated with this exposure. Although toxicity data are available for several salmonid species, there are insufficient data to suggest that any one juvenile salmon species is more sensitive than another; therefore, available toxicity data did not affect selection of juvenile chinook as an ROC over other salmon.

Furthermore, chinook salmon are an icon of the Pacific Northwest. They have been used for centuries by indigenous people as a primary food source and are an economic resource of the region as a commercial fishery species. It is likely that some yearling juvenile chinook salmon (i.e., fish that have reared for one year in fresh water) outmigrate through the LDW (Warner and Fritz 1995; Shannon 2001). Yearling chinook tend to occupy deeper water than subyearling chinook and prey mainly on pelagic organisms, including small fish (Healy 1991). Risks to piscivorous yearling juvenile chinook salmon are assumed to be addressed by the Pacific staghorn sculpin ROC, as discussed below.

A.2.3.2.2 English sole

English sole (*Pleuronectes vetulus*) were selected to represent benthic carnivorous fish in the LDW. English sole live in close proximity to sediment, thus giving them a high potential for direct exposure to sediment-associated chemicals. In addition, English sole feed extensively on sediment-associated invertebrates and thus are subject to exposure to sediment-associated chemicals through their diet.

A number of studies have examined potential effects of sediment-associated chemicals (e.g., polycyclic aromatic hydrocarbons [PAHs]) on flatfish in the LDW, particularly English sole (e.g., Johnson et al. 1997b). Several toxicological studies have used data from English sole collected in the LDW (Casillas et al. 1991b; Johnson and Landahl 1994; Johnson et al. 1988; 1997b; 1998; 1999; Kubin 1997; Malins et al. 1984; 1985a; 1985b; Schiewe et al. 1989). National Marine Fisheries Service (NMFS) data suggest English sole are as sensitive to the effects of PAHs as other flatfish species tested (Myers et al. 1998). English sole are caught recreationally in the LDW and have some value as a commercial fishery species (though not in the LDW). Available toxicity data are not sufficient to suggest that English sole are more or less sensitive than other LDW species represented by English sole. Therefore, except for regionally specific studies conducted with English sole, no preference is given to toxicological data conducted with fish closely related to English sole.

English sole is a surrogate for other benthopelagic, demersal, and benthic fish species. In general, benthic organisms preyed on by other fish in the LDW are similar to those preyed on by English sole (see Section A.2.2.3.3), so their primary exposure route to sediment is similar. Therefore, because of the direct sediment contact, exposure of benthic fish such as English sole to sediment-associated chemicals is assumed to be greater than fish with equivalent prey preferences residing in other LDW habitats.

English sole is one of the most abundant fish in the LDW and is closely related to starry flounder, another of the most abundant fish in the LDW. Exposure and effects studies with English sole should, therefore, be relevant to starry flounder.

Other LDW fish, such as pile perch, ingest organisms that encrust pilings and other vertical structures. However, because these prey organisms do not have direct contact with sediment, this exposure route is not likely to result in greater exposure to sediment-associated chemicals than the ingestion of benthic invertebrates. Similarly, fish species such as herring, surf smelt, longfin smelt, and threespine stickleback that ingest significant quantities of pelagic prey, are likely to have less exposure to sediment-associated chemicals than English sole that consume benthic prey exclusively. The available information thus indicates that the assessment of risks for English sole adequately addresses risks from sediment-associated chemicals to fish with benthopelagic, demersal, and benthic habitat preferences.

A.2.3.2.3 Pacific staghorn sculpin

Pacific staghorn sculpin, a benthic carnivorous fish, were selected to represent upper-trophic-level fish in the LDW. Like English sole, Pacific staghorn sculpin are benthic fish that feed on benthic invertebrates, giving them a high potential for direct and indirect exposure to sediment-associated chemicals.

Pacific staghorn sculpin feed on larger invertebrates than English sole and also feed on fish. Because other upper-trophic-level fish species have foraging ranges that extend beyond the LDW, Pacific staghorn sculpin were selected to represent upper-trophic-level fish in the LDW, thus accounting for exposures to bioaccumulative and biomagnifying chemicals, such as mercury and polychlorinated biphenyls (PCBs). This distinction was made because upper-trophic-level fish may have higher body burdens of biomagnifying chemicals than other fish, such as English sole, that ingest primarily invertebrates at a lower trophic level.

Pacific staghorn sculpin are one of the most abundant fish species in the LDW and likely serve as a food resource to piscivorous wildlife such as bald eagles (Knight et al. 1990), harbor seals (NMFS 1997), and river otters (Strand 1999; as cited in King County 1999a). No toxicity data were identified for Pacific staghorn; therefore, it is unknown whether Pacific staghorn sculpin are more or less sensitive than other higher-trophic fish species.

In the Phase 1 ERA (Windward 2003b), piscivorous bull trout were selected as the ROC to represent upper-trophic-level fish, largely because of their ESA status. Bull trout were replaced as an ROC in Phase 2 because they only occasionally use the LDW,[11] and insufficient tissue could be collected for chemical analysis. Other primarily piscivorous fish, including rockfish and sand sole, were also considered as representative of upper-trophic-level species because adults of these species eat at a high trophic level and rockfish are longer lived than Pacific staghorn sculpin. Available data (presented in Section A.2.2.3 and Attachment 2) and expert opinion (Windward 2004j) suggest that sandsole and rockfish are not abundant in the LDW and their foraging ranges likely extend beyond the LDW.

[11] Bull trout are believed to forage significantly in the LDW only during times of large abundances of small fish, such as just after the live birthing period for perch (Shannon 2001).

Also, subadult rockfish present in the LDW are at a trophic level similar to or lower than Pacific staghorn sculpin. In a study from British Columbia, Murie (1995) found that 80% of subadult copper rockfish (less than 15 cm in length) ingested demersal crustaceans and about 10% of subadult copper rockfish ingested fish. Similarly, about 60% of subadult quillback rockfish (less than 20 cm in length) ingested demersal crustaceans and less than 10% ingested fish. Pacific staghorn sculpin stomach contents analyses from throughout Puget Sound showed that fish constitute from 7 to 48% of the biomass in their diets (Fresh et al. 1979; Wingert et al. 1979; Miller et al. 1977c). Because diets of subadult rockfish are similar to Pacific staghorn sculpin diets, rockfish are assumed to be represented by Pacific staghorn sculpin. Because adult rockfish occur only near Harbor Island (the northern boundary of the LDW) (Windward 2004i) and other piscivorous species (such as bull trout and sand sole) forage extensively outside the LDW, their exposure to LDW-associated chemicals is likely to be less than Pacific staghorn sculpin exposure.

A.2.3.3 Wildlife

The potential wildlife ROCs were grouped into the following three broad categories based on potential sediment exposure at the site:

- **Piscivorous/carnivorous birds**—including species such as great blue heron, western grebe, cormorant, osprey, and bald eagle
- **Benthivorous birds**—including species such as spotted sandpiper, killdeer, and dabbling ducks
- **Piscivorous mammals**—including species such as river otter and harbor seal

Other broad categories of wildlife receptors, such as herbivorous birds, passerine birds, or omnivorous mammals are assumed to be less exposed to COPCs in the LDW than the three categories listed above because of their foraging behavior and diet. Primarily herbivorous birds, such as geese and some diving ducks, may also feed on benthic invertebrates and may incidentally ingest sediment while foraging, but this exposure is assumed to be less than that of benthivorous birds such as shorebirds, which may ingest significant amounts of sediment while probing intertidal sediment for benthic invertebrates. Ingestion of algae is also a potential exposure pathway. The significance of this pathway depends upon the extent to which sediment-associated chemicals migrate through the water column, are taken up by algae, and then ingested in significant quantities. Exposure from ingestion of algae is assumed to be insignificant compared to other more direct pathways examined (e.g., incidental ingestion of sediment or contaminated benthic species). Passerine birds are also likely to experience limited exposure to contaminated sediments in the LDW because they primarily use upland habitat. Other mammals, such as raccoons, are expected to have less exposure to sediment-associated chemicals because their food is largely terrestrial in origin, especially when compared to the primarily piscivorous river otter and harbor seal.

Using the criteria discussed in Section A.2.3, the following wildlife species were selected as ROCs in the LDW:

- Great blue heron—piscivorous birds
- Osprey—piscivorous/carnivorous birds
- Spotted sandpiper—benthivorous birds
- River otter—piscivorous mammals
- Harbor seal—piscivorous mammals

Evaluation of these representative receptors should be protective of other exposed species in the LDW.

The remainder of this section discusses the rationale for selecting each ROC and how these species will serve as representative species for protection of other species within the LDW. Species-specific toxicological data were not available for any of the LDW wildlife species to determine which species might be most sensitive to COPCs, although data for some COPCs are available for mink, which are in the same family as river otter. Therefore, toxicological sensitivity was not considered in the wildlife ROC selection process.

A.2.3.3.1 Great blue heron

The great blue heron (*Ardea herodias*) was selected to represent the piscivorous bird category because they are year-round residents, known to feed in and around the LDW, and nest nearby. Additionally, they are susceptible to biomagnification of certain chemicals because of their trophic position and feeding habits. Site-specific data for chemicals in heron food resources are available. It is assumed that the great blue heron serves as a representative species for waterfowl and seabirds that also feed primarily on fish and aquatic invertebrates (i.e., loons, western grebe, mergansers, double-crested cormorant, pigeon guillemot, Caspian tern, common murre). Great blue herons also attract a high level of societal interest.

A.2.3.3.2 Osprey

The osprey (*Pandion haliaetus*) was selected to represent piscivorous birds in addition to the great blue heron, as well as carnivorous birds such as peregrine falcon and bald eagles. Ospreys were selected because of the high number of nesting pairs along the LDW, their relatively small home range during nesting and fledgling development, and their diet, which is composed almost exclusively of fish. All of these factors can result in higher exposures to sediment-associated chemicals in the LDW when compared to other raptors, such as bald eagles. In addition, site-specific information on osprey feeding preferences is available because ospreys are being studied by USGS in the LDW.

Bald eagle was evaluated in the Phase 1 ERA (Windward 2003b) as the piscivorous bird ROC. Bald eagle has been replaced by osprey as an ROC because of the higher incidence of osprey in the LDW, localized foraging range while nesting in the LDW, diet preferences,[12] and higher ingestion rates (IRs) normalized for body weight. Risk predictions for osprey should be similar or higher than would be estimated for bald eagles because of these factors. In addition, both species have been shown to be sensitive to some chemicals (e.g., DDT and PCBs) (Anthony et al. 1999; Buck 1999; Bowerman et al. 1995; Wiemeyer et al. 1988). While bald eagles are listed under ESA as a federally threatened species, all raptors tend to have high human interest and ecological significance.

A.2.3.3.3 Spotted sandpiper

The spotted sandpiper (*Actitis macularia*) was chosen to represent the benthivorous bird category because it feeds in the intertidal areas of the LDW from June through September and survey data indicate that sandpiper nest near Kellogg Island and perhaps other areas along the LDW. Sandpipers feed on invertebrates by probing the sediment, therefore, potentially ingesting significant quantities of sediment in addition to benthic invertebrates. Sandpipers have a higher incidental rate of sediment ingestion (some sandpiper species can have up to 30% in the diet) than other LDW bird species, including ducks and geese (EPA 1993). It is assumed that because of the high potential exposure through direct ingestion of sediment, protection of the spotted sandpiper will also be protective of other benthivorous birds such as scaup and scoters (i.e., diving ducks), as well as geese and dabbling ducks. Based on available information presented in the *Wildlife Exposure Factors Handbook* (EPA 1993) for birds that ingest aquatic invertebrates, spotted sandpipers have a higher food ingestion rate (FIR) than lesser scaups and mallards, contributing to higher exposure of spotted sandpipers. In addition, spotted sandpiper exposure should be higher than exposure of herbivorous birds such as American coot, American widgeon, mallard, and geese because COPC exposure is likely higher through ingestion of invertebrates than through ingestion of plants.

A.2.3.3.4 River otter

The river otter (*Lutra canadensis*) was chosen from the three semi-aquatic mammals using the LDW (including raccoon and muskrat) to represent the piscivorous mammal group because otters are suspected to be year-round residents that reproduce and feed in and around the LDW. The river otter is susceptible to biomagnification of chemicals because of its high trophic position and feeding habits and is more likely to feed on fish from the LDW than are raccoons or muskrats. Mustelids are also known to be highly sensitive to PCBs and other chlorinated organic compounds. Site-specific data for chemicals in otter food resources are available, as are relevant toxicological data for mustelids. Otters also attract a high level of societal interest.

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[12] Eagles may consume birds and mammals as part of their diet; these dietary components may be less exposed to contaminated sediment in the LDW food web than are fish.

A.2.3.3.5 Harbor seal

The harbor seal (*Phoca vituluna*) was chosen from the three marine mammals using the LDW (including sea lion and harbor porpoise) to represent piscivorous mammals. The harbor seal, like the river otter, is susceptible to biomagnification of chemicals because of its trophic position and feeding habits. Pinnipeds are suspected to be sensitive to PCBs and other chlorinated organic compounds. Site-specific data for chemicals in harbor seal food resources are available. Seals are protected under the Marine Mammal Protection Act, and attract a high level of societal interest. It is assumed that the harbor seal will act as a representative species for other marine mammals, such as sea lions or harbor porpoise, which may infrequently use the LDW.

A.2.3.4 Summary of ROC selection

In summary, the following species were selected as ROCs to represent the range of organisms exposed to sediment-associated chemicals in the LDW:

- Benthic invertebrate community
- Crabs—higher-trophic-level benthic invertebrate
- Juvenile chinook salmon—anadromous juvenile salmon
- English sole—benthivorous fish
- Pacific staghorn sculpin—upper-trophic-level fish
- Great blue heron—piscivorous birds
- Osprey—piscivorous/carnivorous birds
- Spotted sandpiper—benthivorous birds
- River otter—piscivorous mammals
- Harbor seal—piscivorous mammals

The selection criteria for each of the above receptors are presented in Table A.2-10 to summarize the rationale for ROC selection.

Table A.2-10. 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 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 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. 1998)
Juvenile chinook salmon	diet	important prey item for birds/fish; seasonally one of the most abundant juvenile salmonids in the LDW	T&E species; returning adults important to commercial, sport, and tribal fisheries	generally present April-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 in LDW	site-specific data available for food resources	effects data available for other bird species; unknown relative sensitivity of osprey

ROC	Exposure Route	Ecological Significance	Societal Significance	Site Use	Exposure Data Availability	Sensitivity
Spotted sandpiper	diet, incidental sediment ingestion	important role as an intermediate predator	protected under migratory bird treaty	present June to September; nests along LDW	site-specific data available for food resources	effects data available for other bird species; unknown relative sensitivity of sandpiper
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

COPC – chemical of potential concern

LDW – Lower Duwamish Waterway

NMFS – National Marine Fisheries Service

PCB – polychlorinated biphenyl

ROC – receptor of concern

T&E – threatened and endangered (species listed under the Endangered Species Act)

A.2.4 DATA SELECTION, REDUCTION, AND SUITABILITY

This section presents the chemical data available for the LDW and provides an evaluation of the relevance of these data to assess exposure of ROCs to sediment-associated chemicals.

A.2.4.1 Data selection and reduction

Many environmental investigations conducted in the LDW have included the collection of chemistry data from samples of surface sediment, tissue, or water. This section presents the datasets selected for use in the ERA for surface sediments, tissue samples, surface water samples, and porewater samples. The data management rules are presented in Attachment 3.

A.2.4.1.1 Surface sediment chemistry

The baseline surface sediment chemistry memorandum (Windward 2006d) summarizes, in tabular format, all the surface sediment samples that have been collected in the LDW from 1990 to 2005 and identifies those that are included in the baseline surface sediment dataset and those that were excluded, along with the rationale for their exclusion (Windward 2006d). The surface sediment chemistry data used in this baseline ERA are hereafter referred to as the baseline surface sediment chemistry dataset. The baseline surface sediment sampling locations are shown on Map A.2-3. Surface sediment samples (i.e., 15 cm or less) were collected from approximately 570 intertidal locations; the remainder (samples from approximately 750 sampling locations) were collected from subtidal areas.

Surface sediment samples from approximately 1,300 sampling locations were analyzed for PCBs (as Aroclors), 800 for semivolatile organic compounds (SVOCs) (including PAHs and phthalates), and 830 for metals and trace elements. Organochlorine pesticides were analyzed in surface sediment samples from approximately 200 locations. Dioxins and furans were analyzed in sediment samples from 43 locations. A summary of baseline surface sediment chemistry data is provided in Table A.2-11.

Table A.2-11. Summary of studies included in the baseline surface sediment dataset

SAMPLING EVENT	EVENT CODE	YEAR	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	35	Windward (2005b)
LDW RI. Data report: Surface sediment sampling for chemical analyses and toxicity testing.	LDWRI-SurfaceSedimentRound1 & Round2	2005	metals, SVOCs, PCB Aroclors, selected PCB congeners and dioxins and furans on subset of samples, butyltins	160	Windward (2005d; 2005e)

Sampling Event	Event Code	Year	Chemicals	Number of Sampling Locations[a]	Source
Boyer Towing dock replacement	Boyer Towing	2004	metals, SVOCs, PCB Aroclors, TBT	3	WR Consulting (2004)
Slip 4 early action area site characterization	Slip4-EarlyAction	2004	PCB Aroclors, mercury	30	Integral (2004)
Rhône-Poulenc surface/subsurface sampling	RhônePoulenc 2004	2004	metals, organochlorine pesticides, SVOCs	21	EPA (2005b)
Norfolk CSO sediment remediation project five-year monitoring program: Annual monitoring report - year 5, April 2004.	Norfolk-monit7	2004	metals, PCB Aroclors, SVOCs	4	unpublished data from King County
Triad Approach to Characterize PCB in a Washington Riverine Sediment Site	Jorgensen August 2004	2004	metals, PCB Aroclors, SVOCs	43	USACE (2004)
Duwamish/Diagonal Perimeter monitoring – pre-dredge	DuwDiag-October2003	2003	metals, PCB Aroclors, organochlorine pesticides, SVOCs	12	unpublished data from King County
Terminal 117 early action area site characterization	T117 Boundary Definition	2003-2004	PCB Aroclors; metals, TBT and SVOCs on selected samples	54	Windward et al. (2004a; 2004b)
Boeing Plant 2 transformer investigation – Phase 1	Plant 2-Trans-former Phase1	2003	PCB Aroclors	6[b]	Floyd Snider McCarthy (2004)
Norfolk CSO (Duwamish River) sediment cap recontamination. Phase I investigation.	Ecology-Norfolk	2002	PCB Aroclors	17	Ecology (2003a)
Norfolk CSO sediment remediation project five-year monitoring program: Annual monitoring report - year 3, April 2002.	Norfolk-monit5	2002	metals, PCB Aroclors, SVOCs	1	King County (2002)
Norfolk CSO five-year monitoring program, Year Two, April 2001	Norfolk-monit4	2001	metals, PCB Aroclors, SVOCs	1	King County (2001b)
Norfolk CSO five-year monitoring program – Twelve-month post construction	Norfolk-monit3	2000	metals, PCB Aroclors, SVOCs	1	King County (2000c)
Norfolk CSO five-year monitoring program – Supplemental nearshore sampling	Norfolk-monit2b	2000	metals, PCB Aroclors, SVOC	3	King County (2000b)
Outfall and nearshore sediment sampling report, Duwamish Facility	James Hardie Outfall	2000	metals, PCB Aroclors, SVOCs	9	Weston (2000)
Norfolk CSO five-year monitoring program – Six-month post construction	Norfolk-monit2a	1999	metals, PCB Aroclors, SVOCs	2	King County (2000d)
Norfolk CSO five-year monitoring program – Post backfill	Norfolk-monit1	1999	metals, PCB Aroclors, SVOCs	2	King County (1999e)

Sampling Event	Event Code	Year	Chemicals	Number of Sampling Locations[a]	Source
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	251	Weston (1999a)
King County CSO water quality assessment for the Duwamish River and Elliott Bay	KC WQA	1997	metals, PCB Aroclors, SVOCs, tetrabutyltin	14	King County (1999d)
Duwamish Waterway Phase 1 site characterization	Boeing SiteChar	1997	metals, PCB Aroclors, SVOCs	79[c]	Exponent (1998)
Duwamish Waterway sediment characterization study	NOAA SiteChar	1997	total PCBs, selected PCB congeners, total polychlorinated terphenyls	299	NOAA (1997; 1998)
Seaboard Lumber site, Phase 2 site investigation	Seaboard-Ph2	1996	metals, PCB Aroclors, SVOCs	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	36	Weston (1998)
Duwamish/Diagonal cleanup Study – Phase 2	Duw/Diag-2	1996	metals, PCB Aroclors, SVOCs	10	King County (2000a)
Duwamish/Diagonal cleanup Study – Phase 1.5	Duw/Diag-1.5	1995	metals, PCB Aroclors, SVOCs	9	King County (2000a)
Norfolk CSO sediment cleanup study – Phase 3	Norfolk-cleanup3	1995	PCB Aroclors, SVOCs	12	King County (1996)
Norfolk CSO sediment cleanup study – Phase 2	Norfolk-cleanup2	1995	metals, organochlorine pesticides, PCB Aroclors, selected PCB congeners, SVOCs	2	King County (1996)
RCRA Facility Investigation Duwamish Waterway sediment investigation, Plant 2 – Phase 2a	Plant 2 RFI-2a	1995	metals, PCB Aroclors SVOCs	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	66	Weston (1998)
Duwamish/Diagonal cleanup Study – Phase 1	Duw/Diag-1	1994	metals, organochlorine pesticides, PCB Aroclors, SVOCs	31	King County (2001a)
Norfolk CSO sediment cleanup study – Phase 1	Norfolk-cleanup1	1994	metals, organochlorine pesticides, SVOCs, PCB Aroclors	13	King County (1996)
Rhône-Poulenc RCRA Facility Investigation for the Marginal Way facility – Round 2	Rhône-Poulenc RFI-2	1994	SVOCs	6	Rhône- Poulenc (1995)
Results of sampling and analysis, sediment monitoring plan, Duwamish Shipyard, Inc.	Duwamish Shipyard	1993	metals, SVOCs, TBT	1	Hart Crowser (1993)

Sampling Event	Event Code	Year	Chemicals	Number of Sampling Locations[a]	Source
Harbor Island Remedial Investigation	Harbor Island RI	1991	metals, organochlorine pesticides, PCB Aroclors, SVOCs, VOCs, TPH, TBT	9	Weston (1993)

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

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

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

CSO – combined sewer overflow
EPA – US Environmental Protection Agency
KC – King County
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
USACE – US Army Corps of Engineers
VOC – volatile organic compound
WQA – water quality assessment

Since the RI/FS began in December 2000, there have been two sediment removal actions at early action areas within the LDW (Duwamish/Diagonal and Norfolk areas). There is no EPA policy or guidance about whether baseline risk assessments should include or exclude risk reduction achieved by removal actions that occur during the RI/FS. LDWG plans to evaluate the risk reduction achieved at these two areas in the FS as part of the residual risk assessment. Therefore, data characterizing areas prior to early actions are included to represent baseline conditions; post-removal action data collected from within and adjacent to early actions were excluded from the baseline dataset but will be included in the FS.

Both intertidal and subtidal sediment chemistry data were used in this baseline ERA. The elevation dividing intertidal and subtidal locations was approximately –2 ft MLLW, which corresponds to the shoreline (i.e., land/water interface) elevation defined by the aerial photos taken by the US Fish and Wildlife Service (USFWS) in 1999 (2000c).

A.2.4.1.2 Tissue chemistry

Tissue chemistry data for the study area are available for several different tissue types from several sampling events conducted since 1995. Site-specific tissue chemistry data were used in the ERA for the following species: juvenile chinook salmon, English sole, starry flounder, shiner surfperch, Pacific staghorn sculpin, Dungeness crab, slender crab, mussels, clams, and benthic invertebrates (Table A.2-12; Map A.2-2).

Over 200 composite samples of crab, English sole, starry flounder, Pacific staghorn sculpin, shiner surfperch, and benthic invertebrate tissue were used in this ERA (Table A.2-12). PCBs, as Aroclors, were analyzed in almost all samples. Organochlorine pesticides and SVOCs, metals including mercury, and TBT, were also analyzed frequently in approximately 150 to 180 samples. Unless otherwise noted, conversions between wet weight and dry weight were based on sample-specific moisture content. Three studies were conducted with limited analyte lists (King County 1999d; Windward 2002, 2006b). The goals of these studies did not require the analysis of all SMS chemicals. Methylmercury and chromium VI were not analyzed in any of the tissue samples. Measurements of total mercury and chromium were determined to be sufficient in each study. Chemicals that have not been analyzed in LDW tissue samples but were analyzed in surface sediments samples will be discussed in the uncertainty analysis (Section A.6.0). Collection locations for LDW tissue samples listed in Table A.2-12 are presented on Map A.2-2.

Table A.2-12. LDW tissue data used in this baseline ERA

Study	Sample Collection Year	Species	Number of Samples	Individuals per Sample	Sample Type	Chemicals
Chemical analyses of fish and crab tissue samples collected in 2005 (Windward 2006b)	2005	English sole	21[a]	5	whole body	PCB Aroclors
		Dungeness crab	3	5	edible meat[b]	
			3	5	hepatopancreas[b]	
		slender crab	1	5	edible meat[b]	
			1	10	hepatopancreas[b]	
		shiner surfperch	22	10	whole body	
		Pacific staghorn sculpin	4	10	whole body	
Chemical analyses of fish and crab tissue samples collected in 2004 (Windward 2005c)	2004	English sole	21	5	whole body	metals, SVOCs, organochlorine pesticides, PCB Aroclors (PCB congeners in subset of samples), TBT
		starry flounder	3	5	whole body	
		Dungeness crab	7	5	edible meat[b]	
			3	6 – 15	hepatopancreas[b]	
		slender crab	12	5	edible meat[b]	
			4	15 – 18	hepatopancreas[b]	
		shiner surfperch	24	9 – 10	whole body	
		Pacific staghorn sculpin	24	7 – 10	whole body	
Chemical analyses of benthic invertebrate and clam tissue samples and co-located sediment samples (Windward 2005b)	2004	benthic invertebrates	20	nd	whole body	metals, SVOCs, alkylated PAHs, organochlorine pesticides, PCB Aroclors (PCB congeners in subset of samples), TBT
		clams	14	19 – 52	whole body	
Lower Duwamish Waterway remedial investigation. Juvenile chinook salmon data report (Windward 2004c)	2003	chinook salmon	28	9 – 10	whole body	metals, SVOCs, alkylated PAHs, organochlorine pesticides, PCB Aroclors, TBT
			1	74[d]	stomach contents	
Tissue chemistry results for juvenile chinook salmon collected from Kellogg Island and East Waterway (Windward 2002)	2002	chinook salmon	6	6 – 7	whole body	mercury, PCB Aroclors

Study	Sample Collection Year	Species	Number of Samples	Individuals per Sample	Sample Type	Chemicals
King County Combined Sewer Overflow Water Quality Assessment for the Duwamish River and Elliott Bay (King County 1999d) [c]	1996-1997	Dungeness crab	2	3	edible meat[b]	metals, SVOCs, PCB Aroclors, TBT
			1	3	hepatopancreas[b]	
		shiner surfperch	3	10	whole body	
		mussels	22	50 – 100	whole body	
		amphipods	4	nd	whole body	

[a] Whole-body samples include 11 composite samples analyzed as whole bodies and 10 estimated composite samples. Estimated English sole whole-body concentrations were based on the relative weights and total PCB concentrations in skin-on fillet and remainder tissues collected in 2005.

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

[c] Approximately 30 additional mussel samples, beyond those indicated in the table, were analyzed as part of the caged mussel deployment designed to assess effects from the combined sewer overflows (King County 1999d). These data are not included in this ERA because they are not representative of concentrations in mussels that would be ingested by wildlife.

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

nd – no data

PAH – polycyclic aromatic hydrocarbon

PCB – polychlorinated biphenyl

SVOC – semivolatile organic compound

TBT – tributyltin

Not all of the available tissue chemistry data were appropriate for use in this baseline ERA. Juvenile chinook salmon whole-body and stomach content data from Varanasi et al. (1993) and from NMFS (2002) were not used in the ERA because the quality assurance/quality control (QA/QC) data were not readily available for EPA review. The allocation of resources to further review these data was determined to not be warranted because both total PCB and total DDT concentrations in the NMFS dataset were similar to or lower than concentrations in LDWG data (Windward 2005h). However, total PAH concentrations in LDW juvenile chinook salmon stomach contents from three separate NMFS studies (Arkoosh et al. 1998; McCain et al. 1990; Stein et al. 1995) were higher than those in juvenile chinook salmon stomach contents collected in support of the Phase 2 RI from similar locations in the LDW (Windward 2004c).

Data associated with two individual shiner surfperch collected in Slip 4 (NMFS 2002) and 10 individual shiner surfperch collected and analyzed by LDWG in subarea T2E (RM 2.1 to 2.4) (Section 4 of the RI) were also excluded. These data were excluded because only data from composite samples were included in the ERA, as per EPA (2000c) guidance. Total PCB concentrations in the two individual shiner surfperch from Slip 4 were 940 and 2,100 µg/kg ww, and the total PCB concentrations in the 10 individual shiner surfperch from subarea T2E ranged from 172 to 1,140 µg/kg ww. These individual shiner surfperch concentrations were lower than the T2E composite sample concentrations. However, the individual sample concentrations were within the range of concentrations in shiner surfperch composite samples (530 to 18,400 µg/kg). The inclusion of these samples would not have affected the results of the risk assessment. The available data for individual shiner surfperch samples were also deemed to be insufficient to estimate the variability in individuals in composite samples (EPA 2005a).

In addition, some site-specific tissue chemistry data were also excluded from the ERA because they are unrelated to site-specific contamination (e.g., data for adult salmon) or because the samples were not appropriate for ecological receptors (e.g., fillet samples).

The only available bird egg data were heron egg data collected by USFWS (Krausmann 2002a). There were insufficient QA/QC data available to include the data in the risk characterization.

A summary of LDW tissue chemistry data used in this baseline ERA is provided in the Phase 2 RI. The RI report also contains a compact disc with all raw data used in the RI and risk assessments.

A.2.4.1.3 Water chemistry

Water data are available from sampling events conducted by King County in 1996 and 1997 during their WQA (King County 1999d) and during an additional sampling event in 2005 in which water samples were collected and analyzed only for PCBs and conventional parameters (Mickelson and Williston 2006).

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). At the Brandon and Southwest Michigan areas, there were three sampling locations, corresponding to the east and west banks and the center of the channel. Two depths were sampled at each location: 1 m below the surface and 1 m above the bottom. There were only two sampling locations at Norfolk, corresponding to the east and west banks, because of the narrow width and depth of the LDW at this location. Samples were collected weekly from October 1996 to June 1997, except during storm events, in which case sampling was conducted on three successive days following the storm. Chemicals relevant to the risk assessment that were analyzed in these samples are metals and trace elements (total and dissolved concentrations of antimony, arsenic, cadmium, cobalt, copper, lead, mercury, nickel, selenium, silver, thallium, vanadium, and zinc) and SVOCs.

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. 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 16th Avenue South Bridge. All 209 PCB congeners were analyzed using high resolution methods, and the total PCB concentration for the risk assessment was calculated as the total of all detected PCB congener concentrations (data were blank-qualified, and RLs for undetected PCB congeners were not included).

King County also deployed semipermeable membrane devices and analyzed samples for organochlorine pesticides and PAHs. Although predicted water concentrations are available from these data, they were not used in the risk assessment.[13]

A.2.4.1.4 Porewater chemistry

The concentrations of volatile organic compounds (VOCs) in porewater were available from a Phase 2 porewater collection effort conducted by LDWG (Windward 2006a). Groundwater data for VOCs from 11 sites along the LDW were reviewed to determine which sites had the highest potential for VOC concentrations in porewater. As documented in the porewater QAPP (Windward 2005i), two sites (Great Western International [GWI] and Boeing Plant 2/Jorgensen Forge) were selected for porewater sampling and analysis (Windward 2006a). Porewater was sampled at multiple locations within each site using two methods. Piezometers were deployed at 12 locations and peepers were deployed at 16 locations (Windward 2006a). Samples were analyzed for 71 VOCs. No VOCs were detected in the piezometer samples, and 16 VOCs were detected in at least one of the peeper samples.

In 1997, porewater samples were collected at 15 locations throughout LDW subtidal areas as part of the EPA Site Inspection for the LDW (Weston 1999b). The samples were analyzed for a suite of metals and TBT. These data were not used in the risk assessment because benthic invertebrate exposure from metals in porewater was addressed in the sediment exposure evaluation by using SMS, toxicologically based DMMP guidelines, and TRVs. TBT porewater data were not used for the assessment of gastropods because TBT porewater data were not available from the imposex locations. Exposure of other invertebrates to TBT was addressed by the critical tissue-residue approach.

A.2.4.2 Suitability of data for risk assessment

There are several factors to consider in assessing the suitability and sufficiency of environmental data for risk assessments (EPA 1989, 1990). Of primary importance is the degree to which the data adequately represent site-related contamination, and the expected exposure of ecological receptors at the site. Other important considerations are data quality criteria goals, documentation, analytical methods, reporting limits (RLs), and level of review associated with the data. Because data from many different investigations were available for the LDW, these factors were evaluated for each data set to determine whether it was reasonable to combine these data for use in this ERA.

[13] Water data were only used for wildlife exposure estimates. Predicted PAH water concentrations from SPMDs were not used because PAHs did not screen in as COPCs for wildlife. Predicted pesticide water concentrations (i.e., total DDTs) from SPMDs were not used because of uncertainties in calculating the predicted concentration, and because the contribution of COPCs from water to the total exposure dose was very small and was not expected to affect risk calculations (see Section 5.1.3.3).

A.2.4.2.1 Representativeness to site-related contamination and receptor exposure

This section provides an overview of the representativeness of the available sediment, tissue, and water data.

Sediment

Many historical environmental sampling events included collection of sediment from the LDW (Table A.2-11). The studies have been designed for both reconnaissance (e.g., Boeing SiteChar, EPA SI, and NOAA SiteChar) and focused investigation of suspected areas of contamination (e.g., Boeing Resource Conservation and Recovery Act [RCRA] facility investigation [RFI], Rhône-Poulenc RFI). The extensive coverage of the reconnaissance surveys, and the focused intensity of facility investigations, indicate available sediment chemistry data are sufficiently representative of the general range of environmental conditions within the LDW. Because samples of environmental media have often been collected to characterize particularly contaminated areas, the dataset as a whole may have a relatively higher proportion of elevated concentrations than is representative of the LDW as a whole.

Existing data used in Phase 1 were supplemented with Phase 2 surface sediment data. Phase 2 surface sediment samples were collected from more than 165 stations according to the following criteria:

- Low historical spatial coverage, particularly at sites where single SQS or CSL exceedances were observed with few nearby sampling locations
- Special use areas (e.g., intertidal areas with public access or used by wildlife) that had previously been incompletely characterized
- Proximity to potential historical or current chemical sources, including seeps identified as being of concern
- Historical sediment chemical concentrations above the SQS or CSL
- Analyte considerations including chemicals with relatively low numbers of historical samples or historical locations that did not have sufficiently low RLs for certain chemicals

Tissue

To be representative, tissue data must provide a reasonable indication of COPC exposure by ROCs at a site. Key considerations in the representativeness of site data are:

- Representativeness of the tissue data with respect to capture location, timing, and home range of the species relative to the site
- Availability of tissue data for COPCs at the site
- Representativeness of tissue data with respect to ROCs at the site and their primary prey items

The home range of fish collected in the LDW may be greater or smaller than the area of the LDW Superfund site. For English sole, for example, considerable uncertainty exists regarding preferred foraging habitat and home range; no site-specific home range estimates have been published for English sole in the LDW. A few home range estimates have been developed using best professional judgment, such as the 9 km^2 home range of English sole, as reported in the Puget Sound Dredged Disposal Analysis (PSDDA 1988a) and 2 km^2 based on a literature review (Stern et al. 2003). One tagging study (Day 1976) suggests that English sole may have some site fidelity, although the "sites" defined in that study are relatively large compared to the LDW. Also, the extent of migration was not established.

When the home range of a particular species does not match the LDW site boundaries, measured body-burdens may over- or underestimate contamination associated with the site. For example, in winter, English sole migrate to Elliott Bay to spawn (Forrester 1969) and, therefore, could be exposed to some chemicals outside of the LDW. Also, juvenile chinook salmon pass through the LDW in their migration from either upstream spawning locations or hatcheries. Juvenile chinook salmon released from hatcheries have a small contaminant load before entering the LDW, which is generally attributed to the low concentrations of various chemicals (e.g., PCBs) present in hatchery feed (Easton et al. 2002). As such, a portion of their overall contaminant body burden is not associated with LDW exposure (Meador 2000).

Also, the age of the fish captured can influence the body burden. Older fish tend to have higher concentrations of biomagnifying COPCs in their tissues, and thus ingestion of these individuals could result in higher exposures to piscivorous receptors. The available English sole, Pacific staghorn sculpin, and perch data represent adult fish.

Uncertainty in exposure assumptions associated with tissue data used in the ERA is discussed further in the uncertainty analysis (Section A.6.2).

Water

The only use of the surface water data in this baseline ERA was to estimate a relatively small percentage of the exposure of wildlife receptors to COPCs through the ingestion of water. Water data for all COPCs, except PCBs, were based on a large dataset of samples collected on a weekly basis from October 1996 to June 1997 along transects at three areas in the LDW. The PCB data were collected from a downstream location and a location midway through the LDW (RM 3.3), and were collected four times in 2005 to represent a seasonal range of flow conditions. The water data are assumed to represent conditions to which wildlife receptors can be exposed in the LDW because the LDW is a well-mixed system.

Porewater

Porewater data were collected from two areas of the LDW to investigate potential worst case exposure conditions for VOCs to assess benthic invertebrate risk. Porewater samples were collected using peepers deployed 10 cm beneath the sediment surface and left to equilibrate for two 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.

A.2.4.2.2 QA/QC results

All datasets used in this ERA have been validated by the original study authors or by outside third parties. No additional data validation was performed for this ERA. Some results were qualified as unusable[14] by the data validators. Data qualified as unusable were not used in this ERA and were not included in summary information provided in Tables A.2-11 and A.2-12.

Analytical interference with the quantification of organochlorine pesticides from the presence of PCB congeners occurred during the pesticide analysis of benthic invertebrate tissue (Windward 2005b) and fish and crab tissue (Windward 2005c). This issue was identified by both the analytical laboratory and the data validators. The organochlorine pesticides were analyzed using EPA Method 8081 (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 invertebrates and clams (Windward 2005b) and fish and crab samples collected in 2004 (Windward 2005c) 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 1999b). These data were qualified based on the probable interference in the analysis from PCB congeners.

The JN-qualified results are highly uncertain and biased high. The high bias for DDTs was confirmed by reanalyzing 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 that is not susceptible to analytical interference by PCBs for organochlorine pesticides. The GC/MS method is less sensitive than EPA Method 8081, and therefore, could not be used for the original analyses and could only be used for confirmation in the high concentration samples. The only DDT isomers that were detected in the confirmation analyses were 4,4′-DDE and 4,4′-DDD. The four isomers most commonly detected in the GC/ECD analyses were 4,4′-DDT, 2,4′DDT, 4,4′-DDE, and 4,4′-DDD; the highest concentrations were reported for the two DDT isomers.

The confirmation analysis results confirmed the JN-qualification of the original sample results. Specifically, all the results from the confirmation analyses were lower than the original results. The total DDT concentrations in the confirmation analyses ranged from 4% to 60% of the original sediment results (Windward 2005g). The two highest

[14] Approximately 1,000 results were qualified as unusable out of more than 140,000 analytical results.

total DDT concentrations were not detected in the confirmation analyses with RLs that were 3% and 8% of the original results. Eight fish and crab tissue samples (three crab hepatopancreas, four shiner surfperch, and one English sole whole body) were reanalyzed and the reanalysis confirmed the presence of DDTs in six of the samples with total DDT concentrations that were 5 to 34% of the original results (Windward 2005g). Thus, the original reported concentrations of DDT compounds appear to reflect the presence of both PCB congeners and DDT isomers in the sample, and were elevated because of analytical interference.

The DDT confirmation analyses were run using subsamples of the original sample extracts that had been archived frozen. The confirmation analyses were conducted six months after extraction, which greatly exceeds the maximum 40-day extract holding time. Therefore, the results of these analyses were treated as qualitative and useful as an estimate of the DDT isomer concentrations, but were not incorporated into the project database, or used in the risk calculations presented in this ERA.

The JN-qualified organochlorine pesticide results were used in the problem formulation screens to identify COPCs for each receptor (Section A.2.5). However, because of the high uncertainty inherent in the organochlorine pesticide concentrations in tissue samples, the characterization of risks associated with tissue organochlorine pesticide results is presented in the relevant uncertainty analysis sections rather than the risk characterization sections to highlight the uncertainty in the data.

A.2.5 SELECTION OF CHEMICALS OF POTENTIAL CONCERN

This section presents a risk-based screen conducted to identify which chemicals should be identified as COPCs for each of the ROCs. Through this COPC screening process, a clear distinction can be made between chemical/receptor pairs that should be evaluated in more detail using more realistic assumptions (Sections A.3.0 through A.6.0) and pairs for which no additional analysis is warranted. The COPC selection process is presented separately in this section for benthic invertebrates, fish, and wildlife ROCs. A brief summary of the ecotoxicology of potential COPCs was presented in the Phase 1 ERA (Windward 2003b).

Quantitative ecological risk analyses were not conducted for dioxins and furans in this baseline ERA, and dioxins and furans were not selected as COPCs for any ecological receptor. Because dioxin and furan congeners were not analyzed in tissue samples collected from the LDW, risks to ecological receptors associated with exposure to dioxins and furans are unknown.

Risks resulting from exposure to dioxins and furans were not assessed because dioxins and furans were not analyzed in tissue samples collected from the LDW for the following reasons:

- Risks from dioxins and furans were assumed to be unacceptable in the LDW, particularly from a human health seafood-consumption perspective. Dioxins and furans are commonly detected in all media regardless of whether there are any local sources of dioxins and furans, and dioxins and furans are toxic to humans, fish, and wildlife species.
- Because of the ubiquity of dioxins and furans in the environment, it is important to consider the degree of contamination of sediments with dioxins and furans in a regional context. In accordance with MTCA and CERCLA regulations and guidance regarding use of information from background areas in contaminated site investigations, comparisons with background are necessary to understand exposures that may have resulted from site-specific releases relative to exposures associated with the general urban environment. Few tissue data are available for dioxins and furans within the greater Seattle metropolitan area and Puget Sound. Therefore, the collection of a large background tissue dataset, in addition to extensive tissue collection within the LDW, would have been required to conduct a robust comparison between the LDW and the general urban environment.

Sediment data for dioxins and furans are available for both the LDW site and background areas. However, sediment data were not considered sufficient for evaluating risks associated with seafood consumption to either human or ecological receptors because of uncertainties associated with predicting tissue concentrations when only sediment data are available. Although the concentrations of dioxins and furans in most sediment samples from the LDW were comparable to those found throughout the Seattle area (Section 4 of the RI), dioxins and furans were detected in sediment in a few samples at concentrations that were much higher than the concentrations detected at upstream or other locations in the greater Seattle metropolitan area.

Risk management decisions to address the dioxin and furan contamination in LDW sediments will be based on MTCA and CERCLA regulations and guidance. Remedial decisions will be made by EPA and Ecology as part of the FS process and will be documented in the Record of Decision. Additional information regarding the nature and extent of dioxins and furans in LDW sediments is presented in the Phase 2 RI and in the baseline HHRA (Appendix B).

A.2.5.1 Benthic invertebrates

This section presents the screening approach that was used to identify COPCs for the benthic invertebrate community and for crabs. The benthic invertebrate community screen was conducted separately for two media: sediment and porewater. The sediment screen was conducted to identify COPCs in sediment for which SMS or other sediment-based guidelines are available. VOCs were screened using porewater data. VOCs may be present in sediment porewater in areas associated with groundwater discharge to the LDW if there is a continuing source of VOC-contaminated groundwater (Church et al. 2002; Chadwick et al. 1999). VOCs do not have a high affinity for sediment because of their generally low OC/water partition coefficients (K_{OC}) (Mabey et al. 1982). Therefore, the exposure of sediment-dwelling organisms (i.e., benthic invertebrates) to VOCs is most appropriately assessed through analysis of sediment porewater rather than bulk sediment. COPCs for crabs, also presented in this section, were identified based on a critical tissue-residue approach.

A.2.5.1.1 Screening methods and results for benthic communities based on surface sediment chemistry data

The Phase 2 work plan established the use of sediment numerical chemical standards promulgated under the Washington SMS to evaluate whether individual sediment-associated chemicals should be retained as COPCs for benthic invertebrates in this ERA. Thus, surface sediment baseline data described in Section A.2.4.1.1 were compared to SQS for all chemicals listed in the SMS. For chemicals with no SQS and detected in at least 5% of LDW surface sediment samples, maximum concentrations were compared to DMMP guidelines (USACE et al. 2000) that were determined to be toxicologically based (Figure A.2-1). In cases where no DMMP value was available or the available DMMP value was not toxicologically based (i.e., total DDTs), the scientific literature was searched for toxicologically based toxicity reference values (TRVs) for these chemicals. Chemicals exceeding the DMMP guidelines or toxicologically based TRVs were identified as COPCs for benthic invertebrates.

Chemicals detected in greater than 5% of the sediment samples, but without SMS or toxicologically based guidelines or TRVs, are discussed in the uncertainty analysis (Section A.6.1.1.1). Chemicals with RLs greater than the SQS are discussed in the uncertainty analysis and in the Phase 2 RI.

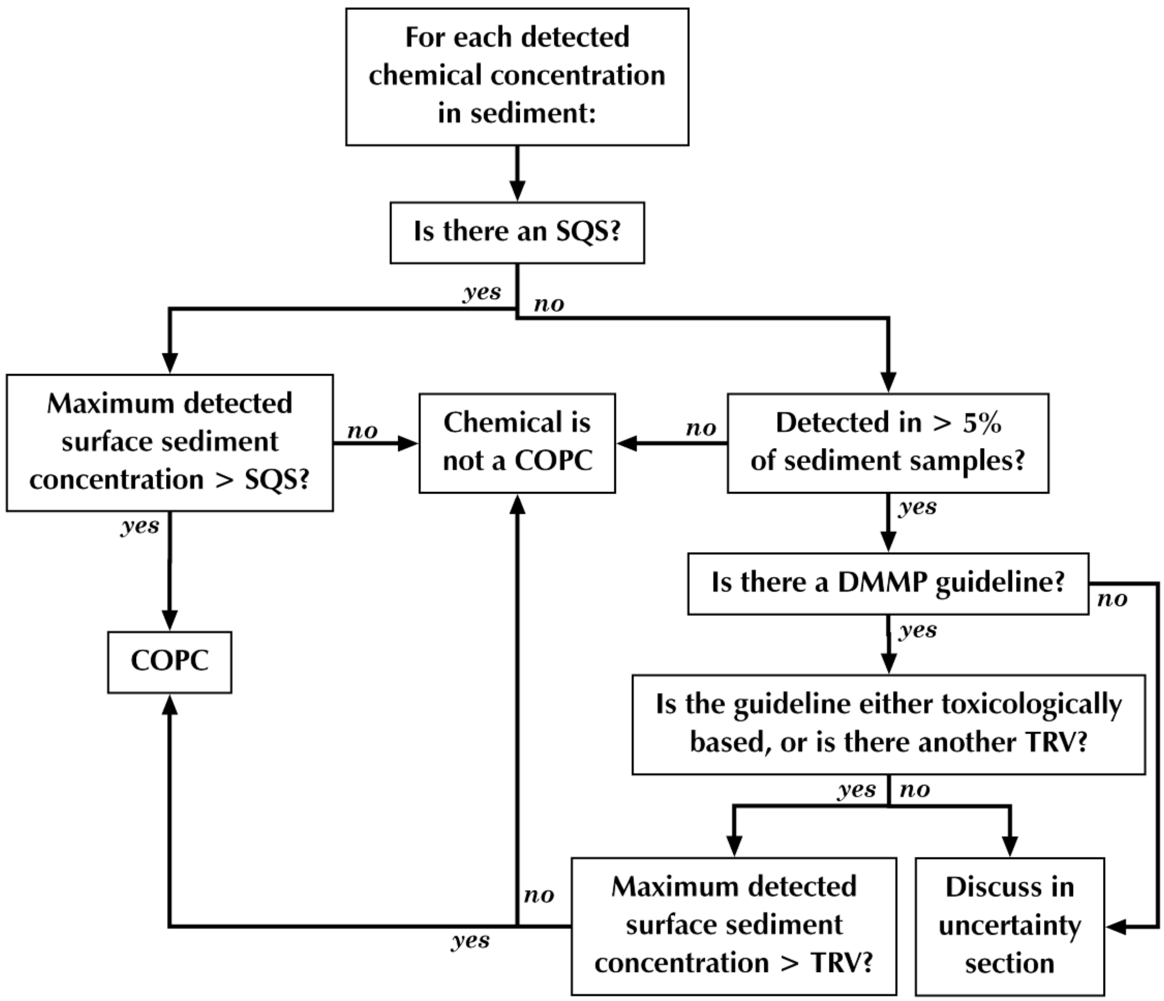

Figure A.2-1. Screening process for benthic invertebrate COPCs using sediment chemistry data

As previously discussed, the SQS were promulgated to address risks to benthic invertebrate communities as a whole, except for higher-trophic-level invertebrates, such as crabs, that may be at greater risk of exposure through bioaccumulation.[15] SQS values are based on apparent effects thresholds (AETs), which are defined as the highest "no effect" chemical concentration above which a significant adverse biological effect always occurred among the several hundred samples used for its derivation. Biological endpoints included in derivation of the SQS chemical standard were field measures of benthic infaunal abundance, laboratory toxicity tests with marine benthic invertebrate organisms (i.e., amphipods [survival] and oysters [percent abnormal development of oyster larvae]), and laboratory toxicity tests with bacteria (Microtox [decrease luminescence from the bacteria *Vibrio fisheri*]). Representatives of these groups are found throughout the LDW. Under the provisions of the SMS, surface sediments with chemical concentrations equal to or less than all the SQS are designated as having no adverse effects on biological resources (Washington Administrative Code [WAC] 173-204-310(1)(a)).[16]

The baseline surface sediment data described in Section A.2.4.1.1 were compared to SQS for all chemicals listed in Ecology's SMS. Many SQS values are expressed as concentrations normalized to total organic carbon (TOC). At very low or high TOC concentrations, normalization is not appropriate (Michelsen and Bragdon-Cook 1993). Concentrations of organic chemicals were not normalized to TOC for samples with TOC concentrations less than or equal to 0.5% or greater than or equal to 4.0%. In these cases, dry weight chemical concentrations were compared to the lowest AET (LAET), which is functionally equivalent to the SQS, and the second lowest AET (2LAET), which is functionally equivalent to the CSL.

Table A.2-13 presents the chemicals identified as COPCs for benthic invertebrates and retained for further evaluation in this baseline ERA. A total of 44 chemicals were retained as COPCs for benthic invertebrates.[17] In addition, TBT was also included as a COPC for benthic invertebrates in this baseline ERA based on Phase 1 ERA (Windward 2003b) results and the focused investigations regarding imposex in gastropods conducted for this baseline ERA.

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[15] Risks to crabs were evaluated using a tissue approach in Sections A.3.3, A.3.4, and A.6.1.2. Also, SMS are not intended to be protective of other receptors (such as fish and wildlife) exposed to sediment-associated COPCs through bioaccumulation. Risks to these receptors are presented in Sections A.6.2 and A.6.3.

[16] Because of the SQS derivation process, there is some uncertainty regarding the prediction of effects based solely on comparison with the SQS, even though this is designated under the provisions of the SMS.

[17] Six SMS chemicals were screened out: 2-methylphenol, acenaphthylene, diethyl phthalate, di-n-butyl phthalate, di-n-octyl phthalate, and hexachlorobutadiene.

Table A.2-13. Summary of COPCs retained for benthic invertebrates based on surface sediment chemistry data

COPC[a]	Maximum Concentration	Unit	Maximum Concentration in Comparable Units	Unit	Screening Criteria: SQS, LAET, SL, or Toxicologically Based TRV	Screening Criteria: Unit
Metals						
Arsenic	1,100	mg/kg dw	1,100	mg/kg dw	57	mg/kg dw
Cadmium	120	mg/kg dw	120	mg/kg dw	5.1	mg/kg dw
Chromium	1,100 J	mg/kg dw	1,100 J	mg/kg dw	260	mg/kg dw
Copper	12,000 J	mg/kg dw	12,000 J	mg/kg dw	390	mg/kg dw
Lead	23,000	mg/kg dw	23,000	mg/kg dw	450	mg/kg dw
Mercury	4.6 J	mg/kg dw	4.6 J	mg/kg dw	0.41	mg/kg dw
Nickel	910	mg/kg dw	910	mg/kg dw	140[b]	mg/kg dw
Silver	270	mg/kg dw	270	mg/kg dw	6.1	mg/kg dw
Zinc	9,700	mg/kg dw	9,700	mg/kg dw	410	mg/kg dw
Organometals						
TBT	3,000	µg/kg dw	na[f]	na[f]	na[f]	na[f]
PAHs						
Acenaphthene	5,200	µg/kg dw	260	mg/kg OC	16	mg/kg OC
Anthracene	10,000	µg/kg dw	380	mg/kg OC	220	mg/kg OC
Benz(a)anthracene	8,400	µg/kg dw	440	mg/kg OC	110	mg/kg OC
Benzo(a)pyrene	7,900	µg/kg dw	420	mg/kg OC	99	mg/kg OC
Benzo(g,h,i)perylene	3,800	µg/kg dw	180	mg/kg OC	31	mg/kg OC
Chrysene	7,700	µg/kg dw	410	mg/kg OC	110	mg/kg OC
Dibenzo (a,h)anthracene	1,500	µg/kg dw	71	mg/kg OC	12	mg/kg OC
Fluoranthene	24,000	µg/kg dw	1,300	mg/kg OC	160	mg/kg OC
Fluorene	6,800	µg/kg dw	260	mg/kg OC	23	mg/kg OC
Indeno (1,2,3,-c,d)pyrene	4,300	µg/kg dw	200	mg/kg OC	34	mg/kg OC
Naphthalene	5,300	µg/kg dw	260	mg/kg OC	99	mg/kg OC

COPC[a]	Maximum Concentration	Unit	Maximum Concentration in Comparable Units	Unit	Screening Criteria: SQS, LAET, SL, or Toxicologically Based TRV	Screening Criteria: Unit
Phenanthrene	28,000	µg/kg dw	1,500	mg/kg OC	100	mg/kg OC
Pyrene	4,400	µg/kg dw	4,400	µg/kg dw	2,600[c]	µg/kg dw
Total benzofluoranthenes	17,000	µg/kg dw	890	mg/kg OC	230	mg/kg OC
HPAH	85,000	µg/kg dw	4,500	mg/kg OC	960	mg/kg OC
LPAH	44,000	µg/kg dw	1,700	mg/kg OC	370	mg/kg OC
Phthalates						
Bis(2-ethylhexyl) phthalate	14,000	µg/kg dw	14,000	µg/kg dw	1,300[c]	µg/kg dw
Butyl benzyl phthalate	7,100	µg/kg dw	340	mg/kg OC	4.9	mg/kg OC
Dimethyl phthalate	1,400 J	µg/kg dw	140 J	mg/kg OC	53	mg/kg OC
Other SVOCs						
1,2-Dichlorobenzene	520 J	µg/kg dw	520 J	µg/kg dw	35[c]	µg/kg dw
1,4-Dichlorobenzene	1,600 J	µg/kg dw	1,600 J	µg/kg dw	110[c]	µg/kg dw
1,2,4-Trichlorobenzene	72 J	µg/kg dw	72 J	µg/kg dw	31[c]	µg/kg dw
2-Methylnaphthalene	3,300	µg/kg dw	160	mg/kg OC	38	mg/kg OC
4-Methylphenol	4,600 J	µg/kg dw	4,600 J	µg/kg dw	670	µg/kg dw
2,4-Dimethylphenol	290 J	µg/kg dw	290 J	µg/kg dw	29	µg/kg dw
Benzoic acid	4,500	µg/kg dw	4,500	µg/kg dw	650	µg/kg dw
Benzyl alcohol	670	µg/kg dw	670	µg/kg dw	57	µg/kg dw
Dibenzofuran	4,200	µg/kg dw	220	mg/kg OC	15	mg/kg OC
Hexachlorobenzene	95 J	µg/kg dw	3.7 J	mg/kg OC	0.38	mg/kg OC
N-Nitrosodiphenylamine	110	µg/kg dw	110	µg/kg dw	28[c]	µg/kg dw
Pentachlorophenol	410	µg/kg dw	410	µg/kg dw	360	µg/kg dw
Phenol	2,800	µg/kg dw	2,800	µg/kg dw	420	µg/kg dw
PCBs						
Total PCBs	220,000	µg/kg dw	10,000	mg/kg OC	12	mg/kg OC

COPC[a]	Maximum Concentration	Unit	Maximum Concentration in Comparable Units	Unit	Screening Criteria	
					SQS, LAET, SL, or Toxicologically Based TRV	Unit
Organochlorine Pesticides						
Total DDTs	2,900 J	µg/kg dw	2,900 J	µg/kg dw	567[d]	µg/kg dw
Total chlordane	230	µg/kg dw	230	µg/kg dw	2.8[e]	µg/kg dw

a COPCs identified based on: 1) a comparison of maximum surface sediment concentrations to SMS SQS, or 2) for chemicals with no SQS, detected in at least 5% of LDW surface sediment samples and a comparison between maximum surface sediment concentrations and DMMP guidelines or toxicologically based TRVs.

b DMMP SL guideline.

c Comparison is based on LAET because TOC normalization was not appropriate for the sample.

d Literature based TRV – (Lotufo et al. 2001b).

e Literature based TRV – PSDDA AET evaluation 1994 (Gries and Waldow 1996).

f TBT was included as a COPC based on the results of the Phase 1 ERA (Windward 2003b), and was evaluated using benthic invertebrate tissue data consistent with EPA (1999a) and Meador et al. (2002). Tissue-residue data integrate multiple exposure pathways and provide a direct measure of exposure (EPA 1999a); therefore, tissue-residue-based TRVs were selected over the porewater-based DMMP SL guideline for TBT.

COPC – chemical of potential concern
DMMP – Dredged Material Management Program
dw – dry weight
HPAH – high-molecular-weight polycyclic aromatic hydrocarbon
J – estimated concentration
LAET – lowest apparent effects threshold
LPAH – low-molecular-weight polycyclic aromatic hydrocarbon
na – not applicable
OC – organic carbon
SL – screening level (DMMP)
SMS – Washington State Sediment Management Standards
SQS – sediment quality standards (SMS)
TBT – tributyltin
TRV – toxicity reference value

A.2.5.1.2 Screening methods and results for benthic communities based on porewater chemistry data

COPCs for benthic invertebrates potentially exposed to VOCs in porewater were also identified. The Phase 2 RI work plan (Windward 2004e) identified the need to assess risks to benthic invertebrates from porewater collected in areas where VOCs have been historically detected in groundwater at upland properties immediately adjacent to the LDW. Available data for VOCs in groundwater at 11 sites along the LDW were reviewed and two sites, GWI and Boeing Plant 2/Jorgensen Forge, were selected for porewater characterization because they had the highest potential for VOC concentrations in porewater based on their proximity to the LDW and existing groundwater data. The two locations are shown on Map A.2-3.

VOCs were detected in samples collected using peepers, but were not detected in deeper samples collected with piezometers. Four VOCs were detected in porewater samples collected at Boeing Plant 2/Jorgensen Forge. In the GWI area, 16 VOCs were detected in the porewater samples. COPCs in porewater for benthic invertebrates were identified by comparing maximum VOC concentrations detected in porewater to literature-based no-observed-effect concentration (NOEC) TRVs.

A literature search was conducted for relevant aquatic toxicity studies on growth, mortality (including immobilization), or reproductive (including developmental) endpoints using two databases, ECOTOX and BIOSIS. Studies with aquatic invertebrate species were preferred because the purpose of the porewater study was to evaluate risks to benthic invertebrates. In many cases, the available toxicity data were uncertain because lowest-observed-effect concentrations (LOECs) were rarely reported, NOECs were dependent on the selected test dilution series, and very few studies reported both effect and no-effect concentrations for a single species and endpoint. Most of the LOECs were based on LC50s (concentrations that result in the death of 50% of a test population). No uncertainty factors were assigned to the toxicity data. In addition, some of the exposure scenarios in the studies were of relatively short duration (48 to 96 hr). Use of these toxicity data may underestimate risks to benthic invertebrates in the LDW. Table A.2-14 presents the TRVs that were selected for 15 of the 16 VOCs. No toxicity data were available for isopropylbenzene. A discussion of all TRVs selected and all toxicological studies evaluated during the TRV search is presented in Attachment 7.

All maximum VOC concentrations in porewater were less than NOECs, except for cis-1,2-dichloroethene (Table A.2-14). The NOEC for cis-1,2-dichloroethene was derived by dividing the LOEC (an LC50 [concentration that causes the death of 50% of a group of test animals]) by 50 because a NOEC was not reported in the literature. Because the maximum concentration of cis-1,2-dichloroethene was greater than the calculated NOEC, it was retained as a COPC.

Table A.2-14. Identification of COPCs for benthic invertebrates in porewater

COPC	Concentration in Porewater		
	Maximum Detected Concentration (µg/L)[a]	NOEC (µg/L)	LOEC (µg/L)[b]
1,1-Dichloroethane	16	7,800[c]	39,600[d]
1,1-Dichloroethene	4.9	2,400	11,600
1,2-Dichlorobenzene	1.2	11[e]	550[f]
1,2-Dichloroethane	15	139[e]	6,927
1,2-Dichloropropane	2.5	840[e]	42,000
1,4-Dichlorobenzene	0.3	3[e]	147
Benzene	9.4	180	1,100[g]
Carbon disulfide	0.7	38[e]	1,900
Chlorobenzene	1.4	1,400	2,500[g]
cis-1,2-Dichloroethene	**2,900**	136[e]	6,785
Isopropylbenzene	0.3	na	na
Tetrachloroethene	1.1	331	332
Toluene	3.5	737	14,700
trans-1,2-Dichloroethene	21 J	136[e]	6,785
Trichloroethene	2.5	2,200	14,000
Vinyl chloride	2,500	12,800[c]	65,300[d]

[a] The maximum detected concentration in the 20 porewater samples analyzed as part of the Phase 2 RI.

[b] Concentration is based on a LC50 (concentration that causes the death of 50% of a group of test animals) unless otherwise noted.

[c] Concentration is a final chronic value (FCV) based on the narcosis model.

[d] Concentration is a final acute value (FAV) based on the narcosis model.

[e] A literature-based NOEC was not available; the NOEC was derived by dividing the LC50 by 50.

[f] Concentration is an EC50 (concentration causing a non-lethal effect in 50% of an exposed population).

[g] Concentration is an LC100 (concentration that causes the death of 100% of a group of test animals).

COPC – chemical of potential concern

HQ – hazard quotient

LOEC – lowest-observed-effect concentration

na – not available

NOEC – no-observed-effect concentration

Bold identifies maximum porewater concentrations greater than the NOEC.

A.2.5.1.3 Crabs

COPCs were identified for crabs using a critical tissue-residue approach and a two-step process. The first step was to identify a list of chemicals of interest (COIs). Chemicals were identified as COIs for crabs if they met at least two of the following three criteria:

- Detection in at least 5% of LDW surface sediment samples
- Identification as a bioaccumulative chemical in EPA (2000)
- Detection in any LDW-collected crab tissue sample

Table A.2-15 presents a summary of the metals, organometals, and organic compounds identified as COIs for crabs. Detailed results of the COI screening step for crabs are presented in Attachment 4.

Table A.2-15. Chemicals identified as COIs for crabs

COIs		
Metals		
Antimony	Copper	Selenium
Arsenic	Lead	Silver
Cadmium	Mercury	Thallium
Chromium	Methylmercury[a]	Vanadium
Chromium VI[a]	Molybdenum	Zinc
Cobalt	Nickel	
Organometals		
Monobutyltin as ion	Dibutyltin as ion	TBT as ion
PAHs		
2-Methylnaphthalene	Benzo(b)fluoranthene	Fluoranthene
Acenaphthene	Benzo(g,h,i)perylene	Fluorene
Acenaphthylene	Benzo(k)fluoranthene	Indeno(1,2,3-cd)pyrene
Anthracene	Chrysene	Naphthalene
Benzo(a)anthracene	Dibenzo(a,h)anthracene	Phenanthrene
Benzo(a)pyrene	Dibenzofuran	Pyrene
Phthalates		
Bis(2-ethylhexyl) phthalate	Diethyl phthalate	Di-n-butyl phthalate
Butyl benzyl phthalate	Dimethyl phthalate	
Other SVOCs		
Benzyl alcohol	Phenol	
Hexachlorobenzene		
PCBs		
Total PCBs		
Dioxins and Furans		
Organochlorine Pesticides		
Total DDTs[b]	gamma-BHC	Heptachlor epoxide
Dieldrin	alpha-Endosulfan	Methoxychlor
alpha-BHC	Endrin aldehyde	Total chlordane[c]

[a] Chromium VI and methylmercury (both bioaccumulative chemicals and detected in at least 5% of surface sediment samples) were not analyzed in any of the tissue samples; these chemicals are assumed to be incorporated in total mercury and total chromium risk analyses.

[b] Includes 2,4-DDE, 2,4-DDD, 2,4-DDT, 4,4-DDD, 4,4-DDE, and 4,4-DDT.

[c] Includes alpha-chlordane and gamma-chlordane.

COI – chemical of interest

PCB – polychlorinated biphenyl

SVOC – semivolatile organic compound

TBT – tributyltin

In the second step of the screening process, the maximum exposure concentration of each COI was compared to a no-observed-adverse-effect level (NOAEL) for that chemical. If the maximum exposure concentration was greater than the NOAEL, the chemical was identified as a COPC for crabs. This step was conducted for all COIs, except dioxins and furans. Dioxins and furans were identified as COIs for crabs because they were detected in at least 5% of LDW surface sediment samples and are bioaccumulative (Attachment 4). However, quantitative ecological risk analysis was not conducted for dioxins and furans in this baseline ERA, as described in Section A.2.5.

For the remaining COIs, the scientific literature was searched[18] to identify TRVs for COIs. The literature search included BIOSIS, EPA's ECOTOX database, aquatic life sciences database, USACE's Environmental Residue Effects Database (ERED), and Jarvinen and Ankley (1999). Original sources of toxicity data were obtained and reviewed to verify effects data summarized in the databases as well as the suitability of the studies. The databases were searched for studies that evaluated effects on survival, growth, and reproduction (including developmental effects).

TRVs were based on the risk evaluation method used. For crabs, the TRV search focused on chemical tissue-residue data associated with effects on decapods to support the critical tissue-residue approach (see Section A.3.4). For critical tissue-residue studies to be acceptable, the concentration in tissue had to be analyzed as part of the study. Acceptable toxicological data that met the following criteria were compiled for crabs.

- All selected TRVs were based on laboratory toxicological studies. Studies using field-collected data (i.e., field-collected crabs) were not considered acceptable. Field studies were not used to derive TRVs because adverse effects observed in organisms from field studies may be attributed to the presence of multiple chemicals and/or other uncontrolled environmental factors, rather than to a single test chemical.
- Selected TRVs were based preferentially on dietary, sediment, or water exposure studies.

After the literature search was conducted, all acceptable studies for TRV derivation were compiled, and presented in Attachment 5.

For each COI, a TRV was selected for both the NOAEL and the lowest-observed-adverse-effect level (LOAEL). The NOAEL represents the level below which adverse effects would not be expected. The NOAEL was compared to the maximum exposure concentration for each COI to identify COPCs. The LOAEL represents the level above

[18] For COIs that were evaluated in the Phase 1 ERA (Windward 2003b), literature published after January 2001 was searched for new toxicity data. If any COIs were not evaluated in the Phase 1 ERA, the literature was searched for appropriate toxicity data and was not limited by date.

which an effect would be expected. LOAELs are presented in this section for informational purposes only; they were not used in the screening process.

The LOAEL was selected from among the list of possible TRVs presented in Attachments 5 through 10 if it was the lowest dose at which an effect was observed for any of the three endpoints evaluated and a clear dose-response relationship was observed. The NOAEL was selected as the highest level below the selected LOAEL with the same endpoint. If no NOAEL with the same endpoint as the selected LOAEL was available, the NOAEL was selected as the highest NOAEL below the selected LOAEL based on another endpoint (survival, growth, or reproduction).

For COIs without NOAELs lower than the selected LOAEL, the NOAEL was determined using the following uncertainty factors following EPA Region 10 guidance (EPA 1997b):

- Acute or subchronic LOAEL/10
- Chronic or critical lifestage[19] LOAEL/5
- LC50 (or similar)/50

Based on the available literature, crab critical tissue-residue TRVs were developed for chemicals identified as COIs and detected in crab tissue according to the guidelines outlined above. TRVs were available for 14 of the 55 COIs (Table A.2-16). No studies with reproductive endpoints were identified; all acceptable studies addressed either growth or survival. TRVs selected from the acceptable studies are presented in Table A.2-17. Chemicals with no TRVs are discussed in the uncertainty analysis (Section A.6.1.2.2).

To identify COPCs for crabs, critical tissue-residue NOAELs were compared to maximum chemical concentrations detected in the corresponding LDW crab tissue samples (e.g., hepatopancreas NOAELs were compared with the maximum concentration detected in hepatopancreas tissue; whole-body NOAELs were compared to the estimated maximum whole-body concentration, as calculated from hepatopancreas and edible muscle tissue concentrations[20]). NOAELs were available in the literature for 10 COIs. NOAELs were estimated for naphthalene, PCBs, and methoxychlor as the LOAEL (based on survival data) divided by 10.

[19] Chronic exposure is defined as >15% of an organism's lifespan (Calabrese and Baldwin 1993). Exposure is assumed to be chronic if the duration is greater than 10 weeks for birds and greater than one year for mammals (Sample et al. 1996). For fish, chronic exposure duration was assumed to be 28 days or greater. A critical lifestage is one that occurs during reproduction, gestation, or development (Sample et al. 1996).

[20] Chemical concentrations in hepatopancreas tissue are likely to provide a conservative estimate of internal organs in general because the hepatopancreas constitutes the great majority of organ mass and has a relatively high lipid content relative to other organs.

Table A.2-16. Results of TRV search for chemicals identified as crab COIs through the initial screening process and detected in crab tissue

COIs for Crabs		
Chemicals with TRVs		
Arsenic	Vanadium	Methoxychlor
Cadmium	TBT	Total chlordane[a]
Chromium	Zinc	Total DDTs[b]
Copper	Naphthalene	Total PCBs[c]
Mercury	Heptachlor epoxide	
Chemicals without TRVs		
Antimony	Di-n-butyl phthalate	Dibenzo(a,h)anthracene
Cobalt	2-Methylnaphthalene	Dibenzofuran
Lead	Benzyl alcohol	Fluoranthene
Molybdenum	Hexachlorobenzene	Fluorene
Nickel	Phenol	Indeno(1,2,3-cd)pyrene
Monobutyltin	Acenaphthene	Phenanthrene
Dibutyltin	Acenaphthylene	Pyrene
Selenium	Anthracene	alpha-BHC
Silver	Benzo(a)anthracene	gamma-BHC
Thallium	Benzo(a)pyrene	Dieldrin
Bis(2-ethylhexyl) phthalate	Benzo(b)fluoranthene	alpha-Endosulfan
Butyl benzyl phthalate	Benzo(g,h,i)perylene	Endrin aldehyde
Diethyl phthalate	Benzo(k)fluoranthene	
Dimethyl phthalate	Chrysene	

[a] Alpha-chlordane and gamma-chlordane were detected in crab tissue. The two chemicals were assessed as total chlordane.

[b] 2,4-DDD, 2,4-DDE, 2,4-DDT, 4,4-DDD, 4,4-DDE, and 4,4-DDT were detected in crab tissue. The five chemicals were assessed as total DDTs.

[c] Aroclor-1248, Aroclor-1254, and Aroclor-1260 were detected in crab tissue, and assessed as total PCBs.

COI – chemical of interest

PCB – polychlorinated biphenyl

TBT – tributyltin

TRV – toxicity reference value

Table A.2-17. Selected critical tissue-residue TRVs for crab COIs

COI	Test Species	Tissue Type	NOAEL	LOAEL	Units	Endpoint	Source
Arsenic	juvenile grass shrimp *(Palaemonetes pugio)*	whole body	1.28 [a]	na	mg/kg ww	growth	Lindsay and Sanders (1990)
Cadmium	grass shrimp *(Palaemonetes pugio)*	whole body	0.6 [a]	na	mg/kg ww	survival	Rule and Alden (1996)
	grass shrimp *(Palaemonetes pugio)*	whole body	na	2.6[a]	mg/kg ww	survival	Vernberg et al. (1977)
Chromium	juvenile (2nd instar) sand crab (*Portunus pelagicus*)	whole body	1	3.2	mg/kg ww	growth	Mortimer and Miller (1994)
Copper	adult crayfish (*Orconectes rusticus*)	whole body	50[a]	na	mg/kg ww	survival	Evans (1980)
Mercury	adult Norway lobster (*Nephrops norvegicus*)	hepatopancreas	0.99[a,b,]	na	mg/kg ww	survival	Canli and Furness (1995)
	adult male shore crab (*Eriocheir sinensis*)	hepatopancreas	na	1[c,d]	mg/kg ww	survival	Bianchini and Giles (1996)
Vanadium	shrimp (*Lysmata seticaudata*)	whole body	0.6	na	mg/kg ww	survival	Miramand et al. (1981)
Zinc	crayfish (*Orconectes virilis*)	whole body	12.7[a]	35.2[a]	mg/kg ww	survival	Mirenda (1986a)
TBT	juvenile blue crab (*Callinectes sapidus*)	whole body	120	na	µg/kg dw	growth	Rice et al. (1989)
Naphthalene	spot shrimp (*Pandalus platyceros*)	whole body	5.0[e]	50	µg/kg ww	survival	Sandborn and Malins (1977)
PCBs (Aroclor 1016)	grass shrimp (*Palaemonetes pugio*)	whole body	110[e]	1,100[f]	µg/kg ww	survival	Hansen et al. (1974b)
Total chlordane	pink shrimp (*Penaeus duorarum*)	whole body	710	1,700	µg/kg ww	survival	Parrish et al. (1976)
Total DDTs	pink shrimp (*Penaeus duorarum*)	whole body	na	60	µg/kg ww	survival	Nimmo et al. (1970)
	crayfish (*Orconectes nais*)	whole body	46	na	µg/kg ww	survival	Johnson et al. (1971)
Heptachlor epoxide	pink shrimp (*Penaeus duorarum*)	whole body	54	180	µg/kg ww	survival	Schimmel et al. (1976)
Methoxychlor	juvenile Dungeness crab (*Cancer magister*)	whole body	15[e]	150	µg/kg ww	survival	Armstrong et al. (1976)

[a] Converted from dry weight to wet weight using a moisture content of 80% (Jarvinen and Ankley 1999) (80% was also the average moisture content of two crab samples collected by King County in 1997 (King County 1999b); the mean % moisture in LDW crabs was 82.7% (Windward 2005c)).

[b] Dietary exposure route

[c] Full equilibrium between water and tissue may not have been reached because of a short exposure time (≤ 48 hrs).

[d] Concentration is lowest of three crab species tested (*Carcinus maenas*, *Eriocheir sinensis*, and *Cancer pagurus*).

[e] Calculated from LOAEL by dividing by 10.

[f] Survival was reduced by 33%.

dw – dry weight
PCB – polychlorinated biphenyl
ww – wet weight
na – not available
TBT – tributyltin

Two chemicals, zinc and PCBs, were identified as COPCs for crabs (Table A.2-18) and retained for further evaluation in this baseline ERA. The maximum concentrations of arsenic, total DDTs, and methoxychlor also exceeded NOAELs for these COIs, but were not identified as COPCs for the following reasons. No LOAEL TRV was identified for arsenic. Because the arsenic NOAEL is not bounded by a LOAEL, this assessment does not indicate a potential threshold above which adverse effects may occur. The two organochlorine pesticides were not identified as COPCs because of the uncertainty associated with JN-qualified organochlorine pesticide tissue data (see Section A.2.4.2.2). Therefore, arsenic, total DDTs, and methoxychlor will be discussed further in the uncertainty analysis (Section A.6.1.2.2). A summary of the COPCs that will be evaluated for benthic invertebrates and crabs in the risk characterization is presented in Table A.2-19.

Table A.2-18. Maximum concentrations of COIs in crab tissue samples compared with NOAEL TRVs

COI	Unit	Maximum Concentration in Crab Tissue[a]	NOAEL TRV
Arsenic	mg/kg ww	**11 M**	1.28
Cadmium	mg/kg ww	0.2951 M	0.6
Chromium	mg/kg ww	0.14 M	1
Copper	mg/kg ww	24 M	50
Mercury	mg/kg ww	0.067[b]	0.99
Vanadium	mg/kg ww	0.2 JM	0.6
Zinc	mg/kg ww	**37.3 M**	12.7
TBT	µg/kg ww	75 M	120
Naphthalene	µg/kg ww	3.2 M	5.0[c]
Total PCBs	µg/kg ww	**1,900 JM**	110[c]
Total chlordane	µg/kg ww	26 JNM	710
Total DDTs	µg/kg ww	**150 JNM**	46
Heptachlor epoxide	µg/kg ww	5.5 JNM	54
Methoxychlor	µg/kg ww	**90 JNM**	15[c]

[a] All whole-body concentrations except as noted.

[b] Hepatopancreas

[c] Calculated from LOAEL by dividing by 10.

COI – chemical of interest

J – estimated concentration

M – estimated concentration based on a weighted mean of hepatopancreas and edible meat data

N – tentative identification

NOAEL – no-observed-adverse-effect level

PCB – polychlorinated biphenyl

TBT – tributyltin

TRV – toxicity reference value

ww – wet weight

Bold identifies maximum crab tissue concentrations greater than the NOAEL.

Table A.2-19. 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		
Bis(2-ethylhexyl) phthalate	X	
Butyl benzyl phthalate	X	
Dimethyl phthalate	X	

COPC	Benthic Invertebrate Community	Crabs
Other SVOCs		
1,2-Dichlorobenzene	X	
1,4-Dichlorobenzene	X	
1,2,4-Trichlorobenzene	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	

COPC – chemical of potential concern
PCB – polychlorinated biphenyl
SVOC – semivolatile organic compound
TBT – tributyltin
VOC – volatile organic compound

A.2.5.2 Fish

COPCs were identified for fish using a two-step process. The first step was to identify a list of COIs. Chemicals were identified as COIs for fish if they met at least two of the following three criteria:

- Detection in at least 5% of LDW surface sediment samples
- Identification as a bioaccumulative chemical in EPA (2000)
- Detection in any LDW-collected tissue sample

Table A.2-20 presents a summary of chemicals identified as COIs as a result of this screen. Detailed results of the COI screening step for fish are presented in Attachment 4.

Table A.2-20. Chemicals identified as COIs for fish ROCs

COIs		
Metals		
Antimony	Copper	Selenium
Arsenic	Lead	Silver
Cadmium	Mercury	Thallium
Chromium	Methylmercury[a]	Vanadium
Chromium VI[a]	Molybdenum	Zinc
Cobalt	Nickel	
Organometals		
Monobutyltin as ion	TBT as ion	
Dibutyltin as ion	Tetrabutyltin as ion	
PAHs		
C1-Chrysenes	C3-Phenanthrenes/anthracenes	Benzo(g,h,i)perylene
C1-Dibenzothiophenes	C4-Naphthalenes	Benzo(k)fluoranthene
C1-Fluoranthene/pyrene	C4-Phenanthrenes/anthracenes	Chrysene
C1-Fluorenes	1-Methylnaphthalene	Dibenzo(a,h)anthracene
C1-Phenanthrenes/anthracenes	2-Methylnaphthalene	Dibenzofuran
C2-Chrysenes	Acenaphthene	Fluoranthene
C2-Dibenzothiophenes	Acenaphthylene	Fluorene
C2-Fluorenes	Anthracene	Indeno(1,2,3-cd)pyrene
C2-Naphthalenes	Benzo(a)anthracene	Naphthalene
C2-Phenanthrenes/anthracenes	Benzo(a)pyrene	Perylene
C3-Fluorenes	Benzo(b)fluoranthene	Phenanthrene
C3-Naphthalenes	Benzo(e)pyrene	Pyrene
Phthalates		
Bis(2-ethylhexyl) phthalate	Diethyl phthalate	Di-n-butyl phthalate
Butyl benzyl phthalate	Dimethyl phthalate	
Other SVOCs		
4-Chlorophenyl phenyl ether	Biphenyl	Hexachlorobenzene
4-Methylphenol	Carbazole	Phenol
Benzoic acid	Dibenzothiophene	
PCBs		
Total PCBs		
Dioxins and Furans		

COIs		
Organochlorine Pesticides		
Total DDTs[b]	delta-BHC	Heptachlor
Aldrin	gamma-BHC	Heptachlor epoxide
Dieldrin	alpha-Endosulfan	Methoxychlor
alpha-BHC	beta-Endosulfan	Total chlordane[c]
beta-BHC	Endrin	

[a] Chromium VI and methylmercury (both bioaccumulative chemicals and detected in at least 5% of surface sediment samples) were not analyzed in any of the tissue samples; these chemicals are assumed to be incorporated in total mercury and total chromium risk analyses.

[b] Includes 2,4'-DDD, 2,4'-DDE, 2,4'-DDT, 4,4'-DDD, 4,4'-DDE, and 4,4'-DDT.

[c] Includes alpha-chlordane, gamma-chlordane, oxychlordane, cis-nonachlor, and trans-nonachlor.

COI – chemical of interest

PCB – polychlorinated biphenyl

SVOC – semivolatile organic compound

TBT – tributyltin

In the second step of the screening process, the maximum exposure concentration of each COI was compared to a NOAEL for that chemical. If the maximum exposure concentration was greater than the NOAEL, the chemical was identified as a COPC for fish. This step was conducted for all COIs, except dioxins and furans, as discussed in Section A.2.5.

For COIs other than dioxins and furans, the scientific literature was searched[21] to identify TRVs for COIs. The literature search included BIOSIS, EPA's ECOTOX database, aquatic life sciences database, USACE's ERED, and Jarvinen and Ankley (1999). Original sources of toxicity data were obtained and reviewed to verify effects data summarized in the databases as well as the suitability of the studies. The databases were searched for studies that evaluated effects on survival, growth, and reproduction (including developmental effects).

TRVs were based on the risk evaluation method used. For fish, databases were searched for dietary studies for metals and PAHs and for critical tissue-residue data for other chemicals (see Section A.4.1 for further explanation of the two methods used). For critical tissue-residue studies to be acceptable, the concentration in tissue had to be analyzed as part of the study. Acceptable toxicological data that met the following criteria were compiled for fish.

- All selected TRVs were based on laboratory toxicological studies. Studies using field-collected data (i.e., field-collected fish or fish fed field-collected diets) were not considered acceptable. Field studies were not used to derive TRVs because

[21] For COIs that were evaluated in the Phase 1 ERA (Windward 2003b), toxicity literature published after January 2001 was searched for new toxicity data. If any COIs not evaluated in Phase 1 ERA, toxicity literature was searched for appropriate toxicity data and was not limited by date.

adverse effects observed in organisms from field studies may be attributed to the presence of multiple chemicals and/or other uncontrolled environmental factors, rather than to a single test chemical.[22]

- Selected TRVs were based preferentially on dietary, sediment, or water exposure studies. Studies conducted using intraperitoneal (IP) or egg injection or oral gavage as exposure routes were not considered representative of the ROC exposure conditions but were used if no other studies were available.
- All selected TRVs were based on whole-body tissue concentrations or egg concentrations that were converted to adult tissue concentrations using adult-to-egg conversion factors from the literature.

After the literature search was conducted, all acceptable studies for TRV derivation were compiled, as presented in Attachment 8. For each COI, TRVs were selected for both the NOAEL and the LOAEL. TRV selection rules and uncertainty factors discussed above for crab in Section A.2.5.1.3 were used for fish as well.

TRVs were available for 45 of the 86 fish COIs (Table A.2-21). Selected TRVs are presented in Tables A.2-22 and A.2-23 for COIs evaluated using the dietary approach and critical tissue-residue approach, respectively. Chemicals with no TRVs are discussed in the uncertainty analysis (Section A.6.2.1.2).

Table A.2-21. Results of TRV search for fish COIs

COIs for Fish		
Chemicals with TRVs		
Arsenic	Benzo(k)fluoranthene[a]	beta-Endosulfan
Cadmium	Chrysene[a]	Total DDTs
Chromium	Dibenzo(a,h)anthracene[a]	Dieldrin
Copper	Fluoranthene[a]	Endrin
Lead	Fluorene[a]	gamma-BHC (lindane)
Selenium	Indeno(1,2,3-cd)pyrene[a]	Heptachlor
Silver	Naphthalene[a]	Heptachlor epoxide
Vanadium	Phenanthrene[a]	Methoxychlor
Zinc	Pyrene[a]	Total chlordane
2-Methylnaphthalene[a]	Mercury	Bis(2-ethylhexyl) phthalate
Acenaphthene[a]	TBT as ion	Butyl benzyl phthalate
Anthracene[a]	4-Methylphenol	Diethyl phthalate
Benzo(a)anthracene[a]	Benzoic acid	Dimethyl phthalate
Benzo(a)pyrene[b]	Hexachlorobenzene	Di-n-butyl phthalate
Benzo(b)fluoranthene[a]	Phenol	Total PCBs

[22] The uncertainty associated with not including TRVs derived from field-collected data is explored in the uncertainty analysis subsections of the risk characterization (Section A.6.2).

COIs for Fish		
Benzo(g,h,i)perylene[a]	alpha-Endosulfan	
Chemicals without TRVs		
Antimony	C1-Fluoranthene/Pyrene	Dibutyltin as ion
Chromium VI	C1-Fluorenes	Monobutyltin as ion
Cobalt	C1-Phenanthrenes/anthracenes	Tetrabutyltin as ion
Molybdenum	C2-Chrysenes	4-Chlorophenyl phenyl ether
Nickel	C2-Dibenzothiophenes	Biphenyl
Thallium	C2-Fluorenes	Carbazole
1-Methylnaphthalene	C2-Naphthalenes	Dibenzothiophene
Acenaphthylene	C2-Phenanthrenes/Anthracenes	Aldrin
Benzo(e)pyrene	C3-Fluorenes	alpha-BHC
Dibenzofuran	C3-Naphthalenes	beta-BHC
Perylene	C3-Phenanthrenes/Anthracenes	delta-BHC
C1-Chrysenes	C4-Naphthalenes	
C1-Dibenzothiophenes	C4-Phenanthrenes/Anthracenes	

[a] PAHs included in the mixtures of PAHs evaluated by Palm et al. (2003) or Meador et al. (2006).

[b] Benzo(a)pyrene was included in the mixtures of PAHs evaluated by Palm et al. (2003) and Meador et al. (2006); TRVs for exposure to benzo(a)pyrene as a single chemical were also available.

COI – chemical of interest

PAH – polycyclic aromatic hydrocarbon

PCB – polychlorinated biphenyl

TBT – tributyltin

TRV – toxicity reference value

Table A.2-22. TRVs selected for fish COIs evaluated using a dietary approach

COI	Test Species	NOAEL (mg/kg dw)	LOAEL (mg/kg dw)	Endpoint	Source
Arsenic	rainbow trout	20	30	growth	Oladimeji et al. (1984)
Cadmium	rockfish	0.1[a]	0.5	growth	Kim et al. (2004); Kang et al. (2005)
Chromium	grey mullet	9.42	na	growth	Walsh et al. (1994)
Copper	rainbow trout	50	100	growth	Kang et al. (2005)
Lead	rainbow trout	7,040	na	growth	Goettl et al. (1976)
Silver	rainbow trout	3,000	na	growth	Galvez and Wood (1999)
Vanadium	rainbow trout	2.04[a]	10.2	growth	Hilton and Bettger (1988)
Zinc	rainbow trout	1,900	nr	growth	Mount et al. (1994)
	rainbow trout	nr	2,000	growth	Takeda and Shimma (1977)
Benzo(a)pyrene	rainbow trout	100	nr	growth	Hart and Heddle (1991)
	English sole	nr	116	growth	Rice et al. (2000)
Total PAHs[c]	chinook salmon	324	951	growth	Meador et al. (2006)

[a] NOAEL estimated using an uncertainty factor of 5 (chronic LOAEL to NOAEL).

[b] NOAEL estimated using an uncertainty factor of 10 (acute/subchronic LOAEL to NOAEL).

[c] Mixture comprises the following 21 PAHs included in Meador et al. (2006) diet: naphthalene, 2-methylnaphthalene, dimethylnaphthalene, dibenzothiophene, acenaphthene, fluorene, 1,8-dimethyl(9H)fluorene, phenanthrene, 9-ethylphenanthrene, 9-ethyl-10-methylphenanthrene, 1-methyl-7-isopropylphenanthrene, anthracene, fluoranthene, pyrene, methyl pyrene, benzo(a)anthracene, chrysene, benz(a)pyrene, benzo(k)fluoranthene, benzo(g,h,i)perylene, dibenzanthracene.

COI – chemical of interest

dw – dry weight

LOAEL – lowest-observed-adverse-effect level

na – not available; no LOAELs identified in the literature search; selected NOAEL is the highest unbounded NOAEL in the literature reviewed.

NOAEL – no-observed-adverse-effect level

nr – not relevant; NOAEL and LOAEL TRVs were derived from separate studies reporting the same endpoint.

PAH – polycyclic aromatic hydrocarbon

Table A.2-23. TRVs selected for fish COIs evaluated using the critical tissue-residue approach

COI	Test Species	NOAEL (µg/kg ww)	LOAEL (µg/kg ww)	Endpoint	Source
Mercury[a]	golden shiner	230	nr	survival	Webber and Haines (2003)
	mummichog	nr	470	survival	Matta et al. (2001)
Selenium	[national criterion]	1,200[b]	1,600[c]	adverse effects	EPA (2004b)
TBT	Japanese flounder	18	159	growth	Shimasaki et al. (2003)
Bis(2-ethylhexyl) phthalate	rainbow trout	390[d]	na	reproduction	Mehrle and Mayer (1976)
Butyl benzyl phthalate	bluegill	6,450	na	survival	Barrows et al. (1980)
Di(n)butyl phthalate	sheepshead minnow	1,170	na	survival	Wofford et al. (1981)
Dimethyl phthalate	bluegill	498	na	survival	Barrows et al. (1980)
Diethyl phthalate	bluegill	1,102	na	survival	Barrows et al. (1980)
4-methylphenol	rainbow trout	1,530[e]	76,500	survival	Kaiser et al. (1984)
Benzoic acid	mosquito fish	3,380	na	survival	Lu and Metcalf (1975)
Hexachlorobenzene	fathead minnow	468,000	na	survival	Schuytema et al. (1990)
Phenol	rainbow trout	1,470[e]	73,400	survival	McKim and Schmeider (1990)
PCBs	common barbel	104 – 528[f]	520 – 2,640[f]	reproduction	Hugla and Thome (1999)
alpha-Endosulfan	spot	0.62[e]	31	survival	Schimmel et al. (1977a)
beta-Endosulfan	spot	0.62[e]	31	survival	Schimmel et al. (1977a)
DDTs (total)	cutthroat trout	1,800[g]	1,800[g]	survival	Allison et al. (1964)
Dieldrin	rainbow trout	120	200	survival	Shubat and Curtis (1986)
Endrin	largemouth bass	1.15[h]	11.5	survival	Fabacher (1976)
gamma-BHC (lindane)	sheepshead minnow	1,580[e]	79,000	survival	Schimmel et al. (1977b)
Heptachlor	spot	30[e]	1,500	survival	Schimmel et al. (1976)
Heptachlor epoxide	bluegill	80[h]	800	growth	Andrews et al. (1966)
Methoxychlor	brook trout	50	300	growth	Oladimeji and Leduc (1975)
Total chlordane	goldfish	710	nr	survival	Moore et al. (1977)
	goldfish	nr	1,360	survival	Feroz and Khan (1979)

[a] TRVs were based on exposure to methylmercury. Methylmercury was not analyzed in any of the tissue samples from the LDW; the total mercury concentration in LDW tissue samples was compared to the methylmercury TRV.

[b] National criterion for selenium in summer-collected fish. Dry weight concentration converted to wet weight assuming 80% moisture content.

[c] National criterion for selenium in winter-collected fish. Dry weight concentration converted to wet weight assuming 80% moisture content.

[d] Fry concentration calculated using a bioconcentration factor and water concentration reported in Mehrle and Mayer (1976).

[e] NOAEL estimated using an uncertainty factor of 50 (LC50 to NOAEL).

[f] A LOAEL range was selected from this study because the specific LOAEL was unclear because of uncertainties associated with this study. The NOAEL range was estimated using an uncertainty factor of 5 (chronic LOAEL to NOAEL) .

[g] The LOAEL is tissue concentration at 111 days (3.7 months) in fish exposed to 0.1 mg/L DDT in water where significant mortality occurred after approximately 4 months (approximately 120 days). The NOAEL (1,800 µg/kg ww) is the highest tissue concentration (at 466 days) in fish exposed to 0.03 mg/L DDT in water at which significant mortality did not occur over the entire exposure duration.

[h] NOAEL estimated using an uncertainty factor of 10 (acute/subchronic LOAEL to NOAEL).

COI – chemical of interest

na – not available; no LOAELs identified in the literature search; selected NOAEL is the highest unbounded NOAEL in the literature reviewed.

nr – not relevant; NOAEL and LOAEL TRVs were derived from separate studies reporting the same endpoint.

PCB – polychlorinated biphenyl

TBT – tributyltin

TRV – toxicity reference value

ww – wet weight

COPCs were screened for fish ROCs using one of two approaches depending on the type of chemical. For chemicals evaluated using a critical tissue-residue approach, maximum detected concentrations of COIs in the whole-body tissue of fish ROCs were compared with effects data reported in the literature. Use of the critical tissue-residue approach integrates all exposure pathways and reduces uncertainty associated with the relative uptake and depuration rates of a chemical. For COPCs that are highly regulated or metabolized by fish, such as PAHs (Varanasi 1989) and most metals (Bury et al. 2003, as cited in Meyer et al. 2005), comparison of chemical concentrations in prey to suitable dietary TRVs is preferable. Therefore, a dietary approach was used to identify COPCs for PAHs and metals (except butyltins, mercury, and selenium). For PAHs and metals, the concentration in the diet more accurately reflects the toxic dose than whole-body tissue residues.

For chemicals evaluated using a dietary approach, maximum detected concentrations of COIs in fish diets (expressed as mg/kg dw) were compared with dietary exposure concentrations reported in the literature (expressed as mg/kg dw). An alternative way of expressing exposure that would have explicitly considered an ROC's rate of chemical uptake would have involved the calculation of "dietary dose" expressed as µg/g fish/day. Because fish prey consumption is variable, use of a dietary dose approach is becoming more prevalent as a way to normalize dietary exposure among species (e.g., Clearwater et al. 2002). This method could also be used to predict a total dose from both water and dietary exposure, although little progress has been made in this regard (e.g., Borgmann et al. 2005). Because use of a dose-based approach for the purpose of estimating effects from dietary exposure is in its infancy, components of dose (such as ration size, feeding frequency, and food wastage) are often not reported in toxicity papers. Therefore, it is difficult to estimate accurate doses from available effects data. In addition, daily food consumption rates are not standardized for fish species as they are for wildlife, making fish dietary dose exposure calculations uncertain. Therefore, a dietary concentration approach rather than a dietary dose approach was used as the dietary approach for fish exposure and effects calculations in this ERA.

To screen COIs using the critical tissue-residue approach, maximum COI concentrations in ROC fish species were compared to NOAELs from the literature. This screen was conducted in two parts. First, the maximum fish tissue concentration for each COI in any fish ROC species was compared to its respective NOAEL. COIs with maximum tissue concentrations greater than the NOAEL were identified as COPCs. Second, a species-specific screen was conducted to determine which ROC/COPC pairs would be evaluated in more detail in the exposure and effects assessment (Section A.5.0) and which pairs would not warrant additional analysis.

Maximum concentrations (or RLs for non-detected COIs) of nine COIs in LDW fish tissue were greater than their respective NOAEL TRVs (Table A.2-24). Of these chemicals, LOAEL TRVs were not identified for benzoic acid, dimethyl phthalate, or di-n-butyl phthalate. Because the NOAELs for these chemicals are not bounded by LOAELs, they do not indicate potential thresholds above which adverse effects may occur. Therefore, risks associated with these chemicals are discussed in the uncertainty analysis (Section A.6.2).

Table A.2-24. Maximum COI concentrations in any fish ROC tissue compared to NOAEL TRVs

COI	Maximum Chemical Concentration in Fish ROC Tissue (µg/kg ww)	NOAEL TRV (µg/kg ww)
Mercury	39	230
Selenium	320	1,200
TBT as ion	**80**	18
Bis(2-ethylhexyl) phthalate	**5,000 U**[a]	390
Butyl benzyl phthalate	1,400	6,450
Diethyl phthalate	900 J	1,102
Dimethyl phthalate	**580 U**	498
Di-n-butyl phthalate	**2,300**	1,170
4-Methylphenol	380 J	1,530
Benzoic acid	**6,800 J**	3,380
Hexachlorobenzene	6.6 J	468,000
Phenol	200 J	1,470
Total PCBs	**4,700**	104 – 528
alpha-Endosulfan	**6.6 J**	0.62
beta-Endosulfan	**18 J**	0.62
Total DDTs	280 J	1,800
Dieldrin	5.7	120
Endrin	**36**	1.15
gamma-BHC (lindane)	5.6 J	1,580
Heptachlor	6.8 J	30
Heptachlor epoxide	45 J	80
Methoxychlor	0.49	50
Total chlordane	59 J	710

[a] RL is elevated because of analytical dilutions. A subset of 49 fish tissue samples with RLs of 7,700 µg/kg was reanalyzed, and BEHP was not detected with RLs ranging from 66 to 130 µg/kg ww.

COI – chemical of interest

J – estimated concentration

NOAEL – no-observed-adverse-effect level

PCB – polychlorinated biphenyl

ROC – receptor of concern

SVOC – semivolatile organic compound

TRV – toxicity reference value

U – not detected at reporting limit shown

ww – wet weight

Bold identifies maximum fish tissue concentrations greater than the NOAEL.

Bis(2-ethylhexyl) phthalate (BEHP) and dimethyl phthalate were not detected in any fish ROC tissue sample but had RLs that were greater than the NOAELs. Uncertainty associated with RLs above NOAELs for these chemicals are described in the uncertainty analysis (Section A.6.2).

Based on the above COPC screen, TBT, alpha-endosulfan, beta-endosulfan, endrin, and total PCBs were identified as COPCs for further evaluation. A juvenile chinook-specific TRV was identified for PCBs because the study providing the selected PCB TRV (Hugla and Thome 1999) reported results following exposures of adults and documented reproductive endpoints. Because exposures of juvenile chinook salmon to chemicals in the LDW are limited to the period during which they migrate through the waterway, the selected TRV for adult fish (Hugla and Thome 1999) would not accurately reflect either the lifestage or exposure regime of the migratory juvenile chinook in the LDW. The lowest LOAEL for a non-reproductive endpoint was 46,000 µg/kg ww for survival of spot (Hansen et al. 1971). The highest NOAEL below this LOAEL (27,000 µg/kg ww) was selected as the screening NOAEL TRV for juvenile chinook salmon. This TRV is likely protective of juvenile chinook salmon growth because no adverse effects on growth of salmonid species have been observed at whole-body tissue concentrations as high as 31,000 µg/kg ww (Mauck et al. 1978).

Based on the ROC-specific screen, where maximum concentrations in each ROC tissue were compared to NOAEL TRVs, Pacific staghorn sculpin/TBT, English sole/total PCBs, and Pacific staghorn sculpin/total PCBs were identified as ROC/COPC pairs (Table A.2-25), and will be evaluated in this baseline ERA. Because of uncertainties in the JN-qualified organochlorine pesticides data (as discussed in Section A.2.4.2), risks associated with alpha-endosulfan and beta-endosulfan in English sole and Pacific staghorn sculpin, and endrin in all three fish ROCs are discussed in the uncertainty analysis (Section A.6.2).

Table A.2-25. Maximum tissue concentrations in each fish ROC compared to NOAEL TRVs for COIs analyzed using a critical tissue-residue approach

ROC	COI	Maximum Detected Chemical Concentration in Fish ROC Tissue (µg/kg ww)	NOAEL TRV (µg/kg ww)
Juvenile chinook salmon	TBT	14	18
	alpha-endosulfan	1.5	3.1
	beta-endosulfan	0.94	3.1
	endrin	**6.5**	1.15
	total PCBs	1,200	27,000

ROC	COI	Maximum Detected Chemical Concentration in Fish ROC Tissue (µg/kg ww)	NOAEL TRV (µg/kg ww)
English sole	TBT	9.9	18
	alpha-endosulfan	**6.6**	3.1
	beta-endosulfan	**18**	3.1
	endrin	**14**	1.15
	total PCBs	**4,700**	104 – 528
Pacific staghorn sculpin	TBT	**80**	18
	alpha-endosulfan	**3.6**	3.1
	beta-endosulfan	**6.4**	3.1
	endrin	**36**	1.15
	total PCBs	**2,800**	104 – 528

COI – chemical of interest

PCB – polychlorinated biphenyl

ROC – receptor of concern

TBT – tributyltin

TRV – toxicity reference value

ww – wet weight

Bold identifies maximum fish tissue concentrations greater than the NOAEL.

For COIs evaluated using a dietary approach, maximum dietary exposure concentrations were represented by a weighted average of 10% maximum sediment concentrations (to account for exposure via incidental sediment ingestion) and 90% maximum benthic invertebrate tissue concentrations (Equation 2-1).[23] Ten percent was selected as an upper-bound estimate of sediment ingestion based on discussions with fish experts (see Section A.4.1.2).

$$\text{Maximum [diet]} = \text{Maximum [sed]} \times 10\% + \text{Maximum [tissue]} \times 90\% \qquad \textbf{Equation 2-1}$$

Where:

diet = dietary concentration (mg/kg dw)

sed = sediment concentration (mg/kg dw)

tissue = tissue concentration (mg/kg dw)

Based on this COPC screen, five COIs evaluated using the dietary approach were selected as COPCs (TableA.2-26). Of these chemicals, a LOAEL TRV was not identified for chromium. Because the NOAEL for chromium is not bounded by a LOAEL, it does not indicate a potential threshold above which adverse effects may occur. Therefore, risks associated with chromium are discussed in the uncertainty analysis

[23] Note that maximum chemical concentrations in benthic invertebrate tissue samples exceeded those in shiner surfperch tissue samples for all dietary COIs. Therefore, this prey assumption conservatively screens COPCs for piscivorous fish, such as Pacific staghorn sculpin.

(Section A.6.2). The remaining four chemicals (arsenic, cadmium, copper, and vanadium) were identified as COPCs for all fish ROCs.

Table A.2-26. Maximum dietary exposure concentrations compared to NOAEL TRVs for fish COIs analyzed using a dietary approach

COI	Maximum LDW-Wide Sediment Concentration (mg/kg dw)	Maximum LDW-Wide Benthic Invertebrate Concentration (mg/kg dw)	Maximum Dietary Exposure Concentration (mg/kg dw)	NOAEL TRV (mg/kg dw)
Arsenic	1,100	116	**210**	20
Cadmium	120	1.3 J	**13**	0.1
Chromium	1,100 J	58.2	**160**	9.42
Copper	12,000 J	170	**1,400**	50
Lead	23,000	217.9	2,500	7,040
Silver	270	2.496	29	3,000
Vanadium	150	22.2	**35**	2.04
Zinc	9,700	384	1,300	1,900
Benzo(a)pyrene	7.9	1.3	2.0	100
Total PAHs[a]	119	21.7 J	34	324

[a] PAH mixture includes the following 21 PAHs: naphthalene, 2-methylnaphthalene, dimethylnaphthalene, dibenzothiophene, acenaphthene, fluorene, 1,8-dimethyl(9H)fluorene, phenanthrene, 9-ethylphenanthrene, 9-ethyl-10-methylphenanthrene, 1-methyl-7-isopropylphenanthrene, anthracene, fluoranthene, pyrene, methyl pyrene, benz(a)anthracene, chrysene, benzo(a)pyrene, benzo(k)fluoranthene, benzo(g,h,i)perylene, dibenzanthracene. These chemicals were evaluated as a PAH mixture by Meador et al. (2006).

COI – chemical of interest
dw – dry weight
J – estimated concentration
NOAEL – no-observed-adverse-effect level
PAH – polycyclic aromatic hydrocarbon
TRV – toxicity reference value

Bold identifies maximum dietary exposure concentrations greater than the NOAEL.

Based on the above COPC screen, PAHs were not selected as a COPC for any fish ROC because the maximum dietary exposure concentration was less than the NOAEL. However, of the 38 chemicals with no TRVs, 15 chemicals were alkylated PAHs, and 5 chemicals were non-alkylated PAHs. These alkylated and non-alkylated PAHs were not included in the calculation of total PAHs in the COPC screen above because they were not included in the PAH mixture used in the TRV study (Meador et al. 2006). To address this uncertainty, an additional PAH screen was conducted by comparing TRVs for benzo(a)pyrene (Hart and Heddle 1991; Rice et al. 2000) and a PAH mixture (Meador et al. 2006) to maximum concentrations of total PAHs (i.e., the sum of all alkylated and non-alkylated PAHs analyzed) calculated in fish diets using Equation 2.1 and detected in juvenile chinook stomach contents (Table A.2-27). Maximum dietary exposure concentrations of all PAHs were lower than the available TRVs (Table A.2-27), thus supporting the exclusion of PAHs as a COPC for all fish species.

Table A.2-27. Maximum dietary PAH exposure concentrations (including all alkylated and non-alkylated PAHs) compared to total PAH and benzo(a)pyrene NOAEL TRVs for fish

Chemical in TRV Study	TRVs		Maximum Exposure Concentration of Total PAHs			
	NOAEL (mg/kg dw)	LOAEL (mg/kg dw)	Juvenile Chinook Salmon Stomach Contents (mg/kg dw)	Sediment (mg/kg dw)	Benthic Invertebrate Prey (mg/kg dw)	Maximum Dietary Exposure Concentration (mg/kg dw)
Benzo(a)pyrene	100[a]	116[b]	14.4 JN	133[d]	36.1J[e]	46[f]
Total PAHs	324[c]	951[c]				

[a] No effects on growth of rainbow trout (Hart and Heddle 1991).

[b] Effects on growth of English sole (Rice et al. 2000).

[c] TRVs based on growth of juvenile chinook salmon exposed to a mixture of naphthalene, 2-methylnaphthalene, dimethylnaphthalene, dibenzothiophene, acenaphthene, fluorene, 1,8-dimethyl(9H)fluorene, phenanthrene, 9-ethylphenanthrene, 9-ethyl-10-methylphenanthrene, 1-methyl-7-isopropylphenanthrene, anthracene, fluoranthene, pyrene, methyl pyrene, benz(a)anthracene, chrysene, benzo(a)pyrene, benzo(k)fluoranthene, benzo(g,h,i)perylene, dibenzanthracene (Meador et al. 2006).

[d] Maximum concentration of total PAHs in LDW sediment.

[e] Maximum total PAH concentration in benthic invertebrate tissue collected from the LDW.

[f] Maximum concentration of total PAHs in diet calculated using Equation 2-1.

dw – dry weight
J – estimated concentration
N – tentative identification
na – not available
LOAEL – lowest-observed-adverse-effect level
NOAEL – no-observed-adverse-effect level
PAH – polycyclic aromatic hydrocarbon
TRV – toxicity reference value

Although PAHs were not selected as a COPC for fish, regional field studies have suggested a correlation between PAH contamination of sediment and adverse effects on reproduction observed in English sole from Puget Sound, including English sole from the LDW (as discussed in Section A.6.2.2). Dietary effects data for PAHs are limited, however, and no laboratory studies investigating effects of ingested PAHs on fish reproduction are available.[24]

The LOAEL of 951 mg/kg dw in diet reported in Meador et al. (2006) was selected as the LOAEL TRV for fish in this section because it was the lowest LOAEL reported. The NOAEL of 324 mg/kg dw from this study was also selected. In the selected study, Meador et al. (2006) exposed juvenile chinook salmon to a mixture of 21 PAHs[25] in a

[24] Four studies that evaluated the dietary toxicity of benzo(a)pyrene to fish were identified (Wu et al. 2003; Hart and Heddle 1991; Hendricks et al. 1985; Rice et al. 2000). Effects investigated included growth or survival for English sole, rainbow trout, and aeriolated grouper. Two studies that evaluated the dietary toxicity of PAH mixtures to fish were identified (Palm et al. 2003; Meador et al. 2006) (see Attachment 8, Table 1). Both studies evaluated effects on growth of juvenile chinook salmon. The uncertainty section (Section A.6.2.1) discusses growth effects on English sole reported in Rice et al. (2000), in which fish were exposed to polychaetes that had previously been exposed to field-collected creosote-contaminated sediments from a Superfund site.

[25] The PAH mixture included the following chemicals: naphthalene, 2-methylnaphthalene, dimethylnaphthalene, dibenzothiophene, acenaphthene, fluorene, 1,8-dimethyl(9H)fluorene,

synthetic diet. Reduced body weight in exposed fish was observed at the LOAEL relative to the control. Palm et al. (2003) observed no adverse effects on juvenile chinook salmon growth following dietary exposure to a mixture of 14 PAHs for 7 weeks at total PAH concentrations of up to 280 mg/kg dw.

Because screening exposure assumptions would be similar for all fish ROCs, no ROC-specific screen was conducted for COPCs evaluated using a dietary approach. Therefore, the COIs that were identified in the dietary screen as COPCs were selected as COPCs for all three fish ROCs. The COPCs that will be evaluated for each fish ROC are presented in Table A.2-28.

Table A.2-28. 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

TBT – tributyltin

A.2.5.3 Wildlife

COPCs were identified for wildlife using a two-step process. The first step was to identify a list of COIs. Chemicals were identified as COIs for wildlife (with the exception of spotted sandpiper) if they met at least two of the following three criteria:

- Detection in at least 5% of LDW surface sediment samples
- Identification as a bioaccumulative chemical in EPA (2000)
- Detection in any LDW-collected tissue sample

For spotted sandpiper, the COI screen was more focused because sandpipers ingest benthic invertebrates only from intertidal areas. Thus, chemicals were identified as COIs for spotted sandpiper if they met at least two of the following three criteria:

- Detection in at least 5% of LDW intertidal surface sediment samples
- Identification as a bioaccumulative chemical in EPA (2000)

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phenanthrene, 9-ethylphenanthrene, 9-ethyl-10-methylphenanthrene, 1-methyl-7-isopropylphenanthrene, anthracene, fluoranthene, pyrene, methyl pyrene, benz(a)anthracene, chrysene, benzo(a)pyrene, benzo(k)fluoranthene, benzo(g,h,i)perylene, dibenzanthracene.

◆ Detection in any LDW-collected benthic invertebrate tissue sample[26]

Table A.2-29 presents a summary of chemicals identified as COIs as a result of this screen. Detailed results of the COI screening step for wildlife are presented in Attachment 4.

Table A.2-29. Chemicals identified as COIs for wildlife ROCs

COIs		
Metals		
Antimony	Copper	Selenium
Arsenic	Lead	Silver
Cadmium	Mercury	Thallium
Chromium	Methylmercury[a]	Vanadium
Chromium VI[a, b]	Molybdenum	Zinc
Cobalt	Nickel	
Organometals		
Monobutyltin as ion	TBT as ion	
Dibutyltin as ion	Tetrabutyltin as ion[b]	
PAHs		
C1-Chrysenes[b]	C3-Phenanthrenes/anthracenes	Benzo(g,h,i)perylene
C1-Dibenzothiophenes	C4-Naphthalenes	Benzo(k)fluoranthene
C1-Fluoranthene/pyrene	C4-Phenanthrenes/anthracenes	Chrysene
C1-Fluorenes	1-Methylnaphthalene	Dibenzo(a,h)anthracene
C1-Phenanthrenes/anthracenes	2-Methylnaphthalene	Dibenzofuran
C2-Chrysenes[b]	Acenaphthene	Fluoranthene
C2-Dibenzothiophenes	Acenaphthylene	Fluorene
C2-Fluorenes	Anthracene	Indeno(1,2,3-cd)pyrene
C2-Naphthalenes	Benzo(a)anthracene	Naphthalene
C2-Phenanthrenes/anthracenes	Benzo(a)pyrene	Perylene
C3-Fluorenes	Benzo(b)fluoranthene	Phenanthrene
C3-Naphthalenes	Benzo(e)pyrene	Pyrene
Phthalates		
Bis(2-ethylhexyl) phthalate	Diethyl phthalate	Di-n-butyl phthalate
Butyl benzyl phthalate[b]	Dimethyl phthalate[b]	
Other SVOCs		
4-Chlorophenyl phenyl ether[b]	Biphenyl	Hexachlorobenzene
4-Methylphenol	Carbazole[b]	Phenol
Benzoic acid	Dibenzothiophene	
PCBs		
Total PCBs		

[26] Includes amphipods, benthic invertebrates, crabs, and clams.

COIs		
Dioxins and Furans		
Organochlorine Pesticides		
Total DDTs[c]	delta-BHC	Endrin aldehyde[d]
Aldrin	gamma-BHC	Heptachlor
Dieldrin	alpha-Endosulfan	Heptachlor epoxide
alpha-BHC	beta-Endosulfan	Methoxychlor
beta-BHC	Endrin	Total chlordane[e]

[a] Chromium VI and methylmercury (both bioaccumulative chemicals and detected in at least 5% of surface sediment samples) were not analyzed in any of the tissue samples; these chemicals are assumed to be incorporated in total mercury and total chromium risk analyses.

[b] Not a COI for spotted sandpiper.

[c] Includes 2,4'-DDD, 2,4'-DDE, 2,4'-DDT, 4,4'-DDD, 4,4'-DDE, and 4,4'-DDT.

[d] COI for spotted sandpiper only.

[e] Includes alpha-chlordane, gamma-chlordane, oxychlordane, cis-nonachlor, and trans-nonachlor.

COI – chemical of interest

PCB – polychlorinated biphenyl

SVOC – semivolatile organic compound

TBT – tributyltin

In the second step of the screening process, the maximum exposure dose of each COI was compared to a NOAEL for that chemical. If the maximum exposure dose was greater than the NOAEL, the chemical was identified as a COPC for wildlife. This step was conducted for all COIs, except dioxins and furans, as discussed in Section A.2.5.

For COIs other than dioxins and furans, the scientific literature was searched[27] to identify toxicity reference values (TRVs) for COIs. The literature search included BIOSIS, EPA's ECOTOX database, the National Library of Medicine's TOXNET database, the US Fish and Wildlife Service's Contaminant Review series, the Oak Ridge National Laboratory's database, and EPA's IRIS database. Original sources of toxicity data were obtained and reviewed to verify effects data summarized in the databases as well as the suitability of the studies. The databases were searched for studies that evaluated effects on survival, growth, and reproduction (including developmental effects).

TRVs were based on the risk evaluation method used. For wildlife, dietary dose studies were identified. In many cases, the toxicity literature presented data only as a concentration in food, so these values were converted to a daily dose (mg/kg bw/day) using the animal's body weight and ingestion rate (IR). The following guidelines were considered in the selection of TRVs for wildlife.

[27] For COIs that were evaluated in the Phase 1 ERA (Windward 2003b), toxicity literature published after January 2001 was searched for new toxicity data. If any COIs were not evaluated in Phase 1 ERA, toxicity literature was searched for appropriate toxicity data and was not limited by date.

- Studies using field-collected data were not used to develop TRVs, but were considered if no other toxicity data were available for a COI.
- Studies conducted using IP injection, intramuscular injection, forced ingestion, or oral gavage as exposure routes were not considered for deriving TRVs unless no other toxicity data are available for a COI.
- Studies using drinking water as the exposure medium were not used to develop TRVs because bioavailability from water may be different from that of food. If no other toxicity data were available, then drinking water studies were considered.
- Studies with egg production endpoints for chicken or quail, such as Edens and Garlich (1983) and Edens et al. (1976) are considered highly uncertain and were only considered if data from other more appropriate studies were not available. These data are considered uncertain because chickens and quail have been bred to have high egg-laying rates. Even with a significant reduction in their baseline egg production, these egg production rates may be much higher than those of any wild avian species. These differences in reproductive physiology result in high uncertainty in extrapolating a reproductive effect threshold from egg production rates for chickens or quails.
- Toxicity studies conducted with chemical forms not likely found in the LDW, such as the fungicide methylmercury dicyandiamide, were not used to develop TRVs. Toxicity of these chemical forms is not comparable to the toxicity of forms of chemicals present in the LDW.

After the literature search was conducted, all acceptable studies for TRV derivation were compiled, as presented in Attachments 9 and 10. For each COI, TRVs were selected for both the NOAEL and the LOAEL. TRV selection rules and uncertainty factors discussed above for crab were used for wildlife as well.

Of the 86 COIs, TRVs were identified for 29 COIs for birds and 41 COIs for mammals (Tables A.2-30 and A.2-31, respectively). Selected TRVs are presented in Table A.2-32 for birds and in Table A.2-33 for mammals.

Table A.2-30. Results of TRV search for COIs for birds

COIs for Birds		
COIs with TRVs		
Arsenic	Thallium	Aldrin
Cadmium	Vanadium	Dieldrin
Chromium	Zinc	gamma-BHC
Cobalt	TBT as ion	alpha-Endosulfan
Copper	Benzo(a)pyrene	beta-Endosulfan
Lead	Total PAH	Endrin
Mercury	Bis(2-ethylhexyl) phthalate	Heptachlor
Molybdenum	Hexachlorobenzene	Methoxychlor
Nickel	Total PCBs	Total chlordane
Selenium	Total DDTs	
COIs without TRVs		
Antimony	Benzo(k)fluoranthene	Di-n-butyl phthalate
Silver	Chrysene	4-Chlorophenyl phenyl ether
Monobutyltin as ion	Dibenzo(a,h)anthracene	4-Methylphenol
Dibutyltin as ion	Dibenzofuran	Benzoic acid
Tetrabutyltin as ion	Fluoranthene	Biphenyl
1-Methylnaphthalene	Fluorene	Carbazole
2-Methylnaphthalene	Indeno(1,2,3-cd)pyrene	Dibenzothiophene
Acenaphthene	Naphthalene	Phenol
Acenaphthylene	Perylene	alpha-BHC
Anthracene	Phenanthrene	beta-BHC
Benzo(a)anthracene	Pyrene	delta-BHC
Benzo(b)fluoranthene	Butyl benzyl phthalate	Heptachlor epoxide
Benzo(e)pyrene	Diethyl phthalate	
Benzo(g,h,i)perylene	Dimethyl phthalate	

COI – chemical of interest

PAH – polycyclic aromatic hydrocarbon

PCB – polychlorinated biphenyl

TBT – tributyltin

TRV – toxicity reference value

Table A.2-31. Results of TRV search for COIs for mammals

COIs for Mammals		
Chemicals with TRVs		
Antimony	Dibutyltin as ion	Phenol
Arsenic	TBT as ion	Total PCBs
Cadmium	1-Methylnaphthalene	Total DDTs
Chromium	2-Methylnaphthalene	Aldrin
Cobalt	Benzo(a)pyrene	Dieldrin
Copper	Naphthalene	beta-BHC
Lead	Bis(2-ethylhexyl) phthalate	gamma-BHC
Mercury	Butyl benzyl phthalate	alpha-Endosulfan
Molybdenum	Diethyl phthalate	beta-Endosulfan
Nickel	Di-n-butyl phthalate	Endrin
Selenium	Benzoic acid	Heptachlor
Thallium	Biphenyl	Methoxychlor
Vanadium	Dibenzothiophene	Total chlordane
Zinc	Hexachlorobenzene	
Chemicals without TRVs		
Silver	Benzo(g,h,i)perylene	Phenanthrene
Monobutyltin as ion	Benzo(k)fluoranthene	Pyrene
Tetrabutyltin as ion	Chrysene	Dimethyl phthalate
Acenaphthene	Dibenzo(a,h)anthracene	4-Chlorophenyl phenyl ether
Acenaphthylene	Dibenzofuran	4-Methylphenol
Anthracene	Fluoranthene	Carbazole
Benzo(a)anthracene	Fluorene	alpha-BHC
Benzo(b)fluoranthene	Indeno(1,2,3-cd)pyrene	delta-BHC
Benzo(e)pyrene	Perylene	Heptachlor epoxide

COI – chemical of interest

PCB – polychlorinated biphenyl

TBT – tributyltin

TRV – toxicity reference value

Table A.2-32. TRVs selected for bird COIs

COI	Test Species	NOAEL (mg/kg bw/day)	LOAEL (mg/kg bw/day)	Endpoint	Source
Metals					
Arsenic	mallard	10	40	reproduction	Stanley et al. (1994)
Cadmium	chicken	1.5	na	growth	Cain et al. (1983)
	Japanese quail	na	4.0	growth	Richardson et al. (1974)
Chromium	black duck	1.0	5.0	reproduction	Haseltine et al. (unpublished), as cited in Sample et al. (1996)
Cobalt	chicken	2.31[a]	23.1	growth	Diaz et al. (1994)
Copper	chicken	ns	29	growth	Smith (1969)
	chicken	21	ns	growth	Poupoulis and Jensen (1976)
Lead	Japanese quail	ns	20	reproduction	Edens et al. (1976)
	American kestrel	5.82	na	reproduction	Pattee (1984)
Mercury	great egret	0.018[b]	0.091	growth	Spalding et al. (2000)
Molybdenum	chicken	6.0[b]	30	reproduction	Lepore and Miller (1965)
Nickel	mallard	77	107	growth	Cain and Pafford (1981)
Selenium	mallard	0.50	0.82	reproduction	Heinz et al. (1987)
Thallium	pheasant	2.4[a]	24	survival	Hudson et al. (1984)
Vanadium	chicken	1.2	2.3	growth	Ousterhout and Berg (1981)
Zinc	chicken	82	124	growth	Roberson and Schaible (1960)
Organometals					
TBT	Japanese quail	1.4	3.6	reproduction	Coenen et al. (1992)
PAHs					
Benzo(a)pyrene	pigeon	0.28[b]	1.4	reproduction	Hough el al. (1993)
Total PAHs	mallard	8.0	40	growth	Patton and Dieter (1980)
Phthalates					
BEHP	chicken	65.8[c]	329	reproduction	Ishida et al. (1982)
PCBs					
PCBs	screech owl	0.49	na	reproduction	McLane and Hughes (1980)
	ringed turtle dove	na	1.4	reproduction	Peakall et al. (1972); Peakall and Peakall (1973)
Organochlorine Pesticides					
Aldrin	quail	0.008[b]	0.04	survival	Dewitt (1956)
Total chlordane	bobwhite quail	0.6	na	growth, survival	Ludke (1976)
	bobwhite quail	na	2.0	survival	Hill et al. (1975); Heath et al. (1972)
Total DDTs	mallard	0.064[d]	0.32	reproduction	Davison and Sell (1974)
Dieldrin	quail	0.08	0.12	survival	Dewitt (1956)
Endosulfan	gray partridge	10	na	reproduction	Abiola (1992)

COI	Test Species	NOAEL (mg/kg bw/day)	LOAEL (mg/kg bw/day)	Endpoint	Source
Endrin	quail	0.070	0.20	survival	DeWitt (1956)
gamma-BHC	mallard	1.6	3.6	reproduction	Chakravarty and Lahiri (1986); Chakravarty et al. (1986)
Heptachlor	bobwhite quail	0.5[a]	5.0	survival	Hill et al. (1975); Heath et al. (1972)
Hexachloro-benzene	Japanese quail	na	1.2	reproduction	Schwetz et al. (1974)
	Japanese quail	1.1	na	reproduction	Vos et al. (1971)
Methoxychlor	zebra finch	34.6	346	reproduction	Gee et al. (2004)[e]
				survival	Millam et al. (2002)[e]

a NOAEL estimated from an acute or subchronic LOAEL using an uncertainty factor of 10.

b NOAEL estimated from a chronic LOAEL using an uncertainty factor of 5.

c There was a NOAEL of 1.45 mg/kg bw/day from a study that reported no effect on eggshell thinning, but this is an unbounded NOAEL at a substantially lower concentration than the study with observed effects. Therefore, the NOAEL was estimated from the reproductive LOAEL using an uncertainty factor of 5.

d There was a NOAEL of 0.19 mg/kg bw/day from a study that reported no effect on eggshell thinning from exposure of barn owls to DDT (Mendenhall et al. 1983). However, as discussed in Section A.6.3.1.2, there is evidence indicating that p,p'-DDE rather than DDT is the likely cause of eggshell thinning (Lundholm 1997). Therefore, the NOAEL was estimated from the DDE LOAEL for eggshell thinning using a factor of 5.

e Both studies had the same LOAEL and NOAEL.

BEHP – bis(2-ethylhexyl) phthalate

bw – body weight

COI – chemical of interest

LOAEL – lowest-observed-adverse-effect level

na – not available

NOAEL – no-observed-adverse-effect level

ns – NOAEL or LOAEL not selected from this study

PAH – polycyclic aromatic hydrocarbon

PCB – polychlorinated biphenyl

TBT – tributyltin

TRV – toxicity reference value

Table A.2-33. TRVs selected for mammal COIs

COI	Test Species	NOAEL (mg/kg bw/day)	LOAEL (mg/kg bw/day)	Endpoint	Source
Metals					
Antimony	rat	1,489	na	growth, survival	Hext et al. (1999)
Arsenic	rat	2.6	5.4	growth	Byron et al. (1967)
Cadmium	rat	3.5	13	growth	Machemer and Lorke (1981)
Chromium	rat	1,466	na	growth, survival	Ivankovic and Preussman (1975)
Cobalt	rat	0.1	1.0	growth	Chetty et al. (1979)
Copper	mink	18	26	reproduction	Aulerich et al. (1982)
Lead	rat	11	90	growth	Azar et al. (1973)
Mercury	rat	0.0017[a]	0.0084	growth	Verschuuren et al. (1976)
Molybdenum	mouse	0.258[b]	2.58	reproduction, survival	Schroeder and Mitchener (1971)
Nickel	rat	na	20	reproduction	Ambrose et al. (1976)
	rat	8.4	ns	growth	
Selenium	rat	0.055	0.08	growth	Halverson et al. (1966)
Thallium	rat	0.74	na	growth	Formigli et al. (1986)
Vanadium	mouse	1.05	na	growth	Schroeder and Balassa (1967)
	rat	na	2.7	growth	Adachi et al. (2000)
Zinc	rat	160	320	reproduction	Schlicker and Cox (1968)
Organometals					
TBT	rat	0.4	2.0	reproduction	Omura et al. (2001)
Dibutyltin	rat	na	7.6	reproduction, growth	Ema et al. (2003)
	rat	3.8	na	growth	Harazono and Ema (2003)
PAHs					
1-Methylnaphthalene	mouse	150	na	growth	Murata et al. (1993)
2-Methylnaphthalene	mouse	54	114	growth	Murata et al. (1997)
Benzo(a)pyrene	mouse	2.0[a]	10	reproduction	MacKenzie and Angevine (1981)
Naphthalene	mouse	133	na	growth, survival	Shopp et al. (1984)
Phthalates					
BEHP	mouse	44	91	reproduction	Tyl et al. (1988)
Butyl benzyl phthalate	rat	250	750	growth, reproduction	Tyl et al. (2004)
Diethyl phthalate	mouse	1,860	3,721	growth, reproduction	Lamb et al. (1987)
Di-n-butyl phthalate	rat	16[a]	80	reproduction	Wine et al. (1997)
Other SVOCs					
Benzoic acid	rat	80	na	growth, survival	Ignat'ev (1965), as cited in IRIS (EPA 2006)
	rat	na	750	growth	Marquardt 1980

COI	Test Species	NOAEL (mg/kg bw/day)	LOAEL (mg/kg bw/day)	Endpoint	Source
Biphenyl	rat	50	250	survival	Ambrose et al. (1960), as cited in IRIS (EPA 2006)
Dibenzothiophene	mouse	4.7[b]	47[c]	survival	Leighton (1989)
	mouse	na	300	reproduction, survival	Plasterer et al. (1985)
Phenol	rat	60	120	growth	Argus Research Laboratories (1997), as cited in IRIS (EPA 2006)[d]
	rat	60	120	reproduction	Charles River Laboratories (1988) and NTP (1983), as cited in IRIS (EPA 2006)[d]
PCBs					
PCBs	mink	0.045[e]	0.089	reproduction	Brunstrom et al. (2001)
Organochlorine Pesticides					
Aldrin	rat	0.8	4.1	survival	Reuber (1980)
beta-BHC	rat	5.7	31	growth, survival	Van Velsen et al. (1986)
Total chlordane	mouse	0.18	0.92	growth	Khasawinah and Grutsch (1989)
Total DDTs	rat	1.2	na	reproduction	Duby et al. (1971)
	mouse	na	1.3	reproduction	Ware and Good (1967)
Dieldrin	mouse	0.038[a]	0.19	reproduction	Treon and Cleveland (1955)
Endosulfan	mouse	0.84	2.5	growth, survival	Hack et al. (1995)
Endrin	rat	0.40	ns	survival, growth	Treon et al. (1955)
	mouse	na	0.92	survival, reproduction	Good and Ware (1969)
gamma-BHC	rat	64	na	growth	Srinivasan et al. (1991)
Heptachlor	mink	1.0	1.8	growth, survival, reproduction	Crum et al. (1993)
Hexachlorobenzene	mink and ferret	0.026[a]	0.13	reproduction	Bleavins et al. (1984)
Methoxychlor	rat	17	na	growth, reproduction	Masutomi et al. (2003)
	rat	na	56	growth, reproduction	You et al. (2002)

a NOAEL estimated from an chronic LOAEL using an uncertainty factor of 5.

b NOAEL estimated from an acute or subchronic LOAEL using an uncertainty factor of 10.

c LOAEL estimated from an LD50 using an uncertainty factor of 10.

d Both studies had the same LOAEL and NOAEL.

e NOAEL estimated from a chronic LOAEL using an uncertainty factor of 2; the rationale for using this uncertainty factor is discussed in Section A.5.2.2.1.

BEHP – bis(2-ethylhexyl) phthalate
bw – body weight
COI – chemical of interest
LOAEL – lowest-observed-adverse-effect level
na – not available
NOAEL – no-observed-adverse-effect level
ns – NOAEL or LOAEL not selected from this study
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
SVOC – semivolatile organic compound
TBT – tributyltin
TRV – toxicity reference value

To identify COPCs for birds and mammals, NOAEL TRVs were compared to the estimated maximum dietary dose (expressed in mg/kg dw-day) for each of the five wildlife ROCs. The maximum dietary exposure dose was calculated using the following equation:

$$\text{Dose} = \frac{\text{FIR} \times C_{food}}{\text{BW}}$$ **Equation 2-2**

Where:

Dose	=	COIs ingested per day via food or intertidal sediment (mg COI/kg body weight/day)
FIR	=	food ingestion rate (kg dw food/day)
C_{food} (sandpiper)	=	80% maximum [benthic invertebrate tissue] + 20% maximum [intertidal sediment] (mg COI/kg dw)
C_{food} (other ROCs)	=	maximum concentration in prey tissue (mg COI/kg dw)
BW	=	wildlife species body weight (kg ww)

The maximum daily ingested doses for great blue heron, osprey, river otter, and harbor seal were calculated using the maximum detected tissue concentration[28] in any of the appropriate prey species in the Phase 2 ERA dataset (see Table A.2-12). For spotted sandpiper, the maximum dose was calculated using a weighted average of 80% of the maximum concentration in any benthic invertebrate tissue sample plus 20% of the maximum concentration in any intertidal sediment sample.[29]

The body weights and FIRs used to calculate the maximum dietary exposure doses for each wildlife ROC are presented in Table A.5-3. The derivation of these values is described in detail in Section A.5.1.2.

Estimates of the maximum daily ingested doses of 11 metals (arsenic, cadmium, chromium, cobalt, copper, lead, mercury, nickel, selenium, vanadium, and zinc), two organic compounds (total PCBs and total DDTs) were greater than their respective NOAEL TRVs for spotted sandpiper (Table A.2-34). Total DDTs will be discussed in the uncertainty analysis because of the uncertainty associated with organochlorine pesticide tissue data (see Section A.2.4.2.2). Risks from the 11 metals and total PCBs are evaluated in the baseline ERA for spotted sandpiper.

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[28] Incidental sediment ingestion was not considered in the screens for wildlife ROCs (except spotted sandpiper) because of the small amount of sediment assumed to be ingested (2% or less of diet).

[29] In risk calculations presented in Section 5.1, sediment ingestion rates are calculated separately from food ingestion rates. In those calculations, food is ingested at 100% of the food ingestion rate, with sediment comprising an additional component of the diet.

Table A.2-34. Results of COPC screen for spotted sandpiper

COI	Maximum Concentration: Benthic Invertebrate Prey (mg/kg dw)	Maximum Concentration: Intertidal Sediment (mg/kg dw)	Calculated Maximum Dietary Exposure Dose (mg/kg bw/day)[a]	NOAEL TRV (mg/kg bw/day)
Metals				
Arsenic	43 J	1,100	**40**	10
Cadmium	0.91	120	**3.9**	1.5
Chromium	58	1,100	**43**	1.0
Cobalt	5.2	140	**5.2**	2.31
Copper	170	12,000	**400**	21
Lead	220	23,000	**750**	5.82
Mercury	0.44	4.6	**0.21**	0.018
Molybdenum	2.6	49	1.9	6.0
Nickel	6.7	910	**29**	77
Selenium	2.4	20	**0.92**	0.50
Thallium	0.026	30	0.94	2.4
Vanadium	164	150	**6.7**	1.2
Zinc	380	9,700	**350**	82
Organometals				
TBT as ion	4.2	0.099	0.53	1.4
PAHs				
Benzo(a)pyrene	1.3	2.9	0.25	0.28
Total PAH	24	130	7.0	8.0
Phthalates				
Bis(2-ethylhexyl) phthalate	14 J	14	2.2	65.8
PCBs				
Total PCBs	21	220	**9.5**	0.49
Organochlorine Pesticides				
Aldrin	0.0073 JN	0.00081	0.00094	0.008
alpha-Endosulfan	0.040 JN	0.071	0.0072	10
beta-Endosulfan	0.082 JN	0.010	0.011	10
Total chlordane	0.11 JN	0.23	0.021	0.6
Total DDTs	1.9 JN	2.9	**0.33**	0.064
Dieldrin	0.032 JN	0.28	0.013	0.08
Endrin	0.0099 JN	0.0091	0.0015	0.070
gamma-BHC	0.018 JN	0.0067	0.0025	1.6
Heptachlor	0.11 JN	0.00089	0.014	0.5
Methoxychlor	0.058 JN	0.099	0.010	34.6

[a] Calculated using Equation 2-2.

COI – chemical of interest
dw – dry weight
J – estimated concentration
N – tentative identification
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
TBT – tributyltin
TRV – toxicity reference value

Bold identifies maximum dietary exposure concentrations greater than the NOAEL.

Maximum dietary exposures of great blue heron and osprey to chromium, lead, mercury, total PCBs, and total DDTs exceeded their respective NOAEL TRVs (Table A.2-35). Total DDT is discussed in the uncertainty assessment because of the uncertainty associated with JN-qualified organochlorine pesticide tissue data (see Section A.2.4.2.2). Risks from chromium, lead, mercury, and total PCBs are evaluated in the exposure and effects assessment and risk characterization for great blue heron and osprey (Sections A.6.3.2 and A.6.3.3, respectively).

Table A.2-35. Results of COPC screen for great blue heron and osprey

COI	Maximum Chemical Concentration in Fish (mg/kg dw)	Calculated Maximum Dietary Exposure Dose (mg/kg bw/day)[a]		NOAEL TRV (mg/kg bw/day)
		Great Blue Heron	Osprey	
Metals				
Arsenic	120	5.0	5.9	10
Cadmium	6.5	0.27	0.32	1.5
Chromium	58	**2.4**	**2.8**	1.0
Cobalt	5.2	0.22	0.25	2.31
Copper	380	16	19	21
Lead	220	**9.2**	**11**	5.82
Mercury	0.49 M	**0.020**	**0.024**	0.018
Molybdenum	2.7	0.11	0.13	6.0
Nickel	23	0.96	1.1	17
Selenium	3.6	0.15	0.18	0.50
Thallium	0.054	0.0023	0.0026	2.4
Vanadium	22	0.92	1.1	1.2
Zinc	380	16	19	82
Organometals				
TBT as ion	4.2	0.18	0.21	1.4
PAHs				
Benzo(a)pyrene	1.3	0.054	0.063	0.28
Total PAHs	24 J	1.0	1.2	8.0
Phthalates				
Bis(2-ethylhexyl) phthalate	14	0.58	0.68	65.8
PCBs				
Total PCBs	69	**2.9**	**3.4**	0.49
Organochlorine Pesticides				
Aldrin	0.025 JN	0.0010	0.0012	0.008
alpha-Endosulfan	0.21 JN	0.0088	0.010	10
beta-Endosulfan	0.18 JN	0.0075	0.0088	10
Total chlordane	1.2 JN	0.050	0.059	0.60
Total DDTs	3.8 JN	**0.16**	**0.19**	0.064
Dieldrin	0.032 JN	0.0013	0.0016	0.08
Endrin	0.17 JN	0.0071	0.0083	0.070

COI	Maximum Chemical Concentration in Fish (mg/kg dw)	Calculated Maximum Dietary Exposure Dose (mg/kg bw/day)[a]		NOAEL TRV (mg/kg bw/day)
		Great Blue Heron	Osprey	
gamma-BHC	0.033 JNM	0.033	0.0016	1.6
Heptachlor	0.11 JN	0.0046	0.0054	0.5
Hexachlorobenzene	0.032 JNM	0.0013	0.0016	1.1
Methoxychlor	0.70 JNM	0.029	0.034	34.6

[a] Calculated using Equation 2-2.

bw – body weight

COI – chemical of interest

COPC – chemical of potential concern

dw – dry weight

J – estimated concentration

M – mean concentration

N – tentative identification

NOAEL – no-observed-adverse-effect level

PAH – polycyclic aromatic hydrocarbon

PCB – polychlorinated biphenyl

TBT – tributyltin

TRV – toxicity reference value

Bold identifies maximum dietary exposure concentrations greater than the NOAEL.

For river otter, estimated maximum exposures to arsenic, cobalt, mercury, selenium, and PCBs were greater than their respective NOAEL TRVs (Table A.2-36). For harbor seal, estimated maximum exposures to mercury and PCBs exceeded their respective NOAEL TRVs (Table A.2-37). Risks associated with these COPCs are evaluated in this baseline ERA. A summary of the COPCs that will be evaluated for each wildlife ROC in the risk characterization is presented in Table A.2-29.

Table A.2-36. Results of COPC screen for river otter and harbor seal

COI	Maximum Chemical Concentration in Fish (mg/kg dw)	Calculated Maximum Dietary Exposure Dose (mg/kg bw/day)[a]		NOAEL TRV (mg/kg bw/day)
		River Otter	Harbor Seal	
Metals				
Antimony	1.9	0.062	0.014	1,489
Arsenic	120	**3.9**	0.089	2.6
Cadmium	6.5	0.21	0.048	3.5
Chromium	58	1.9	0.43	1,466
Cobalt	5.2	**0.17**	0.039	0.1
Copper	380	12	2.8	18
Lead	218	7.2	1.6	11
Mercury	0.49 M	**0.016**	**0.0036**	0.0017
Molybdenum	2.78	0.088	0.020	0.258
Nickel	238	0.75	0.17	8.4
Selenium	3.68	**0.12**	0.027	0.055
Thallium	0.054	0.0018	0.00040	0.74
Vanadium	228	0.72	0.16	1.05
Zinc	380	12	2.8	160

COI	Maximum Chemical Concentration in Fish (mg/kg dw)	Calculated Maximum Dietary Exposure Dose (mg/kg bw/day)[a]		NOAEL TRV (mg/kg bw/day)
		River Otter	Harbor Seal	
Organometals				
Dibutyltin	0.51	0.017	0.0038	3.8
TBT as ion	4.2	0.14	0.031	0.4
PAHs				
1-Methylnaphthalene	0.075	0.0024	0.00056	150
2-Methylnaphthalene	0.084	0.0027	0.00063	54
Benzo(a)pyrene	1.3	0.042	0.010	2.0
Naphthalene	0.13 J	0.0042	0.0010	133
Phthalates				
Bis(2-ethylhexyl) phthalate	14	0.46	0.10	44
Butyl benzyl phthalate	16	0.52	0.12	250
Diethyl phthalate	5.7 JM	0.19	0.042	1,860
Di-n-butyl phthalate	9.3	0.30	0.069	16
Other SVOCs				
Benzoic acid	220	7.2	1.6	80
Biphenyl	0.076 J	0.0025	0.00057	50
Dibenzothiophene	0.27 J	0.0088	0.0020	4.7
Phenol	12	0.39	0.089	60
PCBs				
Total PCBs	69	**2.2**	**0.51**	0.045
Organochlorine Pesticides				
Aldrin	0.025 JN	0.00081	0.00019	0.8
alpha-Endosulfan	0.21 JN	0.0068	0.0016	0.84
beta-BHC	0.13 JN	0.0042	0.0010	5.7
beta-Endosulfan	0.18 JN	0.0059	0.0013	0.84
Total chlordane	1.2 JN	0.039	0.0089	0.18
Total DDTs	3.8 JN	0.12	0.028	1.2
Dieldrin	0.032 JN	0.0010	0.00024	0.038
Endrin	0.17 JN	0.0055	0.0013	0.40
gamma-BHC	0.033 JNM	0.0011	0.00025	64
Heptachlor	0.11 JN	0.0036	0.00082	1.0
Hexachlorobenzene	0.032 JNM	0.0010	0.00024	0.026
Methoxychlor	0.70 JNM	0.023	0.0052	17

[a] Calculated using Equation 2-2.

bw – body weight
COI – chemical of interest
COPC – chemical of potential concern
dw – dry weight
J – estimated concentration
M – mean concentration
N – tentative identification
NOAEL – no-observed-adverse-effect level
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
SVOC – semivolatile organic compound
TBT – tributyltin
TRV – toxicity reference value

Bold identifies maximum dietary exposure concentrations greater than the NOAEL.

Table A.2-37. COPCs evaluated in the risk characterization for birds and mammals

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

This COPC screen for birds evaluated exposure and effects using a dietary approach. An alternative method to assess exposure of birds involves chemical analysis of bird eggs. In 1998, the USGS collected eggs from the great blue heron colony in West Seattle and analyzed them for PCBs (Krausmann 2002a). The results indicated that great blue herons were exposed to PCBs, although the source of the exposure was not established. Because the available QA/QC data were insufficient to use these data quantitatively in this baseline ERA, exposure of great blue herons to PCBs was evaluated using a dietary exposure approach. Osprey eggs collected from nests along the LDW have also been analyzed for PCBs by USGS. Because these data are not yet available from USGS, COPC concentrations in osprey eggs were estimated using biomagnification factors (BMFs) in the uncertainty analysis. In the uncertainty analysis, estimated PCB concentrations in osprey eggs are compared to egg TRVs from the literature.

A.2.6 Conceptual Site Model

A CSM is a graphical representation of chemical sources, transport mechanisms, exposure pathways, exposure routes, and potentially exposed receptors. This section presents the CSM that synthesizes pathways of exposure of ROCs to chemical stressors. Based on this model and assessment endpoints for this risk assessment, measures of exposure and effect are selected and discussed. These assessment endpoints determine which endpoints will be examined in detail in this baseline ERA for each ROC/COPC combination that was retained for further analysis based on the analyses in Section A.2.0.

Although chemical sources other than sediment exist in the LDW, the exposure assessment for each ROC focused on scenarios that include a direct (i.e., ingestion or direct contact) or indirect (i.e., ingestion of fish or benthic invertebrates) pathway for sediment-associated chemicals. Sources of chemical contamination to the sediments are discussed further in the Phase 2 RI.

Ecological risks from exposure to surface water were previously evaluated quantitatively by King County (1999b). The surface water risk estimates from that ERA have been incorporated in this ERA for fish. Additionally, surface water exposure data have been incorporated into bird and mammal exposure calculations in this ERA.

A.2.6.1 Exposure pathways

This section discusses the potential for ROCs in the LDW to be exposed significantly to COPCs. For COPCs to pose risk to ROCs, the exposure pathway must be complete. Identifying complete exposure pathways prior to a quantitative evaluation allows the assessment to focus on only those chemicals that can reach ecological receptors (EPA 1997a, b). An exposure pathway is considered complete if a chemical can travel from a source to ecological receptors and the receptor is exposed via one or more exposure routes (EPA 1997a, b). Complete pathways can be of varying importance, so key pathways that reflect maximum exposures to ecological receptors sensitive to that chemical (EPA 1997a, b) are identified as having more importance than pathways likely to provide a very low fraction of the total exposure of an ROC to a chemical.

Pathways for the exposure of ROCs to sediment-associated chemicals in the LDW were designated in one of four ways: complete and significant, complete and significance unknown, complete and insignificant, or incomplete. Each of the four designations is defined below, including whether it will be further evaluated in this ERA. This section also presents a brief rationale for each designation by receptor. The CSM is presented in Figures A.2-2 and A.2-3 for aquatic species and wildlife, respectively.

- **Complete and significant:** There is a direct link between the receptor and chemical via this pathway, and the specific pathway is considered to be potentially important. Pathways classified as complete and significant will be addressed in greater detail in the exposure and effects assessment (Sections A.3.0, A.4.0, and A.5.0).
- **Complete and significance unknown:** There is a direct link between the receptor and the chemical via this pathway; however, there is insufficient data available to quantify the significance of the pathway in the overall assessment of exposure. Pathways classified as complete and significance unknown will be discussed qualitatively in the uncertainty analysis (Section A.6.0).
- **Complete and insignificant:** There is a direct link between the receptor and the chemical via this pathway; however, the significance of this pathway in terms of

overall exposure is considered to be very low. Pathways classified as complete and insignificant will not be evaluated further in this baseline ERA.

- **Incomplete:** There is no direct pathway between the receptor and the chemical. Pathways classified as incomplete will not be evaluated further in this baseline ERA.

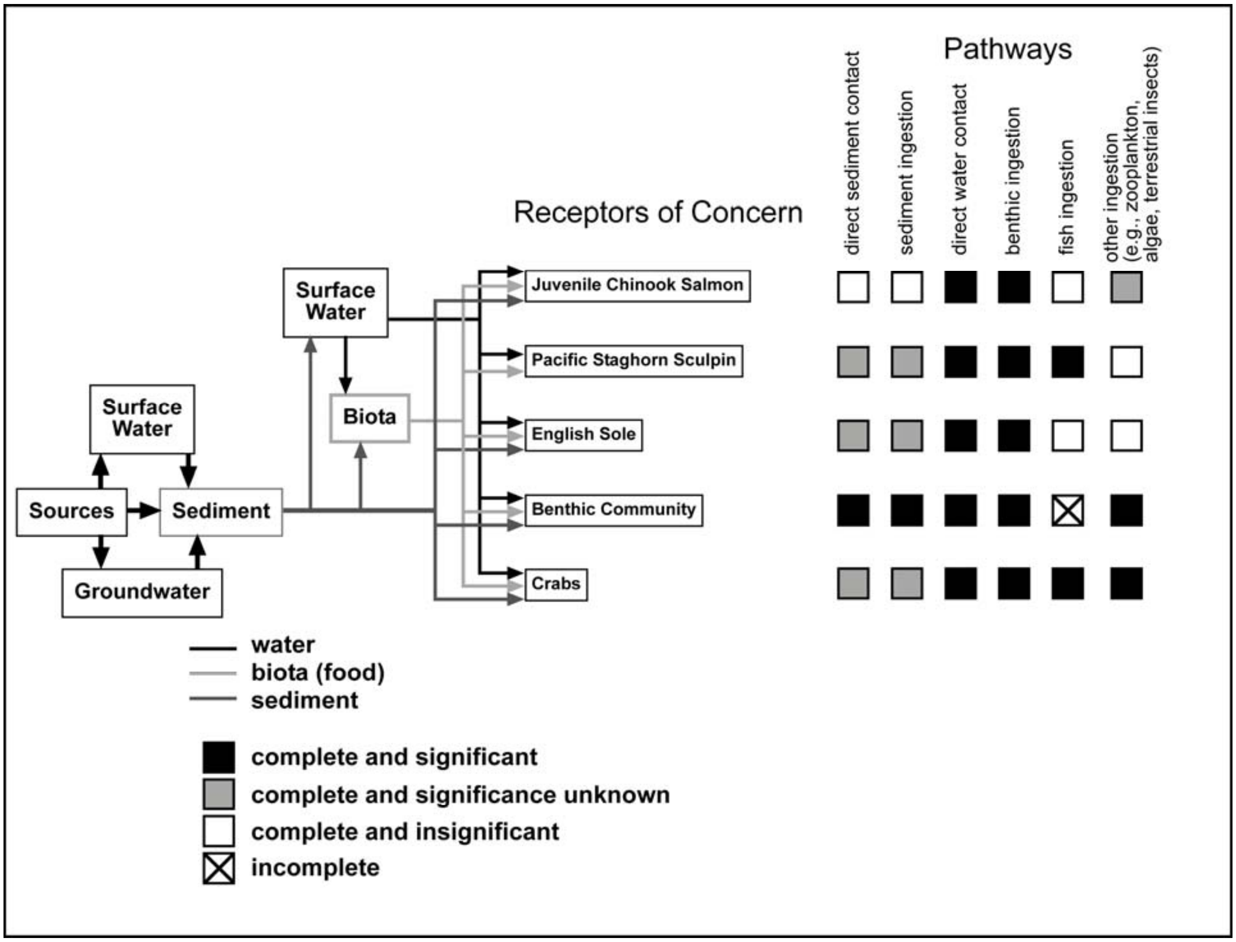

Figure A.2-2. Conceptual site model for fish and the benthic invertebrate community

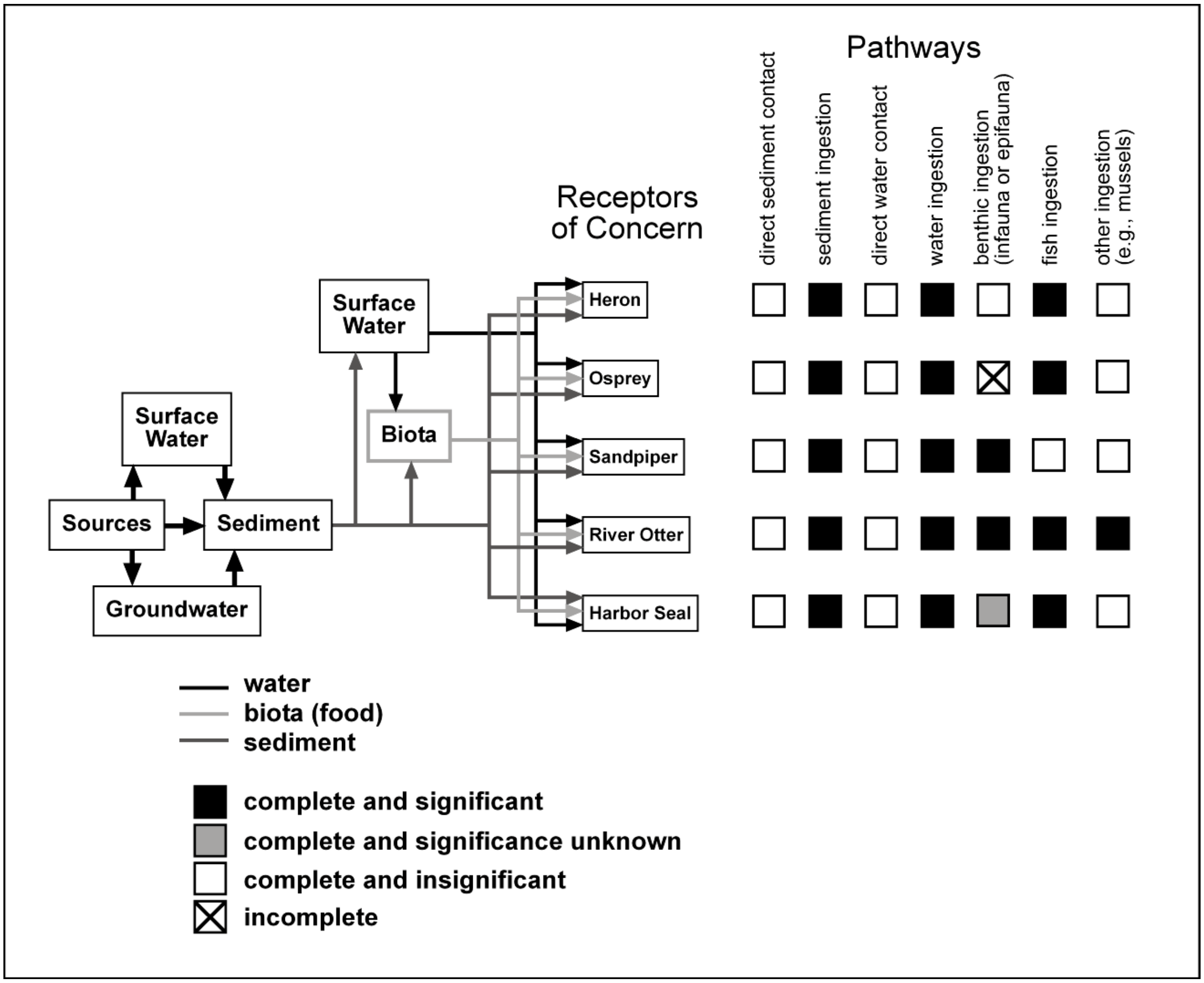

Figure A.2-3. Conceptual site model for wildlife

A.2.6.2 Food web diagram

To understand the potential exposure pathways of a sediment-associated chemical to upper trophic-level ROCs, knowledge of food web relationships is important. The generalized food web diagram for the LDW shows the relationship between major trophic groups, and lists several representative species (Figure A.2-4).[30] The relationship among trophic groups illustrates the pathways for chemical transfer through the food web through the ingestion of prey; Figure A.2-4 provides additional detail for the prey ingestion pathways identified in Figures A.2-2 and A.2-3.

[30] Note that some organisms could have representatives in more than one box, depending on their life stage.

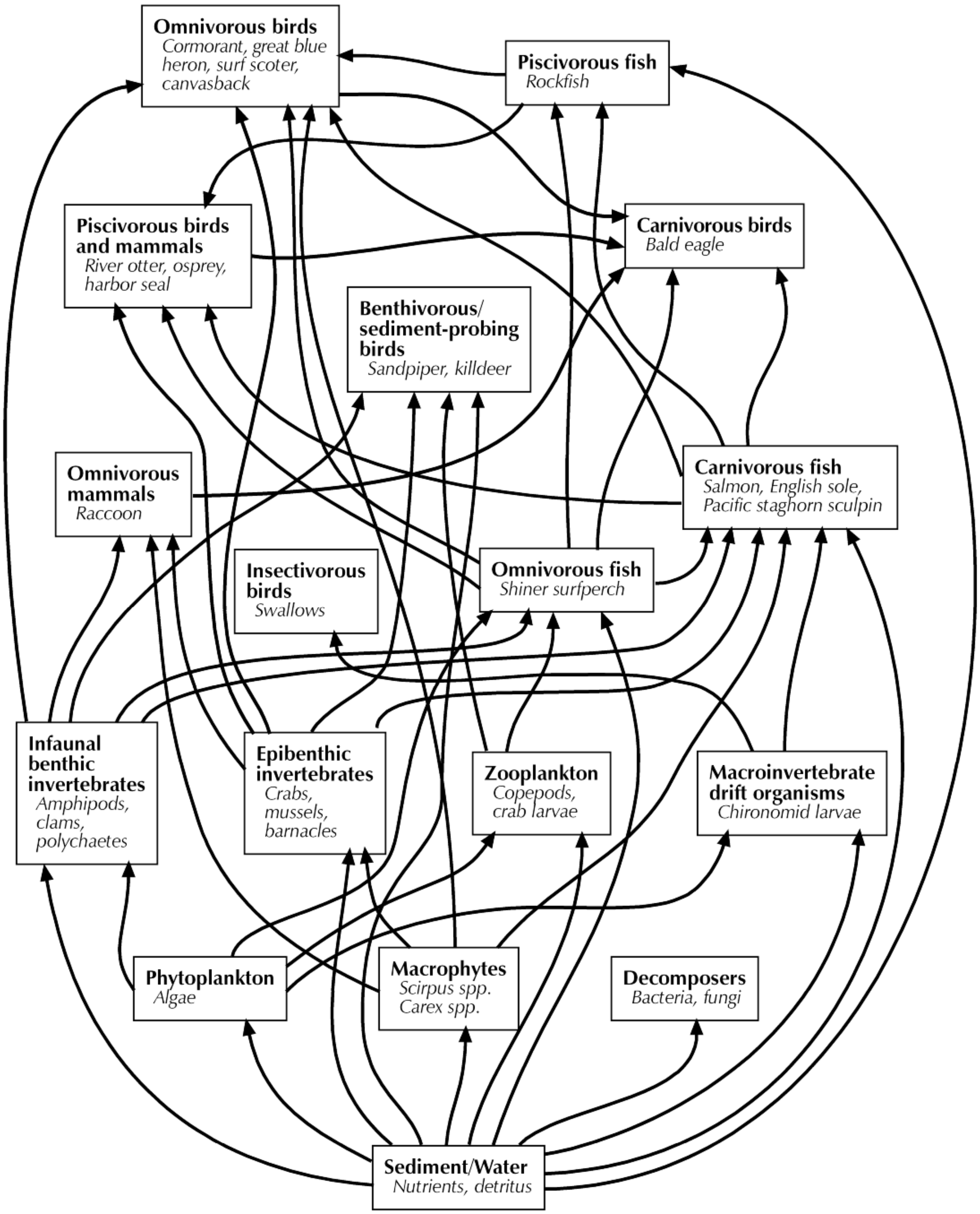

Figure A.2-4. Generalized food web diagram for the LDW

A.2.6.3 Assessment endpoints and measures of effect and exposure

An assessment endpoint is defined as an explicit expression of the ecological value that is to be protected (EPA 1992). Ecological values include those roles and processes vital to ecosystem function, those providing critical resources such as habitat and fisheries, and the perception of value by humans (e.g., threatened or endangered [T&E] species). An assessment endpoint must define both the valued entity and the characteristic of the entity to be protected. They provide direction for the risk assessment and are the basis for the analyses. Unless an ecological receptor is listed as a T&E species, assessment endpoints are selected that are relevant to population-level rather than individual effects. For T&E species, risks to individuals are important to evaluate (EPA 1998), although specific guidance regarding this approach is not available. At other EPA Region 10 Superfund sites, such as Coeur d'Alene and Blackbird Mine, greater emphasis has been placed on the NOAELs than on the LOAELs for the protection of T&E species.

Selection of assessment endpoints was based on available information regarding the ecological relevance of the endpoint and on societal values. In addition, assessment endpoints were evaluated to ensure that their protection would likely result in protection of other valued entities within the system. Finally, endpoints selected must be amenable to assessment either through previously existing data or data that were collected as part of the Phase 2 RI.

Assessment endpoints for each ROC are listed in Table A.2-38 along with the measures of exposure and effect used in the exposure and effects assessments (Sections A.3.0 through A.5.0). Survival, growth, and reproduction[31] are the key endpoints under review for most species in this assessment. Biomarker and histological endpoints are 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. For biomarkers to be useful in determining sediment-associated risk, they must have clear dose-response data relating exposure to ecologically significant effects. Other responses, such as biliary fluorescent aromatic compounds and deoxyribonucleic acid (DNA) adducts, are categorized as a measure of exposure rather than as an assessment endpoint. Research is ongoing in the area of biomarkers to better understand their significance for potential use in ERAs.

[31] The fish reproductive endpoint includes early life stage (i.e., egg and embryo) survival and growth through the fry stage. The wildlife reproduction endpoint includes survival and growth of offspring after exposure of parents.

Table A.2-38. Assessment endpoints for ROCs and measures of effect and exposure

ROC	Assessment Endpoint	Assessment Scale	Measures of Effect	Measures of Exposure
Benthic				
Benthic invertebrate communities	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: crab 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 will 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

ROC	Assessment Endpoint	Assessment Scale	Measures of Effect	Measures of Exposure
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
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

ROC	Assessment Endpoint	Assessment Scale	Measures of Effect	Measures of Exposure
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 (1999a) and Meador et al. (2002) and through a direct assessment of effects (i.e., imposex) on gastropods collected in the LDW.

DMMP – Dredged Material Management Program
LDW – Lower Duwamish Waterway
ROC – receptor of concern
SL – screening level
SMS – Washington State Sediment Management Standards
TBT – tributyltin
TRV – toxicity reference value
VOC – volatile organic compound

A.3.0 Exposure and Effects Assessment: Benthic Invertebrates

The benthic invertebrate community as a whole and crabs were selected as ROCs in the problem formulation to represent benthic invertebrates that may be exposed to sediment-associated chemicals in the LDW (Section A.2.3.1). In addition, site-specific measures of effect were evaluated for one benthic invertebrate group, meso- and neogastropods, because they are more sensitive to TBT exposure then other benthic invertebrates.

COPCs for the benthic invertebrate community were identified in the problem formulation (Section A.2.5.1). Forty-four chemicals based on the sediment chemistry data and one chemical based on porewater chemistry data were retained as COPCs for the benthic invertebrate community. TBT was also retained as a COPC based on the Phase 1 ERA (2003b). Two chemicals (total PCBs and zinc) were identified as COPCs for crabs in the problem formulation (Section A.2.5.1.3)

This section is divided into a benthic invertebrate community exposure assessment (Section A.3.1), benthic invertebrate community effects assessment (Section A.3.2), crab exposure assessment (Section A.3.3), and crab effects assessment (Section A.3.4). The exposure and effects data presented in this section are combined in the risk characterization (Section A.6.1).

A.3.1 BENTHIC INVERTEBRATE COMMUNITY EXPOSURE ASSESSMENT

This section presents the exposure assessment for the benthic invertebrate community based on surface sediment chemistry data for all COPCs except two: cis-1,2-dichloroethene and TBT. Exposures to cis-1,2-dichloroethene were assessed using porewater data. Exposures to TBT were assessed using a critical tissue-residue approach. TBT effects were also evaluated using a direct measure of effect on meso- and neogastropods, as discussed further in Section A.3.2.4. The sediment exposure assessment for all COPCs, except TBT, is presented in Section A.3.1.1, the porewater exposure assessment is presented in Section A.3.1.2, and the TBT exposure assessment is presented in Section A.3.1.3.

A.3.1.1 Sediment exposure assessment

In this section, surface sediment data for COPCs are presented to characterize the exposure regime for the benthic invertebrate community. Concentrations and detection frequencies of COPCs in the baseline surface sediment dataset[32] are

[32] In addition to surface sediment grab samples, representing chemical concentrations within the upper 10 cm of sediment at a particular location, the baseline dataset also includes co-located sediment samples from the clam and benthic invertebrate sampling events (Windward 2005b, d, e). These co-

presented in Table A.3-1. Benthic invertebrates have small home ranges. Therefore, exposure is assessed based on the concentration of a COPC at a particular location. The spatial distribution of concentrations is also relevant from a risk perspective for the benthic invertebrate community as a whole as a food resource for other species in the LDW. Spatial scale is discussed in the risk characterization (Section A.6.1.1) and uncertainties associated with assessing spatial scale are presented in the uncertainty analysis (Section A.6.1.1). Chemicals with RLs greater than the SQS or CSL chemical criteria are discussed in the uncertainty analysis and are also discussed in the Phase 2 RI. The locations of surface sediment samples are shown on Map A.2-3.

Table A.3-1. Chemical concentrations and detection frequencies in LDW surface sediments for COPCs identified for the benthic invertebrate community

COPC	Unit	Detection Frequency (%)[a]	Detected Concentration		
			Minimum	Maximum	Median
Metals					
Arsenic	mg/kg dw	754 / 814 (93)	1.2	1,100	12
Cadmium	mg/kg dw	565 / 797 (71)	0.030 J	120	0.5
Chromium	mg/kg dw	811 / 811 (100)	4.8	1,100 J	29
Copper	mg/kg dw	814 / 814 (100)	5	12,000 J	52.3
Lead	mg/kg dw	814 / 814 (100)	2	23,000	36
Mercury	mg/kg dw	715 / 831 (86)	0.021	4.6 J	0.16
Nickel	mg/kg dw	771 / 773 (100)	5	910	22
Silver	mg/kg dw	481 / 782 (62)	0.020	270	0.42
Zinc	mg/kg dw	810 / 811 (100)	16	9,700	114
Organometals					
TBT[b]	µg/kg dw	143 /159 (90)	0.28J	3,000	29
PAHs					
2-Methylnaphthalene	µg/kg dw	139 / 780 (18)	1.0 J	3,300	27
Acenaphthene	µg/kg dw	301 / 790 (38)	1.0 J	5,200	40
Anthracene	µg/kg dw	552 / 790 (70)	2.0	10,000	82
Benzo(a)anthracene	µg/kg dw	717 / 790 (91)	3.6 J	8,400	195
Benzo(a)pyrene	µg/kg dw	716 / 784 (91)	5.8 J	7,900	200
Benzo(g,h,i)perylene	µg/kg dw	648 / 785 (83)	6.1	3,800	120
Total benzofluoranthenes	µg/kg dw	725 / 784 (92)	4.6	17,000	450
Chrysene	µg/kg dw	739 / 790 (94)	12	7,700	290
Dibenzo(a,h)anthracene	µg/kg dw	400 / 790 (51)	1.6 J	1,500	50
Dibenzofuran	µg/kg dw	246 / 789 (31)	1.0 J	4,200	33
Fluoranthene	µg/kg dw	759 / 790 (96)	18	24,000	430

located sediment samples consisted of surface sediment grab samples that were collected over a small area and composited.

COPC	Unit	Detection Frequency (%)[a]	Detected Concentration		
			Minimum	Maximum	Median
Fluorene	µg/kg dw	371 / 790 (47)	1.4 J	6,800	42
Indeno(1,2,3-cd)pyrene	µg/kg dw	692 / 785 (88)	6.5	4,300	130
Naphthalene	µg/kg dw	148 / 780 (19)	3.0 J	5,300	31
Phenanthrene	µg/kg dw	724 / 790 (92)	7.1	28,000	190
Pyrene	µg/kg dw	750 / 790 (95)	7 J	16,000	410
Total HPAH	µg/kg dw	767 / 790 (97)	20	85,000	2,100
Total LPAH	µg/kg dw	729 / 790 (92)	9.1	44,000	270
Phthalates					
Bis(2-ethylhexyl) phthalate	µg/kg dw	635 / 794 (80)	5.4	14,000	335
Butyl benzyl phthalate	µg/kg dw	390 / 784 (50)	2.0	7,100	40
Dimethyl phthalate	µg/kg dw	136 / 784 (17)	2.0 J	1,400 J	22
Other SVOCs					
1,2,4-Trichlorobenzene	µg/kg dw	5 / 778 (0.64)	1.6 J	72 J	2.9
1,2-Dichlorobenzene	µg/kg dw	18 / 778 (2.3)	1.3 J	520 J	2.2
1,4-Dichlorobenzene	µg/kg dw	35 / 778 (4.5)	0.74 J	1,600 J	6.4
2,4-Dimethylphenol	µg/kg dw	1 / 773 (0.13)	290 J	290 J	290
4-Methylphenol	µg/kg dw	78 / 793 (9.8)	4.8 J	4,600 J	33
Benzoic acid	µg/kg dw	69 / 781 (8.8)	54 J	4,500	220
Benzyl alcohol	µg/kg dw	14 / 771 (1.8)	8.2 J	670	25
Hexachlorobenzene	µg/kg dw	46 / 781 (5.9)	0.4 J	95 J	1.4
N-Nitrosodiphenylamine	µg/kg dw	23 / 780 (3.0)	6.5	230	8
Pentachlorophenol	µg/kg dw	12 / 747 (1.6)	14 J	410	97
Phenol	µg/kg dw	254 / 793 (32)	10 J	2,800	70
PCBs					
Total PCBs	µg/kg dw	1,203 / 1,288 (93)	1.6 J	220,000	150
Organochlorine Pesticides					
Total DDTs	µg/kg dw	78 / 197 (40)	0.72 J	2,900 J	8.6
Total chlordane[c]	µg/kg dw	33 / 197 (17)	0.20 J	230	3.7

[a] Number of detected concentrations per number of surface sediment samples analyzed for that chemical in the baseline dataset.

[b] TBT was evaluated using both a tissue-residue approach and imposex evaluation of gastropods. Sediment data are presented here for completeness.

[c] Total chlordane includes the calculated total chlordane for Phase 2 data and chlordane as reported in a subset of historical data (King County 1999b).

COPC – chemical of potential concern

dw – dry weight

HPAH – high-molecular-weight polycyclic aromatic hydrocarbon

J – estimated concentration

LPAH – low-molecular-weight polycyclic aromatic hydrocarbon

PAH – polycyclic aromatic hydrocarbon

PCB – polychlorinated biphenyl

SVOC – semivolatile organic compound

TBT – tributyltin

A.3.1.2 Porewater exposure assessment

As discussed in Section A.2.5.1.2 and in the *Quality Assurance Project Plan: Porewater Sampling of Lower Duwamish Waterway* (Windward 2005i), two sites, GWI and Boeing Plant 2/Jorgensen Forge, were selected as porewater sampling locations (Maps A.3-1 and A.3-2).

Cis-1,2-dichloroethene was the only VOC with a maximum detected concentration in porewater greater than a NOEC, and thus was the only COPC identified in porewater for the benthic invertebrate community. Tables A.3-2 and A.3-3 present the concentrations of cis-1,2-dichloroethene detected in porewater samples collected using peepers at GWI and Boeing Plant 2/Jorgensen Forge locations, respectively. Cis-1,2-dichloroethene was detected at all locations at GWI and at six locations at Boeing Plant 2/Jorgensen Forge. Porewater from GWI peeper PE-06 had the highest concentration of cis-1,2-dichloroethene (2,900 µg/L). This peeper was adjacent to a known seep (seep S-13), where cis-1,2-dichloroethene was detected at a concentration of 5,400 µg/L in an earlier sampling event (Windward 2005i). The other porewater samples collected at GWI had detected concentrations of cis-1,2-dichloroethene ranging from 0.5 µg/L to 630 µg/L. The concentrations of cis-1,2-dichloroethene in porewater samples collected at Boeing Plant 2/Jorgensen Forge were equal to or less than 1.7 µg/L. The evaluation is based on individual data points because benthic invertebrates have limited home ranges.

Table A.3-2. Concentrations of cis-1,2-dichloroethene in porewater samples from GWI

Sample ID	cis-1,2-Dichloroethene Concentration (µg/L)
LDW-PW-G-PE-01	6.1
LDW-PW-G-PE-02	46
LDW-PW-G-PE-03	0.5
LDW-PW-G-PE-04	2.4
LDW-PW-G-PE-05	630
LDW-PW-G-PE-06	2,900
LDW-PW-G-PE-07	18
LDW-PW-G-PE-08	20
LDW-PW-G-PE-203[a]	41
LDW-PW-G-PE-204[a]	27

[a] Field replicates from LDW-PW-G-PE-08.

ID – identification

Table A.3-3. Concentrations of cis-1,2-dichloroethene in porewater samples from Boeing Plant 2/Jorgensen Forge

Sample ID	cis-1,2-Dichloroethene Concentration (µg/L)
LDW-PW-B-PE-09	0.4
LDW-PW-B-PE-10	0.2 U
LDW-PW-B-PE-201[a]	0.4
LDW-PW-B-PE-202[a]	1.0
LDW-PW-B-PE-11	1.7
LDW-PW-B-PE-12	0.9
LDW-PW-B-PE-13	0.5
LDW-PW-B-PE-14	0.2
LDW-PW-B-PE-15	0.2 U
LDW-PW-B-PE-16	0.2 U

[a] Field replicates from LDW-PW-G-PE-10.

ID – identification

U – not detected at reporting limit shown

A.3.1.3 TBT exposure assessment

TBT was identified as a COPC for the benthic invertebrate community in Phase 1. The Phase 2 RI work plan (Windward 2004e) identified two approaches to evaluate risks to the benthic invertebrate community from TBT. The first approach included a direct measurement of an effect endpoint on meso- and neogastropods, which are the marine invertebrates that are most sensitive to TBT. The potential for effects was directly evaluated through site-specific assessments of imposex in gastropods collected from the LDW over a range of TBT concentrations in sediment (see Section A.3.2.4 for details on this approach). The second approach evaluated potential effects from TBT exposure on survival, growth, and reproduction of benthic invertebrates using a critical tissue-residue approach.

Gastropod sampling events, which were conducted in 2004 and 2005, are detailed in the gastropod pilot survey (Windward 2004f) and 2005 gastropod imposex study (Windward 2005j). Gastropod sampling locations are shown on Map A.3-3. Attempts were made to collect gastropods in both intertidal and subtidal habitats in the LDW. Gastropods were not found at the intertidal locations[33] surveyed but were found in most of the subtidal locations surveyed. Gastropod abundances were higher in the downstream, more saline portions of the LDW (Windward 2006c).

[33] A wide range of intertidal habitats with a range of chemical concentrations were surveyed for gastropods. It is not known why no gastropods were found at intertidal locations.

To assess exposure of the benthic invertebrate community to TBT, co-located benthic invertebrate tissue and sediment samples were collected and analyzed[34] as part of Phase 2 from 20 locations (Table A.3-4) (10 intertidal and 10 subtidal locations throughout the LDW [Map A.3-4]). All benthic invertebrates, including polychaetes, amphipods, and mollusks collected at each of the 20 locations, were combined into one composite tissue sample for each of the 20 locations to meet the biomass requirements for chemical analyses.

The cumulative frequency distribution of TBT concentrations in the co-located sediment samples is provided in Figure A.3-1. This figure demonstrates the range of TBT sediment concentrations sampled. A significant non-linear regression relationship was observed between the TBT concentrations in benthic invertebrate tissue and co-located sediment samples (Attachment 11). The regression equation was:

$$\text{tissue} = 145.4 \times \text{sediment}^{0.1801} \quad (r^2 = 0.587) \qquad \textbf{Equation 3-1}$$

The maximum TBT concentration estimated for tissue (610 µg/kg dw) was estimated using the regression equation and the maximum TBT concentration in sediment (3,000 µg/kg dw). The estimated maximum tissue concentration was similar to the empirical maximum concentration (550 µg/kg dw) because of the distribution of the tissue sampling locations relative to the entire sediment dataset for TBT (Figure A.3-1).

Table A.3-4. Tributyltin concentrations in benthic invertebrate tissue samples and co-located sediment samples

Sample Location	Benthic Invertebrate Tissue TBT Concentration (µg/kg dw)	Sediment TBT Concentration (µg/kg dw)
Intertidal locations		
B1a	60 U	0.35 J
B2a	280	22
B3a	120	2.1
B4a	320	32
B5a-2	140	6.4
B6a	96	2.3
B7a	450	5.6
B8a	270	5.8
B9a	97	1.6 J
B10a	38 J	3.6
Subtidal locations		
B1b	550	2,300 J
B2b	500	63

[34] The benthic invertebrate tissue samples were also analyzed for a variety of metals, SVOCs, PCBs, and pesticides (Windward 2005b).

Sample Location	Benthic Invertebrate Tissue TBT Concentration (µg/kg dw)	Sediment TBT Concentration (µg/kg dw)
B3b	310	320
B4b	360	96
B5b	280	30
B6b	250	20
B7b	270	13
B8b[a]	140	1.7 J
B9b[a]	170	6.7
B10b[a]	74	2.3

[a] Originally designated subtidal in the quality assurance project plan (QAPP) for the collection effort (Windward 2004d) but designated intertidal in the baseline dataset used in the ERA and HHRA.

dw – dry weight

J – estimated concentration

TBT – tributyltin

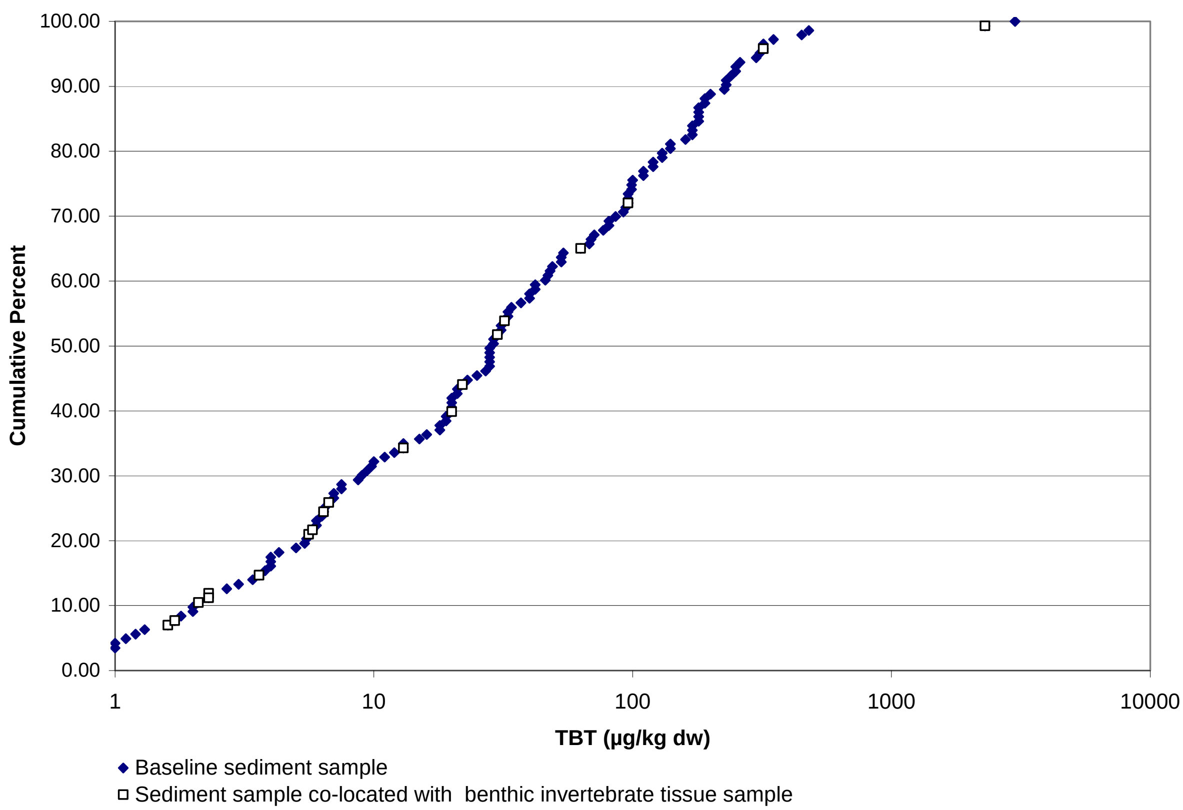

Figure A.3-1. Frequency distribution of TBT concentrations in LDW sediment samples

A.3.2 BENTHIC INVERTEBRATE COMMUNITY EFFECTS ASSESSMENT

This section presents the effects assessment for the benthic invertebrate community. The potential effects of sediment-associated COPCs on the benthic invertebrate community were evaluated through:

- Comparisons of surface sediment chemical concentrations with SQS and CSL chemical criteria from the Washington State SMS (Section A.3.2.1)
- Site-specific sediment toxicity tests (Section A.3.2.2.)

Possible effects of the single COPC identified in porewater were evaluated using NOECs and LOECs selected from the literature (Section A.3.2.3).

Effects of exposure to TBT were evaluated using:

- Direct assessment of imposex on LDW-collected gastropods (Section A.3.2.4.1)
- Tissue-based toxicological data in the literature (for benthic invertebrates) (Section A.3.2.4.2)

Information on effects of COPCs presented in this section is combined with the exposure data in the risk characterization (Section A.6.1), and the uncertainties are discussed in the uncertainty analysis (Section A.6.1).

A.3.2.1 Sediment effects assessment

Effects on the benthic invertebrate community were assessed by comparing the COPC concentrations in LDW surface sediment to the SQS CSL chemical criteria derived in 1991 when Ecology adopted the SMS (WAC 173-204). These criteria are based on AETs (defined in Section A.2.5.1.1) developed for 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. The methods used to calculate the AETs are described by Barrick et al. (1988) and Gries and Waldow (1996).

AETs were empirically derived using data from field-collected sediment samples that contained diverse chemical mixtures analyzed simultaneously for chemistry and toxicity. The data used to derive the 1988 AETs were collected from various locations in Puget Sound between March 1982 and September 1986. AETs for four endpoints[35] (i.e., amphipod mortality, abnormal development of oyster larvae, benthic invertebrate community abundance, and Microtox® bioluminescence) were developed for 47 chemicals. In general, the lowest AET for each chemical was identified as the SQS; the second lowest AET was identified as the CSL. The SQS corresponds to a sediment quality that will result in no adverse effects to biological resources; the CSL

[35] The specific tests associated with each of these endpoints are described in greater detail in the SMS rule (WAC 173-204).

corresponds to a sediment quality that will result in minor adverse effects (WAC 173-204). Table A.3-5 presents the biological effect endpoints that provide the basis for the SQS and CSL chemical criteria for COPCs.

Table A.3-5. Biological effect endpoints used to determine the SQS and CSL for COPCs

Chemical	SQS	CSL	Unit	Biological Endpoint Used to Establish SQS	Biological Endpoint Used to Establish CSL
Metals					
Arsenic	57	93	mg/kg dw	community abundance	amphipod mortality
Cadmium	5.1	6.7	mg/kg dw	community abundance	amphipod mortality
Chromium	260	270	mg/kg dw	community abundance	amphipod mortality
Copper	390	390	mg/kg dw	oyster abnormality	Microtox®
Lead	450	530	mg/kg dw	community abundance	Microtox®
Mercury	0.41	0.59	mg/kg dw	Microtox®	oyster abnormality
Silver	6.1	6.1	mg/kg dw	amphipod mortality	amphipod mortality
Zinc	410	960	mg/kg dw	community abundance	amphipod mortality
PAHs					
2-Methylnaphthalene	38	64	mg/kg OC	na	community abundance
Acenaphthene	16	57	mg/kg OC	oyster abnormality	community abundance
Anthracene	220	1,200	mg/kg OC	community abundance	amphipod mortality
Benzo(a)anthracene	110	270	mg/kg OC	oyster abnormality	amphipod mortality
Total benzofluoranthenes	230	450	mg/kg OC	oyster abnormality	amphipod mortality
Benzo(a)pyrene	99	210	mg/kg OC	oyster abnormality	amphipod mortality
Benzo(g,h,i)perylene	31	78	mg/kg OC	oyster abnormality	amphipod mortality
Chrysene	110	460	mg/kg OC	oyster abnormality	amphipod mortality
Dibenzo (a,h)anthracene	12	33	mg/kg OC	na	Microtox®
Fluoranthene	160	1,200	mg/kg OC	oyster abnormality	community abundance
Fluorene	23	79	mg/kg OC	oyster abnormality	community abundance
Indeno (1,2,3,-c,d)pyrene	34[a]	88	mg/kg OC	oyster abnormality	amphipod mortality
Naphthalene	99	170	mg/kg OC	oyster abnormality	community abundance
Phenanthrene	100[b]	480	mg/kg OC	oyster abnormality	community abundance
Pyrene	1,000	1,400	mg/kg OC	amphipod mortality	community abundance
HPAH	960	5,300	mg/kg OC	oyster abnormality	amphipod mortality
LPAH	370	780	mg/kg OC	oyster abnormality	community abundance
Phthalates					
Bis(2-ethylhexyl) phthalate	47	78	mg/kg OC	Microtox®	amphipod mortality
Butyl benzyl phthalate	4.9	64	mg/kg OC	Microtox®	community abundance
Dimethyl phthalate	53	53	mg/kg OC	amphipod mortality	community abundance
Other SVOCs					
1,2,4-Trichlorobenzene	0.81	1.8	mg/kg OC	Microtox®	amphipod mortality
1,2-Dichlorobenzene	2.3	2.3	mg/kg OC	oyster abnormality	community abundance and Microtox®
1,4-Dichlorobenzene	3.1	9	mg/kg OC	oyster abnormality	amphipod mortality
2,4-Dimethylphenol	29	29	µg/kg dw	oyster abnormality	Microtox®
4-Methylphenol	670	670	µg/kg dw	oyster abnormality	Microtox®

Chemical	SQS	CSL	Unit	Biological Endpoint Used to Establish SQS	Biological Endpoint Used to Establish CSL
Benzoic acid	650	650	µg/kg dw	oyster abnormality	community abundance, Microtox®
Benzyl alcohol	57	73	µg/kg dw	Microtox®	oyster abnormality
Dibenzofuran	15	58	mg/kg OC	oyster abnormality	community abundance
Hexachlorobenzene	0.38	2.3	mg/kg OC	community abundance	Microtox®
n-Nitrosodiphenylamine	11	11	mg/kg OC	community abundance	community abundance
Pentachlorophenol	360	690	µg/kg dw	amphipod mortality	community abundance
Phenol	420	1,200	µg/kg dw	oyster abnormality	amphipod mortality, community abundance, and Microtox®
PCBs					
Total PCBs	12	65	mg/kg OC	Microtox®	community abundance

Source: Washington State Sediment Management Standards (WAC 173-204); Barrick et al. (1988).

[a] The SQS for indeno(1,2,3,-c,d)pyrene is 34 mg/kg OC; the lowest AET, based on oyster abnormality, is 33 mg/kg OC.

[b] The SQS for phenanthrene is 100 mg/kg OC; the lowest AET, based on oyster abnormality, is 120 mg/kg OC.

CSL – cleanup screening level
dw – dry weight
HPAH – high-molecular-weight polycyclic aromatic hydrocarbon
LPAH – low-molecular-weight polycyclic aromatic hydrocarbon
na – not available
OC – organic carbon
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
SQS – sediment quality standard
SVOC – semivolatile organic compound

Nickel, total DDTs, and total chlordane were identified as additional COPCs in the screening process. 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,[36] TRVs were selected from the scientific literature. A search was conducted for relevant sediment toxicity studies for DDT and its metabolites and for chlordane using two databases, ECOTOX and BIOSIS. The databases were searched for relevant toxicity studies involving invertebrate species with growth, mortality, or reproductive endpoints. No toxicity studies were available in the scientific literature for chlordane. However, sediment quality guidelines (SQGs) and AETs other than those used to set SMS were available and were used to derive TRVs for chlordane (Table A.3-6). The lowest SQG (4.79 µg/kg dw) was selected as the LOAEL TRV. The highest NOAEL (2.8 µg/kg dw) below the LOAEL was selected as the NOAEL TRV. For DDTs, the lowest LOAEL (1,063 µg/kg dw) based on an individual laboratory toxicity study was selected as the LOAEL TRV. The NOAEL (567 µg/kg dw) for the same endpoint (survival) from the same study was selected as the NOAEL TRV. Table A.3-7 presents the basis for the screening level (SL) and maximum level (ML) guidelines for nickel and the selected TRVs for total DDTs and total chlordane. All reviewed sediment toxicity studies are presented in Attachment 6.

[36] The DMMP SLs for pesticides were selected to be approximately equal to the limit of quantification (i.e., 5 times the instrument detection limit). Insufficient data were available to establish the MLs for pesticides (PSDDA 1988b).

Table A.3-6. Available toxicity studies for selection of chlordane and total DDTs TRVs

Analyte	NOAEL (µg/kg dw)	LOAEL (µg/kg dw)	Biological Endpoint Determining NOAEL	Biological Endpoint Determining LOAEL	Source
Chlordane	2.8	na	amphipod AET	na	Gries and Waldow (1996)
Chlordane	2.26	4.79	All components of the aquatic ecosystem (e.g., bacteria, algae, macrophytes, invertebrates, fish) were considered, if data were available.	All components of the aquatic ecosystem (e.g., bacteria, algae, macrophytes, invertebrates, fish) were considered, if data were available.	Canadian environmental quality guidelines (CCME 2002)
p,p-DDE	9	na	benthic AET	na	Gries and Waldow (1996)
p,p-DDD	16	na	benthic AET	na	Gries and Waldow (1996)
p,p-DDT	12	na	echinoderm AET	na	Gries and Waldow (1996)
Total DDTs	24	na	amphipod AET	na	Gries and Waldow (1996)
Total DDT	8510	na	*Neanthes arenaceodentata*; full life-cycle toxicity test (survival, growth, fecundity, reproduction)	na	Chapman (1996)
DDTs	567	1,063	*Hyalella azteca* survival	*Hyalella azteca* survival	Lotufo et al. (2001b)
DDTs	na	1,985	na	*Leptocheirus plumulosus* survival	Lotufo et al. (2001a)
DDTs	na	2,910	na	*Diporeia* spp. survival	Lotufo et al. (2001b)
DDTs	na	3,510	na	*Hyalella azteca* survival	Lotufo et al. (2001b)
DDD	1,200	4,000[a]	*Hyalella azteca* survival	*Hyalella azteca* survival	Ingersoll et al. (2005)
DDD	1,200	4,000[a]	*Hyalella azteca* reproduction	*Hyalella azteca* reproduction	Ingersoll et al. (2005)
DDT	na	6,180	na	*Rhepoxynius abronius* survival	Murdoch et al. (1997)
Total DDT	na	7,500	na	significant reduced feeding rate in marine polychaete (*Heteromastus filiformis*)	Mulsow and Landrum (1995)
Total DDT	na	11,000	na	*Hyalella azteca* LC50	Nebeker et al. (1989)
DDTs	na	308,000	na	*Neanthes arenaceodentata* growth	Lotufo et al. (2000)

[a] The Ingersoll study also evaluated a growth endpoint. There was no dose response for the growth endpoint with the lowest effect value at 30 µg/kg dw.

AET – apparent effects threshold

CCME – Canadian Council of Ministers of the Environment

dw – dry weight

na – not available

LOAEL – lowest-observed-adverse-effect level

NOAEL – no-observed-adverse-effect level

TRV – toxicity reference value

Table A.3-7. Biological effect endpoints for DMMP guidelines and selected TRVs

Chemical	Unit	SL or NOAEL	ML or LOAEL	Biological Endpoint Determining SL or NOAEL	Biological Endpoint Determining ML or LOAEL
Nickel	mg/kg dw	140[a]	370[a]	amphipod mortality and community abundance	nr
Total DDTs	µg/kg dw	567[b]	1,063[b]	amphipod mortality	amphipod mortality
Total chlordane	µg/kg dw	2.8[c]	4.79[d]	amphipod mortality	CCME (2002)

[a] Source: USACE et al. (2000); Barrick et al. (1988).

[b] Literature-based TRV – (Lotufo et al. 2001b).

[c] Literature-based TRV – PSDDA AET evaluation 1994 (Gries and Waldow 1996).

[d] Literature-based TRV – probable effect levels for chlordane (CCME 2002).

CCME – Canadian Council of Ministers of the Environment
DMMP – Dredged Material Management Program
dw – dry weight
LC50 – concentration that causes the death of 50% of a group of test animals
LOAEL – lowest-observed-adverse-effect level
ML – maximum level (DMMP)
nr – not reported
NOAEL – no-observed-adverse-effect level
SL – screening level (DMMP)

A.3.2.2 Site-specific toxicity tests assessment

This section describes the results of site-specific toxicity tests conducted on LDW sediment samples to assess the potential effects of sediment-associated chemicals on benthic invertebrates. In general, toxicity testing was performed on sediment samples with at least one detected chemical exceeding SQS.[37] The SMS provide both chemical and biological effects criteria. Because AETs, which form the basis for the chemical criteria, are based on sediment samples with a mixture of chemicals from a large number of locations, toxicity tests either confirm or overrule the SMS designation based on sediment chemistry.

To generate more specific information about the nature and severity of effects on benthic invertebrates exposed to sediments with at least one chemical concentration exceeding the SQS, three toxicity tests were conducted with surface sediments (0 to 10 cm) collected at 48 locations (Map A.3-5) (Windward 2005d, e). The toxicity tests included:

- Acute 10-day amphipod (*Eohaustorius estuarius*) mortality test
- Acute 48-hr bivalve larvae (*Mytilus galloprovincialis*) normal survival test
- Chronic 20-day juvenile polychaete (*Neanthes arenaceodentata*) survival and growth test

[37] Two samples were also tested that did not have any SQS exceedances because they were near sources and were requested by Ecology. In addition, several samples were not tested if they were heavily contaminated; these samples were assumed to be toxic.

The results from the three sediment toxicity tests were evaluated using the SMS rules for marine toxicity tests (Ecology 2003b). The performance criteria and biological effects criteria (SQS and CSL of the SMS) are summarized in Table A.3-8.

Table A.3-8. SMS biological effects criteria for marine sediment toxicity tests

Toxicity Test	Biological Effects Criteria	
	SQS	CSL
Amphipod	mean mortality > 25% on an absolute basis, and statistically different from the reference sediment ($p \le 0.05$)	mean mortality greater than the value in the reference sediment plus 30%, and statistically different from the reference sediment ($p \le 0.05$)
Polychaete[a]	mean individual growth rate < 70% of that of the reference sediment and statistically different ($p \le 0.05$)	mean individual growth rate < 50% of that of the reference sediment and statistically different ($p \le 0.05$)
Bivalve larvae	mean normal survivorship < 85% of that of the reference sediment and statistically different ($p \le 0.10$)	mean normal survivorship <70% of that of the reference sediment and statistically different ($p \le 0.10$)

[a] The mortality endpoint for the polychaete toxicity test is not used for determination of SMS compliance.

CSL – cleanup screening level

SMS – Washington State Sediment Management Standards

SQS – sediment quality standard

For the amphipod mortality endpoint, 11 of the 48 samples failed the biological effects criteria of the SMS at the CSL level, and 5 of the 48 samples failed the biological effects criteria at the SQS level (Table A.3-9). For the polychaete growth endpoint, 8 of the 48 samples failed the biological effects criteria at the SQS level; no samples failed the polychaete criteria at the CSL level. For the bivalve survival/development endpoint, 8 of the 48 samples failed the biological effects criteria at the CSL level, and 12 of the 48 samples failed the biological effects criteria at the SQS level.

An exceedance of the SQS biological effects criteria in any two toxicity tests at one location is considered a CSL exceedance at that location (WAC 173-204-420(3)). Based on this guideline, of the 48 samples tested, 18 sediment samples did not exceed the biological effects criteria, 11 sediment samples exceeded the SQS biological effects criteria, and 19 sediment samples exceeded the CSL biological effects criteria (Table A.3-9; Map A.3-5).

Table A.3-9. Results of Phase 2 site-specific toxicity testing of surface sediment samples from the LDW

Sample ID	Amphipod Toxicity Test		Polychaete Toxicity Test			Bivalve Larval Toxicity Test		Overall SMS Exceedance
	Percent Mean Mortality ± SD	SMS Exceedance[a, b]	Mean Mortality ± SD	Mean Individual Growth Rate (mg/day) ± SD	SMS Exceedance[a, c, d]	Percent Mean Normal Survivorship ± SD	SMS Exceedance[a, e]	
LDW-SS2-010	39.0 ± 16.0	CSL	0.0 ± 0.0	0.76 ± 0.20	no exceedances	31.8 ± 15.1	CSL	CSL
LDW-SS6-010	47.0 ± 21.7	CSL	4.0 ± 8.9	0.81 ± 0.04	no exceedances	23.6 ± 16.3	CSL	CSL
LDW-SS15-010	28.0 ± 8.4	SQS	0.0 ± 0.0	0.73 ± 0.10	no exceedances	75.4 ± 8.7	no exceedances	SQS
LDW-SS16-010	16.0 ± 10.2	no exceedances	4.0 ± 8.9	0.96 ± 0.13	no exceedances	64.3 ± 3.2	SQS	SQS
LDW-SS17-010	35.0 ± 20.9	SQS	0.0 ± 0.0	0.92 ± 0.22	no exceedances	62.9 ± 7.6	no exceedances	SQS
LDW-SS21-010	37.0 ± 23.6	CSL	0.0 ± 0.0	0.99 ± 0.10	no exceedances	72.9 ± 3.6	no exceedances	CSL
LDW-SS22-010	32.0 ± 10.4	SQS	4.0 ± 8.9	0.70 ± 0.13	SQS	51.9 ± 1.7[f]	SQS	CSL
LDW-SS24-010	7.0 ± 4.5	no exceedances	0.0 ± 0.0	0.77 ± 0.17	SQS	18.3 ± 3.3	CSL	CSL
LDW-SS26-010	23.0 ± 11.0	no exceedances	0.0 ± 0.0	0.86 ± 0.17	no exceedances	76.7 ± 7.4	no exceedances	no exceedances
LDW-SS29-010	12.0 ± 7.6	no exceedances	0.0 ± 0.0	0.90 ± 0.08	no exceedances	64.7 ± 6.8	no exceedances	no exceedances
LDW-SS31-010	43.0 ± 5.7	CSL	4.0 ± 8.9	0.81 ± 0.10	no exceedances	62.9 ± 7.6	SQS	CSL
LDW-SS32-010	34.0 ± 11.9	SQS	0.0 ± 0.0	0.69 ± 0.10	no exceedances	78.9 ± 15.7[f]	no exceedances	SQS
LDW-SS37-010	45.0 ± 14.6	CSL	0.0 ± 0.0	0.78 ± 0.05	no exceedances	65.8 ± 19.2[g]	SQS	CSL
LDW-SS39-010	29.0 ± 11.4	SQS	0.0 ± 0.0	0.76 ± 0.12	SQS	83.4 ± 10.1	no exceedances	CSL
LDW-SS40-010	36.0 ± 12.9	CSL	0.0 ± 0.0	0.78 ± 0.14	no exceedances	79.7 ± 4.9	no exceedances	CSL
LDW-SS49-010	49.0 ± 19.5	CSL	0.0 ± 0.0	0.79 ± 0.22	no exceedances	55.1 ± 17.4	SQS	CSL
LDW-SS50-010	39.0 ± 10.8	CSL	0.0 ± 0.0	0.73 ± 0.11	no exceedances	70.2 ± 10.2	no exceedances	CSL
LDW-SS56-010	6.0 ± 4.2	no exceedances	0.0 ± 0.0	0.85 ± 0.14	no exceedances	67.6 ± 7.2	SQS	SQS
LDW-SS57-010	13.0 ± 12.5	no exceedances	0.0 ± 0.0	0.78 ± 0.13	no exceedances	55.3 ± 15.1	CSL	CSL
LDW-SS58-010	5.0 ± 5.0	no exceedances	0.0 ± 0.0	0.69 ± 0.07	SQS	61.0 ± 4.0	SQS	CSL
LDW-SS60-010	7.0 ± 5.7	no exceedances	4.0 ± 8.9	0.77 ± 0.17	no exceedances	84.7 ± 6.0	no exceedances	no exceedances
LDW-SS63-010	5.0 ± 6.1	no exceedances	0.0 ± 0.0	0.68 ± 0.19	no exceedances	80.0 ± 1.6	no exceedances	no exceedances

Sample ID	Amphipod Toxicity Test		Polychaete Toxicity Test			Bivalve Larval Toxicity Test		Overall SMS Exceedance
	Percent Mean Mortality ± SD	SMS Exceedance[a, b]	Mean Mortality ± SD	Mean Individual Growth Rate (mg/day) ± SD	SMS Exceedance[a, c, d]	Percent Mean Normal Survivorship ± SD	SMS Exceedance[a, e]	
LDW-SS68-010	12.0 ± 9.1	no exceedances	0.0 ± 0.0	0.85 ± 0.10	no exceedances	71.6 ± 12.5	no exceedances	no exceedances
LDW-SS69b-010	37.0 ± 15.7	CSL	0.0 ± 0.0	0.85 ± 0.29	no exceedances	59.0 ± 13.6	SQS	CSL
LDW-SS70-010	15.0 ± 7.1	no exceedances	4.0 ± 8.9	0.78 ± 0.12	no exceedances	60.7 ± 12.0	SQS	SQS
LDW-SS71-010	5.0 ± 6.1	no exceedances	0.0 ± 0.0	0.92 ± 0.16	no exceedances	61.8 ± 8.7	no exceedances	no exceedances
LDW-SS73-010	12.0 ± 9.1	no exceedances	0.0 ± 0.0	0.86 ± 0.11	no exceedances	56.8 ± 13.3	SQS	SQS
LDW-SS75-010	8.0 ± 7.6	no exceedances	4.0 ± 8.9	0.69 ± 0.16	no exceedances	76.5 ± 7.5	no exceedances	no exceedances
LDW-SS77-010	16.0 ± 9.6	no exceedances	4.0 ± 8.9	0.95 ± 0.12	no exceedances	10.1 ± 4.0	CSL	CSL
LDW-SS85-010	1.0 ± 2.2	no exceedances	0.0 ± 0.0	0.87 ± 0.18	no exceedances	86.8 ± 5.3	no exceedances	no exceedances
LDW-SS88-010	48.0 ± 25.9	CSL	0.0 ± 0.0	0.68 ± 0.14	no exceedances	11.9 ± 5.3	CSL	CSL
LDW-SS89-010	5.0 ± 3.5	no exceedances	0.0 ± 0.0	0.84 ± 0.26	no exceedances	86.9 ± 5.1[g]	no exceedances	no exceedances
LDW-SS92-010	1.0 ± 2.2	no exceedances	0.0 ± 0.0	0.79 ± 0.17	no exceedances	78.3 ± 14.3	no exceedances	no exceedances
LDW-SS106-010	6.0 ± 4.2	no exceedances	0.0 ± 0.0	0.91 ± 0.11	no exceedances	61.4 ± 8.8	no exceedances	no exceedances
LDW-SS112-010	4.0 ± 4.2	no exceedances	0.0 ± 0.0	0.82 ± 0.06	no exceedances	65.4 ± 9.6	no exceedances	no exceedances
LDW-SS114-010	85.0 ± 7.1	CSL	4.0 ± 8.9	0.77 ± 0.19	no exceedances	56.6 ± 8.5	SQS	CSL
LDW-SS115-010	9.0 ± 4.2	no exceedances	0.0 ± 0.0	0.76 ± 0.21	no exceedances	77.6 ± 11.0	no exceedances	no exceedances
LDW-SS119-010	3.0 ± 2.7	no exceedances	0.0 ± 0.0	0.79 ± 0.06	no exceedances	68.8 ± 7.6	no exceedances	no exceedances
LDW-SS120-010	3.0 ± 2.7	no exceedances	0.0 ± 0.0	0.71 ± 0.09	no exceedances	56.3 ± 11.2	SQS	SQS
LDW-SS121-010	4.0 ± 4.2	no exceedances	0.0 ± 0.0	0.90 ± 0.12	no exceedances	76.4 ± 8.0	no exceedances	no exceedances
LDW-SS122-010	7.0 ± 4.5	no exceedances	0.0 ± 0.0	0.83 ± 0.14	no exceedances	68.5 ± 14.9	no exceedances	no exceedances
LDW-SS143-010	6.0 ± 5.5	no exceedances	0.0 ± 0.0	0.75 ± 0.07	no exceedances	72.8 ± 3.2	no exceedances	no exceedances
LDW-SS144-010	1.0 ± 2.2	no exceedances	0.0 ± 0.0	0.72 ± 0.11	SQS	66.4 ± 12.1	no exceedances	SQS[h]
LDW-SS148-010	6.0 ± 6.5	no exceedances	0.0 ± 0.0	0.78 ± 0.08	SQS	29.9 ± 6.6	CSL	CSL
LDW-SS157-010	8.0 ± 7.6	no exceedances	4.0 ± 8.9	0.78 ± 0.14	SQS	71.6 ± 8.5	no exceedances	SQS[h]
LDW-SS158-010	12.0 ± 4.5	no exceedances	0.0 ± 0.0	0.81 ± 0.10	no exceedances	67.5 ± 6.5	no exceedances	no exceedances

Sample ID	Amphipod Toxicity Test		Polychaete Toxicity Test			Bivalve Larval Toxicity Test		Overall SMS Exceedance
	Percent Mean Mortality ± SD	SMS Exceedance[a, b]	Mean Mortality ± SD	Mean Individual Growth Rate (mg/day) ± SD	SMS Exceedance[a, c, d]	Percent Mean Normal Survivorship ± SD	SMS Exceedance[a, e]	
LDW-SSB2b-010	25.0 ± 12.2	no exceedances	0.0 ± 0.0	1.02 ± 0.10	no exceedances	42.1 ± 20.0	CSL	CSL
LDW-SSB6a-010	2.0 ± 4.5	no exceedances	0.0 ± 0.0	0.82 ± 0.14	SQS	60.1 ± 13.3	no exceedances	SQS[h]

[a] Statistical analyses in SedQual Release 5 included Wilk-Shapiro test for normality and Levene's test for equality of variances followed by the appropriate statistical test for significance (i.e., Student's t-test, approximate t-test, or Mann-Whitney).

[b] SQS – mean mortality > 25% on an absolute basis and statistically different from the reference sediment ($p \leq 0.05$); CSL – mean mortality greater than the value in the reference sediment plus 30% and statistically different from the reference sediment ($p \leq 0.05$). Reference sediment results are presented in the Round 1 and Round 2 surface sediment data reports (Windward 2005d, e).

[c] SQS – mean individual growth rate <70% of that of the reference sediment and statistically different ($p \leq 0.05$).

[d] No exceedance was reported for the polychaete growth endpoint for some of the sediment samples because of high variability in the reference and/or test samples.

[e] SQS – mean normal survivorship < 85% of that of the reference sediment and statistically different ($p \leq 0.10$); CSL – mean normal survivorship < 70% of that of the reference sediment and statistically different ($p \leq 0.10$).

[f] One of the five replicates from each of these tests was double-inoculated, so those replicates were not used in calculating mean normal survivorship and mean effective mortality for those test sediments.

[g] One of the five replicates from each of these tests was not inoculated at test initiation, so those replicates were not used in calculating mean normal survivorship and mean effective mortality for those test sediments.

[h] Exceeded the SQS criterion based on reduction in polychaete growth alone.

CSL – cleanup screening level

ID – identification

SMS – Washington State Sediment Management Standard

SD – standard deviation

SQS – sediment quality standard

Twelve other site-specific toxicity studies have been conducted in the LDW since 1990[38] (Table A.3-10). However, only 1 of these 12 studies (with a total of 7 samples, as shown in bold in Table A.3-10) was conducted on surface sediments (0 to 15 cm) that remained in place and are thus relevant to this study. One other study, *Sediment Quality in Puget Sound* (NOAA and Ecology 2000), was a monitoring program intended to characterize only the most recently deposited surface sediment (0 to 2 cm). The remainder of the studies were dredged material characterization studies that tested sediments from the 0- to 4-ft (or deeper) horizons, and those sediments were subsequently dredged and removed from the LDW.

Collection locations for samples analyzed for sediment toxicity in the one relevant historical study (King County 2000a) are presented in Map A.3-5. The seven locations sampled by King County were selected to help determine the removal boundary for the Duwamish/Diagonal cleanup. Part of the area for the Duwamish/Diagonal cleanup study (King County 2000a) was dredged and capped in 2004. Three of the seven locations sampled were within the removal area and were dredged. However, because the baseline dataset includes pre-removal conditions, a summary of all results from that study is included.

The seven sediment samples collected for the Duwamish/Diagonal cleanup study (King County 2000a) were tested for toxicity using three standard SMS confirmatory tests (i.e., amphipod [*Rhepoxynius abronius*] mortality, echinoderm embryo effective mortality,[39] and polychaete growth). One of seven samples (L9443-7) failed the biological effects criteria of the SMS at the SQS level for both the echinoderm embryo effective mortality and the polychaete growth endpoints (Table A.3-11). The results of the polychaete and echinoderm embryo tests for sample L9443-7 were similar to the results from several other samples tested concurrently. The control sample to which the results from sample L9443-7 were compared had a higher polychaete growth rate (0.77 vs. 0.60 mg/day) and lower effective echinoderm embryo mortality (15 vs. 29%) compared to the reference sample to which all other test sediments were compared.[40]

Thus, based on the available data, one of the seven sediment samples analyzed by King County (2000a) exhibited toxicity in exceedance of the state's biological effects criteria.

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[38] At the beginning of the RI process, EPA and Ecology agreed that data collected within the past 10 years would be suitable to characterize the LDW; therefore, data collected in 1990 or later were acceptable. While some sampling events are now older than 10 years, they are included in Table A.3-9 to remain consistent with the list of suitable data sets developed at the beginning of the process.

[39] Combined mortality and abnormal development.

[40] This sample was not compared to the other reference samples because the grain size was not comparable. The percent fines for sample L9443-7 (7.9%) was much lower than in the remaining samples (49.5% to 91.3%) and in-batch reference sediment (54.5%). Therefore, the SMS comparison was performed on the West Beach control sediment, which had < 10 % fines.

Table A.3-10. Sediment toxicity datasets meeting project data quality objectives

Study	Year Conducted	Number of Samples[a]	Source
Sediment Quality in Puget Sound	1998	3	Ecology (2000)
Duwamish/Diagonal CSO/SD Site Assessment Report – Draft[b]	1996	7	King County (2000)
Sediment Sampling and Analysis, James Hardie Gypsum Inc., Duwamish Waterway, Seattle, Washington	1998 – 1999	7	Spearman (1999)
Dredge Material Characterization, Hurlen Construction Company and Boyer Alaska Barge Lines Berthing Areas, Duwamish Waterway, Seattle, Washington	1998	4	Hart Crowser (1998)
Proposed Dredging of Slip No. 4, Duwamish River, Seattle, Washington	1995	4	PTI (1996)
PSDDA Chemical Characterization of Duwamish Waterway and Upper Turning Basin. FY97 Operations and Maintenance Dredging, Seattle, Washington	1996	3	Striplin Environmental (1996)
Lone Star Northwest and James Hardie Gypsum Kaiser Dock Upgrade, Duwamish Waterway PSDDA Sampling and Analysis Results	1995	4	Hartman Associates (1995)
Lone Star Northwest West Terminal, Duwamish River PSDDA Sampling and Analysis Results	1992	1	Hartman Associates (1992)
Sediment Sampling and Analysis, Brown Morton Properties, Duwamish Waterway, Seattle, Washington	1991	1	Spearman (1991a)
Sediment Sampling and Analysis, South Park Marina, Duwamish Waterway, Seattle, Washington	1991	2	Spearman (1991b)
PSDDA Bioassays for Duwamish Channel Sediments	1991	14	SAIC (1992)
Duwamish Channel and Settling Basin Sediment Bioassays	1990	4	PTI (1990)

a With the exception of the *Sediment Quality in Puget Sound* report and the *Duwamish/Diagonal CSO/SD Site Assessment Report, s*ubsurface sediment samples from these studies were collected for dredged material characterization studies. The tested sediments have been removed.

b Surface sediment was characterized in this study; all other studies were dredged material characterizations and are not discussed further.

CSO/SD – combined sewer overflow/storm drain

PSDDA – Puget Sound Dredged Disposal Analysis

Table A.3-11. Summary of site-specific sediment toxicity test results for surface sediment samples collected by King County at Duwamish/Diagonal CSO/SD site

Sampling Location[a]	Sample ID	Echinoderm Embryo Effective Mortality (%)	SQS or CSL Exceedance[b]	Polychaete Growth Rate (mg/day)	SQS or CSL Exceedance[c]	Amphipod Mortality (%)	SQS or CSL Exceedance[d]	Overall SMS Exceedance
West Beach	control	15	na	0.77	na	1.0	na	na
Carr Inlet	reference	29	na	0.60	na	8.0	na	na
DUD200	L9443-1	32	no exceedances	0.60	no exceedances	13	no exceedances	no exceedances
DUD201	L9443-2	35	no exceedances	0.55	no exceedances	21	no exceedances	no exceedances
DUD202	L9443-3	35	no exceedances	0.62	no exceedances	18	no exceedances	no exceedances
DUD203	L9443-4	33	no exceedances	0.59	no exceedances	22	no exceedances	no exceedances
DUD204	L9443-5	17	no exceedances	0.51	no exceedances	26	no exceedances	no exceedances
DUD205	L9443-6	16	no exceedances	0.54	no exceedances	19	no exceedances	no exceedances
DUD206	L9443-7[e]	34	SQS	0.52	SQS	4.0	no exceedances	CSL

Source: King County (2000a)

[a] Sampling locations DUD200 through DUD206 are shown on Map A.3-5.

[b] SQS – mean normal survivorship < 85% of that of the reference sediment and statistically different ($p \leq 0.10$); CSL – mean normal survivorship < 70% of that of the reference sediment and statistically different ($p \leq 0.10$).

[c] SQS – mean individual growth rate <70% of that of the reference sediment and statistically different ($p \leq 0.05$).

[d] SQS – mean mortality > 25% on an absolute basis and statistically different from the reference sediment ($p \leq 0.05$); CSL – mean mortality greater than the value in the reference sediment plus 30% and statistically different from the reference sediment ($p \leq 0.05$).

[e] Test results for this sample were statistically compared to control sample results rather than the reference sample results because reference sample was not a suitable grain size match.

CSL – cleanup screening level

CSO/SD – combined sewer overflow/storm drain

ID – identification

na – not applicable

SMS – Washington State Sediment Management Standards

SQS – sediment quality standard

A.3.2.3 Porewater effects assessment

As discussed in Section A.2.5.1.2, a literature search was conducted for relevant aquatic toxicity studies for VOCs using two databases, ECOTOX and BIOSIS. The databases were searched for relevant toxicity studies involving invertebrate species with growth, mortality (including immobilization), or reproductive (including developmental) endpoints. Studies with invertebrates were preferred because the purpose of the porewater study was to evaluate risks to the benthic invertebrate community. However, if there were no invertebrate data, then fish studies with growth, survival, or reproductive endpoints were included in the search. Toxicity data for both freshwater and marine species were included because of the wide salinity range in the LDW and the paucity of toxicity data. Guidelines for the selection of NOECs and LOECs were:

- The study identified with the lowest effect level (preferably a LOEC) for each VOC was selected if the study was acceptable.
- The study with the lowest NOEC for each VOC was selected provided that more than one NOEC was not available for a given test species and endpoint. When there were multiple NOECs for the same test species/endpoint for a given VOC, the study with the highest NOEC for that test species/endpoint was selected if the study was acceptable.
- The study must include negative control tests. For no-effect results, the study must have used an exposure period of no less than 48 hrs for daphnids and no less than 96 hrs for fish. In addition, to represent conditions relevant to those found in the LDW, the salinity must be no greater than 35 parts per thousand in tests using *Artemia salina* (brine shrimp).

Two studies evaluated the toxicity of 1,2-dichloroethene. The lowest toxicity value for 1,2-dichloroethene was derived from a study testing toxicity to *Artemia salina* (Sanchez-Fortun et al. 1997). This study was reviewed in detail and found to be acceptable. In this study, *Artemia* shrimp larvae were added to synthetic seawater at a salinity of 35 parts per thousand (ppt) in plastic petri dishes containing the appropriate concentration of 1,2-dichloroethene. Controls containing untreated synthetic seawater were also tested. The petri dishes were incubated at 25°C. Larvae were evaluated for mortality after 72 hours. The reported LC50 was 6,785 µg/L. The only other toxicity data available for this chemical is an LC50 of 140,000 µg/L from a bluegill study. Therefore, the lower LC50 for *Artemia* (6,785 µg/L) was selected as the TRV for 1,2-dichloroethene. Because no NOAELs were available for this chemical, the LC50-based LOAEL was divided by 50 to derive a NOAEL TRV of 136 µg/L.

A.3.2.4 TBT effects assessment

Potential adverse effects to benthic invertebrates from TBT exposure were evaluated using a direct measure of imposex in LDW-collected gastropods (the benthic invertebrate most sensitive to TBT based on the imposex endpoint) and a critical tissue-residue approach for benthic invertebrates. The imposex assessment is presented in Section A.3.2.4.1 and the critical tissue-residue assessment is presented in Section A.3.2.4.2.

A.3.2.4.1 Imposex assessment

Of the benthic invertebrates that have been identified in the LDW, meso- and neogastropods[41] have been reported to be particularly sensitive to toxic effects of TBT (Meador et al. 2002). Specifically, at sufficiently high tissue concentrations, TBT is known to cause the development of male sexual organs in females in some meso- and neogastropod species. This condition, known as imposex (Gibbs et al. 1988), may interfere with gastropod reproduction and potentially results in population-level effects (Meador et al. 2002). Therefore, a site-specific evaluation of imposex in gastropods was conducted in the LDW to directly assess the toxic endpoint. This direct approach is preferable because it reduces uncertainty in the risk analysis. Imposex evaluation was performed on as many different meso-and neogastropods collected in the LDW as possible, including the three most abundant neogastropod species, *Nassarius mendicus*, *Astyris gausapata*, and *Olivella baetica*.

Mature female gastropods were examined for imposex using the methods reported in Oehlmann et al. (1991) and Spence et al. (1990). Oehlmann et al. (1991) determined the imposex stage based on the presence of male reproductive organs in females, including vas deferens[42] and penis. Table A.3-12 presents the imposex stage criteria for this evaluation. Complete sterilization is associated with stages 5 and 6. Stage 4 is generally referred to as transitional, and stages 1 through 3 are described as early stages.

Table A.3-12. Imposex stage criteria

Imposex Stage	Characteristics
1	development of small penis or small section of vas deferens
2	development of either: 1. larger penis with a penis duct, or 2. two sections of vas deferens, or 3. both a penis and a vas deferens section
3	development of either: 1. larger penis with vas deferens section, or 2. a complete vas deferens, or 3. a larger penis with a penis duct and a vas deferens section
4	development of a larger penis with penis duct and a complete vas deferens (last fertile imposex stage)

[41] Mesogastropods and neogastropods are snails in the taxonomic orders of Mesogastropoda and Neogastropoda, respectively.

[42] A sperm-carrying duct.

Imposex Stage	Characteristics
5	development of a prostate gland or occlusion of the vulva (infertile stage)
6	infertile stage with aborted capsules

Source: Oehlmann et al. (1991)

In imposexed specimens where the vas deferens could not be seen, penis length was measured in both genders to assess the level of imposex using the relative penis size (RPS) approach (Gibbs et al. 1988). The RPS index was calculated as follows:

$$\text{RPS index} = (\text{mean length of female penis}^3/\text{mean length of male penis}^3) \times 100 \qquad \textbf{Equation 3-2}$$

Imposex evaluation was performed on as many different meso- and neogastropods as possible collected in LDW. Three neogastropods, *Olivella baetica*, *Astyris gausapata,* and *Nassarius mendicus,* were collected in large numbers and assessed for imposex. All other meso- and neogastropods were collected in low numbers (one to five); and because the majority of the specimens were immature, only two mature female mesogastropods (one of each *Polinices* sp. and *Lacuna vincta*) could be assessed. Imposex was not present in these mesogastropods.

No signs of imposex were found in 127 female *A. gausapata,* which was the most abundant gastropod species in the LDW, over a range of sediment TBT concentrations from 94 to 3,000 µg/kg dw.[43] Similarly, no signs of imposex were found in 19 female *O. baetica* collected at one location with a sediment TBT concentration of 2,300 µg/kg dw. Imposex was observed only in *N. mendicus* females collected at locations with sediment TBT concentrations ranging from 34 to 358 µg/kg dw. The degree of imposex in all female *N. mendicus* examined was found to be similar to Stage 2, except that in all cases penises were present but no vasa deferentia[44] were observed (Windward 2006c). Because the vas deferens could not be seen, the RPS index was calculated. The index ranged from 0.1% to 2.6% (Table A.3-13). A site-wide RPS index of 1.8% was also calculated for all of the locations sampled in 2005 by using the mean female penis length and the mean male penis length from all locations. A station-specific RPS index was also calculated for the two stations that had males and at least one female with imposex. The RPS indices for these two stations (G17b and G18b) were 2.0% and 3.4%, respectively. According to Spence et al. (1990), in general, sterile females are absent at RPS indices below 5%, between 5% and 40% the percentage of sterility increases, and at RPS indices exceeding 40%, most or all females are sterile.

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[43] The sediment TBT concentrations were not analyzed synoptically with the imposex evaluation but were analyzed during previous sampling events.

[44] Plural of vas deferens.

Table A.3-13. TBT concentrations in sediment and *Nassarius mendicus* RPS indices

LOCATION ID	TBT CONCENTRATION IN SEDIMENT (µg/kg dw)[a]	MEAN FEMALE PENIS LENGTH (mm) (±st dev) (number measured)	MEAN MALE PENIS LENGTH (mm) (number measured)[b]	RPS INDEX (%)
2004				
G1b	358	0.86 (1)	6.83 (7)	0.2
G2b	144	1.56 ± 0.49 (4)	6.83 (7)	1.2
G3b	94	1.8 (1)	6.83 (7)	1.8
G6b	34	0.70 ± 0.56 (3)	6.83 (7)	0.1
G8b	117	1.1 (1)	6.83 (7)	0.4
2005				
G17b	320	3.3 (1)	11.2 ± 1.5 (5)	2.6
G18b	350	3.1 ± 0.9 (2)	11.2 ± 1.5 (5)	2.1
G19b	250	2.6 ± 0.1 (2)	11.2 ± 1.5 (5)	1.2

[a] TBT concentration in sediment from a single grab or composite sample previously collected at target location, as reported in the Phase 1 RI (Windward 2003a) or in sediment data reports (Windward 2005b, d, e).

[b] Because penis length in males is not known to be affected by TBT exposures, the mean penis length was calculated for the two separate sampling events and used to calculate the RPS index for each location.

dw – dry weight

ID – identification

RPS – relative penis size

TBT – tributyltin

A.3.2.4.2 Critical tissue-residue assessment

Potential effects from TBT exposure on survival, growth, and reproduction of benthic invertebrates were also evaluated using a critical tissue-residue approach. Excluding studies involving the imposex endpoint for gastropods, which was addressed through direct measurement of the imposex endpoint as discussed in Section A.3.2.4.1 above, five studies were identified that reported tissue concentrations of TBT associated with adverse effects (Table A.3-14). The LOAELs for effects on growth and reproduction ranged from 2.36 to 5.44 mg/kg dw. The lowest LOAEL (2.36 mg/kg dw) was selected as the TRV because of the relevance of the sediment exposure as well as the polychaete growth endpoint. The juvenile polychaetes exhibited a reduction in growth of 25% relative to the control sediment at a TBT concentration of 101 ng/g dw in sediment, which resulted in a tissue concentration of 2.36 mg/kg dw (Meador and Rice 2001). The associated NOAEL of 0.97 mg/kg dw was the only NOAEL below the LOAEL.

Table A.3-14. TBT critical tissue-residue toxicity studies for benthic invertebrates

TEST SPECIES	NOAEL mg/kg dw)	LOAEL (mg /kg dw)	EFFECT	EXPOSURE CONDITIONS	SOURCE
Polychaete (*Armandia brevis*)	0.97	2.36	reduced growth	sediment 42 days	Meador and Rice (2001)
Polychaete (*Neanthes arenaceodentata*)	2.99	6.27	impaired reproduction	aqueous 10 weeks	Moore et al. (1991)
Blue mussel (*Mytilus edulis)*	3.96	5.44	reduced growth	aqueous 4 days	Widdows and Page (1993)
Amphipod (*Hyalella azteca*)	na	32	reduced survival (LC50)	aqueous 10 days	Borgmann et al. (1996)
Polychaete (*Armandia brevis*)	na	41	reduced survival (LC50)	aqueous 10 days	Meador (1997)
Amphipod (*Eohaustorius estuarius*)	na	59	reduced survival (LC50)	aqueous 10 days	Meador (1997)
Amphipod (*Rhepoxynius abronius*)	na	54	reduced survival (LC50)	aqueous 10 days	Meador (1997)

dw – dry weight

LC50 – concentration that causes the death of 50% of a group of test animals

LOAEL – lowest-observed-adverse-effect level

na – not available

NOAEL – no-observed-adverse-effect level

A.3.3 CRAB EXPOSURE ASSESSMENT

This section presents the assessment of crab exposure to COPCs identified in the problem formulation (Section A.2.5.1.3). Two chemicals, zinc and total PCBs, were identified as COPCs for crabs.

As part of Phase 2 investigations, 10 Dungeness crab and 13 slender crab edible meat samples were analyzed for butyltins, SVOCs (including PAHs), metals, total PCBs as Aroclors, and organochlorine pesticides. In addition, chemical analyses were performed on hepatopancreas samples from six Dungeness crabs and five slender crabs. The edible meat samples were analyzed as composite samples created by homogenizing edible meat from five[45] individual specimens together. Similarly, the hepatopancreas samples were created by homogenizing the hepatopancreas from 15[46] individual specimens together (Windward 2005c). The compositing plan took into consideration sampling area in LDW, specimen size, and gender.

[45] One composite sample included edible meat from six Dungeness crabs to provide enough tissue for analysis.

[46] One composite sample included hepatopancreas tissue from 16 slender crabs to provide enough tissue for analysis.

One other study, conducted by King County in 1997, was included in the exposure dataset. King County collected two composite Dungeness crab edible meat samples and one composite hepatopancreas sample (King County 1999b). The samples were analyzed for total PCBs as Aroclors, metals, and SVOCs. The crab samples were collected between Slip 1 and Harbor Island. Map A.2-2 presents the crab tissue sampling location for these samples.

The exposure data were evaluated as edible meat, hepatopancreas, and whole-body depending on the available effects data. Because effects data for several chemicals, including zinc and total PCBs, were available only for whole-body crabs, the whole-body concentrations for Dungeness and slender crabs were estimated using the following equation:

$$C_{wb} = (C_h \times F_h) + (C_{em} \times F_{em}) \qquad \textbf{Equation 3-3}$$

Where:

C = concentration
wb = whole-body
h = hepatopancreas
F = fraction
em = average edible meat

The hepatopancreas and edible meat fractions were estimated to be 0.31 and 0.69, respectively, based on the ratio of masses of these tissues in a 16.6 cm[47] Dungeness crab dissected at Windward[48] with 158 g edible meat and 49 g hepatopancreas tissue mass. Similar results were presented in Atar and Secer (2003). Based on width/mass relationship data presented in Atar and Secer (2003), 50 g of edible meat mass and 15 g hepatopancreas mass is predicted for a 9-cm crab, assuming a constant relationship among the weights of edible meat, hepatopancreas, and whole body. A summary of whole-body tissue-residue estimates for zinc and total PCBs is presented in Table A.3-15. Data for Dungeness and slender crabs were combined for the entire LDW to estimate risks to the crab community.

Table A.3-15. Estimated COPC concentrations in whole-body crab tissue

COPC	Number of Samples	Unit	Minimum Concentration	Maximum Concentration	Mean Concentration	UCL
Zinc	21	mg/kg ww	24.6	37.3	30.8	32
Total PCBs	25	µg/kg ww	250	1,900	888	1,100

COPC – chemical of potential concern
PCB – polychlorinated biphenyl
UCL – upper confidence limit on the mean
ww – wet weight

[47] Maximum width of the shell from tip of spine to tip of spine.
[48] A live Dungeness crab was purchased and dissected at Windward to determine the relative weights of edible meat and hepatopancreas.

A.3.4 CRAB EFFECTS ASSESSMENT

This section presents toxicological studies that reported critical tissue residues of zinc and total PCBs associated with potential effects on survival, growth, and reproduction in decapods.

The literature search for effects data was performed as described in the problem formulation (Section A.2.5.1.3). Guidelines for the selection of NOAELs and LOAELs were:

- Studies with decapod tissue-residue data were retained and preference was given to whole-body tissue data.
- Controlled laboratory studies of single chemical exposure with statistical significance were given preference.
- Chronic exposure duration (30-plus days) studies were preferred if available.

Table A.3-16 presents all of the toxicological studies for the two crab COPCs. All toxicological studies evaluated during the TRV search are presented in Attachment 5.

Only one toxicity study that evaluated the effects of zinc exposure on decapods was identified. Crayfish (*Orconectes virilis*) were exposed to five concentrations of zinc (5.2, 12.2, 26.8, 63.3, and 130 mg/L) for 2 weeks (Mirenda 1986b). Following exposure to zinc concentrations of 26.8 mg/L and 12.2 mg/L, crayfish mortality was 23% and 5.7%, respectively. These two concentrations were selected as the LOAEL and NOAEL because the mortality observed in the 12.2 mg/L exposure was not significantly different than mortality in the control. At the end of the exposure duration, the mean whole-body zinc concentration in crayfish was 35.2 mg/kg ww (LOAEL) and 12.7 mg/kg ww (NOAEL).

Four studies were identified that evaluated the effects of PCBs on decapod species (Duke et al. 1970; Hansen et al. 1974b; Nimmo et al. 1974; Sanders and Chandler 1972). Three shrimp species (brown, grass, and pink shrimp) were exposed to Aroclor 1254 and two shrimp species (grass and brown shrimp) were exposed to Aroclor 1016 in water for 2 to 20 days. Effects on survival were assessed in all studies. Reported NOAELs ranged from 1 µg/L to 10 µg/L and reported LOAELs ranged from 1 µg/L to 100 µg/L. Whole-body tissue concentrations associated with NOAEL exposures ranged from 1,300 µg/kg ww to 18,000 µg/kg ww, and ranged from 1,100 µg/kg ww to 42,000 µg/kg ww for LOAEL exposures. The effects of PCBs (Aroclor 1254) on the survival of two other decapod species, crayfish and blue crab, were also evaluated. No effects were observed in either species following water exposure for 20 to 21 days. NOAELs ranged from a whole-body tissue concentration of 1,220 µg/kg ww (crayfish) to 23,000 µg/kg ww (blue crab).

Table A.3-16. Zinc and PCB critical tissue-residue toxicity studies for crabs and other decapods

Chemical	Test Species	Tissue Type	NOAEL	LOAEL	Unit	Exposure Route and Duration	Effect	Source
Zinc	crayfish (*Orconectes virilis*)	whole body	**12.7**[a]	**35.2**[a]	mg/kg ww	12.2 and 26.8 mg/L in water for 2 weeks	reduced survival	Mirenda (1986a)
PCBs (Aroclor 1016)	grass shrimp (*Palaemonetes pugio*)	whole body	**110**[b]	**1,100**[c]	µg/kg ww	0.4 µg/L in water for 96 hours	reduced survival	Hansen et al. (1974b)
PCBs (Aroclor 1016)	brown shrimp (*Penaeus aztecus*)	whole body	3,800	42,000	µg/kg ww	1 and 10 µg/L in water for 96 hours	reduced survival	Hansen et al. (1974b)
PCBs (Aroclor 1254)	crayfish (*Orconectes nais*)	whole body	1,220[a, d]	na	µg/kg ww	1.2 µg/L in water for 96 hours	no effect on survival	Sanders and Chandler (1972)
PCBs (Aroclor 1254)	pink shrimp (*Panaeus duorarum*)	whole body	1,300	3,900	µg/kg ww	10 and 100 µg/L in water for 48 hours	reduced survival	Duke et al. (1970)
PCBs (Aroclor 1254)	pink shrimp (*Panaeus duorarum*)	whole body	na	16,000	µg/kg ww	5.0 µg/L in water for 20 days	reduced survival	Duke et al. (1970)
PCBs (Aroclor 1254)	blue crab (*Callinectes sapidus*)	whole body	23,000	na	µg/kg ww	5.0 µg/L in water for 20 days	no effect on survival	Duke et al. (1970)
PCBs (Aroclor 1254)	grass shrimp *(Palaemonetes pugio)*	whole body	18,000	27,000	µg/kg ww	1.3 and 4.0 µg/L in water for 16 days	reduced survival	Nimmo et al. (1974)

[a] Converted from dry weight to wet weight using a moisture content of 80% (Jarvinen and Ankley 1999); 80% is also the average moisture content of two crab samples collected by King County in 1997 (King County 1999b). Mean % moisture in LDW crabs was 82.7% (Windward 2005c).

[b] The LOAEL TRV was divided by an uncertainty factor of 10 to obtain the NOAEL TRV.

[c] Survival reduced 33%.

[d] In the study, the data from the 96-hour test were extrapolated to a 21-day exposure using a biomagnification factor.

LOAEL – lowest-observed-adverse-effect level

na – not available

NOAEL – no-observed-adverse-effect level

PCB – polychlorinated biphenyl

ww – wet weight

Bold identifies the NOAEL and LOAEL selected as the TRVs.

The lowest LOAEL of 1,100 µg/kg ww based on a study of grass shrimp and Aroclor 1016 was selected as the TRV (Hansen et al. 1974b). Because this LOAEL was lower than any of the available NOAELs, a NOAEL of 110 µg/kg ww was calculated from the same study by dividing this LOAEL by 10.

The selected NOAELs and LOAELs for zinc and total PCBs were based on acute exposure studies and no uncertainty factor was applied to derive chronic values. This uncertainty is discussed in the uncertainty analysis (Section A.6.1.2).

A.3.5 SUMMARY OF BENTHIC INVERTEBRATE COMMUNITY ASSESSMENT

A.3.5.1 Exposure assessment

Exposure of the benthic invertebrate community to COPCs was evaluated based on surface sediment, porewater, and benthic invertebrate tissue data, depending on the COPC. Exposure of the benthic invertebrate community to 44 COPCs in sediment was assessed based on the magnitude of detected concentrations at a particular location. Exposure of the single COPC in porewater (cis-1,2-dichloroethene) was evaluated based on the concentrations of this COPC in peeper samples at two sites in the LDW (Tables A.3-2 and A.3-3). To evaluate exposure to TBT, concentrations of TBT in benthic invertebrate tissue were estimated using a regression between benthic invertebrate tissue and co-located sediment to derive an estimated UCL concentration in tissue (Table A.3-4). For meso- and neogastropods, which are particularly sensitive to TBT, a direct measurement of the most sensitive endpoint, imposex, was used (Table A.3-13).

A.3.5.2 Effects assessment

In the effects assessment, the type of biological endpoints used to establish the SMS and DMMP guidelines for the COPCs were discussed. TRVs for most sediment COPCs were based on SMS chemical criteria. However, TRVs for three sediment COPCs were derived from either toxicologically based DMMP guidelines or the scientific literature because no chemical criteria were available. In addition, measures of site-specific toxicity were discussed. The results of site-specific toxicity testing were presented for 48 locations tested by LDWG. Of the 48 samples tested, 18 sediment samples did not exceed the biological effects criteria, 11 sediment samples exceeded the SQS biological effects criteria, and 19 sediment samples exceeded the CSL biological effects criteria. In addition, one of seven sediment samples from a historical surface sediment toxicity study (conducted by King County) was toxic.

The literature was searched for toxicity studies that evaluated effects associated with cis-1,2-dichloroethene in porewater. The lowest toxicity value for 1,2-dichloroethene was derived from a study testing toxicity to *Artemia salina* (Sanchez-Fortun et al. 1997). The LC50 for *Artemia* (6,785 µg/L) from this was selected as the LOAEL TRV for 1,2-dichloroethene. Because no NOAELs were available for this chemical, the LC50-based LOAEL was divided by 50 to derive a NOAEL of 136 µg/L (Sanchez-Fortun et al. 1997).

Possible effects of TBT were evaluated using two approaches: 1) a site-specific study providing a direct assessment of imposex in LDW-collected meso- and neogastropods, and 2) a critical tissue-residue approach for benthic invertebrates. Imposex analysis was performed on three neogastropod species, *Nassarius mendicus, Astyris gausapata*, and *Olivella baetica*, and on a limited number (2 to 5) of four mesogastropod species (*Natica* sp., *Polinices* sp., *L. vincta*, and *Melanella* sp.). Signs of imposex (stage 2 and maximum RPS index of 3.4) were observed only in *N. mendicus*.

To assess effects of TBT using the critical tissue-residue approach, the lowest LOAEL (2.36 mg/kg dw), which was associated with reduced growth in the polychaete, *Armandia brevis,* following exposure to TBT-spiked sediment (Meador and Rice 2001), was selected. The associated NOAEL of 0.97 mg/kg dw from the same study was selected as the NOAEL.

A.3.6 Summary of Crab Assessment

A.3.6.1 Exposure assessment

Two chemicals, zinc and total PCBs, were identified as COPCs for crabs in the problem formulation. Exposure of crabs to these COPCs in sediment was assessed as the concentrations of these chemicals in crab tissue. The concentrations of zinc and total PCBs in whole-body tissue were estimated from LDW crab edible meat and hepatopancreas samples (Section A.3.3; Table A.3-15). UCL concentrations of 32 mg/kg ww and 1,100 µg/kg ww were calculated for zinc and total PCBs, respectively.

A.3.6.2 Effects assessment

The effects assessment discussed the effects on crabs from exposure to zinc and total PCBs based on critical tissue-residue TRVs for crabs and decapods. A summary of selected NOAEL and LOAEL TRVs is presented in Table A.3-17.

Table A.3-17. Selected critical tissue-residue TRVs for crabs

COPC	Test Species	NOAEL	LOAEL	Unit	Effect	Source
Zinc	crayfish (*Orconectes virilis*)	12.7[a]	35.2[a]	mg/kg ww	reduced survival	Mirenda (1986a)
Total PCBs	grass shrimp (*Palaemonetes pugio*)	110[b]	1,100[c]	µg/kg ww	reduced survival	Hansen et al. (1974b)

[a] Converted from dry weight to wet weight using a moisture content of 80%(Jarvinen and Ankley 1999); 80% is also the average moisture content of two crab samples collected by King County in 1997 (King County 1999b). Mean % moisture in LDW crabs was 82.7% (Windward 2005c).

[b] The LOAEL TRV was divided by an uncertainty factor of 10 to obtain the NOAEL TRV.

[c] Survival was reduced by 33%.

COPC – chemical of potential concern
LOAEL – lowest-observed-adverse-effect level
NOAEL – no-observed-adverse-effect level
PCB – polychlorinated biphenyl
ww – wet weight

A.4.0 Exposure and Effects Assessment: Fish

Three ROCs were selected in the problem formulation to represent fish that use the LDW (Section A.2.3) and may be exposed to sediment-associated chemicals:

- Juvenile chinook salmon
- Pacific staghorn sculpin
- English sole

ROC/COPC pairs identified in the problem formulation are summarized in Table A.4-1.

Table A.4-1. ROC/COPC pairs evaluated for fish

COPC	ROCs		
	Juvenile Chinook Salmon	English Sole	Pacific Staghorn Sculpin
Arsenic[a]	X	X	X
Cadmium[a]	X	X	X
Copper[a]	X	X	X
Vanadium[a]	X	X	X
TBT[b]			X
PCBs[b]		X	X

[a] Evaluated using the dietary approach.

[b] Evaluated using the critical tissue-residue approach.

COPC – chemical of potential concern

ROC – receptor of concern

PCB – polychlorinated biphenyl

TBT – tributyltin

In this ERA, risks to fish were evaluated using two approaches, depending on the COPC. Exposures to TBT and PCBs were evaluated based on concentrations of COPCs in fish tissue, referred to as a critical tissue-residue approach. Exposures to arsenic, cadmium, copper, and vanadium were evaluated based on concentrations in fish prey or stomach contents, referred to as a dietary approach. With this latter approach, the concentration of the relevant COPC in the fish diet was computed and compared to a LOAEL or NOAEL expressed as a concentration in food.

The application of these approaches is presented in this section, which is divided into an exposure assessment (Section A.4.1) and an effects assessment (Section A.4.2). Data presented in these sections are synthesized in the risk characterization (Section A.6.2) to assess risks to fish in the LDW, and uncertainties are also discussed.

A.4.1 EXPOSURE ASSESSMENT

In this section, relevant data were analyzed to determine representative exposure concentrations in the LDW for each ROC/COPC pair identified in Table A.4-1. The LDW dataset used to estimate exposure of fish to COPCs is summarized in Section A.2.4.1.

This section is divided into two subsections to assess exposure based on the approaches discussed above. The first subsection (Section A.4.1.1) presents whole-body concentrations of TBT and PCBs in ROC tissue for the critical tissue-residue approach. The second subsection (Section A.4.1.2) presents concentrations in dietary items and concentrations in juvenile chinook salmon stomach contents for the dietary approach.

A.4.1.1 Critical tissue-residue exposure assessment

The whole-body tissue concentration integrates exposure from all pathways (e.g., direct sediment and water contact and diet) to a fish over its foraging range. The following ROC/COPC pairs were evaluated using a critical tissue -residue approach:

- Pacific staghorn sculpin/TBT
- Pacific staghorn sculpin/total PCBs
- English sole/total PCBs

Foraging ranges for the fish ROCs are not precisely known. The English sole foraging range is assumed to be the entire LDW, whereas the Pacific staghorn sculpin foraging range is more uncertain. Thus, Pacific staghorn sculpin exposure was estimated assuming two different foraging ranges: 1) the entire LDW, and 2) four segments, each representing approximately one-quarter of the LDW. Therefore, for both species, exposures occurring across the entire LDW were estimated as the UCL on the mean concentration of all composite tissue samples from each ROC collected throughout the LDW.

To estimate exposures of Pacific staghorn sculpin in smaller areas, separate UCLs were also calculated for the PCB and TBT sculpin tissue samples from each of the four Phase 2 fish collection areas of the LDW (Map A.2-2). UCL calculation methods and results, along with minimum, mean, and maximum concentrations, are presented in Attachment 11. Resulting exposures for each ROC/COPC pair are presented in Table A.4-2.

Table A.4-2. TBT and total PCB exposure concentrations in whole-body fish tissue

ROC	COPC	Location	UCL Concentration (µg/kg ww)
English sole[a]	total PCBs	LDW-wide	2,600
Pacific staghorn sculpin	TBT	LDW-wide	36
		T1	37
		T2	36
		T3	28
		T4	53
	total PCBs	LDW-wide	1,100
		T1	800
		T2	920
		T3	2,000
		T4	940

[a] Includes 42 English sole and 3 starry flounder whole-body/composite samples; starry flounder were used as a surrogate for English sole because insufficient numbers of English sole were collected in Area T4 during Phase 2 sampling (see Map A.2-2). Ten of the English sole whole-body sample concentrations were estimated based on the relative weights and total PCB concentrations detected in corresponding skin-on fillet and remainder samples.

COPC – chemical of potential concern
PCB – polychlorinated biphenyl
ROC – receptor of concern
TBT – tributyltin
UCL – upper confidence limit on the mean
ww – wet weight

Composite tissue samples were collected in all four Phase 2 tissue collection areas in order to maximize the number of individuals analyzed to provide the best estimate of the population mean for each species. It should be noted, however, that the use of composite samples may have resulted in a lower estimate of the variance of the population than if individual fish tissue samples had been analyzed. The UCL of the mean of composite samples may therefore underestimate the UCL that would be calculated from an analysis of the same number of individual samples. Because the actual variance at the individual level is unknown, the effect on the estimate of the UCL of analyzing composite tissue samples rather than individual tissue samples is unknown.

A.4.1.2 Dietary exposure

This section presents the approach used to estimate exposure through the diet of the three fish ROCs to arsenic, cadmium, copper, and vanadium. Comparison of chemical concentrations in prey to suitable dietary TRVs is preferable for COPCs that are highly regulated or metabolized by fish. Most aquatic organisms have specific mechanisms for uptake, internal transport, sequestration, and depuration of metals (Meyer et al. 2005). Essential metals are regulated because they are necessary for normal metabolic function, whereas other metals appear to be regulated because they mimic essential elements and are transported by the same mechanisms (Bury et al. 2003 as cited in Meyer et al. 2005).

The primary exposure route of these COPCs was assumed to be ingestion of food. Dermal contact and incidental sediment ingestion were also considered complete exposure pathways for English sole and Pacific staghorn sculpin (see Figure A.2-2) but with unknown significance. Direct water contact and water ingestion were considered complete and significant pathways for all three fish ROCs. Risks associated with water exposure were evaluated in the King County WQA (King County 1999d); the results of that risk assessment are summarized in the risk characterization (Section A.6.2.1).

The dietary exposure approach requires an approximation of the COPC concentration in an ROC's diet. This section presents the dietary exposure assumptions for each fish ROC, the dietary exposure calculation methods, and the results of the exposure calculations. To approximate the dietary concentrations for each fish ROC, the feeding habits of each ROC were considered. ROC-specific exposure assumptions are described below.

Stomach contents analyses of juvenile chinook salmon from the LDW indicate that they typically ingest benthic invertebrates such as amphipods, worms, and clam siphons,[49] as well as drift organisms and zooplankton (Cordell et al. 1997, 1999, 2001). Because no zooplankton or drift organism tissue concentration data were available, juvenile chinook salmon were assumed to ingest only benthic invertebrates. Because benthic invertebrates live in close contact with sediments, they have greater potential for sediment exposure than do other juvenile chinook salmon prey items; therefore, these exposure assumptions are conservative (i.e., may overestimate exposures but unlikely to underestimate them). Juvenile chinook from the LDW were found to have no appreciable amounts of sediment in their stomachs (Cordell 2001); therefore, they were assumed to have no incidental sediment ingestion. Juvenile chinook salmon generally do not use deep-water habitats (Tabor et al. 2004); exposure was estimated assuming juvenile chinook salmon are exposed primarily in intertidal areas. Therefore, juvenile chinook salmon dietary exposures were based on intertidal benthic invertebrate tissue data. In addition to benthic invertebrate tissue data, exposure was also separately estimated based on chemical analysis of stomach contents of juvenile chinook salmon collected from the LDW.

Stomach contents analyses of English sole collected from Puget Sound show that English sole almost exclusively ingest benthic invertebrates such as marine worms, amphipods, bivalves, and mollusks (Fresh et al. 1979; Wingert et al. 1979). Based on

[49] Windward (2005b) clam data were not included in exposure calculations because benthic invertebrate samples included clams less than 2.0 cm, which are assumed to represent clam tissues consumed by fish ROCs. Mollusks constituted from 0 to 41% (median 17%) of tissue mass in benthic invertebrate samples (Windward 2005b).

these stomach contents analyses, all prey of English sole were assumed to be represented by the benthic invertebrate tissue chemistry data. In addition, incidental sediment ingestion of 1% was assumed based on anecdotal stomach contents observations of English sole and other bottom-feeding fish (Johnson 2006; Lange 2006).[50] Although English sole foraging ranges are uncertain, the available information (Day 1976; Stern et al. 2003) suggests that English sole forage in an area as large as or larger than the LDW. Therefore, English sole dietary exposure was calculated only on a site-wide basis.

Stomach contents analyses of Pacific staghorn sculpin collected from Puget Sound show that they ingest small fish and benthic invertebrates such as crabs, shrimp, marine worms, and amphipods (Wingert et al. 1979; Fresh et al. 1979; Miller et al. 1977c). Based on the relative proportions of biomass reported in stomach contents analyses, the average proportions of fish and benthic invertebrates in dietary exposure estimates were assumed to be 44 and 55%, respectively (Wingert et al. 1979; Fresh et al. 1979; Miller et al. 1977c). Shiner surfperch were selected as representative prey fish because they are numerically dominant in the fish community in the LDW (Windward 2005c, 2006b) and, thus, are likely to be ingested by Pacific staghorn sculpin. They also have a primarily benthic diet so they represent bioaccumulation of sediment-associated chemicals through the food chain (Wingert et al. 1979; Fresh et al. 1979; Miller et al. 1977c). All invertebrate prey were assumed to be represented by the benthic invertebrate tissue chemistry data. Incidental sediment ingestion of 1% was assumed based on Pacific staghorn sculpin's primarily epifaunal diet (Lange 2006). No data are available to determine Pacific staghorn sculpin foraging areas; however, based on opinions expressed at an LDW fish experts meeting with LDWG, EPA, and Ecology that took place on March 31, 2004, Pacific staghorn sculpin may have foraging areas as large as or smaller than the size of the LDW. Therefore, two Pacific staghorn sculpin dietary exposure scenarios were evaluated (an LDW-wide scenario and an area-specific scenario). For the area-specific scenario, exposures were estimated for four modeling areas (M1, M2, M3, and M4), defined as the four fish and crab tissue sampling areas, which extended out to the center points between each adjacent pair of tissue sampling areas (Map A.2-2). Modeling areas, rather than tissue sampling areas, were used in this exposure scenario in order to ensure that all available sediment chemistry and benthic invertebrate tissue concentrations were used in calculating dietary exposure for Pacific staghorn sculpin.

[50] Uncertainty in incidental sediment ingestion is explored in the uncertainty analysis (Section A.6.2.2.2) where exposure is also calculated assuming 10% sediment ingestion.

Using the ROC-specific assumptions described above, COPC concentrations in the diet of each fish ROC were calculated as the weighted average of COPC concentrations in sediment and prey tissue using Equation 4-1.

$$C_{diet} = \sum_{i=1}^{n} X_i C_i \qquad \textbf{Equation 4-1}$$

Where:

C_{diet} = COPC concentration in the diet (mg/kg dw)

X_i = proportion of a particular food item (or sediment) in the diet (unitless)

C_i = COPC concentration in the prey item (mg/kg dw)

n = number of dietary items

The relative proportions of each prey item in each fish ROC's diet are summarized in Table A.4-3.

Table A.4-3. Proportions of dietary items in dietary exposure estimates for each fish ROC

ROC	Prey Item	Proportion in Diet (unitless)	Source
Juvenile chinook salmon	benthic invertebrates (intertidal only)	1	Windward (2004c); Cordell et al. (1997)
English sole	benthic invertebrates (LDW-wide)	0.99	Fresh et al. (1979); Wingert et al. (1979)
	sediment (LDW-wide)	0.01	Johnson (2006); Lange (2006)
Pacific staghorn sculpin	shiner surfperch	0.44	Fresh et. al (1979); Wingert et al. (1979); Miller et al. (1977c)
	benthic invertebrates (LDW-wide)	0.55	
	sediment (LDW-wide)	0.01	(Lange 2006)

LDW – Lower Duwamish Waterway

ROC – receptor of concern

Concentrations in each prey item were estimated as the UCL on the mean concentration, which was calculated as described in Attachment 11. For benthic invertebrate prey, UCLs were calculated in two ways (Attachment 11). First, if the regression between co-located sediment and benthic invertebrate tissue data was significant for a given COPC, then the UCL in tissue was predicted from the regression equation using the arithmetic mean concentration in sediment. LDW-wide arithmetic mean concentrations in sediment were used to estimate UCL prey concentrations for English sole and for the Pacific staghorn sculpin LDW-wide exposure scenario. The mean sediment concentrations for the four modeling areas were used for the Pacific staghorn sculpin area-specific exposure scenarios, and the mean concentration of intertidal sediment samples was used for juvenile chinook salmon. If the regression between co-located sediment and benthic invertebrate tissue data was not significant, then the UCL was calculated using the benthic invertebrate tissue dataset.[51] All LDW benthic invertebrate tissue data were included in the calculations for English sole and for the Pacific staghorn sculpin LDW-wide exposure scenario. Benthic invertebrate tissue data collected from within the individual modeling areas were used for the Pacific staghorn sculpin area-specific exposure scenarios, and intertidal benthic invertebrate tissue data were used for juvenile chinook salmon exposure calculations.

Resulting exposure concentrations for each ROC/COPC pair are presented in Table A.4-4. Dietary exposure concentrations for juvenile chinook salmon based on the single composite stomach contents sample from 72 fish[52] are presented in Table A.4-5.

[51] The regression was significant for arsenic but not for cadmium, copper, or vanadium (Attachment 11).

[52] Note that an unknown fraction of these stomachs were empty. Ruggerone et al. (2006) found that 4 to 6% of juvenile chinook from the LDW had empty stomachs.

Table A.4-4. Dietary exposure calculations and resulting COPC concentrations in fish ROC diets

Chemical	Exposure Area	Concentration in Dietary Component (UCL) (mg/kg dw)			Fraction of Dietary Component in Diet			Concentration in Diet (mg/kg dw)
		Sediment	Benthic Invertebrates	Shiner Surfperch	Sediment	Benthic Invertebrates	Shiner Surfperch	
Juvenile chinook salmon								
Arsenic	LDW-wide	na	22[a]	na	0	1	na	22
Cadmium	LDW-wide	na	0.50[b]	na	0	1	na	0.50
Copper	LDW-wide	na	93[b]	na	0	1	na	93
Vanadium	LDW-wide	na	8.1[b]	na	0	1	na	8.1
English sole								
Arsenic	LDW-wide	30	24[c]	na	0.01	0.99	na	24
Cadmium	LDW-wide	2.4	0.60	na	0.01	0.99	na	0.61
Copper	LDW-wide	200	92	na	0.01	0.99	na	93
Vanadium	LDW-wide	60	12	na	0.01	0.99	na	12
Pacific staghorn sculpin								
Arsenic	LDW-wide	30	24[c]	4.2	0.01	0.55	0.44	15
Arsenic	M1	34	23	4.7	0.01	0.55	0.44	15
Arsenic	M2	46	25	5.0	0.01	0.55	0.44	16
Arsenic	M3	40	23	3.7	0.01	0.55	0.44	15
Arsenic	M4	11	18	4.1	0.01	0.55	0.44	12
Cadmium	LDW-wide	2.4	0.60	0.066	0.01	0.55	0.44	0.38
Cadmium	M1	1.1	0.81	0.079	0.01	0.55	0.44	0.49
Cadmium	M2	0.55	0.48	0.071	0.01	0.55	0.44	0.30
Cadmium	M3	6.8	0.77	0.068	0.01	0.55	0.44	0.52
Cadmium	M4	0.33	0.54	0.056	0.01	0.55	0.44	0.32

Chemical	Exposure Area	Concentration in Dietary Component (UCL) (mg/kg dw)			Fraction of Dietary Component in Diet			Concentration in Diet (mg/kg dw)
		Sediment	Benthic Invertebrates	Shiner Surfperch	Sediment	Benthic Invertebrates	Shiner Surfperch	
Copper	LDW-wide	200	92	7.0	0.01	0.55	0.44	56
	M1	97	110	7.6	0.01	0.55	0.44	65
	M2	170	130	7.5	0.01	0.55	0.44	77
	M3	510	65	8.3	0.01	0.55	0.44	45
	M4	41	81	6.4	0.01	0.55	0.44	48
Vanadium	LDW-wide	60	12	2.0	0.01	0.55	0.44	8.1
	M1	63	12	3.6	0.01	0.55	0.44	8.8
	M2	60	18	3.7	0.01	0.55	0.44	12
	M3	60	20	2.0	0.01	0.55	0.44	12
	M4	60	10	1.2	0.01	0.55	0.44	6.6

Note: Calculated using Equation 4-1.

[a] UCL on the mean tissue concentration predicted from the intertidal arithmetic mean sediment concentration using a sediment:tissue regression (see Attachment 11).

[b] UCL on the arithmetic mean of intertidal tissue concentration data (see Attachment 11).

[c] UCL on the mean tissue concentration predicted from the LDW-wide arithmetic mean sediment concentration using a sediment:tissue regression (see Attachment 11).

COPC – chemical of potential concern

dw – dry weight

na – not applicable

ROC – receptor of concern

UCL – upper confidence limit on the mean

Table A.4-5. Juvenile chinook salmon stomach contents data for dietary COPCs

COPC	Exposure Concentration (mg/kg dw)
Arsenic	3.9
Cadmium	0.46
Copper	42
Vanadium	na

COPC – chemical of potential concern

dw – dry weight

na – not analyzed

A.4.2 Effects Assessment

This section presents the toxicological data and describes the selection of TRVs for the COPCs identified for fish ROCs. The literature search and guidelines for TRV selection for fish ROCs are described in detail in Section A.2.5.2. Toxicological studies reporting critical tissue residues of PCBs and TBT associated with potential adverse effects on survival, growth, or reproduction[53] were reviewed. Dietary studies involving these endpoints and arsenic, cadmium, copper, and vanadium were also reviewed. Growth and survival were the only endpoints evaluated for assessing risks to juvenile chinook salmon.[54] TRVs based on reproductive studies were not selected for juvenile chinook salmon 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. Toxicological data presented in this section are assessed in combination with exposure data (presented in Section A.4.1) in the risk characterization (Section A.6.2.1).

The results of several site-specific and region-specific studies that have assessed potential toxicological effects on juvenile chinook salmon and English sole are presented in the uncertainty analysis (Sections A.6.2.1.2 and A.6.2.1.3, respectively). These studies were not used as a primary source of TRVs because they involved exposure to chemical mixtures; thus, chemical-specific NOAELs and LOAELs cannot be determined from these studies.

A.4.2.1 COPCs evaluated using the critical tissue-residue approach

The following ROC/COPC pairs were evaluated using a critical tissue-residue approach:

- Pacific staghorn sculpin/TBT

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[53] The reproductive endpoint is inclusive of early life stage developmental effects (e.g., growth from egg through fry stage, embryo development/viability).

[54] PAH and PCB studies evaluating effects on immunocompetence are also discussed in the uncertainty analysis (Section A.6.2.2).

- Pacific staghorn sculpin/total PCBs
- English sole/total PCBs

The toxicological studies identified for these COPCs are summarized in the following subsections, and the selected NOAEL and LOAEL TRVs are presented. The guidelines for determining the acceptability of toxicity studies for TRV selection are described in detail in Section A.2.5.2. TRVs to evaluate risks to fish from dioxin-like PCB congeners are discussed in the uncertainty analysis (Section A.6.2) because of the uncertainty in the TRV and in fish toxic equivalency factors (TEFs).[55]

A.4.2.1.1 PCBs

Eighteen papers on the potential adverse effects of PCB mixtures on fish were reviewed (Fisher et al. 1994; Freeman and Idler 1975; Hansen et al. 1971; 1974a; 1974b; 1975; Hattula and Karlog 1972; Hendricks et al. 1981; Hugla and Thome 1999; Lieb et al. 1974; Matta et al. 2001; Mauck et al. 1978; Mayer et al. 1977; 1985; McCarthy et al. 2003; Nebeker et al. 1974; Powell et al. 2003; van Wezel et al. 1995). Concentrations in whole-body fish tissue were reported in 14 of these studies (Hansen et al. 1971; 1974a; 1974b; 1975; Hattula and Karlog 1972; Hugla and Thome 1999; Lieb et al. 1974; Matta et al. 2001; Mauck et al. 1978; Mayer et al. 1977; 1985; Nebeker et al. 1974; Powell et al. 2003; van Wezel et al. 1995), concentrations in fish eggs or embryos were reported in four studies (Fisher et al. 1994; Freeman and Idler 1975; Hendricks et al. 1981; McCarthy et al. 2003).

Critical tissue concentrations of PCBs were reported in the toxicological studies reviewed for selection of TRVs in 16 species (i.e., Atlantic croaker, Atlantic salmon, brook trout, channel catfish, coho salmon, common barbel, fathead minnow, goldfish, chinook salmon, pinfish, rainbow trout, mummichog, sheepshead minnow, and spot) . Adverse effects reported in the toxicological studies reviewed included reduced body weight, mortality, reduced early life stage or fry growth and survival, and reduced fecundity, hatchability and spawning success following exposure to PCBs via diet, water, or maternal transfer to eggs. Table A.4-6 presents a summary of the critical tissue-residue NOAELs and LOAELs reported for PCBs in these studies. NOAEL and LOAEL concentrations in eggs and embryos are presented separately from whole-body NOAELs and LOAELs because they are not directly comparable to the adult whole-body tissue concentrations from the LDW used to characterize exposure. Because egg and embryo NOAELs and LOAELs are not directly comparable, these studies were not selected as TRVs but are discussed below and in the uncertainty analysis (Section A.6.2)

[55] EPA refers to the TEF as a toxicity equivalence factor.

Table A.4-6. PCB critical tissue-residue toxicity studies for fish

Chemical	Test Species	Tissue Analyzed	NOAEL (µg/kg ww)	LOAEL (µg/kg ww)	Exposure Route and Duration	Effect	Source	Notes
Studies Reporting Whole-Body NOAELs and LOAELs								
Aroclor 1260	common barbel	whole body	na	**520**	maternal exposure for 50 days	reduced fecundity	Hugla and Thome (1999)	1
Aroclor 1254	juvenile chinook salmon	whole body	980	na	17 mg/kg ww in food for 4 weeks	no effect on growth or survival	Powell et al. (2003)	2
Aroclor 1260	common barbel	whole body	520	**2,640**	maternal exposure for 75 days	lack of spawning in first reproductive season; egg and larval mortality	Hugla and Thome (1999)	1
Aroclor 1254	rainbow trout (14 weeks)	whole body	8,000	na	15 mg/kg dw food for 32 weeks	no effect on growth or survival	Lieb et al. (1974)	
Aroclor 1254	Sheepshead minnow (adult)	whole body	1,900	9,300	maternal exposure to 0.32 µg/L in water for 28 days	decreased fry survival in the first week after hatch	Hansen et al. (1974a)	3
Aroclor 1268	Mummichog (adult)	whole body	15,000	na	15 µg/g in food for 6 weeks	no effect on fertilization, hatching, or larval survival	Matta et al. (2001)	4
Aroclor 1254	Spot	whole body	27,000	46,000	1 and 5 µg/L in water for 20 days	reduced survival	Hansen et al. (1971)	5
Aroclor 1254	brook trout embryos	whole body	31,000	71,000	0.69 and 1.5 µg/L water for 128 days (10 days prior to hatch and 118 days after)	reduced fry growth	Mauck et al. (1978)	6
Aroclor 1016	Sheepshead minnow	whole body	110,000	na	10 µg/L in water for 4 weeks	no effect on fertilization success, survival of embryos, or fry survival	Hansen et al. (1975)	7
Aroclor 1016	pinfish	whole body	na	106,000	21 µg/L in water for 33 days	50% mortality	Hansen et al. (1974b)	
Aroclor 1254: 1260 mixture	juvenile rainbow trout	whole body	120,000	na	2.9 µg/L in water for 90 days	no effect on survival	Mayer et al. (1985)	8
Aroclor 1254: 1260 mixture	juvenile rainbow trout	whole body	70,000	120,000	1.5 and 2.9 µg/L in water for 90 days	reduced growth	Mayer et al. (1985)	

Chemical	Test Species	Tissue Analyzed	NOAEL (µg/kg ww)	LOAEL (µg/kg ww)	Exposure Route and Duration	Effect	Source	Notes
Aroclor 1254	brook trout embryos	whole body	71,000	125,000	1.5 and 3.1 µg/L water for 128 days (10 days prior to hatch and 118 days after)	reduced fry survival	Mauck et al. (1978)	
Aroclor 1016	Sheepshead minnow fry	whole body	57,000	200,000	10 and 32 µg/L in water for 4 weeks	reduced fry survival	Hansen et al. (1975)	
Clophen A50	Goldfish	whole body	na	250,000	4,000 µg/L in water for 5 to 21 days	reduced survival	Hattula and Karlog (1972)	9
Aroclor 1254	fathead minnow	whole body	na	429,000 (female)	1.8 µg/L in water for 8 months	reduced spawning	Nebeker et al. (1974)	10
Aroclor 1242, 1254, or 1260	fathead minnow (6 months)	whole body	na	1,860 – 749,000	0.006 to 0.54 µmol/L in water for100 to 300 hours	range of lethal body burdens (concentration associated with mortality of individuals)	van Wezel et al. (1995)	11
Studies Reporting Only Egg and Embryo NOAELs and LOAELs								
1:1:1:1 Aroclor 1016, 1221, 1254, 1260 mixture	Atlantic salmon	embryo	na	857	embryos exposed to 625 µg/L PCB in water for 48 hours and observed through fry stage	reduced fry body weight	Fisher et al. (1994)	12
Aroclor 1254	rainbow trout	embryos	na	1,640	maternal exposure to 200 mg/kg in food for 60 days	reduced fry growth in offspring	Hendricks et al. (1981)	
Aroclor 1254	Atlantic croaker	egg	na	3,200	maternal transfer	reduced larval growth	McCarthy et al. (2003)	13
Aroclor 1254	brook trout	embryo	na	77,900	200 µg/L in water for 21 days	reduced hatchability (75%)	Freeman and Idler (1975)	

Notes:

1. Whole-body tissue residues were the weighted sum of 10 different tissues (i.e., blood, brain, muscle, skin, liver, gonads, adipose tissues, kidney, digestive tract, and skeleton) (Leroy 2007 [pers. comm.]). Tissue concentrations were converted from dry weight to wet weight assuming 20% solids; all endpoints except first reproductive season spawning were evaluated 1 year after exposure.
2. Whole-body tissue concentrations ranged from 740 to 980 µg/kg ww following the 28-day treatment.
3. Concentrations in maternal adults.
4. Two generations of progeny were observed.
5. Mortality did not appear to be directly related to PCB tissue concentration because tissue concentration increased with exposure duration.

6. At the LOAEL, growth was significantly less than controls at 48 days post hatch, but not at 118 days after hatching. At NOAEL and LOAEL concentrations, study provides tissue concentrations only after 7 days and 118 days of exposure. LOAEL and NOAEL are tissue concentrations in fry at 118 days post hatch. Tissue concentrations at 7 days post-hatch associated with no effects (1,800 µg/kg ww) and low effects (3,200 µg/kg ww) were lower than the concentration at 118 days.
7. For juvenile fish, the LOAEL and NOAEL for reduced survival were 220,000 and 57,000 µg/kg ww, respectively.
8. Survival was not significantly different from control; exposure dose was 1:2 ratio of Aroclor 1254:1260.
9. LOAEL is lethal tissue concentration.
10. LOAEL is average terminal tissue concentration of two female replicate groups. The number of eggs and number of spawnings per female were highly variable (up to 70-fold difference) between duplicate exposures and were not dose responsive at treatment levels below concentrations where no spawning occurred.
11. Tissue concentrations of individual fish that died in less than 20 hours ranged from 1,860 to 30,000 µg/kg ww; tissue concentrations of individual fish that died at 100 to 300 hours ranged from 120,000 to 749,000 µg/kg ww.
12. Growth data from 176 days post-exposure. Effects were not dose responsive; no effects were observed in fish from eggs with 1,534 µg/kg ww PCBs.
13. Only a single dose was evaluated.

LOAEL – lowest-observed-adverse-effect level

na – not available

NOAEL – no-observed-adverse-effect level

PCB – polychlorinated biphenyl

ww – wet weight

Bold identifies the NOAEL and LOAEL selected as TRVs.

Whole-body effect-level concentrations ranged over three orders of magnitude across the fish species included in the toxicological studies reviewed. Whole-body tissue LOAELs ranged from 520µg/kg ww for reduced barbel fecundity (Hugla and Thome 1999) to 429,000 µg/kg ww for reduced spawning in fathead minnows (Nebeker et al. 1974).

In the study reporting the lowest LOAEL, Hugla and Thome (1999) exposed 3- to 5-year-old common barbel from the University of Liege hatchery to 2,500 µg/kg PCBs in food for 50 days or to 12,500 µg/kg PCBs in food for 75 days (nominal concentrations) and analyzed effects on reproduction. Fish were reared at elevated temperatures (Leroy 2007 [pers. comm.]). Treatments were not replicated;16 fish in each treatment were exposed in a single tank (Leroy 2007 [pers. comm.]). Spawning success was monitored in the first reproductive season, and fish were kept in PCB-free water for 1 year and evaluated for additional adverse effects. PCB concentrations in whole fish[56] were reported following 50 or 75 days of exposure. In the first reproductive season, no spawning was reported at the high exposure level. No adverse effects were reported for the lower exposure level associated with the first reproductive season. One year following exposure, significant reductions in fecundity were reported at both exposure levels corresponding to whole-fish concentrations of 520 and 2,640 µg/kg ww for the low and high exposure levels, respectively. Mortality of eggs from the high dietary exposure group was close to 100% and was significantly higher than controls (which had a mean egg mortality of 52.4%), and egg and larval mortality significantly increased as PCB concentrations increased in eggs. At the lower dose, egg mortality was not significantly different from controls.

The fecundity LOAEL associated with the lower dose is uncertain because fecundity as measured after the first two spawning seasons was not dose responsive. Fecundity comparisons are complicated by the fact that the higher-dosed fish did not spawn during the first season and whole-body tissue concentrations were not measured 1 year later when the high-dose fish finally spawned. After the second spawning, average fecundity was similar between the high and low doses, but variance in fecundity was greater at the higher dose. In addition, the number of fish exposed at each treatment level and evaluated for effects is unclear. Because of these and additional uncertainties discussed in the uncertainty analysis (Section A.6.2.2.2), the range of effects concentrations reported in this paper for the fecundity and the spawning and egg hatchability endpoints was considered to represent the range of exposures over which the lowest adverse effects may occur in fish. Thus, the selected LOAEL for PCBs was set at a range of 520 to 2,640 µg/kg ww. Additional effects data are discussed below for comparison.

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[56] The reported whole-body fish tissue PCB concentrations were a weighted average of dry weight PCB concentrations in various tissues, specifically, blood, brain, muscle, skin, liver, gonads, adipose tissue, kidney, digestive tract, and skeleton (Leroy 2007 [pers. comm.]).

In the study reporting the next higher LOAEL, Hansen et al. (1974a) exposed 20 female and 10 male adult sheepshead minnows for four weeks to four concentrations of PCBs ranging from 0.1 to 3.2 µg/L. Eggs from five female fish from each exposure level were fertilized using a male from the same exposure level, and 25 successfully fertilized eggs from each exposure group were raised to the fry stage in PCB-free water and evaluated for survival. Reduced survival was reported for eggs from fish with maternal adult tissue concentrations of 9,300 µg/kg ww and greater; but to enhance egg production, fish were injected with human chorionic gonadotrophic hormone, which may have affected reproduction in the fish. No effects were observed at the next lower exposure level corresponding with a tissue concentration of 1,900 µg/kg ww. Uncertainties in this study are discussed in the uncertainty analysis (Section A.6.2.2.2).

Among the studies reviewed for this ERA, effects concentrations reported in eggs and embryos ranged from 857 to 77,900 µg/kg ww. The lowest value was for reduced growth of Atlantic salmon fry held in PCB-free water for 176 days following exposure of eggs to aqueous PCB concentrations of 625 to 62,500 µg/L for 48 hours (Fisher et al. 1994). The highest value was for brook trout embryos exposed to 200 µg/L of PCBs in water for 21 days (Freeman and Idler 1975). NOAELs were not identified.

Although these egg and embryo effects concentrations were generally lower than effects concentrations reported in the literature for more mature fish, egg/embryo and adult tissue-residue data are not directly comparable. Uncertainties associated with comparison of exposure concentrations to egg and embryo studies are discussed in the uncertainty analysis (Section A.6.2.2).

Whole-body NOAELs ranged from 980 µg/kg ww, at which no effect on growth or survival was reported in juvenile chinook salmon (Powell et al. 2003), to 120,000 µg/kg ww for no effect on survival of juvenile rainbow trout (Mayer et al. 1985). Because there were no NOAELs identified that were lower than the identified LOAEL range from Hugla and Thome (1999), a NOAEL range of 104 to 528 µg/kg ww was estimated by applying an uncertainty factor of 5 to the range of chronic reproductive effects concentrations reported in Hugla and Thome (1999).

A.4.2.1.2 Tributyltin

Three toxicological studies that evaluated the toxicity of TBT to fish were reviewed (Shimasaki et al. 2003; Nirmala et al. 1999; Nakayama et al. 2005). Table A.4-7 summarizes the results of these studies. Critical tissue residues of TBT were reported in Japanese flounder larvae and Japanese medaka associated with adverse effects on growth and reproductive success, respectively, following exposure to TBT in water.

Table A.4-7. TBT critical whole-body tissue-residue toxicity studies for fish

CHEMICAL	TEST SPECIES	NOAEL (µg TBT/kg ww)	LOAEL (µg TBT/kg ww)	EXPOSURE ROUTE AND DURATION	EFFECT	SOURCE	NOTES
Tributyltin oxide	Japanese flounder larvae	**18**	**159**	0.1 and 1 mg/kg dw in food for approximately 65 days	reduced body weight	Shimasaki et al. (2003)	1
Tributyltin oxide	Japanese medaka	na	1,054	maternal exposure to 5 mg/kg dw in food for 3 weeks	reduced swim-up success and hatchability	Nakayama et al. (2005)	2
Tributyltin oxide	Japanese medaka	na	2,390	maternal exposure to 1 mg/kg dw in food for 3 weeks	reduced hatching, swim-up, and embryonic success	Nirmala et al. (1999)	3

Notes:

1. No replication in study. Survival was not significantly affected at any dose; however, some reduced survival was observed in all groups, including the control group.
2. LOAEL is the TBT concentration in adult female fish estimated using an adult:egg conversion factor of 8.57, based on Nirmala et al. (1999).
3. LOAEL is the TBT concentration in adult female fish.

LOAEL – lowest-observed-adverse-effect level

na – not available

NOAEL – no-observed-adverse-effect level

TBT – tributyltin

ww – wet weight

Bold identifies the NOAEL and LOAEL selected as TRVs.

Whole-body tissue-residue[57] LOAELs ranged from 159 µg TBT/kg ww for reduced body weight in Japanese flounder larvae following 65 days of dietary exposure (Shimasaki et al. 2003) to 2,390 µg TBT/kg ww for reduced hatchability and early-life stage mortality of Japanese medaka offspring spawned from exposure of both parents to dietary TBT for three weeks during reproduction (Nirmala et al. 1999). All three toxicological studies were conducted over a chronic exposure duration or critical life stage (reproduction). The lowest LOAEL reported (159 µg TBT/kg ww) was selected as the LOAEL TRV. There is significant uncertainty associated with this LOAEL because there was no replication of test groups and high mortality was observed consistently across the control and low- and high-dose groups. However, because few toxicological data were available (only three studies were identified) and a clear effect on growth was observed in fish exposed to TBT in the study by Shimasaki et al. (2003), the LOAEL TRV was derived from this study to represent a conservatively based TRV. The other two studies by Nirmala et al. (1999) and Nakayama et al. (2005) present adverse effects to Japanese medaka reproduction following exposure to higher TBT concentrations.

Only one NOAEL was reported in the three studies reviewed (Shimasaki et al. 2003). At 18 µg/kg ww, no effect on Japanese flounder growth was observed following 65 days of dietary exposure to tributyltin oxide. This NOAEL was selected as the NOAEL TRV.

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

A.4.2.2 COPCs evaluated using a dietary approach

The following COPCs were evaluated using a dietary approach: arsenic, cadmium, copper, and vanadium. The toxicological studies identified for these COPCs are summarized in the following subsections, and the selected NOAEL and LOAEL TRVs are presented.

A.4.2.2.1 Arsenic

Six toxicity studies that evaluated the effects of dietary arsenic on fish were identified (Blazer et al. 1997; Cockell and Hilton 1988; Cockell and Bettger 1993; Cockell et al. 1991, 1992; Oladimeji et al. 1984). Table A.4-8 summarizes all dietary fish NOAELs and LOAELs for arsenic reported in the reviewed literature.

Table A.4-8. Arsenic dietary toxicity studies for fish

Chemical	Test Species	NOAEL (mg/kg dw)[a]	LOAEL (mg/kg dw)[a]	Exposure Duration	Effect	Source	Notes
Sodium arsenite	juvenile rainbow trout	**20**	**30**	6 weeks	reduced body weight	Oladimeji et al. (1984)	1, 2
Disodium arsenate heptahydrate	juvenile rainbow trout	8	44	16 weeks	reduced body weight	Cockell et al. (1991)	3
Disodium arsenate heptahydrate	juvenile rainbow trout	na	49	24 weeks	reduced body weight	Cockell et al. (1991)	3
Disodium arsenate heptahydrate	juvenile rainbow trout	na	55	8 days	reduced body weight	Cockell et al. (1992)	3
Disodium arsenate	juvenile rainbow trout	na	58	12 days	reduced body weight	Cockell and Bettger (1993)	3
Disodium arsenate heptahydrate	juvenile rainbow trout	32	60	12 days	reduced body weight	Cockell et al. (1992)	3
Disodium arsenate heptahydrate	juvenile rainbow trout	33	65	24 weeks	reduced body weight	Cockell et al. (1991)	4
Disodium arsenate	juvenile rainbow trout	na	137	8 days	reduced body weight	Cockell and Hilton (1988)	3
Arsenic trioxide	juvenile rainbow trout	na	180	8 days	reduced body weight	Cockell and Hilton (1988)	
Disodium arsenate heptahydrate	juvenile striped bass	52.3	188.8	6 days	reduced body weight	Blazer et al. (1997)	3

[a] Concentrations are for elemental arsenic.

Notes:

1. Only the nominal concentration was reported.
2. Concentrations in figure and text in study do not agree: 20 mg/kg dw is mentioned both as an effect level and a no-effect level in the text; however, it is shown in the figure to be not significant. The NOAEL was assumed to be 20 mg/kg dw.
3. Feed refusal accompanied growth effects.
4. Body weight gain reduced at 12 weeks in fish fed 33 mg/kg arsenic in diet but not at 24 weeks (body weight was recovered).

dw – dry weight

LOAEL – lowest-observed-adverse-effect level

na – not available

NOAEL – no-observed-adverse-effect level

Bold identifies the NOAEL and LOAEL selected as TRVs.

No dietary toxicity data were available for survival or reproductive endpoints. All studies reviewed reported reductions in growth among juvenile rainbow trout and striped bass following dietary exposure to arsenic. Dietary LOAELs based on growth ranged from 30 mg/kg dw for juvenile rainbow trout (Oladimeji et al. 1984) to 188.8 mg/kg dw for juvenile striped bass (Blazer et al. 1997). Oladimeji et al. (1984) reported that juvenile rainbow trout exposed for 2, 4, and 6 weeks (a chronic exposure period) to 30 mg/kg of dietary arsenic (as sodium arsenite) had significantly less weight gain than control fish.

Based on the available data, the LOAEL reported in Oladimeji et al. (1984) (30 mg/kg diet) was selected as the LOAEL TRV, representing the lowest reported effect level associated with chronic exposure. Dietary NOAELs ranged from 8 to 52.3 mg/kg dw for growth of juvenile rainbow trout and juvenile striped bass, respectively (Blazer et al. 1997; Cockell et al. 1991). The highest NOAEL (20 mg/kg) that was below the selected LOAEL was selected as the NOAEL TRV. No significant effect on rainbow trout growth was observed in fish fed this dietary concentration.

The results of Oladimeji et al. (1984) indicated slightly greater toxicity of disodium arsenate to juvenile rainbow trout than did studies by Cockell et al. (1991, 1992). Cockell et al. (1991) presented the results of three studies conducted to differentiate effects on growth attributable to arsenic toxicity from those attributable to reduced palatability of arsenic-contaminated food. LOAELs based on Cockell et al. (1991, 1992) ranged from 44 to 65 gm/kg dw. NOAELs based on the same studies range from 8 to 33 mg/kg dw.

A.4.2.2.2 Cadmium

Nine studies that evaluated the effects of dietary cadmium on fish were identified (Baldisserotto et al. 2005; Franklin et al. 2005; Handy 1993; Hatakeyama and Yasuno 1982, 1987; Kang et al. 2005; Kim et al. 2004; Lundebye et al. 1999; Mount et al. 1994; Szebedinsky et al. 2001).[58] Table A.4-9 summarizes the dietary NOAELs and LOAELs for cadmium reported in the reviewed literature.

Table A.4-9. Cadmium dietary toxicity studies for fish

Chemical	Test Species	NOAEL (mg/kg dw)[a]	LOAEL (mg/kg dw)[a]	Exposure Duration	Effect	Source	Notes
Cadmium nitrate	juvenile rockfish	na	**0.5**	60 days	reduced growth rate and condition factor	Kim et al. (2004); Kang et al. (2005)	1, 7
Cadmium chloride	rainbow trout fry	55	na	60 days	no effect on body weight, length, or survival	Mount et al. (1994)	2, 3
Cadmium nitrate	juvenile rockfish	125	na	60 days	no effect on survival	Kim et al. (2004) Kang et al. (2005)	1, 7
Cadmium	guppy	171	na	10 – 30 days	no effect on growth	Hatakeyama and	4, 5

[58] Note that Kim et al. (2004) and Kang et al. (2005) are the same study reported in two separate publications.

Chemical	Test Species	NOAEL (mg/kg dw)[a]	LOAEL (mg/kg dw)[a]	Exposure Duration	Effect	Source	Notes
chloride						Yasuno (1982)	
Cadmium chloride	adult guppy	210	na	2 months	no effect on fry survival or premature embryos	Hatakeyama and Yasuno (1987)	6
Cadmium	Atlantic salmon	250	na	4 weeks	no effect on growth rate (body weight)	Lundebye et al. (1999)	1, 7
Cadmium chloride	guppy (2 months old)	274	na	30 days	no effect on body weight	Hatakeyama and Yasuno (1987)	6
Cadmium chloride	juvenile rainbow trout	294	na	15 – 30 days	no effect on growth rate or survival	Baldisserotto et al. (2005)	7
Cadmium chloride	juvenile rainbow trout	471	na	28 days	no effect on growth rate or survival	Franklin et al. (2005)	
Cadmium chloride	guppy (1 month old)	500	800	7 months	reduced number of fry produced	Hatakeyama and Yasuno (1987)	6, 8
Cadmium chloride	guppy (1 month old)	na	1,250	7 months	reduced female growth and survival	Hatakeyama and Yasuno (1987)	6, 9
Cadmium nitrate	juvenile rainbow trout	786	1,395	30 days	57% survival	Szebedinsky et al. (2001)	7, 10
Cadmium nitrate	juvenile rainbow trout	1,395	2,265	30 days	reduced specific growth rate (weight)	Szebedinsky et al. (2001)	7
Cadmium sulfate	rainbow trout (130 g)	na	10,000	28 days	39% mortality	Handy (1993)	1, 7, 11

[a] Concentrations are for elemental cadmium.

Notes:

1. Only the nominal concentration was reported.
2. Fish were exposed to copper, cadmium, lead, and zinc in water at 23.0, 0.97, 3.32, and 46.3 µg/L, respectively, at the same time as the dietary exposure to cadmium chloride. No effect on growth or survival was observed following exposure; therefore, a dietary NOAEL TRV was obtained from this study.
3. Fish fed live *Artemia* exposed to cadmium chloride in water. Dietary dose corrected for a theoretical 20% loss related to cadmium depuration from the *Artemia* food source.
4. Fish fed live *Moina macrocopa* exposed to cadmium chloride in water.
5. Significant reduction in growth effect was noted on day 10 and recovered at day 20.
6. Fish fed live *Chironomus yoshimatsui* exposed to cadmium chloride in water.
7. Dietary dose was not reported as ww or dw and was assumed to be a dw concentration.
8. Cumulative number of fry produced decreased to about 60% of the control following exposure to 80 and 160 µg/L cadmium chloride. Study did not present statistical significance of data and no replication of treatment was conducted. The LOAEL and NOAEL were estimated using a figure presented in the study.
9. Female body weight decreased to 68% of control on the 48th day of exposure and six of the seven females died in the group exposed to 1,250 mg/kg dw dietary cadmium. No effect on male growth in fish exposed to the same dose. Study did not present statistical significance of data and no replication of treatment was conducted. The LOAEL was estimated using a figure presented in the study.
10. In a separate experiment reported in this study, 92% survival was reported for juvenile rainbow trout exposed to dietary cadmium concentrations of 1,419 mg/kg dw over a 39-day exposure period. Survival data were not statistically analyzed in either experiment. LOAEL – lowest-observed-adverse-effect level.
11. Fish expelled food so the ingested dose is unknown.

na – not available

NOAEL – no-observed-adverse-effect level

Bold identifies the NOAEL and LOAEL selected as TRVs.

LOAELs ranged from 0.5 mg/kg dw for growth of juvenile rockfish fed dietary cadmium for 60 days (Kang et al. 2005; Kim et al. 2004) to 10,000 mg/kg dw for mortality of adult rainbow trout following dietary exposure for 28 days (Handy 1993).

The lowest LOAEL was derived from Kang et al. (2005) and Kim et al. (2004). In this study (reported in two separate publications), juvenile rockfish were treated with 0.5, 5, 25, or 125 mg/kg dw of cadmium as cadmium nitrate for 60 days. Significant effects on growth (identified as condition factor, body weight growth rate, and body length growth rate) were reported for fish exposed to all four dietary concentrations. The lowest LOAEL, 0.5 mg/kg dw, was selected as the LOAEL TRV because it was the most conservative LOAEL reported in the reviewed studies. Reported effects on growth are somewhat uncertain because in one of the two papers in which the results of this study were reported (Kim et al. 2004), the observed growth effect was partially attributed to reduced food intake, which may be the result of food avoidance rather than toxicological effects.

NOAELs ranged from 55 mg/kg dw for rainbow trout fry, as reported in Mount et al. (1994), to 1,395 mg/kg dw for growth of rainbow trout (Szebedinsky et al. 2001). No NOAEL lower than the selected LOAEL was reported, so a NOAEL was estimated by applying an uncertainty factor of 5 to the chronic LOAEL. The resulting NOAEL of 0.1 mg/kg dw was selected.

There was high variability in the toxicological data reviewed for dietary exposure of fish to cadmium. The lowest LOAEL of 0.5 mg/kg dw was two to three orders of magnitude lower than the NOAELs reported in the eight other studies (which ranged from 55 to 1,395 mg/kg dw) and was three to four orders of magnitude lower than the LOAELs reported in the three other studies that reported LOAELs (which ranged from 800 to 10,000 mg/kg dw). The lack of agreement in the toxicological studies results in increased uncertainty associated with the selected TRV.

Of the nine studies available for cadmium, six studies were conducted with salmonid species (Baldisserotto et al. 2005; Franklin et al. 2005; Handy 1993; Lundebye et al. 1999; Mount et al. 1994; Szebedinsky et al. 2001). These studies provide species-specific information for juvenile chinook salmon, which serves as an ROC representing all out-migrating juvenile salmonids. LOAELs were reported in two of these studies, ranging from 1,395 mg/kg dw for mortality of rainbow trout (Szebedinsky et al. 2001) to 10,000 mg/kg dw, also for mortality of rainbow trout (Handy 1993). Salmonid-specific NOAELs were reported in five studies (Baldisserotto et al. 2005; Franklin et al. 2005; Lundebye et al. 1999; Mount et al. 1994; Szebedinsky et al. 2001), ranging from 55 mg/kg dw for growth of rainbow trout (Mount et al. 1994) to 1,395 mg/kg dw, also for growth of rainbow trout (Szebedinsky et al. 2001). These NOAEL TRVs, in particular, suggest that the TRVs selected for fish ROCs in general may result in overestimates of risk for juvenile chinook salmon.[59]

[59] TRVs for salmonids are discussed further in the uncertainty analysis (Section A.6.2.1.2).

A.4.2.2.3 Copper

Fifteen toxicity studies that exposed fish to dietary copper were evaluated for TRV selection (Baker et al. 1998; Berntssen et al. 1999a; 1999b; Gatlin and Wilson 1986; Handy 1992, 1993; Kamunde et al. 2001; Kang et al. 2005; Lanno et al. 1985a; 1985b; Lorentzen et al. 1998; Lundebye et al. 1999; Miller et al. 1993; Mount et al. 1994; Murai et al. 1981). Table A.4-10 summarizes the dietary NOAELs and LOAELs for copper reported in the reviewed literature.

Table A.4-10. Copper dietary toxicity studies for fish

Chemical	Test Species	NOAEL (mg/kg dw)[a]	LOAEL (mg/kg dw)[a]	Exposure Duration	Effect	Source	Notes
Copper sulfate	channel catfish fingerling	8	16	16 weeks	reduced growth	Murai et al. (1981)	1, 2
Copper sulfate pentahydrate	channel catfish fingerling	40	na	13 weeks	no effect on growth	Gatlin and Wilson (1986)	
Copper sulfate	juvenile rockfish	**50**	**100**	60 days	reduced growth rate	Kang et al. (2005)	2
Copper sulfate pentahydrate	Atlantic salmon parr	98	na	12 weeks	no effect on survival or growth	Lorentzen et al. (1998)	
Copper sulfate	rainbow trout (138 g)	200	na	32 days	no effect on survival	Handy (1992)	2
Copper sulfate	juvenile rainbow trout	684	na	42 days	no effect on growth	Miller et al. (1993)	
Copper sulfate pentahydrate	Atlantic salmon parr	691.3	na	4 weeks	no effect on body length, weight, or condition factor	Berntssen et al. (1999b)	
Copper sulfate pentahydrate	Atlantic salmon fry	500	700	3 months	reduced growth	Lundebye et al. (1999)	2
Copper sulfate pentahydrate	juvenile rainbow trout	287	730	8 weeks	reduced growth	Lanno et al. (1985b)	3
Copper sulfate pentahydrate	juvenile rainbow trout	730	na	8 weeks	no effect on survival	Lanno et al. (1985b)	4
Copper sulfate pentahydrate	juvenile rainbow trout	na	796	16 weeks	reduced growth	Lanno et al. (1985a)	
Copper chloride	rainbow trout fry	352	na	60 days	reduced survival	Mount et al. (1994)	5
Copper sulfate	Atlantic salmon fry	638	868	3 months	reduced growth	Berntssen et al. (1999a)	
Copper chloride	rainbow trout fry	800	na	60 days	no effect on body weight or length	Mount et al. (1994)	5
Copper sulfate pentahydrate	juvenile rainbow trout	1,042	na	28 days	no effect on survival or growth	Kamunde et al. (2001)	
Copper sulfate pentahydrate	juvenile grey mullet	na	2,397	67 days	reduced growth	Baker et al. (1998)	6
Copper sulfate	rainbow trout (130 g)	10,000	na	28 days	no effect on survival	Handy (1993)	2

[a] Concentrations are for elemental copper.

Notes:

1. Significant effects on growth were reported following a 4-week dietary exposure to 16 and 32 mg/kg dw. At the NOAEL (8 mg/kg dw), no effect on weight gain was reported.

2. Only the nominal concentration was reported.
3. Significant reduction in growth was reported in fish fed 664 mg/kg dw following 16 weeks of exposure; however, growth reduction was recovered at 24 weeks.
4. Mortality (5.2%) in fish fed 730 mg/kg dw was similar to mortality (3%) reported in fish fed control diet.
5. Fish were fed live *Artemia* previously exposed to copper in water. Dietary concentrations corrected for a theoretical 20% loss resulting from depuration of copper from *Artemia.* Fish were exposed to copper, cadmium, lead, and zinc in water at 23.0, 0.97, 3.32, and 46.3 µg/L, respectively, at the same time as the dietary exposure to copper chloride. No effect on growth or survival was observed following exposure; therefore, a dietary NOAEL TRV was obtained from this study. Increased mortality observed at a dietary concentration of approximately 700 mg/kg dw was attributed to elevated copper concentrations in water; therefore, no dietary LOAEL was identified from this study.
6. Growth effects were associated with a reduction in feeding.

dw – dry weight

LOAEL – lowest-observed-adverse-effect level

na – not available

NOAEL – no-observed-adverse-effect level

Bold identifies the NOAEL and LOAEL selected as TRVs.

Adverse effects on growth or survival were reported in channel catfish, Atlantic salmon, rainbow trout, and grey mullet following exposure to dietary copper; no dietary studies evaluating reproductive effects were found. LOAELs ranged from 16 mg/kg dw for growth of channel catfish (Murai et al. 1981) to 2,397 mg/kg dw for growth of grey mullet (Baker et al. 1998). NOAELs ranged from 8 mg/kg dw for growth of channel catfish (Murai et al. 1981) to 10,000 mg/kg dw for mortality of rainbow trout (Handy 1993).

The lowest LOAEL reported in the reviewed literature was based on Murai et al. (1981). In this study, a significant decrease in body weight was reported for channel catfish fingerlings exposed to 16 mg/kg dw of copper as copper sulfate in a prepared diet for 16 weeks compared to the control group, but a significant reduction in body weight was not observed in fish fed 8 mg/kg dw relative to controls. However, the sensitivity of channel catfish fingerlings documented by Murai et al. (1981) has not been confirmed in subsequent studies using similar exposures and fish of similar age (Erickson et al. 2003; Gatlin and Wilson 1986). Gatlin and Wilson (1986) attempted to reproduce the exposure conditions used by Murai et al. (1981). However, they used larger fingerling catfish (5.5 g/fish versus 1 g/fish in Murai et al.). Gatlin and Wilson (1986) did not report any difference in weight gain in their highest dietary exposure of 40 mg/kg dw. Likewise, Erickson et al. (2003) did not report differences in weight gain following exposure for 30 days to copper-contaminated prey at dietary concentrations of 157 and 246 mg/kg dw using much smaller (0.2 g/fish) fingerling channel catfish. However, that study has not been published in manuscript form and thus was not included in the studies evaluated for TRV derivation. Although the results of Erickson et al. (2003) are not published, they do help bracket the size of fingerlings tested and confirm that the Murai et al. (1981) study results are anomalous. The sensitivity of catfish to copper reported by Murai et al. (1981) has also been characterized as atypical by other studies of copper in fish (Lorentzen et al. 1998).

The next lowest LOAEL was presented in Kang et al. (2005). In that study, juvenile rockfish were exposed to 50, 100, 250, or 500 mg/kg dw of copper as copper sulfate for

60 days. Significant effects on growth (identified as body weight growth rate) were reported for fish exposed to dietary concentrations of 100 mg/kg dw or greater. No adverse effects were observed in fish exposed to 50 mg/kg dw. The NOAEL and LOAEL of 50 and 100 mg/kg dw, respectively, were thus selected as the NOAEL and LOAEL TRVs.

A.4.2.2.4 Vanadium

One study that evaluated the toxicological effects of dietary vanadium to fish (Hilton and Bettger 1988) was identified (Table A.4-11).

Table A.4-11. Vanadium dietary toxicity studies for fish

Chemical	Test Species	NOAEL (mg/kg dw)[a]	LOAEL (mg/kg dw)[a]	Exposure Duration	Effect	Source	Notes
Sodium orthovanadate	juvenile rainbow trout	na	**10.2**	12 weeks	reduced body weight	Hilton and Bettger (1988)	1

[a] Concentrations are for elemental vanadium.

Note:

1. Reduced food intake by fish fed 10.2 mg/kg dw compared to fish fed control diet.

dw – dry weight

LOAEL – lowest-observed-adverse-effect level

na – not available

NOAEL – no-observed-adverse-effect level

Bold identifies the LOAEL selected as the TRV.

Body weights of juvenile rainbow trout fed vanadium were significantly lower than those of the control group following 12 weeks of exposure to 10.2, 80, and 493 mg/kg dw vanadium in a prepared diet (Hilton and Bettger 1988). The LOAEL, 10.2 mg/kg dw, was selected as the LOAEL TRV. Food intake was reduced in fish fed 10.2 mg/kg dw; however, the ratio of food ingestion:body weight was significantly greater for fish in this group than for fish in the control group. Therefore, the reduction in body weight observed at this LOAEL was not attributed to reduced food intake. No NOAEL was reported by Hilton and Bettger (1988); therefore, a NOAEL was extrapolated from the chronic LOAEL using an uncertainty factor of 5. The resulting NOAEL of 2 mg/kg dw was selected as the NOAEL TRV. Because of the paucity of toxicological data on dietary effects of vanadium to fish (only one study was reviewed), there is uncertainty associated with the selected TRVs. Orthovanadate is in a +5 oxidation state, which is more toxic than the +3 redox state. The +5 form is commonly used in toxicity testing; the prevalence of this form in the LDW is unknown.

A.4.3 SUMMARY OF FISH ASSESSMENT

A.4.3.1 Exposure assessment

In Section A.4.1, exposure of fish to COPCs was assessed using one of two approaches depending on the COPC:

- A critical tissue-residue approach was used for COPCs that persist in fish tissue. ROC/COPC pairs evaluated included TBT and total PCBs in Pacific staghorn sculpin tissue and total PCBs in English sole tissue.
- Concentrations in the diet were used for chemicals that are highly regulated or metabolized by fish. COPCs evaluated using this approach include arsenic, cadmium, copper, and vanadium for all fish ROCs.

Whole-body concentrations of TBT and total PCBs were summarized in Table A.4-2 for English sole and Pacific staghorn sculpin.

Concentrations in the diet were estimated based on assumptions regarding ROC-specific prey items and incidental sediment ingestion. Resulting concentrations in the diet were summarized in Tables A.4-4 and A.4-5.

A.4.3.2 Effects assessment

The effects assessment was divided into two sections. Section A.4.2.1 presented the critical tissue-residue approach, and Section A.4.2.2 presented the dietary approach. Each section presented an overview of the available literature from laboratory studies with controlled exposures to COPCs. Ranges of NOAELs and LOAELs from these studies were reviewed, and TRVs were selected. A summary of selected TRVs for COPCs evaluated using the critical tissue-residue approach is presented in Table A.4-12. A summary of selected dietary TRVs is presented in Table A.4-13.

Table A.4-12. TRVs selected for COPCs evaluated using the critical tissue-residue approach

COPC	Test Species	NOAEL (µg/kg ww)	LOAEL (µg/kg ww)	Effect	Source
Total PCBs	common barbel	104 – 528	520 – 2,640	fecundity, egg hatchability, spawning	Hugla and Thome (1999)
TBT	Japanese flounder	18	159	reduced body weight	Shimasaki et al. (2003)

COPC – chemical of potential concern
LOAEL – lowest-observed-adverse-effect level
na – not available
NOAEL – no-observed-adverse-effect level
PCB – polychlorinated biphenyl
TBT – tributyltin
ww – wet weight

Table A.4-13. TRVs selected for COPCs calculated using the dietary approach

COPC	Test Species	NOAEL (mg/kg dw)	LOAEL (mg/kg dw)	Effect	Source
Arsenic	rainbow trout	20	30	reduced body weight	Oladimeji et al. (1984)
Cadmium	rockfish	0.1[a]	0.5	reduced growth rate and condition factor	Kim et al. (2004); Kang et al. (2005)
Copper	juvenile rockfish	50	100	reduced growth rate	Kang et al. (2005)
Vanadium	rainbow trout	2	10.2	reduced body weight	Hilton and Bettger (1988)

[a] NOAEL estimated using an uncertainty factor of 5 (chronic LOAEL to NOAEL).

COPC – chemical of potential concern

dw – dry weight

ROC – receptor of concern

LOAEL – lowest-observed-adverse-effect level

na – not available

NOAEL – no-observed-adverse-effect level

A.5.0 Exposure and Effects Assessment: Wildlife

This section presents the exposure and effects assessment for birds and mammals. In the problem formulation, three bird and two mammal species were selected as ROCs in the LDW: spotted sandpiper, great blue heron, osprey, river otter, and harbor seal. COPCs for these ROCs were selected based on the COPC screen described in Section A.2.5.3, as summarized in Table A.5-1.

Table A.5-1. ROC/COPC pairs evaluated for wildlife receptors

	Receptors of Concern			
COPC[a]	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	
Zinc	X			
Vanadium	X			
PCBs[b]	X	X	X	X

a Total DDTs also screened in as a COPC for spotted sandpiper, great blue heron, and osprey, but will be evaluated in the uncertainty analysis because of uncertainties associated with the exposure data for organochlorine pesticides (see Section A.2.4.2.2).

b PCBs were evaluated in two ways: as total PCBs and as a toxic equivalent (TEQ) for dioxin-like PCB congeners, as discussed in Section A.5.1.1.

COPC – chemical of potential concern

PCB – polychlorinated biphenyl

ROC – receptor of potential concern

This section presents methods and assumptions for estimating doses of COPCs (Section A.5.1) for each ROC, and provides a review of toxicity data identified in the literature (Section A.5.2). Data presented in these sections are synthesized in the risk characterization, and uncertainties are discussed in Section A.6.3.

A.5.1 EXPOSURE ASSESSMENT

This exposure assessment presents the methods used and results of calculated daily exposure doses of COPCs to wildlife receptors in the LDW.

A.5.1.1 Approach

In this assessment, estimates of daily doses of each chemical for each receptor are calculated for three ingested media: food, water, and sediment. Other pathways considered in the conceptual site model in the problem formulation were determined to be insignificant relative to these primary exposure pathways.[60] The daily doses were estimated using the following equation:

$$\text{Daily Dose} = \frac{\left[\left(\text{FIR} \times C_{\text{food}}\right) + \left(\text{WIR} \times C_{\text{water}}\right) + \left(\text{SIR} \times C_{\text{sed}}\right)\right] \times \text{SUF}}{\text{BW}}$$ **Equation 5-1**

Where:

Daily 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)

Site use factors, body weights, and food, water, and sediment ingestion rates vary among ROCs. The body weights and food, water, and sediment ingestion rates were obtained from the literature for each receptor, as described in Section A.5.1.2. Incidental sediment ingestion rates are calculated as a specified percentage of the food ingestion rate for each ROC. As shown in Equation 5-1, food alone represents 100% of the food ingestion rate, with sediment comprising an additional component of the diet. All COPCs were conservatively assumed to have the same bioavailability in the field as in the laboratory toxicity study that provides the basis for the TRV in all media. The chemical concentration in each receptor's diet was calculated from concentrations in each component of the ROC diet and estimates of each component's

[60] Direct (or dermal) contact with sediment was considered a complete exposure pathway. However, risks from sediment contact are considered to be insignificant relative to those from ingestion (EPA 2000b). Direct contact with water was also considered a complete exposure pathway, but also was assumed to be insignificant because feathers on birds and fur on mammals limit direct contact of skin with contaminated media.

fraction of the total diet. For example, the concentration in food for an ROC that might ingest both fish and benthic invertebrates would be estimated as follows:

$$C_{food} = (C_f \times F_f) + (C_a \times F_a) \qquad \textbf{Equation 5-2}$$

Where:

C_{food} = concentration in prey items (mg COPC/kg food dw)
C_f = concentration in fish tissue (mg COPC/kg tissue dw)
C_a = concentration in benthic invertebrate tissue (mg COPC/kg tissue dw)
F_f = fraction of the ROC diet consisting of fish (kg fish/kg food)
F_a = fraction of the ROC diet consisting of benthic invertebrates (kg benthic invertebrates/kg food)

The dietary fraction of each component in each ROC's diet was based on information from the literature. The dietary fractions assumed for each ROC and the assumptions used to derive them are described in detail in Section A.5.1.2.

For PCBs, dietary risks to birds and mammals were evaluated in two ways: as exposure to total PCBs and as exposure to dioxin-like PCB congeners. Dioxin-like PCB congeners show structural and toxic similarities to dioxins and furans. The potency of each individual dioxin-like PCB congener relative to that of 2,3,7,8-tetrachlorodibenzo-*p*-dioxin (2,3,7,8-TCDD) is quantified using TEFs. These factors and the concentrations of each dioxin-like PCB congener are multiplied by the congener-specific TEF, and the products are then summed to calculate a toxic equivalent (TEQ) for the dioxin-like PCB congeners in each sample, as follows:

$$TEQ = \sum_{i=1}^{n} C_i TEF_i \qquad \textbf{Equation 5-3}$$

Where:

TEF_i = the TEF for an individual PCB congener
C_i = concentration of an individual PCB congener

Using the assumption that the combined effect of these PCB congeners is additive, the total TCDD-like toxicity of the PCB congeners is estimated by summing the TEF-concentration products for individual PCB congeners.

The TEFs used in this ERA were developed by the World Health Organization (WHO) in 1998, with updated values for mammals developed in 2005, from a database containing all available mammalian, bird, and fish studies involving the relative toxicity of dioxin-like compounds (Van den Berg et al. 1998; 2006).

The TEQ approach accounts for the toxicity of a subset of PCB congeners that have dioxin-like modes of toxic action. All PCB congeners may elicit other toxic responses with different toxic mechanisms. Therefore, the TEQ approach is not used as a surrogate for the total PCB approach, which captures all PCB toxicity mechanisms or modes of action because it contains multiple classes of congeners including coplanar PCB congeners. It should also be noted that the TEQs calculated in this ERA were used only for assessing the toxicity of the dioxin-like PCB congeners and do not account for TEQ contributions from dioxin and furan congeners; dioxin and furan concentrations in tissue were not available for the LDW. Risk estimates for wildlife based on TEQs calculated on the basis of using only dioxin-like PCB congeners are lower than the actual risk resulting from the cumulative exposure of wildlife to dioxins, furans, and dioxin-like PCB congeners in the LDW; the degree of underestimation is uncertain. Uncertainties in the TEQ approach are discussed in Section A.6.3.1.2.

A.5.1.2 Exposure assumptions

This section presents values used in Equations 5-1 and 5-2 to calculate the daily exposure dose for each ROC, including dietary fractions of prey items, ingestion rates of food, water, and incidental sediment, site use factors, and body weights. Tables A.5-2 and A.5-3 summarize these values and the following sections provide details of exposure factor assumptions and sources of information for each ROC.

Table A.5-2. Dietary fractions of prey items used in wildlife exposure calculations

ROC	COPC	Fraction of Prey Item in Diet (by weight)							
		Shiner Surfperch	English Sole	Juvenile Chinook Salmon	Pacific Staghorn Sculpin	Crab	Benthic Invertebrates	Clams	Mussels
Spotted sandpiper	all COPCs	0	0	0	0	0	1.0	0	0
Great blue heron	total PCBs and mercury	0.24	0.24	0.24	0.24	0.05	0	0	0
	other COPCs	0.32	0.32	na[a]	0.32	0.05	0	0	0
Osprey	total PCBs and mercury	0.34	0.21	0.45	0	0	0	0	0
	other COPCs	0.61	0.39	na[a]	0	0	0	0	0
River otter	total PCBs and mercury	0.22	0.22	0.22	0.22	0.10	0	0.01	0.01
	other COPCs	0.29	0.29	na[a]	0.29	0.10	0	0.01	0.01[b]
Harbor seal	total PCBs and mercury	0.25	0.25	0.25	0.25	0	0	0	0
	other COPCs	0.33	0.33	na[a]	0.33	0	0	0	0

[a] COPCs other than total PCBs and mercury were not analyzed in juvenile chinook salmon; therefore, the dietary fractions of other prey were adjusted to compensate.

[b] Selenium and PCB congeners were not analyzed in mussels, so the clam fraction of the diet was assumed to be 0.02 for selenium and PCB TEQ exposure calculations.

COPC – chemical of potential concern

na – not available

PCB – polychlorinated biphenyl

ROC – receptor of concern

Table A.5-3. Exposure factor values for each ROC

ROC	Gender[a]	Body Weight (kg ww)	Food Ingestion Rate (kg dw/day)	Water Ingestion Rate (kg/day)	Incidental Sediment Ingestion Rate (kg dw/day)	Site Use Factor (unitless)[b]
Spotted sandpiper	M	0.038	0.0060	0.0066	0.0011	1
	F	0.047	0.0074	0.0076	0.0013	
	average	0.043	0.0067	0.0071	0.0012	
Great blue heron	M	2.6	0.11	0.11	0.0022	0.5
	F	2.2	0.093	0.10	0.0019	
	average	2.4	0.10	0.11	0.020	
Osprey	M	1.5	0.076	0.077	0.00076	0.75
	F	1.8	0.091	0.087	0.00091	
	average	1.7	0.083	0.083	0.00083	
River otter	M	9.2	0.30	0.73	0.0060	1
	F	7.9	0.26	0.64	0.0052	
	average	8.6	0.28	0.68	0.0056	
Harbor seal	M	85	0.62	5.4	0.012	0.33
	F	77	0.58	4.9	0.012	
	average	81	0.60	5.1	0.0012	

[a] Female values were used for COPCs with a TRV based on a reproductive endpoint and average values were used for COPCs with a TRV based on a growth or survival endpoint. Average values were used for the COPC screen (Equation 2-2 in Section A.2.5.3).

[b] Site use factor is the fraction of a receptor's total foraging time that is spent in the LDW.

dw – dry weight

M – male

F – female

ROC – receptor of concern

ww – wet weight

A.5.1.2.1 Spotted sandpiper

Body Weight

Representative body weights for adult male and female spotted sandpiper (0.0379 and 0.0471 kg, respectively) were obtained from a study by Maxson and Oring (1980), as cited in EPA (1993).

Food Ingestion Rate

The FIR for spotted sandpiper was estimated as a function of the metabolic rate and the caloric content of the prey using the following equation:

$$\mathrm{FIR} = \frac{\mathrm{FMR}}{\mathrm{ME}} \times \frac{0.001\,\mathrm{kg\ food}}{\mathrm{g\ food}} \qquad \textbf{Equation 5-4}$$

Where:

FIR = food ingestion rate (kg food dw/day)
FMR = free-living metabolic rate (kcal/day)
ME = average metabolizable energy of the total diet (kcal/g food dw)

The body-weight-normalized free-living metabolic rate (FMR) for the common sandpiper (0.676 kcal/g bw/day) from Nagy et al. (1999) was multiplied by male and female spotted sandpiper body weights to derive male and female FMRs used in Equation 5-4 for spotted sandpiper (25.7 and 31.8 kcal/day, respectively). An average metabolizable energy (ME) value of 4.3 kcal/g dw was used in Equation 5-4. This value was derived from Nagy (1987) as the average ME of insects that are ingested by birds. Using these FMR and ME values, the calculated male and female food consumption rates are 0.0060 and 0.0074 kg dw/day, respectively (Table A.5-3).

Water Ingestion Rate

The water ingestion rate (WIR) was estimated as a function of the spotted sandpiper's body weight, using an allometric equation developed by Calder and Braun (1983), as recommended in EPA (1993), as follows::

$$\mathrm{WIR} = 0.059 \times \mathrm{BW}^{0.67}$$ **Equation 5-5**

Where:

WIR = water ingestion rate (L water/day)
BW = body weight (kg)

The calculated male and female WIR values are 0.0066 and 0.0076 L/day, respectively.

Incidental Sediment Ingestion Rate

Site-specific data on incidental sediment ingestion by the spotted sandpiper were not available. However, data were available from Beyer et al. (1994) regarding incidental sediment ingestion by four other sandpiper species that feed on mud-dwelling invertebrates. On a dry-weight basis, the incidental sediment ingestion ranged from 7.3 to 30% of the diet, with an average of 18%, which was used in exposure calculations. The uncertainty in estimated sediment ingestion for spotted sandpiper is addressed in the uncertainty analysis (Section A.6.3.1.2). It was assumed that spotted sandpiper would ingest sediment only from intertidal areas.

Composition of Diet

Spotted sandpipers feed along the sandy or muddy edges of inlets, creeks, and ponds. Their diet is composed primarily of terrestrial and marine invertebrates (Bent 1929), but they also may feed on crustaceans, leeches, mollusks, small fish, and carrion (Oring et al. 1983). In the LDW, it was assumed that spotted sandpipers feed solely on benthic invertebrates, such as amphipods and polychaetes, in the intertidal mudflats along the LDW.

Exposure of spotted sandpipers via ingestion of prey was calculated using benthic invertebrate and amphipod tissue data. The possibility that spotted sandpipers could also ingest fish, mussels, or crab was addressed in the uncertainty assessment in the Phase 1 ERA (Windward 2003b).

Site Use

Spotted sandpiper are a common bird in western Washington, and are known to nest along the LDW. They have been observed in the LDW from late June through September (Cordell et al. 1996), but have been known to overwinter locally (Paulson 1993). Spotted sandpipers breed in open habitats along the margins of water bodies (Oring and Lank 1986). Canning et al. (1979) recorded seven spotted sandpiper nests located on Kellogg Island and at least three additional nest sites were thought to be present on the island based on the behavior of adult or juvenile birds in those areas.

A survey was conducted in June 2004 to identify the presence and quality of spotted sandpiper habitat along the LDW (Windward 2004h). Although no nests were observed during this survey, high-quality nesting habitat was observed at the following areas:

- Fringe of salt marsh on the north side of Kellogg Island (RM 0.7 to RM 0.9)
- T-105 restored salmon rearing channel (western shoreline, RM 0.4)
- Herring's House restored off-channel emergent marsh (western shoreline, RM 0.5)
- Channel beneath First Avenue South Bridge (western shoreline, RM 2.1)
- Hamm Creek restoration site (western shoreline, RM 4.3)
- Western shoreline of the Upper Turning Basin (RM 4.6 to RM 4.9).

The sandpiper survey also found that approximately 65% of the LDW shoreline contained foraging habitat; with about 40% of the shoreline considered to be high-quality habitat and about 25% considered to be poor-quality habitat. The intertidal areas associated with these shoreline segments are identified on Map A.5-1 as high- or poor-quality foraging habitat.

Based on the survey observations of potential nesting and foraging habitat, and an estimated foraging range of about 1 mile from the nest (Norman 2002b), six exposure scenarios for spotted sandpiper exposure were developed. Assuming a nest in the center of each area at RM 0.8, RM 2.1, and RM 4.3, and a foraging range of 1 mile upstream and downstream of the nest, three hypothetical exposure areas were created (Map A.5-1). Within each of the three exposure areas, two foraging scenarios were assessed; one in which spotted sandpipers forage only in 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 areas resulted in a total of six exposure scenarios.

A.5.1.2.2 Great blue heron

Body Weight

Representative body weights of adult male and female great blue heron (2.6 and 2.2 kg, respectively) were obtained from Hartman (1961), as cited in EPA (1993).

Food Ingestion Rate

The FIR values for males and females were calculated using an allometric equation for wading birds (Kushlan 1978):

$$FIR = 10^{0.966\log(BW)-0.64} \times \frac{0.001\ kg}{g}$$ **Equation 5-6**

Where:

FIR = food ingestion rate (kg ww/day)
BW = ROC body weight (g)

The FIR values were converted from wet weight to dry weight assuming a moisture content in prey of 76%, which was the average moisture content in fish collected from the LDW during Phase 2. The body weights and calculated FIR values are presented in Table A.5-3.

Water Ingestion Rate

The WIR was estimated as a function of the great blue heron's body weight, using the allometric equation recommended in *Wildlife Exposure Factors Handbook* (EPA 1993). This equation is the same one presented in Section A.5.1.2.1 (Equation 5-5). WIR values of 0.11 and 0.10 L/day were calculated for male and female great blue heron, respectively.

Incidental Sediment Ingestion Rate

Information on rates of incidental sediment ingestion by herons was not available. Great blue herons may ingest a small amount of sediment while foraging for food. Sediment ingestion was estimated to be 2% of the heron's diet. It was assumed that only intertidal sediment would be ingested.

Composition of Diet

Great blue herons feed in shallow water primarily on fish, but they also ingest crustaceans, insects, amphibians, reptiles, and occasionally small mammals (Kushlan 1978; Butler 1993). Great blue herons hunt by sight and stalk or ambush their prey. They will also feed by probing, quickly moving their bills in and out of the water and substrate. Great blue herons feed on small fish that range in length from 8 to 33 cm (Kirkpatrick 1940; Alexander 1977; Hoffman 1978). Butler (1993) reported that shiner surfperch is a major food source for female and hatchling great blue herons and may be important for juvenile survival. Butler (1997) also reported that sculpin and starry flounder are important prey species for herons along the British Columbia coast, and that shrimp and crabs may also be ingested. The predominant prey items identified at the West Seattle colony were tails from sculpin (Krausmann 2002a). However, sculpin may not be representative of actual prey captured because the large lateral and dorsal spines make them more likely to be rejected by the heron chicks (Krausmann 2002a). In addition to perch and sculpin, herons may feed on other small fish in the LDW, including juvenile salmonids.

For exposure calculations, it was estimated that crabs account for a small portion of the heron's diet (5%), with shiner surfperch, Pacific staghorn sculpin, juvenile chinook salmon, and English sole/starry flounder accounting for equal amounts of the remaining 95 % of the diet.

Site Use

A great blue heron colony of at least 37 nests (presumably more than 74 birds) occupied a site in West Seattle in 1998 (Norman 2002). In 1999, there were only 21 active nests. No successful nesting occurred in 2000 or 2001, and since then, the colony has remained empty (Norman 2002, 2006). It is suspected that the colony was abandoned because of eagle predation or human disturbance from construction at the Herring's House restoration area (Norman 2006). Nevertheless, because of suitable habitat present in the West Seattle area, another colony may be established (Norman 2006). Other colonies are located 12 km south in Renton (Black River colony) and 12 km northwest between the Ship Canal and Discovery Park (Kiwanis Ravine colony). The Black River colony contains about 120 to 135 active nests (Heron Working Group 2006b). The Kiwanis Ravine colony contained 44 nests in 2005, with an estimated 100 young herons successfully fledged (Heron Habitat Helpers 2005).

Information presented in *Wildlife Exposure Factors Handbook* (EPA 1993) indicates that foraging grounds are generally close to breeding colonies, and that 15 to 20 km is the farthest great blue herons might regularly travel from the colony to the foraging area. Herons from the Black River colony are known to forage in the Black River Riparian Forest, along the Cedar River, the Green-Duwamish River, and the Lake Washington shoreline (Heron Working Group 2006b), but the relative time spent foraging in each of these areas is unknown. Studies have not been conducted to determine where herons from the Kiwanis Ravine colony forage, but they have been observed flying in the direction of both saltwater and freshwater during the most intense feeding periods (Heron Habitat Helpers 2005) and great blue herons are regularly observed along the LDW. Based on observations of individual birds from the West Seattle colony when it was active, it was estimated that at least half of the birds from this colony used the LDW to forage, primarily in the Kellogg Island area (Krausmann 2002b). A site use factor of 0.5 was assumed for herons based on observations of the abandoned West Seattle colony; limited foraging information was available for Black River and Kiwanis Ravine herons. The uncertainty of this site use factor is discussed in the uncertainty analysis.

A.5.1.2.3 Osprey

Body Weight

Representative body weights for adult male and female osprey (1.5 and 1.8 kg, respectively) were obtained from Poole (1989), as cited in Poole et al. (2002).

Food Ingestion Rate

The FIR in wet weight was estimated as 21% of the body weight (Poole 1983; as cited in EPA 1993). FIR values were converted from wet weight to dry weight assuming a moisture content in prey of 76%, which was the average moisture content in fish collected from the LDW during Phase 2. FIR values of 0.076 and 0.091 kg dw/day were calculated for male and female osprey, respectively.

Water Ingestion Rate

The WIR was estimated as a function of the osprey's body weight, using the allometric equation recommended in *Wildlife Exposure Factors Handbook* (EPA 1993). This equation is presented in Section A.5.1.2.1 (Equation 5-5). WIR values of 0.077 and 0.087 L/day were calculated for male and female osprey, respectively.

Incidental Sediment Ingestion Rate

Data on incidental sediment ingestion by osprey were not available. Osprey may ingest a small amount of sediment while foraging for fish in shallow water. Sediment ingestion was estimated to be 1% of the osprey's diet. It was assumed that only intertidal sediment would be ingested.

Composition of Diet

Osprey feed almost exclusively on live fish; at least 99% of their prey items were live fish in most published accounts (Poole et al. 2002). Ospreys can penetrate about 1 m below the water surface. Therefore, they generally catch pelagic fish or those that frequent shallow flats and shorelines. Ospreys may infrequently ingest other types of vertebrate prey, such as birds, reptiles, and small mammals. A west-central Idaho osprey study reported 89% of fish ingested by osprey were 11 to 30 cm long, suggesting a preference for medium-sized fish (Van Daele and Van Daele 1982). During a USGS study, osprey were observed while returning to their nests along the LDW with prey. Seventy-three percent of the prey observed were marine fish species (Henny 2005). Salmon, perch, and sole/flounder accounted for approximately 33, 25, and 15% of total prey, respectively. Twenty-five percent of the total prey was peamouth, a freshwater fish, and 2% were not reported. Preliminary data indicate that the size of some of the salmonids taken by osprey from the LDW during the pre-egglaying period ranged from 18 to 30 cm in length (Henny 2006). This dietary information was collected during only a portion of the nesting season. Additional dietary data have been collected and are currently being analyzed but are not yet available (Kaiser 2006). The types of fish ingested by osprey for this assessment were assumed to be the same as those observed in the USGS study, with the proportions reallocated to represent 100% of the diet. Juvenile chinook salmon, shiner surfperch, and English sole/starry flounder were assumed to account for 45, 34, and 21% of the diet, respectively. The juvenile chinook salmon collected in the LDW had an average length of 8 cm, so tissue data for these fish may not represent salmon captured by osprey in the LDW. However, they were used as surrogates for larger juvenile salmon. Uncertainties associated with the composition of the osprey's diet are addressed in the uncertainty analysis (Section 6.3.3.2).

Site Use

There are five known osprey nests along the LDW between RM 0.0 and 5.0, one nest on Harbor Island, and three nests along the Duwamish and Green Rivers within about 4 miles of the Upper Turning Basin. The distance osprey travel from their nests to forage depends on the availability of fish near the nest (Van Daele and Van Daele 1982). Osprey were given a site use factor of 0.75, because preliminary USGS data on prey for LDW osprey indicated that approximately 25% of fish in the diet were freshwater; the remaining 75% marine fish were all assumed to be caught from the LDW.

A.5.1.2.4 River otter

Body Weight

Adult body weights of 9.2 and 7.9 kg were assumed for male and female river otter, respectively (Table A.5-3) based on a study by Melquist and Hornocker (1983), as cited in EPA (1993).

Food Ingestion Rate

The FIR was estimated as a function of the metabolic rate and the caloric content of the prey using the following equation:

$$FIR = \frac{FMR}{ME} \times \frac{0.001\,\text{kg food}}{\text{g food}}$$ **Equation 5-7**

Where:

FIR = food ingestion rate (kg food dw/day)
FMR = free-living metabolic rate (kilocalories [kcal]/day)
ME = average metabolizable energy of the total diet (kcal/g food dw)

The FMRs for males and females were calculated to be 1,340 and 1,180 and kcal/day, using an equation developed by Nagy (1987), as cited in EPA (1993) for placental mammals:

$$FMR\ (kcal/day) = 0.800 \times BW^{0.813}$$ **Equation 5-8**

where body weight is expressed in grams. The ME value used for mammals on a diet of fish was that calculated by Nagy (1987), as cited in EPA (1993) (4.47 kcal/g dw). The calculated FIRs for males and females were 0.30 and 0.26 kg dw/day, respectively (Table A.5-3).

Water Ingestion Rate

The WIR was estimated as a function of the river otter's body weight, using an allometric equation recommended in EPA (1993). This equation was developed by Calder and Braun (1983), as cited in EPA (1993):

$$WIR = 0.099 \times BW^{0.90}$$ **Equation 5-9**

Where:

WIR = water ingestion rate (L water/day)

BW = body weight (kg)

WIR values for males and females (0.73 and 0.64 L/day, respectively) were calculated.

Incidental Sediment Ingestion Rate

Data were not available to estimate the amount of sediment ingested incidentally by river otters. A small amount of sediment could be ingested when river otters forage on crabs and benthic fish species; therefore, the incidental sediment ingestion rate was estimated to be 2% of the diet. It was assumed that river otters incidentally ingest sediment from both intertidal and subtidal areas of the LDW.

Composition of Diet

River otters are opportunistic carnivores that take advantage of food that is most abundant and easiest to catch. Fish are their primary prey (Wise et al. 1981; Kurta 1995; Larsen 1984; Stenson et al. 1984). River otters catch fish by diving and ambushing or chasing, and obtain invertebrates by digging in the substrate (Coulter et al. 1984). Slower-moving fish, such as suckers, carp, chubs, and bullheads, are generally eaten most frequently (Wise et al. 1981; Kurta 1995). Studies in coastal southeast Alaska and British Columbia found that river otters feed primarily on sculpin, surfperch, and flatfish, with greenling, salmon, and rockfish making up lesser portions of the diet (Larsen 1984; Stenson et al. 1984). Other components of the river otter diet include aquatic invertebrates (including crayfish, mussels, clams, and aquatic insects), frogs, snakes, and occasionally mammals and birds (Coulter et al. 1984). River otters generally ingest fish ranging from 7.6 to 41 cm in length (Gilbert and Nancekivell 1982; Greer 1955; both as cited in EPA 1993), although Toweill (1974) found that many of the salmon preyed upon by river otters in western Oregon were up to an estimated 80 cm in length. These salmon were taken in coastal streams where fish enter the rivers to spawn. Local river otters feed largely on fish, but will also feed on crabs and sometimes mussels and clams (Strand 1999).

The proportion of prey types ingested by river otter for this assessment was based on Larsen's (1984) study of river otters in southeastern Alaska. This study was used because it was the only study from the Pacific Northwest that reported remains in scat on a volume basis rather than as a frequency of occurrence. Larsen (1984) reported the following proportions of prey ingested by river otters: 86% fish, 10% crabs, 2% invertebrates other than crabs, 1% birds, and 1% mammals and plant material. Thus, for this assessment, it is assumed that river otters ingest 88% fish, 10% crabs, and 2% mussels. Based on feeding habits of river otters documented in coastal southeast Alaska and British Columbia (Larsen 1984; Stenson et al. 1984), any of the four types of fish tissue for which chemistry data were available in the LDW might be ingested. Because no site-specific information was available on fish preference of river otters, it was assumed that shiner surfperch, English sole/starry flounder, juvenile chinook salmon, and Pacific staghorn sculpin are ingested in equal proportions of the 88% of their diet that is fish. The juvenile chinook salmon collected in the LDW had an average length of 8 cm, so tissue data for these fish may not represent salmon captured by river otters in the LDW. However, they were used as surrogates for larger salmon; uncertainties associated with this assumption are addressed in the uncertainty analysis (Section 6.3.4.2).

Site Use

Anecdotal information indicates that a river otter family lives year-round on Kellogg Island in the LDW, although otters were not observed during wildlife surveys by Cordell (2001). River otters are almost exclusively aquatic and prefer food-rich habitats such as the lower portions of streams and rivers, estuaries, and lakes and tributaries that feed rivers (Tabor and Wight 1977; Mowbray et al. 1979). In streams, the river otter's home range can average 30 km (19 mi) (Melquist and Hornocker 1983). At any given time, river otters generally occupy only a few kilometers of stream, but often move from one area to another (Nebraska Game and Parks Commission 2000). A radio-tracking study of relocated river otters was conducted as part of the New York State Department of Environmental Conservation river otter reintroduction program. This study showed that river otter ranges were from 1.5 to 22.4 km long, with an average length of 10 km (6 mi) for individuals monitored in western New York State (Spinola et al. 1999; as cited in EPA 2000d).

No studies were found that document usage of the LDW by river otters. Because of the average 10 km linear length documented in the literature, and because the extent of the LDW study area is approximately 10 km, it was assumed that river otters could potentially ingest prey solely from within the LDW. Therefore, a site use factor of 1.0 was assumed.

A.5.1.2.5 Harbor seal

Body Weight

Body weights for adult male and female harbor seals (84.6 and 76.5 kg, respectively) were based on a study by Pitcher and Calkins (1979), as cited in the *Wildlife Exposure Factors Handbook* (EPA 1993).

Food Ingestion Rate

The FIR for harbor seals was calculated using an allometric equation developed by Boulva and McLaren (1979) for harbor seals from eastern Canada, as cited in the *Wildlife Exposure Factors Handbook* (EPA 1993):

$$FIR = 0.089 \times BW^{0.76} \qquad \textbf{Equation 5-10}$$

Where:

FIR = food ingestion rate (kg food ww/day)
BW = ROC body weight (kg)

The calculated wet weight FIR values were converted to dry weight using the average moisture content (76%) in whole-body fish from the LDW. FIRs calculated for males and females were 0.62 and 0.58 kg dw/day, respectively (Table A.5-3).

Water Ingestion Rate

The WIR was estimated as a function of the harbor seal's body weight, using the allometric equation recommended in EPA (1993). This equation is presented in Section A.5.1.2.4 (Equation 5-9). Using the male and female body weights of the harbor seal, the calculated WIR values were 4.9 and 5.4 L/day, respectively.

Incidental Sediment Ingestion Rate

Data on incidental sediment ingestion by harbor seals were not available, but it is possible that a small amount of sediment could be incidentally ingested while foraging on bottom fish. Therefore, the sediment ingestion rate was assumed to be 2% of the FIR. It was assumed that harbor seals ingest sediment from both intertidal and subtidal areas of the LDW.

Composition of Diet

Harbor seals are opportunistic feeders, selecting prey based on availability and ease of capture (Pitcher and Calkins 1979; Pitcher 1980; Schaffer 1989). Their diet can vary seasonally with local abundance and includes bottom-dwelling fishes, invertebrates such as octopus and squid, and species that congregate for spawning (Pitcher and Calkins 1979; Everitt et al. 1981; Lowry and Frost 1981; Roffe and Mate 1984). In Washington, the most important prey include Pacific whiting, tomcod, walleye pollock, flatfish, Pacific herring, shiner surfperch, plainfin midshipman, and sculpin (NMFS 1997). Fish ingested were generally between 4 and 28 cm in length (Brown and Mate 1983). Harbor seals may also prey on salmon during upriver spawning migrations of adults or downriver outmigrations of juveniles, although site-specific data were not available on the dietary importance of migrating salmon to local seal populations. Because site-specific information was not available on the amount of each type of prey ingested, it is assumed that juvenile chinook salmon, English sole/starry flounder, shiner surfperch, and Pacific staghorn sculpin are ingested in equal proportions. The juvenile chinook salmon collected in the LDW had an average length of 8 cm, so they may or may not be representative of salmon ingested by harbor seals in the LDW. However, they are used as surrogates for larger salmon; the absence of specific information on types of prey ingested by harbor seals is discussed in the uncertainty analysis (Section A.6.3.5.2).

Site Use Factor

Harbor seals are commonly seen in Elliott Bay and occasionally enter the LDW (Kenney 1982). Harbor seals have been shown to forage over large distances, ranging from 5 km (3 mi) (Stewart et al. 1989) to 55 km (34 mi) (Beach et al. 1985). In Puget Sound, harbor seals generally forage within 8 to 13 km (5 to 8 mi) of their haulout areas established as pupping sites (Jeffries 2001). The closest known pupping site to the LDW is located at Blakely Rocks off the southeast end of Bainbridge Island, approximately 12 km (7 mi) from the LDW. Site-specific information on harbor seal usage of the LDW is limited. The WDFW observed harbor seals infrequently in the LDW during an intensive survey conducted from December 1998 to June 1999 (WDFW 1999). This survey monitored the waterway for the presence of sea lions and seals up to the 16th Avenue South Bridge for a total of 307 hours on 52 days. Harbor seals were observed on 17 occasions, and were most frequently seen north of the 1st Avenue South Bridge. While harbor seals have been observed in Elliott Bay and may use log booms to haul out, they are not known to aggregate in large numbers (Jeffries 2001). The LDW may be a preferential feeding area during salmonid outmigration from March through August. For example, in the Columbia River, salmonids appear to be targeted as prey by seals in the spring and fall when they are abundant and available in the river (NMFS 1997).

Data from the WDFW survey discussed above were used to establish a site use factor for risk calculations. The following conservative assumptions were used: 1) a harbor seal was observed in the LDW on 17 of 52 days of observation, and it was assumed that all observations were of a single seal; 2) the seal was assumed to obtain all of its food for those days in the LDW; and 3) site usage from December through June accurately represents usage during other times of the year. Based on these assumptions, the site use factor was equal to 17/52 or 0.33. This approach does not include exposure of harbor seals in Elliott Bay to fish that may have been exposed to chemicals in the LDW or through a food web connected to the LDW.

A.5.1.3 Prey tissue, sediment, and water data

This section presents the COPC concentrations in prey tissue, sediment, and water that were used in Equations 5-1 and 5-2 to calculate exposure doses. Attachment 11 presents summary statistics (i.e., minimum, maximum, and mean COPC concentrations) for the prey tissue, sediment, and water data.

A.5.1.3.1 Prey tissue

Tissue data were available for eight tissue types that are potential prey of wildlife ROCs in the LDW: shiner surfperch, English sole, Pacific staghorn sculpin, juvenile chinook salmon, benthic invertebrates,[61] clams, crabs, and mussels. These data are described in Section A.2.4.1.2.

For all wildlife ROCs except spotted sandpiper, it was assumed that the foraging area includes the entire LDW based on home range information presented in Section A.5.1.2. Thus, for each prey type ingested by those species (i.e., all prey types except benthic invertebrates), the exposure concentration was calculated using all tissue data from the LDW as one exposure data set. The exposure concentration for each tissue type was estimated as the UCL,[62] which was calculated as described in Attachment 11. For example, for great blue heron, all Pacific staghorn sculpin composite tissue samples collected in the LDW were used to estimate the UCL on the mean for this prey type. The calculated concentrations for COPCs in each tissue type are shown in Table A.5-4.

Table A.5-4. COPC concentrations in tissues of prey species ingested by great blue heron, osprey, river otter, and harbor seal

COPC	Tissue Concentration in Prey Species (mg/kg dw)						
	Shiner Surfperch	English Sole	Juvenile Chinook Salmon	Pacific Staghorn Sculpin	Clams	Crabs	Mussels
Arsenic	4.2	13	na	4.0	25	25	5.9
Chromium	0.82	4.4	na	0.40	5.2	0.30	1.2
Cobalt	0.18	0.41	na	0.13	2.9	0.39	0.41
Lead	0.53	1.7	na	0.30	19	0.53	3.3
Mercury	0.15	0.070	0.14	0.15	0.12	0.29	0.094
Selenium	0.77	0.82	na	0.87	2.0	1.2	na
Total PCBs	14	10	3.4	5.0	4.0	6.0	0.27
PCB TEQs (birds)[a]	7.2×10^{-4}	4.2×10^{-4}	na	1.2×10^{-4}	1.5×10^{-4}	3.6×10^{-4}	na
PCB TEQs (mammals)[b]	1.6×10^{-4}	8.2×10^{-5}	na	3.8×10^{-5}	2.7×10^{-5}	5.0×10^{-5}	na

[a] PCB TEQs were calculated using TEFs for birds presented in Van den Berg et al. (1998).These TEFs are listed in Attachment 3, and uncertainties associated with application of these TEFs are discussed in Section A.6.3.1.

[b] PCB TEQs were calculated using TEFs for mammals presented in Van den Berg et al. (2006). These TEFs are listed in Attachment 3.

COPC – chemical of potential concern

dw – dry weight

na – not available

PCB – polychlorinated biphenyl

TEQ – toxic equivalent

[61] Includes amphipod samples collected for the King County Water Quality Assessment (King County 1999d) and benthic invertebrate samples collected for the Phase 2 RI (Windward 2005b).

[62] The UCL was used even if it was higher than the maximum detected concentration.

For spotted sandpipers, which forage over an area smaller than the entire LDW (see Section A.5.1.2.1), concentrations of COPCs in the fraction of the diet consisting of benthic invertebrate tissues were calculated separately for each of six exposure scenarios described in Section A.5.1.2.1. For COPCs with a linear relationship between concentrations in sediment and tissue (i.e., arsenic and total PCBs), the COPC concentrations in benthic invertebrate tissue were estimated using a linear regression, as described in Section 4.0 of Attachment 11. For arsenic, the arithmetic average surface sediment concentration within each intertidal exposure area was used to estimate the benthic invertebrate tissue concentration for each exposure area using the linear regression. For total PCBs, spatially weighted average concentrations (SWACs) for surface sediments have been developed for each exposure area, so these SWACs were used to estimate the total PCB benthic invertebrate tissue concentrations from the linear regression. The SWACs were used for total PCBs because the arithmetic mean of the sediment concentrations in a specified area may be biased high if more highly contaminated areas were more intensively sampled than less-contaminated areas. In the LDW, this bias is most pronounced for total PCBs, which are the focus of several ongoing early action investigations in the LDW. The 95th UCLs on the total PCB and arsenic tissue concentrations estimated from the linear regression were used as the benthic invertebrate concentrations in the exposure assessment (Table A.5-5). The statistical methods used to calculate the benthic tissue concentrations are described in detail in Attachment 11. The locations of intertidal sediment samples included in each exposure scenario are shown in Map A.5-1.

For the remaining COPCs, a significant statistical correlation was not observed between benthic invertebrate tissue and sediment. Therefore, concentrations of these COPCs in benthic invertebrate tissue were calculated as the UCL of the available tissue data collected from each sandpiper intertidal exposure area to estimate spotted sandpiper exposure. Locations of benthic invertebrate tissue samples included in these calculations are shown on Map A.5-1. COPC concentrations in benthic invertebrate tissues that were used in the spotted sandpiper exposure calculations are summarized in Table A.5-5.

Table A.5-5. COPC concentrations in benthic invertebrate tissues used to estimate spotted sandpiper exposure

COPC	Tissue Concentration in Benthic Invertebrates (mg/kg dw)					
	Exposure Area 1 [a]		Exposure Area 2[a]		Exposure Area 3 [a]	
	High-Quality Foraging Habitat	High- and Poor-Quality Foraging Habitat	High-Quality Foraging Habitat	High- and Poor-Quality Foraging Habitat	High-Quality Foraging Habitat	High- and Poor-Quality Foraging Habitat
Arsenic	23	21	25	21	18	26
Cadmium	0.56	0.56	0.40	0.49	0.64	0.64
Chromium	3.0	3.0	18	51	5.0	5.0
Cobalt	1.8	1.8	2.0	2.2	2.2	2.2
Copper	120	120	140	140	70	70
Lead	31	31	15	660	5.6	5.6
Mercury	0.10	0.10	0.066	0.070	0.50	0.50
Nickel	3.5	3.5	5.8	5.3	7.4	7.4
Selenium	1.9	1.9	1.6	1.8	1.4	1.4
Zinc	190	190	300	380	270	270
Vanadium	6.4	6.4	10	9.8	9.6	9.6
BEHP	2.9	2.9	14	14	61	61
Total PCBs	1.5	1.5	5.9	3.8	2.7	3.4
PCB TEQs (bird)[b]	1.6×10^{-4}	1.6×10^{-4}	5.6×10^{-4}	5.6×10^{-4}	3.9×10^{-4}	3.9×10^{-4}

[a] Six exposure scenarios were evaluated; in each of three exposure areas, foraging only in high-quality habitats and foraging in both high- and poor-quality habitats were evaluated. These exposure scenarios are described in detail in Section A.5.1.2.1.

[b] PCB TEQ concentrations were calculated using TEFs for birds based on TEFs presented in Van den Berg et al. (1998). These TEFs are listed in Attachment 3, and uncertainties associated with application of these TEFs are discussed in Section A.6.3.1.

BEHP – bis(2-ethylhexyl) phthalate

COPC – chemical of potential concern

dw – dry weight

PCB – polychlorinated biphenyl

TEQ – toxic equivalent

A.5.1.3.2 Sediment

Surface sediment data were used to estimate COPC exposure resulting from incidental sediment ingestion. For river otter and harbor seal, it was assumed that the foraging area includes the entire LDW; thus, COPC concentrations in sediment were calculated as the UCL of all surface sediment data.[63] For osprey and great blue heron, it was assumed that the foraging area includes only intertidal areas (areas exposed during

[63] The UCL was used even if it was higher than the maximum detected concentration.

low tide) of the LDW; thus, sediment COPC concentrations were calculated as the UCL of only intertidal surface sediment data. For spotted sandpiper, surface sediment COPC concentrations were calculated as the UCL for each of the six intertidal exposure scenarios discussed in Section A.5.1.2.1 (Map A.5-1).

For PCBs, the sediment concentration was calculated as the UCL of the SWAC concentration for surface sediments discussed above. Statistical methods for calculating this UCL are presented in Attachment 11. For other COPCs, the concentration in sediment was calculated as the UCL of the point data. The calculated concentrations in sediment are presented in Tables A.5-6 (for all ROCs, except spotted sandpiper) and Table A.5-7 (for spotted sandpiper).

Table A.5-6. COPC concentrations in LDW sediment used to estimate exposure of wildlife ROCs, except spotted sandpiper

COPC	Sediment Concentration (mg/kg dw)	
	Entire LDW[a]	Intertidal Areas[b]
Arsenic	30	na
Chromium	50	70
Cobalt	10	na
Lead	300	600
Mercury	0.30	0.40
Selenium	5.0	na
Total PCBs	0.72	0.98
PCB TEQs (bird)[c]	5.4×10^{-4}	9.2×10^{-4}
PCB TEQs (mammal)[d]	7.2×10^{-5}	na

[a] Used for exposure calculations for river otter and harbor seal.

[b] Used for exposure calculations for osprey and great blue heron.

[c] PCB TEQ concentrations were calculated using TEFs for birds as presented in Van den Berg et al. (1998). These TEFs are listed in Attachment 3, and uncertainties associated with application of these TEFs are discussed in Section A.6.3.1.

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

COPC – chemical of potential concern

dw – dry weight

LDW – Lower Duwamish Waterway

na – not applicable; not a COPC for ROCs using intertidal areas (i.e., great blue heron and osprey) based on the COPC screen presented in Section A.2.5.3.

Table A.5-7. COPC concentrations in sediment used to estimate spotted sandpiper exposure

COPC	Sediment Concentration (mg/kg dw)					
	Exposure Area 1[a]		Exposure Area 2[a]		Exposure Area 3[a]	
	High-Quality Foraging Habitat	High- and Poor-Quality Foraging Habitat	High-Quality Foraging Habitat	High- and Poor-Quality Foraging Habitat	High-Quality Foraging Habitat	High- and Poor-Quality Foraging Habitat
Arsenic	36	18	49	18	20	60
Cadmium	0.45	0.65	1.0	0.93	0.62	7.8
Chromium	32	30	43	32	31	120
Cobalt	9.0	8.0	9.5	8.0	8.0	10
Copper	94	80	120	79	63	730
Lead	90	90	160	96	90	1,000
Mercury	0.17	0.16	0.43	0.29	0.60	0.40
Nickel	21	30	22	20	22	90
Selenium	0.59	0.56	7.2	3.0	7.0	6.6
Zinc	180	170	240	180	150	710
Vanadium	53	52	58	55	57	59
BEHP	0.12	2.5	2.7	2.1	0.41	0.41
Total PCBs	0.34	0.33	2.5	1.5	0.72	1.1
PCB TEQs (bird)[b]	3.7×10^{-5}	3.4×10^{-5}	4.3×10^{-3}	1.8×10^{-3}	9.6×10^{-5}	3.0×10^{-3}

a Six exposure scenarios were evaluated; in each of three exposure areas, foraging only in high quality habitat and foraging in both high and poor quality habitat were evaluated. These exposure scenarios are described in detail in Section A.5.1.2.1.

b PCB TEQs were calculated using TEFs for birds as presented in Van den Berg et al. (1998). These TEFs are listed in Attachment 3, and uncertainties associated with application of these TEFs are discussed in Section A.6.3.1.

BEHP – bis(2-ethylhexyl) phthalate

COPC – chemical of potential concern

dw – dry weight

PCB – polychlorinated biphenyl

TEQ – toxic equivalent

A.5.1.3.3 Water

Water data used to estimate exposure of wildlife to COPCs in the LDW were collected at three locations in 1996 and 1997 as part of the King County WQA (King County 1999d) and at two locations during an additional sampling event by King County in 2005, in which water data were collected for PCB congeners only (Mickelson and Williston 2006), as described in Section A.2.4.1.3. Surface water ingestion represents a very small proportion of the exposure of wildlife to COPCs in the LDW (e.g., less than 0.01% of the overall dose of PCBs is from water).[64]

Metals and BEHP

Metals and BEHP[65] were the only chemicals detected in water samples collected along transects from the three LDW locations sampled as part of the WQA: RM 1.1 in the vicinity of the Brandon CSO, RM 1.9 in the vicinity of the Southwest Michigan CSO, and RM 4.9 in the vicinity of the Norfolk CSO. The Brandon CSO and Southwest Michigan CSO transects contained three sampling locations and the Norfolk CSO transect contained two sampling locations. Each location was sampled at 1 m below the surface and 1 m above the bottom. For estimates of exposure to ROCs expected to forage throughout the LDW (great blue heron, osprey, river otter, and harbor seal), data from all locations and depths were combined into one dataset and each data point was treated equally. The exposure concentration for each metal COPC was estimated as the UCL of the combined data, as presented in Table A.5-8.

Exposure concentrations of total DDTs in water were not calculated because of the uncertainties in predicting organochlorine pesticide concentrations using data from semi-permeable membrane devices. The contribution of COPCs from water to the total exposure dose is very small and is not expected to affect risk calculations (e.g., the contribution of total PCBs via water exposure was less than 0.01 percent of the contribution from food and sediment exposure).

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[64] There is some uncertainty associated with exposure to PCBs through water. Exposure to PCBs in water was based on a summation of PCB congener concentrations in grab samples collected from the channel at two locations in the LDW; these samples were collected by King County in 2005. It is very unlikely that risk estimates would be affected by the uncertainty associated with this limited sampling because ingestion of PCBs through water is less than 0.01% of the amount ingested from food and sediment.

[65] BEHP was detected in laboratory blank water samples; therefore, BEHP concentrations in water are likely overestimated.

Table A.5-8. Concentrations of metals and BEHP in surface water used to estimate exposure of great blue heron, osprey, river otter, and harbor seal

COPC	LDW-Wide Concentration in Water (mg/L)
Arsenic	8.8×10^{-4}
Cadmium	4.7×10^{-5}
Chromium	7.1×10^{-4}
Cobalt	2.7×10^{-4}
Copper	1.6×10^{-3}
Lead	5.0×10^{-4}
Mercury	2.7×10^{-6}
Nickel	8.2×10^{-4}
Selenium[c]	2.7×10^{-4}
Vanadium	1.4×10^{-3}
Zinc	3.2×10^{-3}
BEHP	2.1×10^{-3}

BEHP – bis(2-ethylhexyl) phthalate

COPC – chemical of potential concern

For spotted sandpiper, water data collected at the Brandon, Southwest Michigan, and Norfolk locations were used for Exposure Areas 1, 2, and 3, respectively. The UCL was based upon the available data from each location (Table A.5-9).

Table A.5-9. Concentrations of metals and BEHP in surface water used to estimate exposure of spotted sandpiper

COPC	Concentration in Water (mg/L)		
	Exposure Area 1	Exposure Area 2	Exposure Area 3
Arsenic	9.7×10^{-4}	9.3×10^{-4}	5.3×10^{-4}
Cadmium	5.4×10^{-5}	5.5×10^{-5}	1.6×10^{-5}
Chromium	6.1×10^{-4}	7.3×10^{-4}	1.1×10^{-3}
Cobalt	2.2×10^{-4}	2.5×10^{-4}	4.6×10^{-4}
Copper	1.5×10^{-3}	1.6×10^{-3}	2.3×10^{-3}
Lead	3.8×10^{-4}	4.9×10^{-4}	8.9×10^{-4}
Mercury	2.0×10^{-6}	2.0×10^{-4}	4.3×10^{-6}
Nickel	7.1×10^{-4}	8.2×10^{-4}	1.3×10^{-3}
Selenium	2.7×10^{-4}	1.6×10^{-4}	1.6×10^{-4}
Zinc	3.0×10^{-3}	3.0×10^{-3}	4.6×10^{-3}
Vanadium	1.4×10^{-3}	1.5×10^{-3}	1.9×10^{-3}
BEHP	3.5×10^{-3}	2.4×10^{-2}	1.8×10^{-4}

BEHP – bis(2-ethylhexyl) phthalate

COPC – chemical of potential concern

PCBs

PCBs were analyzed in water samples collected from two locations: RM 0.0 (LTKE03) and RM 3.3 (LUTM03), as described in Section A.2.4.1.3. For the ROCs expected to forage throughout the LDW (great blue heron, osprey, river otter, and harbor seal), all data from both locations were combined into one dataset. The PCB water exposure concentration for these four ROCs was estimated as the UCL of these combined data, as presented in Table A.5-10.

Table A.5-10. Total PCB concentrations in surface water used to estimate exposure of wildlife ROCs

COPC	Concentration in Water (mg/L)		
	Osprey, Great Blue Heron, River Otter, and Harbor Seal	Spotted Sandpiper	
	LDW-Wide	Exposure Area 1	Exposure Areas 2 and 3
Total PCBs	1.6×10^{-6}	1.7×10^{-6}	1.9×10^{-6}

COPC – chemical of potential concern

LDW – Lower Duwamish Waterway

PCB – polychlorinated biphenyl

ROC – receptor of concern

For spotted sandpiper, PCB data from the LTKE03 location were used for Area 1, and PCB data from the LUTM03 location were used for Areas 2 and 3. The UCLs were based on all data from each location, as presented in Table A.5-10.

PCB TEQs in water were not calculated because PCB exposure via water is a negligible component of overall PCB exposure, based on total PCB exposure calculations. The contribution of total PCBs via water exposure was less than 0.01% of the contribution from food and sediment exposure.[66]

A.5.1.4 Estimated dietary doses

Exposures as dietary doses were estimated for each wildlife ROC based on the information presented in preceding sections. Estimated dietary exposures for spotted sandpiper are presented in Table A.5-11, and those for great blue heron, osprey, river otter, and harbor seal are presented in Table A.5-12. Tables containing all data used in these calculations for each COPC/ROC pair are presented in Attachment 12.

[66] There is some uncertainty in not including undetected PCB congeners in the calculation of total PCBs, but the contribution of total PCBs via water exposure would remain less than 0.01% if undetected congeners were included at one-half the reporting limit; therefore, uncertainty associated with undetected PCB congeners is very low.

Table A.5-11. Estimated dietary doses of COPCs for spotted sandpiper

COPC	Dietary Dose (mg/kg bw/day)					
	Exposure Area 1[a]		Exposure Area 2[a]		Exposure Area 3[a]	
	High-Quality Foraging Habitat	High- and Poor-Quality Foraging Habitat	High-Quality Foraging Habitat	High- and Poor-Quality Foraging Habitat	High-Quality Foraging Habitat	High- and Poor-Quality Foraging Habitat
Arsenic	4.6	3.8	5.3	3.8	3.4	5.8
Cadmium	0.10	0.11	0.090	0.10	0.12	0.32
Chromium	1.4	1.3	4.0	8.8	1.6	4.1
Cobalt	0.53	0.50	0.58	0.57	0.57	0.62
Copper	21	21	25	24	13	31
Lead	7.4	7.4	6.8	110	3.4	29
Mercury	0.020	0.020	0.022	0.019	0.095	0.090
Nickel	1.1	1.4	1.5	1.4	1.8	3.7
Selenium	0.32	0.31	0.45	0.37	0.41	0.40
Zinc	35	34	53	64	46	62
Vanadium	2.5	2.4	3.2	3.1	3.1	3.1
BEHP	0.46	0.53	2.3	2.3	9.6	9.6
Total PCBs	0.25	0.25	1.0	0.64	0.45	0.57
PCB TEQs	2.6×10^{-5}	2.6×10^{-5}	2.1×10^{-4}	1.4×10^{-4}	6.4×10^{-5}	1.4×10^{-4}

[a] Six exposure scenarios were evaluated; in each of three exposure areas, foraging only in high quality habitats and foraging in both high and poor quality habitats were evaluated. These exposure scenarios are described in detail in Section A.5.1.2.1.

BEHP – bis(2-ethylhexyl) phthalate

bw – body weight

COPC – chemical of potential concern

PCB – polychlorinated biphenyl

TEQ – toxic equivalent

Table A.5-12. Dietary exposure doses of COPCs for great blue heron, osprey, river otter, and harbor seal

COPC	Dietary Exposure Dose (mg/kg bw/day)			
	Great Blue Heron	Osprey	River Otter	Harbor Seal
Arsenic	ne	ne	0.32	ne
Chromium	0.067	0.11	ne	ne
Cobalt	ne	ne	0.016	ne
Lead	0.27	0.26	ne	ne
Mercury	0.0031	0.0051	0.0048	3.3×10^{-4}
Selenium	ne	ne	0.032	ne
Total PCBs	0.17	0.32	0.26	0.020
PCB TEQs	9.3×10^{-6}	2.3×10^{-5}	6.4×10^{-6}	5.1×10^{-7}

bw – body weight

COPC – chemical of potential concern

ne – not evaluated; not a COPC for this ROC

PCB – polychlorinated biphenyl

TEQ – toxic equivalent

A.5.2 Effects Assessment

This section presents a summary of the toxicity literature for each of the COPCs and describes the selection of TRVs for birds and mammals. The literature search and guidelines for TRV selection for wildlife ROCs was described in detail in Section A.2.3.3. Toxicological data presented in this section are assessed in combination with exposure data (presented in Section A.5.1) in the risk characterization (Section A.6.3).

In many of the studies reviewed, the toxicity data were presented as a concentration in the diet rather than as an ingested dose. These dietary concentrations were converted to ingested doses using the toxicity test species-specific FIRs and body weights, as follows:

$$\text{ingested dose} = \frac{(C_{food} \times FIR)}{BW} \qquad \textbf{Equation 5-11}$$

Where:

ingested dose	=	COPCs ingested per day via food (mg COPC/kg body weight/day)
FIR	=	food ingestion rate (kg food dw/day)
C_{food}	=	COPC concentration in dietary items (mg COPC/kg food dw)
BW	=	test species body weight (kg ww)

If the FIR and no-effect or effect concentrations were in different units (i.e., wet weight vs. dry weight), the units of the no-effect or effect concentrations were converted to the units of the FIR, using the moisture content in food as presented in table footnotes in this section. The values for FIR, BW, and food moisture content were obtained from the specific toxicity study, if available.

A.5.2.1 TRVs for birds

This section presents results from laboratory toxicity studies for COPCs identified for bird ROCs (Table A.5-1), and selects TRVs for estimating risks. TRVs for PCBs were selected for both total PCBs (generally based on Aroclors) and for 2,3,7,8-TCDD (as TEQs for dioxin-like PCB congeners). Toxicity data are not sufficient to develop unique TRVs for each of the bird species used as ROCs in this risk assessment. Therefore, a single NOAEL and LOAEL TRV was selected for each COPC and used for all bird ROCs.

A.5.2.1.1 Total PCBs

Effects on birds from exposure to dietary PCBs include disruption of normal patterns of growth, reproduction, metabolism, and behavior (Eisler 1986b). The most sensitive effects are related to reproduction, and include egg production, fertility, and hatching success (Eisler 1986b). Of the bird laboratory species used to examine reproductive endpoints, chickens and other galliformes, such as pheasant and quail, have been found to be the most sensitive to PCB toxicity (Kennedy et al. 1996). Because of concerns with poultry laboratory studies, only data from wildlife laboratory studies were considered in choosing a bird PCB TRV. This approach is consistent with an EPA-sponsored peer review panel charged with reviewing an ERA for the Hudson River. This panel evaluated use of PCB TRVs derived from chicken studies to assess risks to wild birds. Reviewers considered PCB TRVs developed from chicken studies to be “unrealistically low and excessively conservative” and found that “using the chicken as a representative species for wild birds was not defensible” (EPA 2000b). The use of chicken reproductive toxicity data to assess risks to ROCs should be considered protective, but it is not likely to predict risks accurately. Therefore, chicken toxicity data for reproductive endpoints were not used in this ERA.

Thirteen studies were identified that evaluated the dietary toxicity of PCBs to birds (Table A.5-13). Various species were studied, including American kestrels, screech owls, turtle doves, Japanese quail, and mallard ducks. All studies reviewed involved the assessment of reproductive endpoints following dietary exposure to PCBs. These endpoints included fertility, hatchability, eggshell thickness, egg production, eggshell weight, embryo development, clutch size, and embryo mortality and viability.

Table A.5-13. PCB dietary toxicity studies for birds

Chemical	Test Species	NOAEL (mg/kg bw/day)	LOAEL (mg/kg bw/day)	Exposure Duration	Effect	No-Effect Conc. (mg/kg ww)[a]	Effect Conc. (mg/kg ww)[a]	Body Weight (kg)	Food Ingestion Rate[b]	Source	Notes
Aroclor 1248	American kestrel	na	0.35	5.5 months	reduced eggshell thickness	na	3	0.13 (Pattee 1984)	0.0136 kg dw/day, Eurasian kestrel, Nagy (2001)	Lowe and Stendell (1991)	1
Aroclor 1248	screech owl	**0.49**	na	two generations	no effect on eggshell thickness, egg production, or hatching and fledging success	3	na	0.181 (Dunning 1993)	0.0266 kg dw/day, carnivorous birds, Nagy (2001)	McLane and Hughes (1980)	1
Aroclor 1242	Japanese quail	na	0.60	45 days	reduced eggshell thickness	na	10	0.09 (Dunning 1993)	0.0048 kg dw/day, galliformes, Nagy (2001)	Hill et al. (1976)	1
Aroclor 1254	ringed turtle dove	na	**1.4**	two generations	reduced hatching success in second generation	na	10	0.155 (Sample et al. 1996)	0.0202 kg dw/day, all birds, Nagy (2001)	Peakall et al. (1972); Peakall and Peakall (1973)	2
Aroclor 1254	mallard	2.5	na	~ 1 month	no effect on reproductive success	25	na	1.082 (Dunning 1993)	0.1082 kg ww/day (Heinz et al. 1987)	Custer and Heinz (1980)	
Aroclor 1254	mallard	3.9	na	4 months	no effect on egg production or eggshell thickness	39	na	1.082 (Dunning 1993)	0.1082 kg ww/day (Heinz et al. 1987)	Risebrough and Anderson (1975)	
Aroclor mixture	American kestrel	na	7	100 days until eggs hatched	reduced egg laying in second generation (exposed *in ovo*); reduced clutch size and fledgling success	na	na	na	na	Fernie et al. (2000; 2001)	3, 4
Aroclor mixture	American kestrel	na	7	three generations	increased incidence of embryo abnormalities and cracked eggs, reduced F1 offspring survival, reduced offspring growth rate in F2 nestlings	na	na	na	na	Fernie et al. (2003a; 2003b; 2003c)	3, 4

CHEMICAL	TEST SPECIES	NOAEL (mg/kg bw/day)	LOAEL (mg/kg bw/day)	EXPOSURE DURATION	EFFECT	NO-EFFECT CONC. (mg/kg ww)[a]	EFFECT CONC. (mg/kg ww)[a]	BODY WEIGHT (kg)	FOOD INGESTION RATE[b]	SOURCE	NOTES
Aroclor 1242	mallard	na	15	12 weeks	reduced hatchability, embryo mortality, and egg viability, and increased incidence of embryo abnormalities	na	150	1.082 (Dunning 1993)	0.1082 kg ww/day (Heinz et al. 1987)	Haseltine and Prouty (1980)	

a No-effect and effect concentrations are presented in the units given in the studies reviewed. Table notes indicate how units were converted to wet weight or dry weight to correspond to the FIR units for calculating NOAELs and LOAELs.

b Ingestion rates are from equations for bird groups presented in Nagy (2001), from data presented for individual bird species (Nagy 2001), or from other sources as noted.

Notes:

1. Effect and/or no-effect concentration converted into dry weight assuming 10% moisture in prepared diet.
2. Effect concentration converted into dry weight assuming 9% moisture contents of seeds (EPA 1993b).
3. Body weight-normalized dose was estimated in study.
4. Fifty-two percent of first-generation offspring died within 3 days of hatching.

F1 – first generation
F2 – second generation
bw – body weight
dw – dry weight
LOAEL – lowest-observed-adverse-effect level
na – not available or not applicable
NOAEL – no-observed-adverse-effect level
PCB – polychlorinated biphenyl
ww – wet weight
Bold identifies the NOAEL and LOAEL selected as the TRVs.

LOAELs ranged from 0.35 mg/kg bw/day for eggshell thickness in American kestrels (Lowe and Stendell 1991) to 15 mg/kg bw/day for reproduction in mallards (Haseltine and Prouty 1980). The lowest calculated LOAELs (0.35 and 0.60 mg/kg bw/day) were based on eggshell thinning in American kestrels and Japanese quail (Lowe and Stendell 1991; Hill et al. 1976). These LOAELs were not selected as TRVs because the eggshell thinning results were not associated with reduced hatchability. The next lowest TRV (1.4 mg/kg bw/day) was based on an endpoint of reduced hatching success, which was measured in second generation offspring of ringed turtle doves following dietary exposure to Aroclor 1254 (Peakall et al. 1972; Peakall and Peakall 1973). This dose (1.4 mg/kg bw/day) was selected as the LOAEL TRV for total PCBs.

NOAELs ranged from 0.49 mg/kg bw/day, at which no adverse effect on reproduction was reported in screech owls (McLane and Hughes 1980), to 3.9 mg/kg kg/d, at which egg production and eggshell thinning was unaffected in mallards (Risebrough and Anderson 1975). The NOAEL of 0.49 mg/kg bw/day was selected as the NOAEL TRV.

A.5.2.1.2 PCB TEQs

PCB TEQs are expressed as 2,3,7,8-TCDD equivalents; therefore, toxicity studies involving exposing birds to 2,3,7,8-TCDD were reviewed. Effects of dioxins and furans reported in laboratory studies with various species of animals include developmental toxicity, hepatotoxicity, endocrine disruption, immunotoxicity, and death (Kennedy et al. 1996).

No studies in the published literature involving dietary exposure of birds to 2,3,7,8-TCDD were found. Two studies that evaluated the exposure of birds to 2,3,7,8-TCDD were identified. One study exposed birds through IP injection and the other study used oral intubation. Although these studies used less relevant forms of exposure, they were the only studies available, and thus data from these studies are presented in Table A.5-14.

The lowest dose resulting in effects was from a study of reproductive effects by Nosek et al. (1992), which exposed ring-necked pheasants to weekly IP injections of 2,3,7,8-TCDD for 10 weeks. These weekly injections were converted to daily doses. There are significant uncertainties in assuming that effects resulting from this acute high-level exposure would be similar to effects from chronic dietary exposure. The other study (Schwetz et al. 1973) resulted in effects to cockerels exposed through oral intubation at a higher dose (0.001 mg/kg bw/day) than the Nosek et al. (1992) study, but did not involve reproductive endpoints. Therefore, the LOAEL from Nosek et al. (1992) (0.00014 mg/kg bw/day) was selected as the LOAEL TRV. The highest NOAEL below the LOAEL was from the same study with the same endpoints. This dose of 0.000014 mg/kg bw/day was selected as the NOAEL TRV. Uncertainties associated with the absence of relevant toxicological data for chronic exposure of birds to 2,3,7,8-TCDD, as well as uncertainties associated with the PCB TEQ approach, are discussed in Section A.6.3.1.2.

Table A.5-14. 2,3,7,8-TCDD toxicity studies for birds for use in PCB TEQ risk analysis

Chemical	Test Species	NOAEL (mg/kg bw/day)	LOAEL (mg/kg bw/day)	Exposure Duration	Effect	No-Effect Conc. (mg/kg ww)	Effect Conc. (mg/kg ww)	Body Weight (kg)	Food Ingestion Rate	Source	Notes
2,3,7,8-TCDD	ring-necked pheasant	**0.000014**	**0.00014**	once per week for 10 weeks	reduced body weight, egg production, and survival of adults and embryos	na	na	na	na	Nosek et al. (1992)	1
2,3,7,8-TCDD	cockerel	0.0001	0.001	20 to 21 days	decreased survival	na	na	na	na	Schwetz et al. (1973)	2

Notes:

1. Exposure route was through IP injection
2. Exposure route was through oral intubation.

bw – body weight

LOAEL – lowest-observed-adverse-effect level

na –not applicable; exposure dose was presented in the study

NOAEL – no-observed-adverse-effect level

PCB – polychlorinated biphenyl

TCDD –tetrachlorodibenzo-*p*-dioxin

TEQ – toxic equivalent

ww – wet weight

Bold identifies the NOAEL and LOAEL selected as the TRVs.

A.5.2.1.3 Arsenic

Chronic dietary exposure to arsenic has been shown to cause growth and reproductive effects in birds based on three studies identified that evaluated the toxicity of dietary arsenic to mallards (Stanley et al. 1994; USFWS 1964) (Table A.5-15). All three of these studies exposed the mallards to sodium arsenate in their diet. Although arsenite is known to be more highly toxic in general than arsenate (Eisler 1988a), there are no available data on the reproductive toxicity of arsenite to birds, resulting in uncertainty in the arsenic TRV.

Adverse effects were reported in two of the studies presented in Table A.5-15. Reproduction was affected in a study that dosed both parents at 40 mg/kg bw/day (Stanley et al. 1994), and the survival of young mallards was affected when they were dosed at a rate of 50 mg/kg bw/day (USFWS 1964). Adverse effects reported in the study with the lowest LOAEL (40 mg/kg bw/day) included delayed onset of egg laying, decreased offspring body weight, decreased egg production, and decreased eggshell thickness. This dietary dose (40 mg/kg bw/day) was selected as the LOAEL TRV.

NOAELs ranged from 6.1 to 28 mg/kg bw/day. Mallard growth, survival, or reproduction were unaffected at these doses (Camardese et al. 1990; USFWS 1964; Stanley et al. 1994). The highest NOAEL below the LOAEL with the same endpoint, 10 mg/kg bw/day, was from the same study as the selected LOAEL TRV (Stanley et al. 1994), and was selected as the NOAEL TRV.

A.5.2.1.4 Cadmium

Chronic dietary exposure to cadmium has been reported to cause growth and reproductive effects in birds as well as histopathological effects on kidneys and testes (White and Finley 1978b). Eight studies that evaluated the toxicity of dietary cadmium to birds with growth, reproduction, or survival endpoints were identified (Table A.5-16). Dietary exposure to cadmium resulted in adverse effects on growth or reproduction in chickens, mallards, and Japanese quail.

LOAELs ranged from 2.9 mg/kg bw/day for eggshell thinning in chickens (Leach et al. 1979) to 47 mg/kg bw/day for reduced growth in mallards (DiGiulio and Scanlon 1984). At the lowest LOAEL, eggshell thickness was significantly reduced in chickens fed 2.9 mg/kg bw/day cadmium sulfate for 48 weeks (Leach et al. 1979), but decreased reproductive success was not reported, so this LOAEL was not selected as the TRV. The next lowest LOAEL (4.0 mg/kg bw/day) was associated with a reduction in body weight in male chicks following a 6-week exposure (Richardson et al. 1974). This LOAEL was selected as the TRV. The NOAEL of 1.5 mg/kg bw/day from Cain et al. (1983) was selected as the NOAEL TRV because it was lower than the selected LOAEL TRV and was based on the same growth endpoint.

Table A.5-15. Arsenic dietary toxicity studies for birds

Chemical	Test Species	NOAEL (mg/kg bw/day)	LOAEL (mg/kg bw/day)	Exposure Duration	Effect	No-Effect Conc. (mg/kg ww)	Effect Conc. (mg/kg ww)	Body Weight (kg)	Food Ingestion Rate (kg ww/day)	Source	Notes
Sodium arsenate	mallard (young)	6.1	na	10 weeks	no effect on female growth or survival	300	na	0.6305	0.0128	Camardese et al. (1990)	1
Sodium arsenate	mallard	**10**	**40**	115 to 128 days	delayed egg laying; depressed egg weight, production, and shell thinning; decreased offspring body weight	100	400	1.082 (Dunning 1993)	0.1082 (Heinz et al. 1987)	Stanley et al. (1994)	2
Sodium arsenite	mallard (young)	25	50	154 days	reduced survival	250	500	1.082 (Dunning 1993)	0.1082 (Heinz et al. 1987)	USFWS (1964)	3

Notes:

1. No-effect concentration was converted into dry weight using 13.1% moisture as reported in study. Body weight and FIR were reported in study.
2. Effect and no-effect concentrations were converted into dry weight assuming 10% moisture in prepared diet.
3. Literature values for body weight and FIR for adult mallard were used because average values for the birds used in the study (1 day old at the start of the 154-day study) were not available. Based on body weight data for control birds presented in Camardese et al. (1990), birds reach the adult weight of 1.082 kg (Dunning 1993) at approximately 50 days of age.
4. Sample et al. (1996) selected 250 mg/kg ww as the effect concentration from the USFWS study. However, mortality at this concentration (12%) was similar to or less than mortality in the controls (8-31%). Therefore, in this assessment, the concentration of 500 mg/kg ww with 60% mortality was selected as the effect concentration.

bw – body weight

LOAEL – lowest-observed-adverse-effect level

na – not available or not applicable

NOAEL – no-observed-adverse-effect level

ww – wet weight

Bold identifies the NOAEL and LOAEL selected as the TRVs.

Table A.5-16. Cadmium dietary toxicity studies for birds

Chemical	Test Species	NOAEL (mg/kg bw/day)	LOAEL (mg/kg bw/day)	Exposure Duration	Effect	No-Effect Conc. (mg/kg ww)	Effect Conc.	Body Weight (kg)	Food Ingestion Rate[a]	Source	Notes
Cadmium chloride	mallard (young females)	**1.5**	na	12 weeks	no effect on body weight	14.6	na	0.8825	0.883 kg ww/day (Heinz et al. 1987)	Cain et al. (1983)	1
Cadmium sulfate	chicken (hens)	0.73	2.9	48 weeks	reduced shell thickness	12.22	48.22 mg/kg ww	0.1019 (NRC 1984)	1.7 kg ww/day (NRC 1984)	Leach et al. (1979)	
Cadmium chloride	Japanese quail (chicks)	na	**4.0**	6 weeks	reduced male body weight	na	75 mg/kg dw	0.093	0.0050 kg dw/day, galliformes (Nagy 2001)	Richardson et al. (1974)	1
Cadmium chloride	mallard	19	na	90 days	no effect on survival	200	na	1.153	0.110 kg ww/day	White and Finley (1978a)	2
Cadmium chloride	mallard	20	na	30 to 90 days	no effect on body weight, adult mortality	210	na	1.153	0.110 kg ww/day	White and Finley (1978b)	2
Cadmium chloride	chicken (chicks)	na	24	21 days	reduced male body weight	na	75 mg/kg dw	0.138	0.0434 kg dw/day (NRC 1994)	Freeland and Cousins (1973)	1
Cadmium chloride	chicken (chicks)	na	40	20 days	reduced male body weight	na	400 mg/kg dw	0.574 (NRC 1994)	0.0577 kg dw/day (NRC 1994)	Pritzl et al. (1974)	
Cadmium chloride	mallard	16	47	42 days	reduced body weight	150	450 mg/kg ww	1.119 (NOAEL), 1.027 (LOAEL)	0.122 (NOAEL), 0.108 (LOAEL) kg ww/day	DiGiulio and Scanlon (1984)	2

[a] Ingestion rates are from equations for bird groups presented in Nagy (2001), from data presented for individual bird species (Nagy 2001), or from other sources as noted.

Notes:

1. Body weight was reported in study.
2. Body weight and FIR were reported in study.

bw – body weight
dw – dry weight
LOAEL – lowest-observed-adverse-effect level
na – not available or not applicable
NOAEL – no-observed-adverse-effect level
ww – wet weight
Bold identifies the NOAEL and LOAEL selected as the TRVs.

A.5.2.1.5 Chromium

In its hexavalent form, chromium is mutagenic, carcinogenic, and teratogenic to a wide variety of organisms under laboratory conditions (Eisler 1986a). Fewer data are available involving the effects of trivalent chromium on birds. Three studies that evaluated the toxicity of dietary chromium to birds were identified (Table A.5-17). Adverse effects were observed in one of the three studies. In that study, Haseltine et al. (unpublished) reported a decrease in offspring survival of adult black ducks treated with 5.0 mg/kg bw/day trivalent chromium over a chronic period (10 months) and critical lifestage (reproduction). This LOAEL (5.0 mg/kg bw/day) was selected as the LOAEL TRV for chromium. The results of this unpublished study were reported in Sample et al. (1996), and the original study could not be obtained. Therefore, there is some uncertainty regarding the quality of the study. The highest NOAEL below the LOAEL with the same endpoint was from the same study. This NOAEL of 1.0 mg/kg bw/day was selected as the NOAEL TRV.

A.5.2.1.6 Cobalt

One study that evaluated the toxicity of dietary cobalt to birds was identified (Table A.5-18). In this study (Diaz et al. 1994), body weights of chickens were significantly less than the control group following 2 weeks of exposure to 116, 251, or 472 mg/kg cobalt in the diet. The lowest dose of 116 mg/kg was used to calculate the LOAEL of 23.1 mg/kg bw/day. There was no lower dose with which to calculate a no-effect dose, so the subchronic LOAEL was divided by an uncertainty factor of 10 to estimate the NOAEL TRV of 2.31 mg/kg dw/day.

Table A.5-17. Chromium dietary toxicity studies for birds

Chemical	Test Species	NOAEL (mg/kg bw/day)	LOAEL (mg/kg bw/day)	Exposure Duration	Effect	No-Effect Conc. (mg/kg ww)[a]	Effect Conc. (mg/kg ww)[a]	Body Weight (kg)	Food Ingestion Rate (kg ww/day)	Source	Notes
Chromium picolinate	chicken (hens)	0.10	na	28 days	no effect on egg weight or shell thickness	1.6	na	0.1019 (NRC 1984)	1.7 (NRC 1984)	Lien et al. (2004)	
Chromium potassium sulfate	black duck	**1.0**	**5.0**	10 months	reduced duckling survival	10	50	1.25 (Dunning 1993)	0.125 (Heinz et al. 1987)	Haseltine et al. (unpublished) as cited in Sample et al. (1996)	1
Sodium chromate	chicken (chicks)	7.7	na	22 days	no effect on adult male survival or male body weight	32.1	na	0.12 (NRC 1994)	0.0286 (NRC 1994)	Romoser et al. (1961)	

[a] No-effect and effect concentrations are presented in the units given in the studies reviewed.

Notes:

1. Original paper (unpublished study) could not be obtained.

bw – body weight
LOAEL – lowest-observed-adverse-effect level
na – not available or not applicable
NOAEL – no-observed-adverse-effect level
ww – wet weight
Bold identifies the NOAEL and LOAEL selected as the TRVs.

Table A.5-18. Cobalt dietary toxicity studies for birds

Chemical	Test Species	NOAEL (mg/kg bw/day)	LOAEL (mg/kg bw/day)	Exposure Duration	Effect	No-Effect Conc.	Effect Conc. (mg/kg ww)	Body Weight (kg)	Food Ingestion Rate (kg ww/day)	Source	Notes
Cobalt chloride	chicken (chicks)	na	**23.1**	14 days	reduced body weight	na	125	0.1462	0.027	Diaz et al. (1994)	1

Notes:

1. Body weight and FIR reported in study.

bw – body weight
LOAEL – lowest-observed-adverse-effect level
na – not available or not applicable
NOAEL – no-observed-adverse-effect level
ww – wet weight
Bold identifies the LOAEL selected as the TRV. A NOAEL TRV was estimated by dividing the subchronic LOAEL TRV by an uncertainty factor of 10. The resulting NOAEL TRV was 2.31 mg/kg bw/day.

A.5.2.1.7 Copper

No data are available to assess the potential effects of chronic dietary exposure of copper to wild birds (Eisler 1997). Seven studies that evaluated the toxicity of dietary copper to chickens were available (Table A.5-19). In studies with chicks, impaired growth was reported at doses ranging from 29 to 66 mg/kg bw/day dietary copper in various forms (copper sulfate, copper chloride, and copper oxide) for 2 to 10 weeks. The lowest dose (29 mg/kg bw/day) was selected as the LOAEL TRV, although there is uncertainty in this growth effect because of the subchronic exposure period of 25 days. The next lowest LOAEL (62 mg/kg bw/day) resulted in a reduction in growth and survival in chicks after exposure for 10 weeks (Mehring et al. 1960).

NOAELs ranged from 2.1 mg/kg bw/day, at which no effect on chick growth or survival was reported (Dozier et al. 2003), to 47 mg/kg bw/day, at which chick growth was not affected (Mehring et al. 1960). The NOAEL of 21 mg/kg bw/day (Poupoulis and Jensen 1976) was selected as the NOAEL TRV for the subchronic LOAEL, because it was the highest NOAEL that was lower than the selected LOAEL TRV based on the same growth endpoint.

A.5.2.1.8 Lead

The acute effects of lead poisoning as a result of lead shot ingestion by birds have been extensively studied, although this exposure route is not relevant to the LDW. Numerous effects have been reported, including mortality, damage to the nervous system, muscular paralysis, kidney and liver damage, internal lesions, enlarged gall bladder, anemia, reduced brain weight, and abnormal skeletal development (Eisler 1988b). Fewer studies have been conducted on chronic effects of dietary exposure to lead.

Four studies that evaluated the chronic toxicity of dietary lead to birds were identified (Table A.5-20). Adverse effects were reported in two of the four studies. One of the studies (Pattee 1984) used metallic lead powder, which is a form of lead that is not likely found in the LDW; adverse effects were not reported in this study. Edens et al. (1976) reported a significant decrease in egg hatchability of Japanese quail exposed to 20 mg/kg bw/day lead over a chronic period (12 weeks) and critical lifestage (reproduction). Japanese quail chicks fed 28 mg/kg bw/day lead for 6 weeks experienced a reduction in body weight (Morgan et al. 1975). The lowest LOAEL of 20 mg/kg bw/day was selected as the TRV.

NOAELs ranged from 2.0 mg/kg bw/day, at which egg hatchability was unaffected in Japanese quail (Edens et al. 1976), to 5.82 mg/kg bw/day, at which no effect on survival or reproduction was reported in American kestrels (Pattee 1984). The NOAEL of 5.82 mg/kg bw/day was selected as the TRV because it was the highest NOAEL below the selected LOAEL based on a reproductive endpoint.

Table A.5-19. Copper dietary toxicity studies for birds

Chemical	Test Species	NOAEL (mg/kg bw/day)	LOAEL (mg/kg bw/day)	Exposure Duration	Effect	No-Effect Conc.[a]	Effect Conc.[a]	Body Weight (kg)	Food Ingestion Rate	Source	Notes
Copper sulfate, copper amino acid complex	chicken (chicks)	2.1	na	17 days	no effect on body weight or survival	18 mg/kg dw	na	0.254 (NRC 1994)	0.0295 kg dw/day (NRC 1994)	Dozier et al. (2003)	
Copper sulfate	chicken (hens)	11.2	na	90 days	no effect on damaged egg ratio, egg weight, or adult survival	200 mg/kg ww	na	0.09585	1.71 kg ww/day (NRC 1984)	Balevi and Coskun (2004)	1
Copper sulfate	chicken (hens)	15	na	28 days	no effect on egg weight and shell thickness	250 mg/kg ww	na	0.1019 (NRC 1984)	1.70 kg ww/day (NRC 1984)	Lien et al. (2004)	
Copper sulfate	chicken (1 day old)	16	29	25 days	reduced growth	200 mg/kg ww	350 mg/kg ww	0.534 (EPA 1993)	0.044 kg ww/day (EPA 1993)	Smith (1969)	
Copper sulfate	chicken (chicks)	**21**	41	4 weeks	reduced growth	250 mg/kg ww	500 mg/kg ww	0.534 (EPA 1993)	0.044 kg ww/day (EPA 1993)	Poupoulis and Jensen (1976)	
Copper oxide	chicken (chicks)	47	**62**	10 weeks	reduced growth and survival	570 mg/kg ww	749 mg/kg ww	0.534 (EPA 1993)	0.044 kg ww/day (EPA 1993)	Mehring et al. (1960)	
Copper chloride	chicken (chicks)	na	66	8 to 22 days	reduced body weight	na	500 mg/kg dw	0.3845	0.0509 kg dw/day	Persia et al. (2004)	2

[a] No-effect and effect concentrations are presented in the units given in the studies reviewed.

Notes:

1. Body weight reported in study.
2. Body weight and FIR reported in study.

bw – body weight

dw – dry weight

LOAEL – lowest-observed-adverse-effect level

na – not available or not applicable

NOAEL – no-observed-adverse-effect level

ww – wet weight

Bold identifies the NOAEL and LOAEL selected as the TRVs.

Table A.5-20. Lead dietary toxicity studies for birds

Chemical	Test Species	NOAEL (mg/kg bw/day)	LOAEL (mg/kg bw/day)	Exposure Duration	Effect	No-Effect Conc.[a]	Effect Conc.[a]	Body Weight (kg)	Food Ingestion Rate[b]	Source	Notes
Lead nitrate	mallard (first year)	2.5	na	12 weeks	no effect on survival	25 mg/kg ww	na	1.082 (Dunning 1993)	0.1082 kg ww/day (Heinz et al. 1987)	Finley et al. (1976)	
Metallic lead powder	American kestrel	**5.82**	na	5 to 7 months	no effect on survival, fertility, egg production, or eggshell thickness	50 mg/kg ww	na	0.13	0.0136 kg dw/day, Eurasian kestrel, Nagy (2001)	Pattee (1984)	1, 2
Lead acetate	Japanese quail	2.0	**20**	12 weeks	reduced egg hatchability	10 mg/kg ww	100 mg/kg ww	0.155 (Edens and Garlich 1983)	0.031 kg ww/day (Edens and Garlich 1983)	Edens et al. (1976)	
Lead acetate	Japanese quail (chicks)	5.5	28	6 weeks	reduced body weight	100 mg/kg dw	500 mg/kg dw	0.0715	0.0040 kg dw/day, galliformes, Nagy (2001)	Morgan et al. (1975)	2

[a] No-effect and effect concentrations are presented in the units given in the studies reviewed. Table notes indicate how units were converted to wet weight or dry weight to correspond to the FIR units for calculating NOAELs and LOAELs.

[b] Ingestion rates are from equations for bird groups presented in Nagy (2001), from data presented for individual bird species (Nagy 2001), or from other sources as noted

Notes:

1. No-effect concentration converted into dry weight assuming 10% moisture in prepared diet.
2. Body weight reported in study.

bw – body weight

dw – dry weight

LOAEL – lowest-observed-adverse-effect level

na – not available or not applicable

NOAEL – no-observed-adverse-effect level

ww – wet weight

Bold identifies the NOAEL and LOAEL selected as the TRVs.

A.5.2.1.9 Mercury

Chronic effects of dietary mercury on birds include adverse effects on growth, development, reproduction, metabolism, and behavior (Eisler 1987). Six studies that evaluated the toxicity of dietary mercury to birds were identified (Table A.5-21). When reviewing the toxicity literature for mercury, only forms of mercury relevant to the LDW were considered. Acceptable forms included inorganic mercury salts, such as mercuric chloride, as well as organic forms, such as methyl mercury chloride and dimethylmercury. Mercury-containing fungicides (e.g., Ceresan, methyl mercury dicyandiamide) were not considered relevant because these forms of mercury are not expected to occur in the LDW. The toxicity of these fungicidal formulations is likely highly influenced by the attached anions that are intended to enhance the toxicity of the fungicide because of the additive effects of these non-mercury components. As a result, laboratory bird studies involving mercury-containing fungicides were not considered for TRV selection.

In the studies reviewed, adverse effects on reproduction, early-life-stage growth, or adult survival were reported for various bird species, including great egrets, Japanese quail, zebra finch, and bobwhite quail from dietary exposure to mercury. LOAELs ranged from 0.091 mg/kg bw/day for reduced growth in young great egrets (Heinz 1980) to 62 mg/kg bw/day for offspring mortality of Japanese quail (Hill and Soares 1987). The lowest LOAEL of 0.091 mg/kg bw/day mercury was selected as the TRV.

NOAELs ranged from 0.43 mg/kg bw/day, at which there was no effect on survival of young bobwhite quail (Spann et al. 1986), to 5.24 mg/kg bw/day, at which there was no effect on eggshell thickness in American kestrels (Peakall and Lincer 1972). None of these NOAELs were lower than the lowest LOAEL. Therefore, the chronic LOAEL was divided by an uncertainty factor of 5 to obtain the NOAEL TRV of 0.018 mg/kg bw/day.

A.5.2.1.10 Nickel

Three studies that evaluated the effects of dietary nickel to birds were identified (Table A.5-22). Adverse effects were reported in two studies, ranging from a dose of 33 mg/kg bw/day that affected growth of broiler chicks (Weber and Reid 1968) to a dose of 107 mg/kg bw/day that affected mallard growth and survival (Cain and Pafford 1981). The lowest LOAEL of 33 mg/kg bw/day was selected as the TRV, although there is uncertainty in this growth effect because of the subchronic exposure period of only 4 weeks. The other LOAEL (107 mg/kg bw/day) resulted in a reduction in growth and survival in mallards after exposure for 90 days (Cain and Pafford 1981).

The NOAEL of 77 mg/kg bw/day from Cain and Pafford (1981) was selected as the NOAEL TRV. It was selected because it is the highest NOAEL that was lower than the selected LOAEL TRV and was from the same study based on the same endpoint.

Table A.5-21. Mercury dietary toxicity studies for birds

Chemical	Test Species	NOAEL (mg/kg bw/day)	LOAEL (mg/kg bw/day)	Exposure Duration	Effect	No-Effect Conc.[a]	Effect Conc.[a]	Body Weight (kg)	Food Ingestion Rate[b]	Source	Notes
Methyl-mercury chloride	great egret (1 day old)	na	**0.091**	14 weeks	reduced growth	na	0.5 mg/kg ww	1.02 (Fish 2002)	0.185 kg ww/day (Kushlan 1978)	Spalding et al. (2000)	
Methyl-mercury chloride	mallard	0.50	na	> 60 days	no effect on eggshell thickness	5 mg/kg ww	na	1.082 (Dunning 1993)	0.1082 kg ww/day (Heinz et al. 1987)	Heinz (1980)	
Methyl-mercury chloride	Japanese quail (chicks)	na	0.9	5 days	reduced hatchling survival of offspring	na	16 mg/kg ww	0.1 (NRC 1994)	0.0053 kg dw/day, galliformes (Nagy 2001)	Hill and Soares (1987)	1
Methyl-mercury chloride	zebra finch	0.72	1.4	76 days	reduced survival	2.5 mg/kg dw	5 mg/kg dw	0.012 (Dunning 1993)	0.0034 kg dw/day, passerines (Nagy 2001)	Scheuhammer (1988)	2
Methyl-mercury chloride	northern bobwhite quail (12 days old)	0.43	1.6	6 weeks	reduced survival	5.4 mg/kg ww	20 mg/kg ww	0.19 (EPA 1993)	0.0150 kg ww/day (EPA 1993)	Spann et al. (1986)	
Mercuric chloride	Japanese quail (1 day old)	0.80	1.6	10 weeks	reduced eggshell thickness	4 mg/kg ww	8 mg/kg ww	0.155 (Edens and Garlich 1983)	0.031 kg ww/day (Edens and Garlich 1983)	Stoewsand et al. (1971)	
Dimethyl mercury	American kestrel	5.24	na	3 months	no effect on eggshell thickness	10 mg/kg ww	na	0.13 (Pattee 1984)	0.0136 kg dw/day, Eurasian kestrel (Nagy 2001)	Peakall and Lincer (1972)	3
Mercuric chloride	Japanese quail (chicks)	na	62	5 days	reduced offspring hatchling survival	na	1,045 mg/kg ww	0.1 (NRC 1994)	0.0053 kg dw/day, galliformes (Nagy 2001)	Hill and Soares (1987)	1

[a] No-effect and effect concentrations are presented in the units given in the studies reviewed. Table notes indicate how units were converted to wet weight or dry weight to correspond to the FIR units for calculating NOAELs and LOAELs.

[b] Ingestion rates are from equations for bird groups presented in Nagy (2001), from data presented for individual bird species (Nagy 2001), or from other sources as noted

Notes:

1. Effect concentration converted into dry weight assuming 10% moisture in prepared diet.
2. No-effect and effect concentration converted into wet weight assuming 10% moisture in prepared diet.
3. Study did not indicate whether the mercury concentration in the diet, which consisted of dead chicks, was reported in wet weight or dry weight. It was assumed to be reported in wet weight and was converted into dry weight using 80% moisture content.

bw – body weight

dw – dry weight

LOAEL – lowest-observed-adverse-effect level

na – not available or not applicable

NOAEL – no-observed-adverse-effect level

ww – wet weight

Bold identifies the LOAEL selected as the TRV. A NOAEL TRV was estimated by dividing the chronic LOAEL TRV by an uncertainty factor of 5. The resulting NOAEL TRV was 0.018 mg/kg bw/day.

Table A.5-22. Nickel dietary toxicity studies for birds

Chemical	Test Species	NOAEL (mg/kg bw/day)	LOAEL (mg/kg bw/day)	Exposure Duration	Effect	No-Effect Conc. (mg/kg ww)[a]	Effect Conc. (mg/kg ww)[a]	Body Weight (kg)	Food Ingestion Rate (kg ww/day)	Source	Notes
Nickel sulfate	chicken (chicks)	15	na	4 weeks	no effect on body weight gain	na	111.7	0.484	0.067	Weber and Reid (1968)	1, 2
Nickel sulfate	chicken (chicks)	**17**	**33**	4 weeks	reduced body weight	117	156	0.467 (NOAEL); 0.39 (LOAEL)	0.069 (NOAEL); 0.083 (LOAEL)	Weber and Reid (1968)	1, 2, 3
Nickel acetate	chicken (chicks)	na	38	4 weeks	reduced body weight	na	165	0.376	0.086	Weber and Reid (1968)	1, 2
Nickel sulfate	mallard	77	107	90 days	reduced body weight and survival	774	1,069	0.561 (NOAEL); 0.178 (LOAEL)	0.0178 (NOAEL); 0.0561 (LOAEL) (Heinz et al. 1987)	Cain and Pafford (1981)	1, 3
Nickel sulfate	mallard	132	na	90 days	no effect on adult survival, body weight, or offspring hatchling weight	800	na	1.082 (Dunning 1993)	0.178	Eastin and O'Shea (1981)	2

[a] No-effect and effect concentrations are presented in the units given in the studies reviewed.

Notes:

1. Body weight was reported in study.
2. FIR was reported in study.
3. Body weight and ingestion rates used were specific to the no-effect and effect concentration test groups.

bw – body weight

LOAEL – lowest-observed-adverse-effect level

na – not available or not applicable

NOAEL – no-observed-adverse-effect level

ww – wet weight

Bold identifies the NOAEL and LOAEL selected as the TRVs.

A.5.2.1.11 Selenium

Selenium is an essential nutrient and deficiency in the diet may cause adverse effects in birds; elevated dietary concentrations have also been observed to cause adverse effects (Eisler 1985). Five studies that evaluated the dietary toxicity of selenium to birds were identified (Table A.5-23). Adverse effects were reported on offspring survival and growth, embryo development, adult growth, reproductive success, and adult survival. LOAELs ranged from 0.82 mg/kg bw/day, which affected growth and reproduction in mallards (Heinz et al. 1989), to 10 mg/kg bw/day, which affected survival in mallards (Heinz et al. 1988, 1989). In the study reporting the lowest LOAEL, offspring survival and growth was significantly affected for mallards fed 0.82 mg/kg bw/day selenium, as selenomethionine, for approximately 100 days (Heinz et al. 1989). This dose was selected as the LOAEL TRV.

NOAELs ranged from 0.025 mg/kg bw/day, which did not affect survival and growth in chicks (Choct et al. 2004), to 4.6 mg/kg bw/day, which did not affect survival in mallards (Heinz et al. 1988). The highest NOAEL that was lower than the selected LOAEL TRV and based on a reproductive endpoint was 0.5 mg/kg bw/day. This dose was selected as the NOAEL TRV.

A.5.2.1.12 Vanadium

Two studies that evaluated the toxicity of dietary vanadium to birds were identified (Table A.5-24). One of these studies reported reduced body weight in chickens exposed to 2.3 mg/kg bw/day vanadium in the form of ammonium metavanadate in the diet for 4 weeks (Ousterhout and Berg 1981). This dose was selected as the LOAEL TRV because this was the only LOAEL reported. The NOAEL from the same study and endpoint was selected as the NOAEL TRV (1.2 mg/kg bw/day). The only other dietary study identified was conducted with mallards for a longer period of 12 weeks using a different form of vanadium (vanadium sulfate); no effects were reported in that study at a dose of 11.4 mg/kg bw/day.

Table A.5-23. Selenium dietary toxicity studies for birds

Chemical	Test Species	NOAEL (mg/kg bw/day)	LOAEL (mg/kg bw/day)	Exposure Duration	Effect	No-Effect Conc.	Effect Conc. (mg/kg ww)	Body Weight (kg)	Food Ingestion Rate	Source	Notes
Sodium selenite or sel-plex 50	chicken (chicks)	0.025	na	~40 days	no effect on body weight or survival	0.25 mg/kg dw	na	0.993	0.0995 (NOAEL) kg dw/day	Choct et al. (2004)	1, 2
Seleno-methionine	mallard	0.42	**0.82**	~100 days	reduced offspring growth and survival	4.15 mg/kg ww	8.15	1.158 (NOAEL); 1.145 (LOAEL)	0.1158 (NOAEL); 0.1145 (LOAEL) kg ww/day (Heinz et al. 1987)	Heinz et al. (1989)	1, 3
Sodium selenite	mallard	**0.50**	1.0	4 weeks before laying to 3 weeks after hatching	increased embryo abnormalities	5 mg/kg ww	10	1.036 (NOAEL); 1.046 (LOAEL)	0.105 kg ww/day	Heinz et al. (1987)	1, 2, 3
Sodium selenite	mallard	1.0	2.5	4 weeks before laying to 3 weeks after hatching	reduced adult growth	10 mg/kg ww	25	1.046 (NOAEL); 0.938 (LOAEL)	0.105 (NOAEL); 0.094 (LOAEL) kg ww/day	Heinz et al. (1987)	1, 2, 3
Seleno-methionine	mallard	1.6	na	~100 days	no effect on body weight or adult survival	16.15 mg/kg ww	na	1.107	0.1107 kg ww/day (Heinz et al. 1987)	Heinz et al. (1989)	1, 3
Seleno-methionine	screech owl	1.0	3.2	~ 3 months	reduced body weight, clutch size, hatching success, offspring survival, egg size, and mass	3.7 mg/kg ww	12.1	0.189	0.050 kg ww/day	Wiemeyer and Hoffman (1996)	1, 2, 3
Sodium selenite	mallard	4.6	na	42 days	no effect on survival	20.1 mg/kg ww	na	0.326	0.075 kg ww/day	Heinz et al. (1988)	1, 2
Sodium selenite	mallard	2.1	4.6	42 days	reduced body weight	10.1 mg/kg ww	20.1	0.326	0.075 kg ww/day	Heinz et al. (1988)	1, 2, 3
Sodium selenite	mallard	2.5	10	4 weeks before laying to 3 weeks after hatching	reduced adult survival	25 mg/kg ww	100	0.718	0.072 kg ww/day	Heinz et al. (1987)	1, 2
Sodium selenite	mallard	na	10	42 days	reduced adult survival	na	40.1	0.146	0.038 kg ww/day	Heinz et al. (1988)	1, 2

Notes:

1. Body weight was reported in study.
2. FIR was reported in study.
3. Body weight and FIRs used were specific to the no-effect and effect concentration test group.

bw – body weight

dw – dry weight

LOAEL – lowest-observed-adverse-effect level

na – not available or not applicable

NOAEL – no-observed-adverse-effect level

ww – wet weight

Bold identifies the NOAEL and LOAEL selected as the TRVs.

Table A.5-24. Vanadium dietary toxicity studies for birds

Chemical	Test Species	NOAEL (mg/kg bw/day)	LOAEL (mg/kg bw/day)	Exposure Duration	Effect	No-Effect Conc. (mg/kg ww)	Effect Conc. (mg/kg ww)	Body Weight (kg)	Food Ingestion Rate (kg ww/day)	Source	Notes
Ammonium metavanadate	chicken (hens)	**1.2**	**2.3**	4 weeks	reduced body weight	20	40	1.71 (Dunning 1993)	0.0997 (NRC 1984)	Ousterhout and Berg (1981)	
Vanadium sulfate	mallard	11.4	na	12 weeks	no effect on body weight or survival	110	na	1.17	0.121	White and Dieter (1978)	1

Notes:

1. Body weight and FIR provided in study.

bw – body weight

LOAEL – lowest-observed-adverse-effect level

na – not available or not applicable

NOAEL – no-observed-adverse-effect level

ww – wet weight

Bold identifies the NOAEL and LOAEL selected as the TRVs.

A.5.2.1.13 Zinc

Most laboratory studies investigating zinc toxicity have been conducted with domestic birds such as chickens; only one wildlife study, using mallards, was found in the literature. Reported effects of zinc exposure on ducks and chickens in laboratory studies included reduced food intake and egg production, cessation of egg laying, weight loss, leg paralysis, pancreatic histopathology, and mortality (Eisler 1993). Adverse effects may also be observed if zinc is deficient in the diet because zinc is a nutrient essential for normal growth, development, and function. Effects from zinc deficiency are generally noted at concentrations below 120 mg/kg ww in food (Eisler 1993).

Six studies that evaluated the toxicity of dietary zinc to birds were identified (Table A.5-25). Adverse effects on growth and survival in young birds (mallards or chickens) were reported in four of these studies. LOAELs ranged from 124 mg/kg bw/day, which resulted in reduced growth of young chickens (Roberson and Schaible 1960), to 659 mg/kg bw/day, which resulted in mortality and reduced growth of young broiler chickens (Oh et al. 1979). The lowest LOAEL of 124 mg/kg bw/day was selected as the TRV. The NOAEL of 82 mg/kg bw/day from the same study was selected as the NOAEL TRV because it was the highest NOAEL below the LOAEL based on a growth endpoint.

Table A.5-25. Zinc dietary toxicity studies for birds

Chemical	Test Species	NOAEL (mg/kg bw/day)	LOAEL (mg/kg bw/day)	Exposure Duration	Effect	No-Effect Conc. (mg/kg ww)	Effect Conc.	Body Weight (kg)	Food Ingestion Rate	Source	Notes
Zinc sulfate or copper amino acid complex	chicken (chicks)	17	na	17 days	no effect on body weight or survival	150	na	0.254 (NRC 1994)	0.0295 kg ww/day (NRC 1994)	Dozier et al. (2003)	1
Zinc oxide, zinc sulfate, or zinc carbonate	chicken (chicks)	**82**	**124**	5 weeks	reduced growth	1,000	1,500 mg/kg ww	0.534 (EPA 1993)	0.044 kg ww/day (EPA 1993)	Roberson and Schaible (1960)	
Zinc sulfate	chicken (hens)	133	na	44 weeks	no effect on egg hatchability	2,028	na	1.9	0.125 kg ww/day	Stahl et al. (1990)	2
Zinc carbonate	mallard (7 weeks old)	na	300	60 days	reduced survival	na	3,000 mg/kg ww	1.082 (Dunning 1993)	0.1082 kg ww/day (Heinz et al. 1987)	Gasaway and Buss (1972)	
Zinc acetate	chicken (chicks)	330	659	5 weeks	reduced growth and survival	4,000	8,000 mg/kg ww	0.534 (EPA 1993)	0.044 kg ww/day (EPA 1993)	Oh et al. (1979)	
Zinc chloride	chicken (chicks)	na	344	8 to 22 days	reduced body weight	na	2,500 mg/kg dw	0.2873	0.0395 kg dw/day	Persia et al. (2004)	2

Notes:

1. NOAEL is based on background concentration in prepared food (30 mg/kg) plus exposure concentration added to food (120 mg/kg).
2. Body weight and FIR provided in study.

bw – body weight

dw – dry weight

LOAEL – lowest-observed-adverse-effect level

na – not available or not applicable

NOAEL – no-observed-adverse-effect level

ww – wet weight

Bold identifies the NOAEL and LOAEL selected as the TRVs.

A.5.2.2 TRVs for mammals

This section presents results from laboratory toxicity studies for COPCs identified for river otter and harbor seal (Table A.5-1), and selects TRVs for estimating risks. TRVs for PCBs were selected for both total PCBs (generally based on Aroclors) and for 2,3,7,8-TCDD (for comparison to PCB TEQs to assess effects from dioxin-like PCB congeners).

A.5.2.2.1 Total PCBs

PCBs have been reported to elicit a broad range of toxic effects in laboratory mammals under controlled exposure conditions, including lethality, hepatotoxicity, porphyria, body weight loss, dermal toxicity, thymic atrophy, immunosuppressive effects, reproductive and developmental effects, carcinogenesis, and neurotoxicity (Safe 1992, 1991, 1984; Seegal 1996; Safe 1990, 1994; Kimbrough 1985, 1987; Silberhorn et al. 1990; WHO 1993; Bolger 1993; Battershill 1994; Delzell et al. 1994). Review of the toxicology literature indicates that the potency of PCB mixtures depends on the chlorine content of the mixture and, in general, mixtures with higher chlorine content (i.e., Aroclors 1242, 1248, 1254, and 1260) are more toxic than mixtures with lower chlorine content (i.e., Aroclors 1221 and 1232). In general, the gastrointestinal tract of most mammals readily absorbs PCBs, but the absorption rate may be affected by the dose level and lipophilicity of the compound (Eisler 1986b; Van den Berg et al. 1998). There is evidence for placental transfer of PCBs in mammals (Eisler 1986b), and PCBs can also accumulate in the lipid portion of milk, resulting in exposure to suckling young.

Adverse reproductive effects (e.g., fertility, litter size, offspring survival) appear to be among the most sensitive *in vivo* endpoints of PCB toxicity in mammals (Golub et al. 1991; Rice and O'Keefe 1995; Hoffman et al. 1996). Reproductive success can be affected directly by toxic action on the differentiated reproductive tract or indirectly on systems that regulate reproduction (e.g., endocrine and central nervous systems). In laboratory studies, PCBs have been reported to elicit a broad range of direct and indirect effects associated with reproductive functions. Direct effects on the gonads and the female reproductive tract have been reported (Fuller and Hobson 1986). The precise mechanism by which PCBs cause reproductive effects in mammals remains unclear, but reproductive success appears to be a sensitive integrated endpoint of *in vivo* toxicity.

The most comprehensive studies of PCB toxicity in a non-domesticated mammal have been conducted with mink. Mink also appears to be one of the most sensitive mammalian species tested (Fuller and Hobson 1986), and is therefore a good surrogate for the assessment of risk to other mammals. Thus, only mink studies were reviewed for the development of PCB TRVs. Monkeys are also sensitive to PCBs, with reproductive effects reported at approximately 0.1 mg/kg bw/day (Allen et al. 1980; Barsotti et al. 1976; Truelove et al. 1982). However, data from mink studies were used instead because of their greater taxonomic similarity to river otter and harbor seal.

Ten studies that evaluated the toxicity of dietary PCBs to mink were identified (Table A.5-26). In the studies reviewed, adverse effects on maternal growth, kit growth, kit survival, whelping success, and reproductive success were reported for captive-bred mink following dietary exposure to PCBs. Reported reproductive effect levels ranged from 0.089 mg/kg bw/day (Brunström et al. 2001) to 2.6 mg/kg bw/day (Bleavins et al. 1980). At the lowest dose, offspring growth was significantly reduced in mink fed 0.089 mg/kg bw/day of a Clophen A50 PCB mixture for 18 months compared to mink in the control group (Brunström et al. 2001). This dose was selected as the LOAEL TRV.

NOAELs ranged from 0.13 mg/kg bw/day, which had no effect on reproduction in mink (Aulerich and Ringer 1977), to 1.5 mg/kg bw/day, which had no effect on growth in mink (Aulerich et al. 1986). There was no NOAEL that was lower than the selected LOAEL TRV in any of the studies reviewed, so a NOAEL TRV was estimated from the selected chronic LOAEL using an uncertainty factor of 2. This uncertainty factor was selected rather than an uncertainty factor of 5 used for other ROC/COPC pairs because of the large toxicity dataset for mink and PCBs that indicates that an uncertainty factor of 5 is high. As shown in Table A.5-26, the LOAELs are higher than the NOAELs from the same studies by factors ranging from 1.5 to 2 in the two studies that had both NOAELs and LOAELs (Aulerich and Ringer 1977; Aulerich et al. 1986). In addition, dose-response plots of toxicity to mink exposed to PCBs show very steep transitions between PCB exposures causing no adverse effects and those resulting in severe adversity (EPA 2003), indicating that an uncertainty factor of 2 is more appropriate than 5. The resulting NOAEL was 0.045 mg/kg bw/day.

Table A.5-26. PCB dietary toxicity studies for mammals

Chemical	Test Species	NOAEL (mg/kg bw/day)	LOAEL (mg/kg bw/day)	Exposure Duration	Effect	No-Effect Conc. (mg/kg ww)	Effect Conc.	Body Weight (kg)	Food Ingestion Rate (kg ww/day)	Source	Notes
Clophen A50	mink	na	**0.089**	18 months	reduced offspring kit growth	na	0.1 mg/day	1.12	na	Brunström et al. (2001)	1, 2
Aroclor 1254	mink	na	0.13	6 months	reduced offspring kit growth rate	na	1 mg/kg ww	1.34 (Bleavins and Aulerich 1981)	0.18 (Bleavins and Aulerich 1981)	Wren et al. (1987)	
Aroclor 1254	mink	na	0.22	4 and 9 months prior to giving birth	reduced number of offspring per female, decrease in offspring kit body weight	na	2 mg/kg ww	1.34 (Bleavins and Aulerich 1981)	0.15	Ringer (1983)	3
Aroclor 1254	mink	0.13	0.26	4 months	no kits born alive at 4 weeks	1	2 mg/kg ww	1.34 (Bleavins and Aulerich 1981)	0.18 (Bleavins and Aulerich 1981)	Aulerich and Ringer (1977)	
Aroclor 1254	mink	na	0.39	88 to 102 days	no kits whelped or born alive	na	2.5 mg/kg ww	0.87 (Bleavins and Aulerich 1981)	0.13 (Bleavins and Aulerich 1981)	Aulerich et al. (1985)	
PCB mixture (compo-sition not reported)	mink	na	0.51	66 days	reduced number of kits born alive	na	3.3 mg/kg ww	0.87 (Bleavins and Aulerich 1981)	0.13 (Bleavins and Aulerich 1981)	Jensen et al. (1977)	
Aroclor 1242	mink	na	0.65	8 months	reduced reproductive success	na	5 mg/kg ww	1.34 (Bleavins and Aulerich 1981)	0.18 (Bleavins and Aulerich 1981)	Bleavins et al. (1980)	
Aroclor 1254	mink	na	1.31	4 weeks	reduced weight gain in adults	na	10 mg/kg ww	1.34 (Bleavins and Aulerich 1981)	0.18 (Bleavins and Aulerich 1981)	Hornshaw et al. (1986)	
Aroclor 1254	mink	na	1.64	3 months	all whelps stillborn	na	na	na	na	Kihlstrom et al. (1992)	4
Aroclor 1254	mink	1.2	1.8	28 days	reduced female growth	na	na	na	na	Aulerich et al. (1986)	4
Clophen A50	mink	na	2.0	3 months	all whelps stillborn	na	na	na	na	Kihlstrom et al. (1992)	4

CHEMICAL	TEST SPECIES	NOAEL (mg/kg bw/day)	LOAEL (mg/kg bw/day)	EXPOSURE DURATION	EFFECT	NO-EFFECT CONC. (mg/kg ww)	EFFECT CONC.	BODY WEIGHT (kg)	FOOD INGESTION RATE (kg ww/day)	SOURCE	NOTES
Aroclor 1254	mink	1.5	2.4	28 days	reduced male and female growth	na	na	na	na	Aulerich et al. (1986)	4
Aroclor 1016	mink	na	2.6	8 months	reduced birth weight and growth rate of offspring kits, and 25 % adult female mortality	na	20 mg/kg ww	1.34 (Bleavins and Aulerich 1981)	0.18 (Bleavins and Aulerich 1981)	Bleavins et al. (1980)	

Notes:

1. Dietary dose determined by dividing daily dose by body weight. Female mink were exposed to 0.24 mg Clophen A50 three times a week, or 0.1 mg/day.
2. Body weight provided in study.
3. FIR provided in study.
4. Dietary dose calculated in study.

bw – body weight

LOAEL – lowest-observed-adverse-effect level

na – not available or not applicable

NOAEL – no-observed-adverse-effect level

PCB – polychlorinated biphenyl

ww – wet weight

Bold identifies the LOAEL selected as the TRV. A NOAEL TRV was not available from the study in which the chronic LOAEL of 0.089 mg/kg bw/day was reported, so it was estimated using an uncertainty factor of 2. The resulting NOAEL TRV is 0.045 mg/kg bw/day.

A.5.2.2.2 PCB TEQs

PCB TEQs are expressed as 2,3,7,8-TCDD equivalents; therefore, toxicity studies involving 2,3,7,8-TCDD effects on mammals were reviewed. Effects of dioxins and furans reported in laboratory studies with various species of mammals include developmental toxicity, hepatotoxicity, endocrine disruption, immunotoxicity, and death (Kennedy et al. 1996).

Seven studies that evaluated the dietary toxicity of 2,3,7,8-TCDD to mammals were identified (Table A.5-27). In these studies, adverse effects on growth, reproduction, and survival of guinea pigs, rats, or mink were reported following exposure to dietary 2,3,7,8-TCDD. The lowest dose at which effects were reported (4.9×10^{-6} mg/kg bw/day) was a subchronic study that resulted in reduced growth in guinea pigs exposed to 2,3,7,8-TCDD for 90 days (DeCaprio et al. 1986). This LOAEL was selected as the LOAEL TRV, although there is some uncertainty associated with this value because it was a short-term growth study. The highest NOAEL below this LOAEL (6.5×10^{-7} mg/kg bw/day) was from the same study with the same endpoint. This dose was selected as the NOAEL TRV.

Table A.5-27. 2,3,7,8-TCDD toxicity studies for mammals

Chemical	Test Species	NOAEL (mg/kg bw/day)	LOAEL (mg/kg bw/day)	Exposure Duration	Effect	No-Effect Conc. (mg/kg ww)	Effect Conc. (mg/kg ww)	Body Weight (kg)	Food Ingestion Rate (kg ww/day)	Source	Notes
2,3,7,8-TCDD	Hartley guinea pig	**6.5×10^{-7}**	**4.9×10^{-6}**	90 days	reduced body weight	1.0×10^{-5}	7.6×10^{-5}	na	na	DeCaprio et al. (1986)	1
2,3,7,8-TCDD	mink	2.6×10^{-6}	9.1×10^{-6}	131 to 132 days	decreased survival in kits at 3 weeks	1.6×10^{-5}	5.3×10^{-5}	1.089 (NOAEL), 1.054 (LOAEL)	0.18 (Bleavins and Aulerich 1981)	Hochstein et al. (2001)	2
2,3,7,8-TCDD	Sprague-Dawley rat	1.0×10^{-6}	1.0×10^{-5}	3 generations	reduced litter size and F2 postnatal survival	na	na	na	na	Murray et al.(1979)	1
2,3,7,8-TCDD	Hartley guinea pig	4.9×10^{-6}	2.85×10^{-5}	90 days	reduced survival	7.6×10^{-5}	4.3×10^{-4}	na	na	DeCaprio et al. (1986)	1
2,3,7,8-TCDD	mink	4.9×10^{-6}	5.0×10^{-5}	125 days	reduced body weight and adult survival	1.0×10^{-4}	1.0×10^{-3}	0.8776 (NOAEL), 0.8183 (LOAEL)	0.049 (NOAEL) 0.050 (LOAEL)	Hochstein et al. (2001)	3
2,3,7,8-TCDD	Sprague-Dawley rat	1.0×10^{-5}	1.0×10^{-4}	2 years	reduced body weight and adult female survival	na	na	na	na	Kociba et al.(1978)	1
2,3,7,8-TCDD	Sprague-Dawley rat	na	3.2×10^{-4}	13 weeks	reduced body weight	na	na	na	na	Van Birgelen et al. (1994)	1
2,3,7,8-TCDD	Sprague-Dawley rat	1.0×10^{-4}	na	3 generations	reduced body weight	na	na	na	na	Murray et al.(1979)	1

Notes:

1. Dietary dose calculated in study.
2. Body weight and ingestion rates used were specific to the no-effect and effect concentration test groups.
3. Body weight and FIR provided in study.

F2 – second generation

bw – body weight

LOAEL – lowest-observed-adverse-effect level

na – not available or not applicable

NOAEL – no-observed-adverse-effect level

TCDD – tetrachlorodibenzo-*p*-dioxin

ww – wet weight

Bold identifies the NOAEL and LOAEL selected as the TRVs.

A.5.2.2.3 Arsenic

Mammalian effects from chronic exposure to inorganic arsenic may include weakness, paralysis, conjunctivitis, dermatitis, decreased growth, liver damage, and developmental effects in offspring (Eisler 1988a). Early developmental stages are most sensitive to arsenic exposure.

One study that evaluated the toxicity of arsenic to mammals from dietary exposure (i.e., via food rather than drinking water or gavage) was identified (Table A.5-28). In this study, female rats fed 5.4 mg/kg bw/day arsenic had reduced body weights following two years of exposure to sodium arsenite (Byron et al. 1967). The results of this study were not statistically evaluated, although the final body weight range reported in rats fed 5.4 mg/kg bw/day (280.4 g ± standard error of 21.38 g) was lower than the final body weight range of the control group (350.9 g ± standard error of 26.36 g). Growth in rats appeared unaffected in rats exposed to 2.6 mg/kg bw/day (body weight at the end of the 2-year study was 322.4 g, with a standard error range of ± 21.38 g).

The LOAEL and NOAEL from Byron et al. (1967) (5.4 and 2.6 mg/kg bw/day, respectively) were selected as the LOAEL and NOAEL TRVs.

A.5.2.2.4 Cobalt

In laboratory studies, chronic exposure of mice and rats to cobalt via food, gavage or drinking water has resulted in adverse effects on reproduction, development, growth, and survival (ATSDR 2004). In addition, cardiovascular, neurological, renal, and endocrine effects have been reported (ATSDR 2004). Three studies that evaluated the toxicity of cobalt to mammals from dietary exposure (i.e., via food rather than drinking water or gavage) were identified for growth, reproductive, or survival endpoints (Table A.5-29). In these studies, adverse effects on survival or growth of laboratory guinea pigs and rats were reported following subchronic exposure to cobalt in food. No chronic studies were available. The lowest dose at which effects were reported (1.0 mg/kg bw/day) was selected as the LOAEL TRV. This LOAEL resulted in reduced growth in rats exposed to cobalt for 4 weeks (Chetty et al. 1979). No NOAELs that were lower than the selected LOAEL TRV were available. Therefore, a NOAEL TRV was estimated by dividing the subchronic LOAEL TRV by an uncertainty factor of 10, resulting in a NOAEL of 0.1 mg/kg bw/day.

Table A.5-28. Arsenic dietary toxicity studies for mammals

Chemical	Test Species	NOAEL (mg/kg bw/day)	LOAEL (mg/kg bw/day)	Exposure Duration	Effect	No-Effect Conc. (mg/kg ww)	Effect Conc. (mg/kg ww)	Body Weight (kg)	Food Ingestion Rate (kg ww/day)	Source	Notes
Sodium arsenite	rat	**2.6**	**5.4**	2 years	reduced female body weight	31.25	62.5	0.302 (NOAEL), 0.278 (LOAEL)	0.025 (NOAEL), 0.024 (LOAEL)	Byron et al. (1967)	1

Notes:

1. Data were not statistically evaluated. Body weight and FIR were reported in study.

bw – body weight

LOAEL – lowest-observed-adverse-effect level

NOAEL – no-observed-adverse-effect level

ww – wet weight

Bold identifies the NOAEL and LOAEL selected as the TRVs.

Table A.5-29. Cobalt dietary toxicity studies for mammals

CHEMICAL	TEST SPECIES	NOAEL (mg/kg bw/day)	LOAEL (mg/kg bw/day)	EXPOSURE DURATION	EFFECT	NO-EFFECT CONC. (mg/kg ww)	EFFECT CONC. (mg/kg ww)	BODY WEIGHT (kg)	FOOD INGESTION RATE (kg ww/day)	SOURCE	NOTES
Cobalt chloride	rat	na	**1.0**	4 weeks	reduced body weight	na	10	0.187	0.018 (EPA 1993)	Chetty et al. (1979)	1
Cobalt sulfate	guinea pig	na	1.4	5 weeks	reduced survival	na	20	0.50	0.035 (EPA 1993)	Mohiudden et al. (1970)	1, 2
Cobalt sulfate	guinea pig	1.4	na	5 weeks	no effect on body weight	20	na	0.50	0.035 (EPA 1993)	Mohiudden et al. (1970)	1
Cobalt chloride	rat	1.9	10	3 days	reduced body weight	20	100	0.196	0.019 (EPA 1993)	Wellman et al. (1984)	1, 3

Notes:

1. Body weight presented in study.
2. Data were not statistically evaluated; 4 of 20 guinea pigs died at LOAEL, and 1 of 20 guinea pigs died in the control group.
3. Data were not statistically evaluated; reduced food intake at LOAEL.

bw – body weight

LOAEL – lowest-observed-adverse-effect level

na – not available or not applicable

NOAEL – no-observed-adverse-effect level

ww – wet weight

Bold identifies the LOAEL selected as the TRV. A NOAEL TRV was not available from the study in which the subchronic LOAEL of 1.0 mg/kg bw/day was reported, so it was estimated using an uncertainty factor of 10. The resulting NOAEL TRV is 0.01 mg/kg bw/day.

A.5.2.2.5 Mercury

Exposure of mammals to mercury has been reported to adversely affect reproduction, growth, development, behavior, blood and serum chemistry, motor coordination, vision, hearing, histology, and metabolism (Eisler 1987). Three studies that evaluated the toxicity of dietary mercury to mammals were identified for growth, reproduction, and survival endpoints (Table A.5-30). In these studies, adverse effects following dietary ingestion of mercury included mortality and depressed growth in laboratory rats and mink. At the lowest LOAEL, growth was significantly reduced in rats fed 0.0084 mg/kg bw/day of mercury as methylmercuric chloride for three generations (Verschuuren et al. 1976). Adverse effects in mink were reported at concentrations two orders of magnitude higher than the LOAEL measured for rats. Growth was significantly reduced, and mortality was observed in mink fed diets with 0.25 mg/kg bw/day methylmercuric chloride (Wobeser et al. 1976) and 0.64 gm/kg bw/day methylmercury (Aulerich et al. 1974) for a subchronic duration. The lowest LOAEL, 0.0084 mg/kg bw, was selected as the LOAEL TRV. While toxicology data based on mink studies may be more representative to the mammals utilizing the LDW, the LOAEL based on rats was selected because it was the most conservative effects threshold reported in the three studies reviewed and was based on a multi-generational study. No controlled laboratory studies were found where mink were exposed to dietary mercury over a chronic period or during a critical lifestage.

No NOAELs lower than the selected LOAEL TRV were available. Therefore, the NOAEL TRV was estimated by dividing the selected chronic LOAEL TRV by an uncertainty factor of 5, resulting in a NOAEL TRV of 0.0017 mg/kg bw/day.

A.5.2.2.6 Selenium

Selenium is an essential nutrient and deficiency in the diet may cause adverse effects in mammals; elevated dietary concentrations have also been reported to cause adverse effects (Eisler 1985). Primary effects on laboratory mice and rats from chronic exposure to selenium via food, gavage, or drinking water include reduced growth and survival; effects to the reproductive, cardiovascular, and hematological systems have also been reported (ATSDR 2003). Four studies that evaluated the toxicity of selenium to mammals from dietary exposure (i.e., via food rather than drinking water or gavage) were identified (Table A.5-31). In these studies, adverse effects on growth or survival were reported following subchronic exposure of laboratory rats or hamsters to selenium in their diet. Rats exhibited a higher sensitivity to dietary selenium than did hamsters. No studies were identified where mammals were exposed to dietary selenium for a chronic exposure period or during a critical lifestage.

LOAELs ranged from 0.080 mg/kg bw/day, resulting in reduced growth of rats (Halverson et al. 1966), to 5.8 mg/kg bw/day, resulting in reduced survival of hamsters (Julius et al. 1983). The lowest LOAEL of 0.080 mg/kg bw/day was selected as the TRV. The only NOAEL below this LOAEL (0.055 mg/kg bw/day) was from the same study with the same endpoint. This dose was selected as the NOAEL TRV.

Table A.5-30. Mercury dietary toxicity studies for mammals

Chemical	Test Species	NOAEL (mg/kg bw/day)	LOAEL (mg/kg bw/day)	Exposure Duration	Effect	No-Effect Conc. (mg/kg ww)	Effect Conc. (mg/kg ww)	Body Weight (kg)	Food Ingestion Rate (kg ww/day)	Source	Notes
Methylmercuric chloride	rat	na	**0.0084**	three generations	reduced growth	na	0.0799	0.16	0.016 (EPA 1993)	Verschuuren et al. (1976)	1
Methylmercuric chloride	rat	0.19	na	three generations	no effect on survival or reproduction	1.997	na	0.20	0.019 (EPA 1993)	Verschuuren et al. (1976)	1
Methylmercuric chloride	mink	0.16	0.25	93 days	reduced growth, 40% mortality	1.2	1.9	1.34 (Bleavins and Aulerich 1981)	0.18 (Bleavins and Aulerich 1981)	Wobeser et al. (1976)	2
Methylmercury	mink	na	0.64	2 months	reduced growth, 100% mortality	na	5	1.2	0.15	Aulerich et al. (1974)	1, 3

Notes:

1. Body weight presented in study.
2. Two out of five mink died at the LOAEL.
3. FIR presented in study.

bw – body weight

LOAEL – lowest-observed-adverse-effect level

na – not available or not applicable

NOAEL – no-observed-adverse-effect level

ww – wet weight

Bold identifies the LOAEL selected as the TRV. A NOAEL TRV was not available from the study in which the chronic LOAEL of 0.0084 mg/kg bw/day was reported, so it was estimated using an uncertainty factor of 5. The resulting NOAEL TRV is 0.0017 mg/kg bw/day.

Table A.5-31. Selenium dietary toxicity studies for mammals

Chemical	Test Species	NOAEL (mg/kg bw/day)	LOAEL (mg/kg bw/day)	Exposure Duration	Effect	No-Effect Conc. (mg/kg ww)	Effect Conc. (mg/kg ww)	Body Weight (kg)	Food Ingestion Rate (kg ww/day)	Source	Notes
Sodium selenite	rat	**0.055**	**0.080**	6 weeks	reduced body weight	3.2	4.8	0.139 (NOAEL), 0.129 (LOAEL)	0.00238 (NOAEL), 0.00215 (LOAEL)	Halverson et al. (1966)	1, 2, 3
Sodium selenite	rat	0.13	0.14	6 weeks	reduced survival	8.0	9.6	0.129 (NOAEL), 0.1255 (LOAEL)	0.00215 (NOAEL), 0.00186 (LOAEL)	Halverson et al. (1966)	1, 2
L-seleno-methionine	rat	na	0.16	110 days	reduced body weight	na	2	0.34	0.027 (EPA 1993)	Behne et al. (1992)	1
Selenite	rat	0.16	na	110 days	no effect on body weight	2	na	0.34	0.027 (EPA 1993)	Behne et al. (1992)	1
Sodium selenite, nano-Se, or organic selenium	rat	0.17	0.28	13 weeks	reduced body weight	na	na	na	na	Jia et al. (2005)	4
Seleno-methionine	hamster	0.36	0.76	21 days	reduced body weight	5.1	10.1	0.092 (NOAEL), 0.091(LOAEL)	0.00655 (NOAEL), 0.0068 (LOAEL)	Julius et al. (1983)	1, 2, 3
Sodium selenite	hamster	na	3.4	21 days	reduced body weight	na	40.25	0.074	0.0062	Julius et al. (1983)	1, 2
Sodium selenite	hamster	na	5.8	21 days	reduced female survival	na	80.24	0.062	0.0045	Julius et al. (1983)	1, 2

Notes:

1. Body weight presented in study.
2. FIR presented in study. Data presented in study were not statistically evaluated.
3. Body weight and ingestion rates used were specific to the no-effect and effect concentration test groups.
4. Dietary dose calculated in study.

bw – body weight

LOAEL – lowest-observed-adverse-effect level

na – not available or not applicable

NOAEL – no-observed-adverse-effect level

ww – wet weight

Bold identifies the NOAEL and LOAEL selected as the TRVs.

A.5.3 SUMMARY OF WILDLIFE ASSESSMENT

A.5.3.1 Exposure assessment

The exposure assessment provided an estimate of each wildlife ROC's exposure to COPCs through ingestion of prey, water, and incidental sediment ingestion. Exposure doses were calculated for each ROC/COPC pair, and expressed as mg COPC ingested per kg body weight per day. Estimates of dietary composition and site usage were made using site-specific information, if available, along with species life history information. Exposure doses were estimated using UCL concentrations in prey tissue, sediment, and water. Exposure doses for wildlife were presented in Tables A.5-11 and A.5-12.

A.5.3.2 Effects assessment

The effects assessment selected TRVs to represent dietary thresholds of effects for each ROC/COPC pair. The toxicity literature was searched and relevant data for birds and mammals were compiled and screened against a set of guidelines to select the most appropriate TRVs. TRVs for both no-effects and low-effects data were selected, as summarized in Table A.5-32.

Table A.5-32. TRVs for ROC/COPC pairs

	NOAEL and LOAEL TRVs (mg/kg bw/day)							
	Spotted Sandpiper		Great Blue Heron and Osprey		River Otter		Harbor Seal	
COPC	NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL
Arsenic	10	40	ne	ne	2.6	5.4	ne	ne
Cadmium	1.5	4.0	ne	ne	ne	ne	ne	ne
Chromium	1.0	5.0	1.0	5.0	ne	ne	ne	ne
Cobalt	2.31	23.1	ne	ne	0.1	1.0	ne	ne
Copper	21	29	ne	ne	ne	ne	ne	ne
Lead	5.82	20	5.82	20	ne	ne	ne	ne
Mercury	0.018	0.091	0.018	0.091	0.0017	0.0084	0.0017	0.0084
Nickel	17	33	ne	ne	ne	ne	ne	ne
Selenium	0.5	0.82	ne	ne	0.055	0.080	ne	ne
Vanadium	1.2	2.3	ne	ne	ne	ne	ne	ne
Zinc	82	124	ne	ne	ne	ne	ne	ne
Total PCBs	0.49	1.4	0.49	1.4	0.018	0.089	0.018	0.089
PCB TEQs	1.4×10^{-5}	1.4×10^{-4}	1.4×10^{-5}	1.4×10^{-4}	2.6×10^{-6}	9.1×10^{-6}	2.6×10^{-6}	9.1×10^{-6}

bw – body weight
COPC – chemical of potential concern
LOAEL – lowest-observed-adverse-effect level
ne – not evaluated; not a COPC for this ROC
NOAEL – no-observed-adverse-effect level
PCB – polychlorinated biphenyl
TEQ – toxic equivalent
TRV – toxicity reference value

A.6.0 Risk Characterization and Uncertainty Analysis

This section presents the risk characterization for each ROC/COPC pair identified in the problem formulation (Section A.2.0) and discussed in the exposure and effects assessments (Sections A.3.0 through A.5.0) of this baseline ERA. The risk characterization section for each receptor group (i.e., benthic invertebrates, fish, and wildlife) consists of a risk estimate, an uncertainty analysis, and a risk conclusion section. The risk estimate section presents the hazard quotients (HQs)[67] calculated for each ROC/COPC pair. Uncertainties inherent in the risk assessment, including calculation of HQs, the problem formulation, and the exposure and effects assessment approach are discussed in the uncertainty analysis. The results of the HQ calculations and the uncertainty analysis are then integrated into risk conclusions.

In ERAs, HQs greater than 1.0 indicate that the exposures of some receptors are estimated to be greater than toxicological benchmarks. Such a finding is generally regarded as indicating a potential for adverse effects, particularly if the benchmark is an effects concentration (or dose) at which adverse effects were observed (i.e., a LOAEL). HQs may also be calculated based on a NOAEL. The potential for adverse effects associated with a NOAEL HQ greater than 1.0 is uncertain unless the LOAEL is also assessed because the true threshold for effects occurs at a concentration (or dose) somewhere between the NOAEL and LOAEL. An exposure falling between the NOAEL and LOAEL may or may not result in any adverse effect. Therefore, both types of HQs are calculated and presented to better describe the potential for adverse effects and to support risk management decisions.

A.6.1 BENTHIC INVERTEBRATES

This section characterizes risks to benthic invertebrates closely associated with sediment, such as amphipods, bivalves, and polychaetes, as well as more mobile, higher-trophic-level benthic invertebrates, such as crabs, that may travel over relatively greater distances than other invertebrates. Results from a direct measure of effect to one specific invertebrate group, gastropods, are also summarized.

Risk characterization for infaunal and epibenthic invertebrates was based primarily on an assessment of effects through the comparison of available surface sediment chemistry data with available sediment chemical criteria and guidelines and through the use of site-specific sediment toxicity tests (Ecology 1995).

Risks to crabs from two COPCs, zinc and PCBs, were characterized using a critical tissue-residue approach. The critical tissue-residue approach was also used to evaluate risks to infaunal invertebrates from TBT. Risks to meso- and neogastropods were

[67] The HQ is the ratio of the exposure concentration (or dose) to a concentration (or dose) associated with adverse effects.

characterized in two site-specific studies using a direct measure of effect (i.e., imposex).

Risks to benthic invertebrates from VOCs in porewater were characterized by comparing detected porewater concentrations to TRVs selected from the available literature. The risk characterization focused on areas where VOCs have been historically detected in groundwater at upland properties immediately adjacent to the LDW.

In this section, the risk characterization for the benthic invertebrate community is presented in Section A.6.1.1, and the risk characterization for crabs is presented in Section A.6.1.2. Risk characterization for the benthic invertebrate community includes risks from sediment-associated chemicals (Section A.6.1.1.1), porewater-associated VOCs (Section A.6.1.1.2), and TBT (Section A.6.1.1.3).

A.6.1.1 Benthic invertebrate community

The risk characterization for the benthic invertebrate community evaluated the following:

- Site-specific sediment toxicity test results and comparisons of surface sediment chemical concentrations to available sediment chemical criteria, guidelines, and TRVs
- Comparisons of VOC concentrations in porewater to toxicity data for benthic invertebrates
- Risks from TBT exposure based on results of the imposex study of meso- and neogastropods and using a critical tissue-residue approach

A.6.1.1.1 Sediment

Risk Estimates

The potential for adverse effects to benthic invertebrate communities resulting from exposure to sediment-associated COPCs was evaluated through site-specific toxicity testing and through a comparison of COPC concentrations in LDW surface sediment to SMS chemical criteria for 41 COPCs for benthic invertebrates. Three COPCs were evaluated based on a comparison of sediment chemical concentrations to toxicologically based sediment guidelines or TRVs.

The SMS regulations (WAC 173-204) provide both chemical- and biological effects-based criteria. The biological effects-based criteria provide an option for conducting site-specific confirmation of the chemical criteria through the use of sediment toxicity tests. Because AETs, which form the basis for the chemical criteria, are based on sediment samples with a mixture of chemicals from various locations in Puget Sound, and exceedance of those criteria is not always an accurate predictor of adverse effects, the regulations state that site-specific toxicity tests supersede site-specific chemistry data. For example, if the concentration of a chemical was greater than the CSL chemical criteria at a location, but the sample was not toxic in the biological testing, then the location would not be classified as a CSL exceedance.

Table A.6-1 presents a summary of the surface sediment chemistry data for the COPCs identified in Section A.2.5.1.1 with at least one concentration greater than the SQS chemical criteria. The toxicity test results and associated sediment chemistry data from the 46 locations tested in Phase 2 and 7 locations tested in earlier studies are presented in Table A.6-2. The table also presents the final SMS classification for each location tested with the toxicity test results superseding the chemistry results. SQS and CSL exceedances based on toxicity test results and detected chemical concentrations above the SQS and CSL chemical criteria are shown as point locations on Map A.6-1. Chemicals not detected in sediments but with reporting limits greater than the SMS chemical criteria are discussed in the uncertainty analysis.

Table A.6-1. Detection frequencies and frequencies of detected concentrations greater than SQS and CSL for all SMS COPCs

COPC	Sediment Chemistry								
	Detection Frequency		Frequency of Detected Concentrations > SQS			Frequency of Detected Concentrations > CSL			Maximum Detected Conc./CSL
	No. of Samples[a]	Percent	No. of Samples[b]	Percent	No. of Samples with RL > SQS[c]	No. of Samples[d]	Percent	No. of Samples with RL > CSL[e]	
Metals									
Arsenic	754/816	92	13/816	1.6	0	8/816	0.98	0	12
Cadmium	565/799	71	13/799	1.6	0	11/799	1.4	0	18
Chromium	813/813	100	9/813	1.1	0	8/813	0.98	0	4.1
Copper	816/816	100	12/816	1.5	0	12/816	1.5	0	31
Lead	816/816	100	21/816	2.6	0	19/816	2.3	0	43
Mercury	717/833	86	37/833	4.4	0	23/833	2.8	0	7.8
Silver	481/784	61	10/784	1.3	0	10/784	1.3	0	44
Zinc	813/813	100	42/813	5.2	0	16/813	2.0	0	10
PAHs									
2-Methylnaphthalene	139/782	18	3/782	0.38	9	3/782	0.38	3	2.5
Acenaphthene	301/792	38	19/792	2.4	13	3/792	0.38	4	4.6
Anthracene	553/792	70	2/792	0.25	0	0/792	0	0	ne[f]
Benzo(a)anthracene	719/792	91	12/792	1.5	0	3/792	0.38	0	1.6
Benzo(a)pyrene	718/786	91	8/786	1.0	0	3/786	0.38	0	2.0
Benzo(g,h,i)perylene	649/787	82	16/787	2.0	7	7/787	0.89	3	2.3
Total benzofluoranthenes	727/786	92	9/786	1.1	0	4/786	0.51	0	2.0
Chrysene	741/792	94	24/792	3.0	0	1/792	0.13	0	1.3
Dibenzo(a,h) anthracene	400/792	51	19/792	2.4	19	4/792	0.51	9	2.2
Dibenzofuran	246/791	31	10/791	1.3	13	3/791	0.38	4	3.8
Fluoranthene	762/792	96	39/792	4.9	0	8/792	1.0	0	2.7
Fluorene	373/792	47	14/792	1.8	9	3/792	0.38	1	3.7

COPC	Sediment Chemistry: Detection Frequency: No. of Samples[a]	Percent	Frequency of Detected Concentrations > SQS: No. of Samples[b]	Percent	No. of Samples with RL > SQS[c]	Frequency of Detected Concentrations > CSL: No. of Samples[d]	Percent	No. of Samples with RL > CSL[e]	Maximum Detected Conc./CSL
Indeno(1,2,3-cd)pyrene	694/787	88	23/787	2.9	3	8/787	1.0	2	2.3
Naphthalene	148/782	19	2/782	0.26	2	2/782	0.26	2	1.7
Phenanthrene	727/792	92	27/792	3.4	0	3/792	0.38	0	3.1
Pyrene	755/792	95	4/792	0.51	0	3/792	0.38	0	1.3
Total HPAH	769/792	97	24/792	3.0	0	3/792	0.38	0	1.4
Total LPAH	731/792	92	6/792	0.76	0	3/792	0.38	0	2.9
Phthalates									
Bis(2-ethylhexyl) phthalate	636/796	80	106/796	13	5	58/796	7.3	2	7.4
Butyl benzyl phthalate	390/786	50	77/786	9.8	79	8/786	1.0	4	8.3
Dimethyl phthalate	136/786	17	2/786	0.25	15	2/786	0.25	7	2.6
Other SVOCs									
1,2,4-Trichlorobenzene	5/780	0.64	1/780	0.13	363	1/780	0.13	131	1.4
1,2-Dichlorobenzene	18/780	2.3	3/780	0.38	113	3/780	0.38	113	10
1,4-Dichlorobenzene	35/780	4.5	3/780	0.38	98	3/780	0.38	21	13
2,4-Dimethylphenol	1/773	0.13	1/773	0.13	224	1/773	0.13	224	10
4-Methylphenol	78/795	9.8	4/795	0.50	12	4/795	0.50	12	6.9
Benzoic acid	69/783	8.8	7/783	0.89	107	7/783	0.89	107	6.9
Benzyl alcohol	14/773	1.8	4/773	0.52	112	2/773	0.26	105	9.2
Hexachlorobenzene	46/783	5.9	6/783	0.77	388	2/783	0.26	108	1.7
n-Nitrosodiphenylamine	23/782	2.9	2/782	0.26	70	2/782	0.26	36	2.8
Pentachlorophenol	12/749	1.6	1/749	0.13	120	0/749	0	32	ne[g]
Phenol	254/795	32	25/795	3.1	5	7/795	0.88	0	2.3
PCBs									
Total PCBs	1205/1290	93	474/1290	37	0	173/1290	13	0	150

a Number of detected concentrations/number of surface sediment samples analyzed for the COPC.

b Number of detected concentrations > SQS/number of surface sediment samples analyzed for the COPC. For individual samples with TOC > 4% or < 0.5%, that sample was tallied as greater than the SQS if the dry weight concentration was greater than the LAET. The number of detected concentrations > SQS includes the number > CSL (i.e., this is not the number of concentrations between the SQS and the CSL).

c Number of samples with RL greater than the SQS. The number of samples with RLs > SQS includes the number > CSL (i.e., this is not the number of samples with RLs between the SQS and the CSL).These chemicals are discussed in the uncertainty analysis.

d Number of detected concentrations > CSL/number of surface sediment samples analyzed for the COPC. For individual samples with TOC > 4% or < 0.5%, the sample was tallied as greater than the CSL if the dry weight concentration was greater than the 2LAET.

e Number of samples with RLs exceeding the CSL. These chemicals are discussed in the uncertainty analysis.

f Maximum concentration of anthracene did not exceed CSL; maximum concentration/SQS equals 1.7.

g Maximum concentration of pentachlorophenol did not exceed CSL; maximum concentration/SQS equals 1.1.

COPC – chemical of potential concern
CSL – cleanup screening level
HPAH – high-molecular-weight polycyclic aromatic hydrocarbon
LPAH – low-molecular-weight polycyclic aromatic hydrocarbon
ne – not exceeded
PCB – polychlorinated biphenyl
SMS – Washington State Sediment Management Standards
SQS – sediment quality standard
SVOC – semivolatile organic compound

Table A.6-2. Sediment chemistry and toxicity test results for samples from the LDW

Location ID	Conc. > SMS Chemical Criteria	COPC(s) with Detected Concentrations Greater than the SQS or CSL[a]	Toxicity Test Exceedance[b]	Agreement in Classification	Final SMS Classification[c]
DUD200	CSL	**4-methylphenol**, bis(2-ethylhexyl) phthalate, butyl benzyl phthalate, total PCBs	no exceedance	no	no exceedance
DUD201	SQS	bis(2-ethylhexyl) phthalate, total PCBs	no exceedance	no	no exceedance
DUD202	CSL	**bis(2-ethylhexyl) phthalate**, butyl benzyl phthalate, total PCBs	no exceedance	no	no exceedance
DUD203	no exceedance	none	no exceedance	yes	no exceedance
DUD204	CSL	**4-methylphenol**, **bis(2-ethylhexyl) phthalate**, butyl benzyl phthalate	no exceedance	no	no exceedance
DUD205	CSL	**4-methylphenol**, **bis(2-ethylhexyl) phthalate**, butyl benzyl phthalate, total PCBs	no exceedance	no	no exceedance
DUD206	no exceedance	none	CSL	no	CSL
LDW-SS2	SQS	fluoranthene	CSL	no	CSL
LDW-SSB2b	SQS	total PCBs	CSL	no	CSL
LDW-SS6	CSL	arsenic, **bis(2-ethylhexyl) phthalate**, **lead**, **total PCBs**, zinc	CSL	yes	CSL
LDW-SSB6a	SQS	total PCBs	SQS	yes	SQS
LDW-SS15	CSL	**mercury**	SQS	no	SQS
LDW-SS16	SQS	total PCBs	SQS	yes	SQS
LDW-SS17	SQS	bis(2-ethylhexyl) phthalate	SQS	yes	SQS
LDW-SS24	CSL	**benzo(a)anthracene**, benzo(a)pyrene, **benzo(g,h,i)perylene**, **total benzofluoranthenes**, **benzyl alcohol**, **chrysene**, dibenzo(a,h)anthracene, **fluoranthene, indeno(1,2,3-cd)pyrene**, **mercury**, total PCBs, phenanthrene, **pyrene**, **total HPAH**, zinc	CSL	yes	CSL
LDW-SS26	SQS	butyl benzyl phthalate, total PCBs	no exceedance	no	no exceedance
LDW-SS29	no exceedance	none	no exceedance	yes	no exceedance
LDW-SS31	CSL	**arsenic**, **zinc**	CSL	yes	CSL
LDW-SS32	SQS	zinc	SQS	yes	SQS
LDW-SS37	CSL	**mercury**, **total PCBs**	CSL	yes	CSL
LDW-SS39	CSL	**mercury**	CSL	yes	CSL

LOCATION ID	CONC. > SMS CHEMICAL CRITERIA	COPC(S) WITH DETECTED CONCENTRATIONS GREATER THAN THE SQS OR CSL[a]	TOXICITY TEST EXCEEDANCE[b]	AGREEMENT IN CLASSIFICATION	FINAL SMS CLASSIFICATION[c]
LDW-SS40	SQS	total PCBs	CSL	no	CSL
LDW-SS49	CSL	**arsenic**, **copper**, zinc	CSL	yes	CSL
LDW-SS50	SQS	total PCBs	CSL	no	CSL
LDW-SS56	CSL	**arsenic**, **total PCBs**, zinc	SQS	no	SQS
LDW-SS57	SQS	total PCBs	CSL	no	CSL
LDW-SS58	SQS	total PCBs	CSL	no	CSL
LDW-SS60	SQS	total PCBs	no exceedance	no	no exceedance
LDW-SS63	no exceedance	none	no exceedance	yes	no exceedance
LDW-SS68	CSL	**hexachlorobenzene**	no exceedance	no	no exceedance
LDW-SS69b	SQS	total PCBs	CSL	no	CSL
LDW-SS70	SQS	bis(2-ethylhexyl) phthalate	SQS	yes	SQS
LDW-SS71	SQS	total PCBs	no exceedance	no	no exceedance
LDW-SS73	CSL	**benzyl alcohol**	SQS	no	SQS
LDW-SS75	SQS	total PCBs	no exceedance	no	no exceedance
LDW-SS77	SQS	arsenic	CSL	no	CSL
LDW-SS85	SQS	total PCBs	no exceedance	no	no exceedance
LDW-SS88	CSL	**mercury**, total PCBs	CSL	yes	CSL
LDW-SS89	CSL	**total PCBs**	no exceedance	no	no exceedance
LDW-SS92	CSL	**total PCBs**	no exceedance	no	no exceedance
LDW-SS106	SQS	total PCBs	no exceedance	no	no exceedance
LDW-SS112	CSL	**arsenic**, butyl benzyl phthalate, fluoranthene, total PCBs	no exceedance	no	no exceedance
LDW-SS114	CSL	**arsenic**, bis(2-ethylhexyl) phthalate, chrysene, fluoranthene, indeno(1,2,3-cd)pyrene, total PCBs	CSL	yes	CSL
LDW-SS115	SQS	chrysene, dibenzo(a,h)anthracene, fluoranthene, phenanthrene, total HPAH	no exceedance	no	no exceedance
LDW-SS119	SQS	butyl benzyl phthalate, total PCBs	no exceedance	no	no exceedance
LDW-SS120	SQS	butyl benzyl phthalate, total PCBs	SQS	yes	SQS
LDW-SS121	CSL	butyl benzyl phthalate, **lead**, total PCBs	no exceedance	no	no exceedance

Location ID	Conc. > SMS Chemical Criteria	COPC(s) with Detected Concentrations Greater than the SQS or CSL[a]	Toxicity Test Exceedance[b]	Agreement in Classification	Final SMS Classification[c]
LDW-SS122	SQS	total PCBs	no exceedance	no	no exceedance
LDW-SS143	CSL	**total PCBs**	no exceedance	no	no exceedance
LDW-SS144	SQS	total PCBs	SQS	yes	SQS
LDW-SS148	SQS	total PCBs	CSL	no	CSL
LDW-SS157	CSL	**benzoic acid**, butyl benzyl phthalate	SQS	no	SQS
LDW-SS158	SQS	total PCBs	no exceedance	no	no exceedance

[a] **Bold** COPCs had detected concentrations greater than the CSL in that sample. Other COPCs had detected concentrations greater than the SQS.

[b] Overall toxicity test exceedance of the SMS (for further details, see Section A.3.2.2).

[c] Overall SMS designation.

COPC – chemical of potential concern

CSL – cleanup screening level

HPAH – high-molecular-weight polycyclic aromatic hydrocarbon

ID – identification

LDW – Lower Duwamish Waterway

PCB – polychlorinated biphenyl

SMS – Washington State Sediment Management Standards

SQS – sediment quality standard

The frequency of COPC concentrations greater than the SQS chemical criteria ranged from 1.1 to 5.2% for metals and trace elements, from 0.25 to 4.9% for PAHs, from 0.25 to 13% for phthalates, and from 0.13 to 3.1% for other SVOCs; the frequency of PCB concentrations greater than the SQS was 37%. All detected chemicals, except anthracene and pentachlorophenol, were above the CSL chemical criteria in at least one sample. The highest ratios of maximum detected concentrations to CSL chemical criteria were for total PCBs (150), silver (44), lead (43), copper (31), cadmium (18), 1,4-dichlorobenzene (13), and arsenic (12). Ratios for all other COPCs were less than or equal to 10.

Table A.6-3 presents the sediment chemistry data for the three COPCs without SMS chemical criteria (nickel, total DDTs, and total chlordane). The frequency of nickel concentrations greater than the SL and ML was 1.0 and 0.52%, respectively. For total DDTs and chlordane, the frequency of concentrations greater than the NOAEL was 0.51 and 9.6%, and the frequency of concentrations greater than the LOAEL was 0.51 and 7.1%, respectively. The ratios of maximum detected concentrations to the ML or NOAEL were 2.5, 2.7, and 48 for nickel, total DDTs, and total chlordane, respectively. The detected chemical concentrations greater than the DMMP guidelines or TRV values are shown as point locations on Map A.6-2.

Application of SMS criteria to assess adverse effects from sediment-associated chemicals requires an assessment of both the magnitude and spatial extent of the contamination. A spatial analysis of potential effects was performed using Thiessen polygons, a method commonly used to illustrate spatial variability in sampling intensity and to extrapolate results from small areas (sample points) to larger areas. The Thiessen polygon associates each point on a plane with the closest sampling location for which an empirical value is available (Burmaster and Thompson 1997). In effect, this algorithm assumes that the concentration at any point where an empirical value is not available has not been made is the same as the concentration in the sample closest to that point. Assumptions regarding spatial homogeneity within polygons introduce uncertainty in all areal percentages discussed in this section. In areas with lower sample density, the uncertainty in this assumption increases (i.e., all points within a polygon are less likely to have the same characteristics as the point that defined the polygon category). The uncertainty section includes a discussion of the spatial analysis using Thiessen polygons relative to another interpolation approach.

Map A.6-3 presents areas of the LDW that are categorized as ≤ SQS (white), > SQS and ≤ CSL (yellow), and > CSL (red) based on a combination of the sediment chemistry data and toxicity test results. The overall SMS designation for each location is presented in Table A.6-2. The percent area in each of those three categories is 75%, 18%, and 7%, respectively.

Table A.6-3. Detection frequencies and frequencies of detected concentrations above the SL/NOAEL and ML/LOAEL for COPCs without SMS chemical criteria

COPC	Sediment Chemistry								
	Detection Frequency		Frequency of Detected Concentrations > SL or NOAEL			Frequency of Detected Concentrations > ML or LOAEL			Maximum Detected Conc./ML or LOAEL
	No. of Samples[a]	Percent	No. of Samples[b]	Percent	No. of Samples with RL > SL or NOAEL[c]	No. of Samples[d]	Percent	No. of Samples with RL > ML or LOAEL[e]	
Nickel	775/775	100	9/775	1.2	0	4/775	0.52	0	2.5
Total DDTs	78/197	40	1/197	0.51	1	1/197	0.51	0	2.7
Total chlordane[f]	33/197	17	19/197	9.6	79	14/ 197	7.1	61	48

[a] Number of detected concentrations/number of surface sediment samples analyzed for the COPC.

[b] [Number of detected concentrations > DMMP SL or NOAEL]/number of surface sediment samples analyzed for the COPC. The number of detected concentrations > SL or NOAEL includes the number > ML or LOAEL (i.e., this is not the number of concentrations between the SL and the ML or between the NOAEL and the LOAEL).

[c] Number of samples with RLs exceeding the DMMP SL or NOAEL. These chemicals are discussed in the uncertainty analysis.

[d] [Number of detected concentrations > DMMP ML or LOAEL]/number of surface sediment samples analyzed for the COPC.

[e] Number of samples with RLs exceeding the DMMP ML or LOAEL. These chemicals are discussed in the uncertainty analysis.

[f] Total chlordane includes the calculated total chlordane for Phase 2 data and chlordane as reported in a subset of historical data (King County 1999b).

COPC – chemical of potential concern

DMMP – Dredged Material Management Program

LOAEL – lowest-observed-adverse-effect level

ML –maximum level (DMMP)

NOAEL – no-observed-adverse-effect level

SL – screening level (DMMP)

SMS – Washington State Sediment Management Standards

TRV – toxicity reference value

The numbers of detected concentrations greater than the SQS and CSL based solely on sediment chemistry are shown in Table A.6-4. As shown in this table, PCBs and BEHP were the two COPCs with the highest numbers of detected concentrations greater than the CSL chemical criteria. Therefore, spatial analyses using Thiessen polygons were also performed for total PCBs and BEHP individually. The spatial distributions of chemical concentrations for these two chemicals relative to SQS and CSL chemical criteria are presented in Maps A.6-4 and A.6-5 and are discussed below.

Table A.6-4. Number of detected concentrations for each COPC greater than SQS and CSL chemical criteria based on sediment chemistry data

COPC	Number of Detected Concentrations > SQS and ≤ CSL	Number of Detected Concentrations > CSL
Total PCBs	301	173
Bis(2-ethylhexyl) phthalate	48	58
Mercury	14	23
Lead	2	19
Zinc	26	16
Copper	0	12
Cadmium	2	11
Silver	0	10
Fluoranthene	31	8
Butyl benzyl phthalate	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
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
Hexachlorobenzene	4	2
Benzyl alcohol	2	2
Chrysene	23	1

COPC	Number of Detected Concentrations > SQS and ≤ CSL	Number of Detected Concentrations > CSL
1,2,4-Trichlorobenzene	0	1
2,4-Dimethylphenol	0	1
Anthracene	2	0
Pentachlorophenol	1	0

COPC – chemical of potential concern

CSL – cleanup screening level

HPAH – high-molecular-weight polycyclic aromatic hydrocarbon

LPAH – low-molecular-weight polycyclic aromatic hydrocarbon

PCB – polychlorinated biphenyl

SQS – sediment quality standard

Using Thiessen polygons and the chemistry data, total PCB concentrations were greater than the SQS and less than or equal to the CSL chemical criteria in approximately 17% of the total LDW area; total PCBs were the only COPC with concentrations greater than the SQS chemical criteria in approximately 14% of the total LDW area. In addition, total PCB concentrations were greater than the CSL chemical criteria in approximately 3% of the total LDW area (Map A.6-3). Total PCB concentrations were less than the SQS or CSL chemical criteria in approximately 80% of the total LDW. Total PCB concentrations greater than the SQS or the CSL chemical criteria were mostly reported between RM 0.4 and RM 0.6, in Slip 4, and between RM 2.9 and RM 3.7 (Map A.6-4). Thiessen polygons associated with locations where total PCBs were not detected are shown on Map A.6-4 in gray.

BEHP concentrations greater than the SQS or the CSL chemical criteria were mostly reported near RM 0.4 (Map A.6-5). BEHP concentrations were greater than the SQS and less than or equal to the CSL in approximately 4% of the total LDW area and greater than the CSL in approximately 2% of the total LDW area. BEHP concentrations were less than the SQS or CSL chemical criteria in approximately 94% of the total LDW. Thiessen polygons associated with locations where BEHP was not detected are shown on Map A.6-5 in gray, regardless of whether the BEHP RL for that location was above or below the SQS or CSL chemical criteria.

Uncertainties Associated with Sediment Risk Estimates

This section presents uncertainties in the sediment-based risk characterization for the benthic invertebrate community. The uncertainties are discussed separately for the problem formulation, exposure assessment, and effects assessment.

Problem Formulation

The benthic invertebrate community as a whole (except for the larger, more mobile species, such as crabs, which were assessed separately) was selected as an ROC because the community encompasses all benthic invertebrates as a functional group, not as individual species. Because the benthic invertebrate community is the selected receptor, this approach does not address risks or toxicity to each individual species that is or could be present in the sediment environment. Instead, the receptor selection addresses effects at the community level, reflecting the diversity of species and ecological functions that are achieved with various benthic invertebrate assemblages. This receptor group, assessment endpoints (survival, growth and reproduction) and the sediment regulatory framework (SMS criteria) are aimed at protecting community function, not individual species.

AETs, which form the basis for SMS criteria and some of the DMMP guidelines, exist for about 20% of the chemicals that have been detected in the LDW. Therefore, chemicals without such criteria, guidelines, or other relevant toxicity information or chemicals with guidelines not derived on the basis of toxicity were not identified as COPCs during the problem formulation. However, it is likely that locations with the highest potential for adverse effects were adequately identified because criteria and guidelines are available for chemicals within most of the chemical groups (e.g., metals, PAHs, phthalates, SVOCs, PCBs, and pesticides) that are generally considered in CERCLA investigations. The importance of chemicals lacking criteria or other TRVs to the overall risk to benthic invertebrate communities was evaluated by identifying locations where the maximum detected concentration of each of these chemicals was greater than 10 times the mean concentration calculated using only detected values. A factor of 10 was selected as an arbitrary means to identify those chemicals with highly variable concentrations. Sixteen chemicals were identified through this process, including polychlorinated terphenyls (PCTs); thallium, barium, and manganese; and SVOCs such as carbazole and dibenzothiophene. All but three of the locations where the concentrations of these chemicals were elevated relative to the mean were already among the stations at which the concentrations of one or more chemicals exceeded the CSL or SQS. Chemicals without criteria at the three locations without any SQS exceedances were PCTs and barium. Therefore, the lack of criteria or TRVs for several chemicals in sediment is not likely to substantially alter the overall risk conclusions.

Finally, by using sediment criteria to judge the potential for adverse effects on the benthic invertebrate community, the risk assessment does not address the potential for other types of more subtle effects, such as biochemical changes. While the SMS is assumed to address the likelihood of adverse effects, such as reduced survival and growth, assessment of biochemical endpoints that may or may not relate to adverse effects on the benthic invertebrate community as a whole, is beyond the scope of this risk assessment.

Exposure Assessment

Uncertainties in the exposure assessment for the benthic invertebrate community were associated with the following factors.

Depth of biologically active zone. Some benthic invertebrate species (e.g., clams) may burrow deeper than 15 cm, which was the surface sediment threshold used in this

Phase 2 ERA to define the biologically active zone where the majority of the benthic invertebrate community resides. A risk characterization for these animals could have a different outcome than that presented in Section A.6.1.1 if: 1) concentrations in sediment above 15 cm were markedly different than those in sediment between 15 and 40 cm (depth that clams have been found in the LDW), or 2) the chemical sensitivity of animals living below 15 cm is markedly different than the chemical sensitivity of animals living above 15 cm, on which the existing chemical criteria and guidelines are based. The Phase 2 RI presents an evaluation of the potential exposure of these subsurface sediments to human and ecological receptors and assesses risks associated with these exposures.

Relationships among sediment chemistry, toxicity, and actual *in-situ* effects. The use of chemical criteria and toxicity testing to assess *in-situ* effects is uncertain. For example, adverse effects on the benthic invertebrate community may occur in areas with chemical concentrations less than SQS chemical criteria and may not occur in areas with concentrations greater than SQS or CSL chemical criteria. Factors such as site-specific bioavailability, mixtures of chemicals with or without criteria, and species-specific sensitivities may contribute to this uncertainty. Moreover, the SMS provides chemical-specific criteria to assess the risks from individual chemicals. Although these criteria were developed from field data in which mixtures of chemicals are common, the chemical-specific criteria do not assess the cumulative risks to benthic invertebrates from exposure to multiple chemicals with potentially synergistic or antagonistic effects.

Frequency of analyses of chemicals in surface sediment samples. In addition, not all chemicals evaluated in this ERA were analyzed in all surface sediment samples. While most COPCs have been analyzed in surface sediment at more than 700 locations, some, such as total DDTs and chlordane, have been analyzed at fewer locations (e.g., organochlorine pesticides were analyzed at 197 locations; Table A.3-1). The certainty regarding the risk characterization for these chemicals is lower compared to chemicals analyzed more frequently. However, because all COPCs were analyzed throughout the LDW as part of one or more reconnaissance-level sampling event and more recent sampling events focused on specific potential sources, the potential effect of this uncertainty on overall risk conclusions is likely to be low.

Elimination of COPCs based on the 5% detection frequency screen. Nineteen chemicals were not selected as COPCs because they were detected in fewer than 5% of the baseline surface sediment samples. To assess the uncertainty resulting from this approach, a cluster analysis was conducted for each of the chemicals that were screened out. The cluster analysis helped to determine whether the locations where these rarely detected chemicals were detected were clustered together. If detections were clustered, these chemicals might be of greater concern than if they had been widely dispersed or randomly distributed. The majority of the screened-out chemicals (14) had fewer than eight detected concentrations; six of the chemicals were detected only once. Among the other eight chemicals with fewer than eight detected concentrations, only endrin aldehyde was detected at more than one location within 0.1 RM of each other, all within an early action area. Four of the five chemicals with more than eight detected concentrations (2-dichlorobenzene, benzyl alcohol, n-nitrosodiphenylamine, and pentachlorophenol) did not cluster but were scattered throughout the LDW. Detected concentrations of the fifth remaining chemical (1,4-dichlorobenzene) exceeded the SQS in three areas of the LDW: along the east bank between RM 0.3 and RM 0.7, between RM 3.3 and RM 3.8, and between RM 4.8 and RM 4.9. All of these locations are in early action areas.

RLs greater than criteria or TRVs. Twenty-seven chemicals had non-detected results with RLs greater than the corresponding SQS/CSL chemical criteria in at least one sediment sample in the baseline surface sediment dataset. These chemicals can be divided into three groups: PAHs (9 chemicals), phthalates (5 chemicals), and other SVOCs (13 chemicals). The detected results and RLs for each of these chemicals were compared to the corresponding SQS and CSL chemical criteria, and the results are presented in Table A.6-5. The results are presented separately for the 2004/2005 data collected as part of the Phase 2 RI, which includes all surface sediment samples collected by LDWG in 2004 and 2005, and the non-LDWG data, which includes the portion of the dataset that was presented in the Phase 1 RI (Windward 2003a) and data collected by parties other than LDWG in the time period since the Phase 1 RI.

The sample-specific RL is based on the lowest point of the calibration curve associated with each analytical batch of samples. The most common reason for elevated RL values is sample extract dilution. For example, elevated RLs for some chemicals in some areas reflect the greater degree of analytical dilution required for quantification of other analytes, such as PCBs. In addition, there are analytes known to be analytically difficult. These compounds tend to have chemical characteristics that differ from those of other analytes being analyzed using the same method. For example, benzoic acid, benzyl alcohol, phenols, and n-nitrosodiphenylamine are all more chemically reactive than the other SVOCs analyzed by EPA (2003a). More reactive compounds can be difficult to extract and often degrade during analysis.

PAHs and phthalates were detected relatively frequently in surface sediments, with detection frequencies ranging from 11 to 93% for PAHs and from 3.0 to 76% for phthalates. The majority of the RL values reported for these compounds were below the SQS and CSL chemical criteria. LDWG data and non-LDWG data can be compared in terms of the frequency with which RL values were greater than the chemical criteria. The LDWG data had much lower frequencies of RL values greater than the chemical criteria. No RLs above the SQS were reported for 2-methyl naphthalene, acenaphthylene, benzo(g,h,i)perylene, indeno(1,2,3-cd)pyrene, and naphthalene in the LDWG dataset because the laboratory, where possible, conducted additional analyses to increase the sensitivity of the analyses. RLs greater than the SQS chemical criteria for PAHs and phthalates primarily resulted from analytical dilution of the sample extracts.

Table A.6-5. Summary of chemical data for COPCs with at least one reporting limit greater than SQS/CSL chemical criteria

Chemical	Dataset[a]	Total No. of Samples Analyzed	Detection Frequency (%)	Detected Results		Non-Detected Results			RL > SQS Frequency (%)
				> SQS and ≤ CSL	> CSL	RL ≤ SQS	RL > SQS and ≤ CSL	RL > CSL	
PAHs									
2-Methylnaphthalene	2004/2005 LDWG	193	24	0	1	147	0	0	0.0
	non-LDWG	589	16	0	2	487	6	3	1.5
Acenaphthene	2004/2005 LDWG	193	34	2	2	126	1	0	0.52
	non-LDWG	599	39	14	1	352	8	4	2.0
Acenaphthylene[b]	2004/2005 LDWG	193	30	0	0	136	0	0	0.0
	non-LDWG	589	11	0	0	522	0	3	0.51
Benzo(g,h,i)perylene	2004/2005 LDWG	193	73	2	2	53	0	0	0.0
	non-LDWG	594	86	7	5	78	4	3	1.2
Dibenzo(a,h)anthracene	2004/2005 LDWG	193	31	3	1	130	3	0	1.6
	non-LDWG	599	57	12	3	243	7	9	2.7
Dibenzofuran	2004/2005 LDWG	193	28	0	2	139	1	0	0.52
	non-LDWG	598	32	7	1	393	8	4	2.0
Fluorene	2004/2005 LDWG	193	39	1	2	118	0	0	0.0
	non-LDWG	599	50	10	1	292	8	1	1.5
Indeno(1,2,3-cd)pyrene	2004/2005 LDWG	193	93	4	2	14	0	0	0.0
	non-LDWG	594	87	11	6	76	1	2	0.51
Naphthalene	2004/2005 LDWG	193	26	0	1	143	0	0	0.0
	non-LDWG	589	17	0	1	489	0	2	0.34
Phthalates									
Bis(2-ethylhexyl) phthalate	2004/2005 LDWG	193	76	5	3	47	0	0	0.0
	non-LDWG	603	81	43	55	108	3	2	0.83
Butyl benzyl phthalate	2004/2005 LDWG	193	35	8	0	115	11	4	7.8
	non-LDWG	593	55	61	8	202	64	0	11

Chemical	Dataset[a]	Total No. of Samples Analyzed	Detection Frequency (%)	Detected Results		Non-Detected Results			RL > SQS Frequency (%)
				> SQS and ≤ CSL	> CSL	RL ≤ SQS	RL > SQS and ≤ CSL	RL > CSL	
Diethyl phthalate	2004/2005 LDWG	193	12	0	0	168	1	0	0.52
	non-LDWG	603	3.0	0	0	580	3	2	0.83
Dimethyl phthalate[b]	2004/2005 LDWG	193	14	0	0	165	0	1	0.52
	non-LDWG	593	18	0	2	470	8	6	2.4
Di-n-octyl phthalate	2004/2005 LDWG	193	3.6	0	0	179	0	0	0.0
	non-LDWG	603	6.8	0	0	499	8	0	1.3
Other SVOC									
1,2,4-Trichlorobenzene	2004/2005 LDWG	193	0.0	0	0	155	16	22	20
	non-LDWG	587	0.17	0	1	257	216	109	55
1,2-Dichlorobenzene[b]	2004/2005 LDWG	193	0.5	0	0	174	0	18	9.3
	non-LDWG	587	2.9	0	3	475	0	95	16
1,4-Dichlorobenzene	2004/2005 LDWG	193	0.5	0	0	175	14	3	8.8
	non-LDWG	587	5.8	0	3	472	63	18	14
2,4-Dimethylphenol[b]	2004/2005 LDWG	193	0.0	0	0	162	0	31	16
	non-LDWG	580	0.2	0	1	386	0	193	33
2-Methylphenol	2004/2005 LDWG	193	1.0	0	0	175	0	16	8.3
	non-LDWG	592	0.2	0	0	490	0	101	17
4-Methylphenol[b]	2004/2005 LDWG	193	8.8	0	0	176	0	0	0.0
	non-LDWG	602	10	0	4	529	0	12	2.0
Benzoic acid[b]	2004/2005 LDWG	193	21	0	1	133	0	20	10
	non-LDWG	590	4.9	0	6	474	0	87	15
Benzyl alcohol	2004/2005 LDWG	193	4.7	1	2	166	0	18	9.3
	non-LDWG	580	0.9	1	0	481	7	87	16
Hexachlorobenzene	2004/2005 LDWG	194	6.2	0	2	157	16	9	13
	non-LDWG	589	5.8	4	0	192	264	99	62
Hexachlorobutadiene	2004/2005 LDWG	193	0.0	0	0	178	8	7	7.8
	non-LDWG	589	0.0	0	0	456	49	84	23

Chemical	Dataset[a]	Total No. of Samples Analyzed	Detection Frequency (%)	Detected Results		Non-Detected Results			RL > SQS Frequency (%)
				> SQS and ≤ CSL	> CSL	RL ≤ SQS	RL > SQS and ≤ CSL	RL > CSL	
n-Nitrosodiphenylamine[b]	2004/2005 LDWG	193	7.8	0	0	177	0	1	0.52
	non-LDWG	589	1.4	0	2	512	34	35	12
Pentachlorophenol	2004/2005 LDWG	193	3.6	1	0	168	7	11	9.3
	non-LDWG	556	0.9	0	0	449	81	21	18
Phenol	2004/2005 LDWG	193	22	2	3	151	0	0	0.0
	non-LDWG	602	35	16	4	385	5	0	0.83

a Non-LDWG refers to data collected by parties other than LDWG.

b The SQS and CSL chemical criteria are the same for these chemicals. Therefore, all values greater than either the SQS or CSL are presented as being above the CSL, except in cases where the AET was used because of TOC values outside the acceptable range.

CSL – cleanup screening level

LDWG – Lower Duwamish Waterway Group

PAH – polycyclic aromatic hydrocarbon

RL – reporting limit

SQS – sediment quality standard

SVOC – semivolatile organic compound

The group of compounds labeled as "other SVOCs" included the following chemicals: chlorobenzenes, phenol, methyl phenols, pentachlorophenol, benzoic acid, benzyl alcohol, hexachlorobutadiene, hexachlorobenzene, and n-nitrosodiphenylamine. This group includes compounds that are analytically difficult to quantify at the levels required for comparison to SQS chemical criteria and are generally very rarely detected. Only benzoic acid and phenol have been detected at frequencies greater than 10% in either dataset. The 2004/2005 LDWG data had consistently lower frequencies of RL values greater than the SQS chemical criteria than did the non-LDWG data because additional selected ion monitoring analyses were conducted to improve the sensitivity of the analyses of these compounds. Except for hexachlorobenzene and n-nitrodiphenylamine, the frequency of RL values above the SQS chemical criteria in the LDWG data was approximately half the frequency observed for the non-LDWG data. For hexachlorobenzene, the frequency of RL values greater than the SQS chemical criteria in the LDWG data was 20% of the frequency in the non-LDWG data. For n-nitrosodiphenylamine, the frequency in the LDWG data was 5% of that observed in the non-LDWG data. Nevertheless, there were many RL values above the SQS and CSL chemical criteria for these compounds in both the 2004/2005 LDWG and the non-LDWG datasets. The highest frequencies of RL values greater than the chemical criteria were reported for 1,2,4-trichlorobenzene, 2,4-dimethylphenol, and hexachlorobenzene. For the LDWG 2004/2005 data, more sensitive analytical methods were used whenever RL values were greater than the SQS with no detected values above the SQS chemical criteria.

Because chemical criteria are not available for organochlorine pesticides, NOAEL and LOAEL values were developed for the two organochlorine pesticide COPCs (total DDTs and total chlordane) for benthic invertebrates.[68] The number of detected results and RLs that exceeded these values are summarized in Table A.6-6. One RL value for total DDTs was greater than the total DDT NOAEL. Sixty-one samples had total chlordane RLs greater than both the NOAEL and LOAEL values. Elevated RL values for organochlorine pesticides generally reflect the presence of probable analytical interference in the analysis because of the presence of PCB congeners. The LDWG dataset represents 91 of the total of 197 samples analyzed for organochlorine pesticides. All of the organochlorine pesticide results for these samples were reported as not detected with elevated RLs because of analytical interference resulting from the presence of PCB congeners. All of the 61 samples with total chlordane RLs above the LOAEL had detected results of other chemicals greater than the SQS. Of the 18 samples with total chlordane RLs above the NOAEL, 15 did not have detected concentrations of other chemicals greater than the SMS.

[68] Chemical criteria were also unavailable for nickel. Nickel was always detected in surface sediment samples, and thus RLs were not an issue for nickel.

Table A.6-6. Summary of detected results and RLs for total DDTs and total chlordane relative to NOAELs and LOAELs

Chemical	Total No. of Samples Analyzed	Detection Frequency (%)	Detected Results > NOAEL and ≤ LOAEL	Detected Results > LOAEL	RL > NOAEL and ≤ LOAEL	RL > LOAEL
Total DDTs	197	40	0	1	1	0
Total chlordane	197	17	5	14	18	61

LOAEL – lowest-observed-adverse-effect level

NOAEL – no-observed-adverse-effect level

RL – reporting limit

The spatial distributions of the three chemicals with the highest frequencies of RL values greater than the chemical criteria (i.e., 1,2,4-trichlorobenzene, 2,4-dimethyl phenol, and hexachlorobenzene) are presented in Maps A.6-6, A.6-7, and A.6-8. For all three compounds, samples with RLs greater than the SQS chemical criteria occurred throughout the LDW with no spatial relationship between the detected results that were greater than the SQS and CSL chemical criteria and the RLs that were greater than the SQS chemical criteria, although these chemicals were detected infrequently. 1,2,4-trichlorobenzene and 2,4-dimethyl phenol each had one detected concentration above the CSL chemical criteria in the non-LDWG dataset. Hexachlorobenzene had four detected values above the SQS chemical criteria in the non-LDWG dataset and two detected values above the CSL chemical criteria in the LDWG dataset.

Locations where there were no detected concentrations greater than the chemical criteria but RLs were greater than the SQS/CSL chemical criteria are shown on Map A.6-9. Only two of the 294 locations where this occurs were from the LDWG 2004/2005 dataset. The majority of the locations were associated with RL values for 1,2,4-trichlorobenzene, hexachlorobenzene, and 2,4-dimethyl phenol. These three chemicals were responsible for 94% of the locations where only RL values were greater than the SQS or CSL chemical criteria. Two of these chemicals, 1,2,4-trichlorobenzene and 2,4-dimethylphenol, were never reported as detected in the LDWG 2004/2005 surface sediment dataset, despite increased efforts to improve the sensitivity of the analytical method, which resulted in a reduced frequency of RL values above the SQS chemical criteria. Hexachlorobenzene was detected in approximately 6% of the samples in both the LDWG and non-LDWG datasets. The increased sensitivity of the analysis in the LDWG 2004/2005 dataset reduced the frequency of RL values above the SQS chemical criteria. The RL values above SQS chemical criteria for these compounds appear to reflect analytical difficulties with the analysis, and the paucity of detected concentrations suggests that these chemicals are not present in the LDW at the concentrations represented by the RL.

There were 18 locations with RL values greater than the SQS chemical criteria associated with other chemicals. The majority of those locations were associated with n-nitrosodiphenylamine RLs above the SQS chemical criteria (14 locations).

For PAHs and phthalates, the RL values greater than the SQS chemical criteria occur infrequently and are associated with samples that required analytical dilutions because of the presence of other chemicals with detected concentrations above the SQS chemical criteria. Thus, RLs for these chemicals had little effect on risk conclusions.

Spatial analysis using Thiessen polygons. Thiessen polygons were used in the ERA to estimate the area potentially affected by exceedances of SMS at each point for which chemical concentrations were compared to SMS, or for which bioassays were performed, after the chemical and biological analyses of LDW sediment at all points were combined (Section A.6.1.1.1). There is uncertainty associated with methods for interpolation of point values to area values, including the Thiessen polygon method. To assess this uncertainty, results of this interpolation method were compared to results of the same analysis using a different method, inverse distance weighting (IDW). IDW estimates chemical concentrations for each model cell in a grid surface as a weighted average of the sample concentrations that are defined as the neighbors of that cell. The weights are a function of the inverse distance of the cell from each neighboring sample concentration. Thiessen polygons are the result of an algorithm (Voronoi tessellation) that defines polygon boundaries such that any arbitrary location within a polygon is closer to its associated sample location than to any other sample location. The sample concentration within each Thiessen polygon is assumed to be uniformly distributed across the area of the polygon, so that any of the aforementioned arbitrary locations is estimated to have the same chemical concentration as its associated sample location. Each method is known to have a degree of uncertainty regarding the reliability of its estimates; comparison of the results of the two methods provides an indication of the uncertainty associated with application of Thiessen polygons to estimate areas affected when only point values are known. There is no way with the existing dataset to quantify the deviation of either method from reality.

Map A.6-10 illustrates differences in estimates of PCBs generated by the two methods in one area within the LDW. This area is representative of the types of relative over- and underestimation that occur at numerous locations throughout the LDW based on a comparison of the approaches. The differences described here can be observed to varying degrees throughout the waterway. As illustrated on Map A.6-10, the estimated concentration within an individual Thiessen polygon (excluding the specific location of the sample itself where the concentration is known) may be high or low relative to IDW-estimated concentrations for any fraction of the area within the polygon. Moreover, estimates for multiple Thiessen polygons can be wholly overestimated by IDW, depending on the point value for the polygon and the density of the sampling locations.

Individual polygons are marked in Map A.6-10 and discussed here as examples of the degree of uncertainty resulting from the use of each method. In the areas marked as "A," sampling stations with chemical concentrations < SQS are surrounded by or adjacent to stations with concentrations > SQS. Within these polygons, the area of chemical exceedance estimated using IDW is greater than the area estimated using Thiessen polygons. Two of these areas are notable because the cells representing the specific sampling locations where concentration < SQS are estimated as locations of chemical exceedances by IDW. The areas marked as "B" are areas estimated to have chemical exceedances using IDW that are smaller than the areas estimated to have exceedances by Thiessen polygons because the concentrations at surrounding points are < SQS. Underestimation of chemical exceedance by IDW relative to Thiessen polygon chemical exceedance estimation is less common throughout the LDW because of the positive skew of chemical concentrations and the fixed nature of exceedance criteria. Extreme values have a significant effect on the moving average of the IDW surface, affecting a relatively large area regardless of nearby samples. For Thiessen polygons, the effect of extreme concentration values is limited to the boundaries of the polygons associated with the extreme values.

Effects Assessment

The uncertainty in the effects assessment for the benthic invertebrate community was associated with the use of SMS chemical criteria, DMMP guidelines, or TRVs to assess the potential for a biological effect. Uncertainties associated with using an alternative approach (a tissue-residue approach) to evaluate PAHs and PCBs were also considered as another line of evidence. These uncertainties are discussed below.

Use of SMS criteria, DMMP guidelines, and TRVs to assess biological effects—The likelihood of adverse effects to benthic organisms from chemicals associated with sediments was assessed using two approaches. In the first approach, surface sediment chemical concentrations were compared to SQS/CSL chemical criteria, DMMP guidelines (nickel), or TRVs (total DDTs and total chlordane). In the second approach, site-specific sediment toxicity test results were compared to SMS biological effects criteria.

The chemical criteria, guidelines, and TRVs used in the first approach were based on test species that represent a small portion of the diverse benthic invertebrate community present in the LDW, although the test species included crustaceans, which are considered to represent one of the taxonomic groups most sensitive to chemical exposure (Hyland et al. 1999). In addition, the benthic invertebrate community AETs, which were the basis for several SQS or CSL chemical criteria, incorporated invertebrates with different feeding strategies and habitat requirements and therefore represent COPC concentrations likely to be protective of the benthic invertebrate community as a whole. There is some uncertainty associated with the benthic invertebrate community AETs because these values are based on the abundance of several major benthic infaunal taxa (mollusks, crustaceans, polychaetes) and do not address the potential for effects on species diversity, relative abundances of different taxa, the success of rare species, and other benthic invertebrate community metrics. Thus, potential effects on some LDW benthic species may not be addressed by these criteria and guidelines; consequently, there is some uncertainty associated with the risk estimates.

The SMS criteria in WAC 173-204 were developed for application to marine sediments. The LDW is an estuarine system with both spatial and temporal variation in salinity. Salinity in the porewater of the surface sediment samples collected for toxicity testing between RM 0.0 and RM 4.9 ranged from 20 to 30 ppt. Thus, although all sediments within LDW are not fully marine, salinity is relatively high because of salt wedge intrusion, and SMS criteria should be applicable.

A fundamental distinction between the two approaches used to estimate risks to the benthic invertebrate community is that one approach (i.e., use of chemical criteria and guidelines) estimates effects based on a comparison to detected chemical concentrations, while the other approach (i.e., sediment toxicity tests) assesses the potential for effects directly through the exposure of test organisms to surface sediment samples collected from the LDW. SMS chemical criteria and some of the DMMP guidelines were developed using the AET approach described in Section A.3.2.1. 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 for its derivation.

Note that SMS chemical criteria were developed for specific chemicals based on AETs empirically derived from a dataset of field-collected sediment samples that contained diverse chemical mixtures and that were analyzed for both chemistry and toxicity. Therefore, the AETs do not reflect a cause-and-effect relationship for specific chemicals.

Two published studies have assessed the ability of selected AETs to estimate adverse effects in Puget Sound (Barrick et al. 1988; Gries and Waldow 1996). The study by Barrick et al. (1988) calculated overall reliability values[69] between 50 and 96% for benthic, amphipod, Microtox, and oyster larvae AETs. The study by Gries and Waldow (1996) calculated overall reliability values between 65 and 85% for amphipod and echinoderm AETs.

For the LDW, sediment toxicity test results from the 46 Phase 2 locations in the baseline ERA dataset were compared to estimates of toxicity made using only chemical criteria (Table A.6-2). Seven locations assessed by King County (2000a) were also included in the comparison. Site-specific sediment toxicity tests were conducted with *Eohaustorius estuarius* (amphipod), *Neanthes arenaceodentata* (polychaete), and *Mytilus galloprovincialis* (bivalve) using sediment from 46 Phase 2 surface sediment locations within the baseline dataset. The King County sediment toxicity tests were

[69] Overall reliability was calculated as the percentage of all "hit" (i.e., > SQS biological effects criteria) and "no hit" (i.e., ≤ SQS biological effects criteria) samples that were correctly predicted, and thus did not distinguish between SQS and CSL levels of toxicity.

conducted with *Rhepoxynius abronius* (amphipod), *Neanthes arenaceodentata* (polychaete), and echinoderm (*Dendraster excentricus*). Toxicity test responses were evaluated using the toxicity test rules established by SMS to classify responses as exceeding either the SQS (WAC 173-204-320(3)) or the CSL (WAC 173-204-520(3)) (for further details, see Section A.3.2.2).

If the individual SMS classifications are compared considering the level of toxicity (i.e., SQS or CSL), then the chemical and biological criteria concurred at 18 of the 53 locations tested (34%). If the comparison is done by counting the number of locations where the biological effects and chemical predicted either "hits" (either SQS or CSL exceedances) or "no hits" (no SQS exceedances), then the chemical and biological criteria concurred at 31 of the 53 locations tested (58%).

Tissue residue approach for PAHs and PCBs. The numeric standards of SMS and site-specific sediment toxicity tests were used to assess the risks from PAHs and PCBs to the benthic invertebrate community. EPA and Ecology requested that an alternative approach, the tissue-residue approach, also be evaluated in the uncertainty analysis for both PAHs and PCBs. This approach is presented below.

Three studies with mollusks (Borchert et al. 1997; Eertman et al. 1995; Roper et al. 1997) evaluated tissue concentrations of individual PAHs (fluoranthene and benzo(a)pyrene) or a mixture of PAHs associated with adverse effects. Concentrations of benzo(a)pyrene in all benthic invertebrate tissue (market basket samples) did not exceed the LOAEL of 302 µg/kg ww (Eertman et al. 1995), and concentrations of LPAH and HPAH in LDW benthic invertebrate tissue (market basket samples) did not exceed the NOAELs based on a mixture of PAHs (Roper et al. 1997). Fluoranthene in LDW benthic invertebrate tissue (market basket samples) exceeded the LOAEL of 222 µg/kg ww (Eertman et al. 1995) at five locations (B6a, B8a, B3b, B4b, and B6b) in the LDW. Sediment concentrations of several chemicals, including PCBs, PAHs, and metals, exceeded their SMS criteria (PCBs exceeded the SQS and CSL, PAHs exceeded the SQS, and metals exceeded the SQS and CSL) at these five locations, indicating that all locations with potential effects were identified by the primary method for the assessment of risks to the benthic invertebrate community in this ERA.

Seven studies (Boese et al. 1995; Duke et al. 1970; Hansen et al. 1974b; Lowe et al. 1972; Nimmo et al. 1974; Peterson et al. 1994; Sanders and Chandler 1972) evaluated the adverse effects of PCBs on decapods or mollusks. The lowest LOAEL was 1,100 µg/kg ww, reported for Aroclor 1016. Total PCB concentrations in benthic invertebrate tissue samples at two intertidal locations (1,700 µg/kg ww at B5a-2 and 5,900 µg/kg ww at B8a) exceeded this LOAEL, although it should be noted that Aroclor 1016 was never detected in LDW benthic invertebrate tissue samples. Concentrations of several other chemicals exceeded their SMS criteria at these two locations as well (PCBs exceeded the SQS and CSL, PAHs exceeded the SQS, and metals exceeded the SQS and CSL).

Therefore, given both the limits of available TRVs and the fact that tissue concentrations were generally below the available critical tissue residue-based TRVs for PAHs and PCBs, use of this alternative method for risk evaluation would not have estimated the potential for effects resulting from PAHs or PCBs at additional locations. The absence of TRVs for other chemicals that might occur in the LDW prevents a full exploration of risks to the benthic invertebrate community with this method. Application of the SMS to identify COPCs and areas of concern for the benthic invertebrate community is likely to have captured any areas or chemicals presenting risk to the benthic invertebrate community.

Risk Conclusions

The potential for adverse effects on benthic invertebrate communities was evaluated based on site-specific toxicity tests and on comparisons of chemical concentrations in surface sediment to SQS/CSL chemical criteria, toxicologically based DMMP guidelines, and TRVs. The potential adverse effects included in the existing criteria are reduced survival, abnormal development, and reduced growth at the individual level and altered ecological function at the community level.

In Phase 2, toxicity was tested using three toxicity tests at 48 point locations within the LDW. Nearly all of these locations were selected for toxicity testing because of exceedances of the SQS or CSL chemical criteria.[70] Based on SMS rules, 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, and 19 samples (39.6%) exceeded the CSL biological effects criteria. Because these tests are direct measures of effect, the uncertainty in their interpretation is low at the specific locations tested.

At other point locations in the LDW with chemical concentrations greater than SQS or CSL chemical criteria and other guidelines, the estimation of effects is more uncertain. According to SMS, locations with all chemical concentrations less than or equal to the SQS chemical criteria are defined as having no acute or chronic adverse effects on biological resources, locations with any chemical concentrations greater than CSL chemical criteria are defined as having adverse effects, and locations with any chemical concentrations between the SQS and CSL chemical criteria have adverse effects between these two definitions.

The potential for adverse effects is more uncertain at locations where no detected chemicals exceeded the SQS/CSL chemical criteria or DMMP guidelines but RLs were greater than criteria and guidelines. However, based on an analysis of these elevated

[70] Two locations with no chemical concentrations greater than SQS chemical criteria were selected for toxicity testing at the request of Ecology because they were located near specific potential sources. In addition, toxicity testing was also conducted at one location (B6A) without chemical concentrations greater than criteria based on the rounding rules approved for this baseline surface sediment dataset. This sample was tested for toxicity because using pre-baseline rounding rules, the concentration of PCBs was greater than the SQS chemical criteria.

RLs, the likelihood of risks from non-detected chemicals with RLs that exceeded their respective SQS chemical criteria is low.

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, the results indicated that:

- No adverse effects to benthic invertebrates living in intertidal and subtidal sediments are expected for 75% of the LDW area (i.e., the area in which chemical concentrations were less than or equal to the SQS chemical criteria and where sediments were non-toxic according to SQS biological effects criteria).
- 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 criteria.
- The remaining 18% of the LDW area had chemical concentrations or biological effects between the SQS and CSL criteria, indicating that risks are less certain in these areas than in areas with chemical concentrations greater than one or more CSL values.

There is some uncertainty associated with these area estimates because areas were estimated by interpolating individual points at which sediments were sampled. The spatial extent of individual samples exceeding chemical and biological effects criteria is relevant to the assessment of overall risks to the benthic invertebrate community, both as an ROC and as a food resource. Uncertainty in the areal extent of effects increases as the size of the polygon increases. In some locations, groups of samples exceeded the SQS/CSL chemical criteria and in other locations only an isolated sample exceeded the chemical criteria. In areas with isolated exceedances, the potential impact to the ecological function of the benthic invertebrate community is likely to be less than in areas with numerous locations with exceedances.

A.6.1.1.2 Porewater

This section presents the risk estimates, uncertainties, and risk conclusions for benthic invertebrates exposed to VOCs in porewater.

Risk Estimates

One COPC was identified based on the exposure of benthic invertebrates to VOCs in porewater (cis-1,2-dichloroethene). All other VOCs detected in the porewater samples were at concentrations below levels of concern. COPCs were identified based on a comparison of maximum VOC concentrations detected in porewater samples at two sites (i.e., GWI and Boeing Plant 2/Jorgensen Forge) in the LDW to no-effects concentrations (Section 2.5.1.2). These two sites were selected to represent worst-case exposure areas (Windward 2005i).

The maximum porewater concentration of cis-1,2-dichloroethene (2,900 µg/L), detected in peeper PE-06 (Windward 2005a), was approximately 21 times higher than the estimated NOEC TRV but was less than the LOEC TRV (Table A.6-7). Other peepers in the vicinity of PE-06 (i.e., PE-03, PE-04, and PE-05) had lower concentrations of cis-1,2-dichloroethene (i.e., 0.5, 2.4, and 630 µg/L, respectively), with only PE-05 exceeding the NOEC TRV. Thus, the spatial extent of groundwater discharge of this chemical into porewater appears to be highly localized. This finding is consistent with the proposed conceptual site model for discharge at the GWI site (Windward 2005i).

Table A.6-7. HQs for and benthic invertebrates exposed to cis-1,2-dichloroethene in porewater

Peeper ID	cis-1,2-Dichloroethene Concentration in Porewater: Detected Concentration (µg/L)	NOEC TRV (µg/L)[a]	LOEC TRV (µg/L)[b]	NOEC-Based HQ	LOEC-Based HQ
PE-01	1.6	136	6,785	0.01	< 0.01
PE-02	46	136	6,785	0.34	< 0.01
PE-03	0.5	136	6,785	< 0.01	< 0.01
PE-04	2.4	136	6,785	0.02	< 0.01
PE-05	630	136	6,785	**4.6**	0.09
PE-06	2,900	136	6,785	**21**	0.43
PE-07	18	136	6,785	0.13	< 0.01
PE-08	20[c]	136	6,785	0.15	< 0.01

a NOEC was calculated from LC50-based LOEC by dividing by an uncertainty factor of 50.

b Concentration is based on an LC50.

c Field replicates for PE-08 had detected concentrations of 41 and 27 µg/L.

HQ – hazard quotient

ID – identification

LC50 – concentration that causes the death of 50% of a group of test animals

LOEC – lowest-observed-effect concentration

NOEC – no-observed-effect concentration

TRV – toxicity reference value

Bold identifies NOEC-based HQs greater than 1.0 and LOEC-based HQs greater than or equal to 1.0.

Uncertainties Associated with Porewater Risk Estimates

This section presents uncertainties in the porewater risk characterization for the benthic invertebrate community. The uncertainties are discussed separately for the exposure and effects assessments.

Exposure Assessment

The assessment of porewater-associated chemicals focused on two areas, GWI (RM 2.3 to RM 2.4) and Boeing Plant 2/Jorgensen Forge (RM 3.5 to RM 3.6). These locations were selected as worst-case exposure scenarios based on an evaluation of groundwater VOC data at high-priority sites identified by EPA and Ecology in Phase 1 (Windward 2005i). The groundwater conceptual site models for these locations showed that fresh groundwater migrating beneath upland areas is likely to discharge primarily into intertidal areas. Analyses of porewater samples collected with piezometers in these areas were consistent with the models. Therefore, even though there is uncertainty associated with the identification of areas with groundwater discharge and resulting porewater exposures, based on this evaluation, the exposure areas sampled with peepers were likely to be the worst-case exposure areas.

Effects Assessment

The primary uncertainty in the porewater assessment is the limited amount of relevant toxicity data available in the literature (i.e., LOECs were rarely reported, NOECs were dependent on the selected test dilution series, and very few studies reported both effect and no-effect concentrations for a single species and endpoint). In addition, no toxicological data were available for isopropylbenzene, and few data were available for 1,1-dichloroethane and vinyl chloride. The NOECs and LOECs for 1,1-dichloroethane and vinyl chloride used in the COPC screen in Section A.2.5.1.2 were obtained by calculating acute and chronic values using the narcosis model from DiToro et al. (2000). Because the calculation of acute and chronic values in the narcosis model followed the EPA water quality criteria guidelines for the protection of aquatic organisms, it is unlikely that the model underestimated effects to benthic invertebrates from 1,1-dichloroethane and vinyl chloride.

There is also uncertainty in the NOEC TRV calculated for cis-1,2-dichloroethene (the single COPC identified). The NOEC TRV was calculated by dividing the LC50-based LOEC TRV by 50. Thus, there is uncertainty regarding the probability of adverse effects in the immediate vicinity of peepers PE-05 and PE-06.

Risk Conclusions

Risks to benthic invertebrates from exposure to VOCs in porewater are expected to be very low for most VOCs and low for one VOC in a localized area in the intertidal area near GWI. These results are based on a risk assessment at two worst-case scenario locations, GWI and Boeing Plant 2/Jorgensen Forge, where only one chemical, cis-1,2-dichloroethene, was identified as a COPC at GWI. Among the peepers sampled, the highest NOEC-based HQ for this compound was 21; the highest LOEC-based HQ was less than 1.0. The NOEC TRV is uncertain because it was derived from the LOEC using a safety factor of 50.

A.6.1.1.3 TBT

This section presents an evaluation of risks to benthic invertebrates from exposure to TBT in the LDW, including uncertainties and risk conclusions. Risks were assessed in two ways:

- Risks to meso- and neogastropods, invertebrates that have been identified as being particularly sensitive to TBT (Meador et al. 2002), were evaluated through direct measurement of effects (i.e., imposex). TBT exposure was based on the TBT concentration in sediment at locations where gastropods were collected.

- Risks to benthic invertebrates other than gastropods were assessed using a critical tissue-residue approach. TBT exposure was based on TBT concentrations in co-located tissue and sediment samples.

Imposex

The potential adverse effects from TBT exposure were assessed for meso- and neogastropods through the direct measurement of imposex in field-collected gastropods from the LDW in two separate investigations. The relative sensitivity of the specific neo- and mesogastropod species found in the LDW to TBT has not been evaluated in the literature.

Meso- and neogastropod imposex data were presented in the *Technical Memorandum: Gastropod Pilot Survey Results* (Windward 2004g) and the *Technical Memorandum: 2005 Gastropod Imposex Study Results* (Windward 2006c) and were summarized in Section A.3.2.4 of this ERA. Imposex was not observed in two of the three neogastropod species examined or in the two mature female mesogastropods collected in the LDW (12 other mesogastropod specimens were collected, but they were either males or immature) (Windward 2004f, 2006c). Imposex was not observed in the abundant neogastropods *Astyris gausapata* or *Olivella baetica*, including specimens collected from areas within the LDW that historically have had high TBT concentrations in surface sediment.

Nassarius mendicus was the only gastropod species among those collected that showed imposex symptoms. All collected females of this species were classified as Stage 2, specifically Stage 2a (large penis with penis duct), according to the methods of Oehlmann et al. (1991). None of the affected specimens had an evident vas deferens. The calculated RPS indices for females of this species ranged from 0.2 to 3.4%. These RPS indices are all below thresholds associated with sterility in female neogastropods (Spence et al. 1990). In all other gastropods that have been studied in Great Britain and other locations, Stage 2 imposex has not interfered with female reproduction (Spence et al. 1990; Gibbs and Bryan 1996).

There is uncertainty in the assessment because: 1) neo-and mesogastropod species may already have been affected by TBT and therefore are no longer present in the LDW, and 2) low numbers of gastropods were collected in the LDW. One to four specimens of *Nassarius mendicus*, the only neogastropod showing imposex characteristics, were collected at five out of the six sampling locations; and females were collected at three of the six locations. No *N. mendicus* females were collected at the two locations with the highest TBT concentrations or at the location with the lowest TBT concentration.

In addition, low numbers (one to six) of specimens from the order Mesogastropoda were assessed for imposex. Because the majority of these specimens were immature, only two females were evaluated for imposex. Neither of these females showed any signs of imposex.

Based on the results of the site-specific imposex studies, it is probable that the level of imposex observed in *N. mendicus* in the LDW does not have an adverse effect on reproduction. The observed imposex was characterized as Stage 2, a stage that is not expected to affect reproduction. Imposex was not observed in any other neo- or mesogastropod collected. Thus, although there were uncertainties in the field studies, TBT risk to gastropods is considered to be low.

Critical Tissue-Residue Approach

The potential adverse effects of TBT exposure were assessed for benthic invertebrates other than gastropods using a critical tissue-residue approach. TBT concentrations in tissue were calculated from the non-linear regression relationship observed between the TBT concentrations in co-located benthic invertebrate tissue and sediment samples, as described in Section A.3.1.3 and Attachment 11. The non-linear regression equation was also used to estimate the TBT concentration in tissue that may be associated with the maximum TBT concentration in surface sediment. The estimated maximum TBT concentration in tissue was 0.61 mg/kg dw. The maximum TBT concentration detected in tissue was 0.55 mg/kg dw. All of the TBT concentrations detected or estimated in benthic invertebrate tissue were less than the NOAEL TRV of 0.97 mg/kg dw (Table A.6-8).

Table A.6-8. TBT risk estimates for benthic invertebrates using the critical tissue-residue approach

Organism	TBT Concentration in Tissue			NOAEL-Based HQ	LOAEL-Based HQ
	Exposure Concentration (mg/kg dw)	NOAEL TRV (mg/kg dw)	LOAEL TRV (mg/kg dw)		
Benthic invertebrates	0.55[a]	0.97	2.36	0.57	0.23
	0.61[b]	0.97	2.36	0.63	0.26

[a] Maximum concentration in benthic invertebrate tissue samples collected from the LDW.

[b] Maximum concentration in benthic invertebrate tissue estimated from the maximum TBT concentration in sediment (3 mg/kg dw) and the non-linear regression equation.

dw – dry weight
HQ – hazard quotient
LOAEL – lowest observed adverse effect level
NOAEL – no observed adverse effect level
TBT – tributyltin
UCL – upper confidence limit on the mean

The largest source of uncertainty associated with the critical tissue-residue evaluation of risk to benthic invertebrates is the TRV value. The selected TRV was based on the response of a single species, the polychaete *Armandia brevis*, in a spiked sediment bioassay. This polychaete is found in marine intertidal mud flats and would not be expected to be present in substantial numbers in the LDW. This species has been shown to bioaccumulate TBT with relatively little ability to metabolize TBT (Meador 1997; Meador et al. 1997). Meador et al. (1997) reported significant differences in TBT accumulation and depuration between *Armandia brevis* and two amphipods, *Rhepoxynius abronius* and *Eohaustorius washingtonianus*. Of the amphipods tested, *E. washingtonianus* was most sensitive to sediment-associated TBT, and *R. abronius* was the least sensitive.

The TRV was compared to exposure concentrations represented by benthic invertebrate tissue samples, which contained a wide range of species, including amphipods, mollusks, and polychaetes. It is unknown how similar the species present in these samples are to *Armandia brevis* in terms of their ability to accumulate TBT or their sensitivity to the effects of TBT.

Locations of the co-located benthic invertebrate tissue and sediment samples analyzed to derive the regression relationship were selected to represent the range of TBT concentrations throughout the LDW. The co-located sediments represent 99% of the range of TBT concentrations found throughout the LDW. The use of *in-situ* sediment and tissue concentrations to derive the regression relationship results in reduced uncertainty with regard to the estimate of exposure when compared to using a sediment/tissue relationship from the literature.

The NOAEL-based HQ for benthic invertebrates was less than 1.0. Thus, although there were uncertainties in this analysis, risks to the benthic invertebrate community from exposure to TBT are considered to be very low.

A.6.1.2 Crabs

This section presents the risk estimates, uncertainties, and risk conclusions for crabs.

A.6.1.2.1 Risk estimates

In this section, risks to crabs from exposures to two COPCs, zinc and PCBs, are assessed using a critical tissue-residue approach. Risks were assessed separately for crabs using a critical tissue-residue approach because crabs are more mobile than infaunal organisms, are not specifically covered by SMS criteria, and have a greater potential for exposure through bioaccumulation because of their higher trophic position. HQs were calculated for zinc and PCBs based on the UCL concentration of these COPCs in whole-body crab tissue relative to tissue-based NOAEL and LOAEL TRVs (Table A.6-9). The NOAEL-based HQ was greater than 1.0 for both COPCs (2.5 for zinc and 10 for total PCBs). LOAEL-based HQs were less than 1.0 for zinc and equal to 1.0 for total PCBs.

Table A.6-9. HQs for crabs using whole-body exposure and effects data

Chemical	Unit	Whole-Body Tissue Concentration			NOAEL-Based HQ	LOAEL-Based HQ
		UCL[a]	NOAEL TRV	LOAEL TRV		
Zinc	mg/kg ww	32	12.7	35.2	**2.5**	0.91
Total PCBs	µg/kg ww	1,100	110[b]	1,100	**10**	**1.0**

[a] Whole-body concentrations were estimated by combining hepatopancreas and edible-meat concentrations, assuming 69% by mass edible meat and 31% by mass hepatopancreas.

[b] Calculated by dividing the LOAEL by 10.

HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
NOAEL – no-observed-adverse-effect level
PCB – polychlorinated biphenyl
UCL – upper confidence limit on the mean
TRV – toxicity reference value

Bold identifies NOAEL-based HQs greater than 1.0 and LOAEL-based HQs greater than or equal to 1.0.

A.6.1.2.2 Uncertainty associated with crab tissue risk estimates

This section presents specific areas of uncertainty in the crab risk estimates related to the problem formulation, exposure assessment, effects assessment, and risk characterization.

Problem Formulation

Crabs were selected as an ROC to represent higher-trophic-level benthic invertebrates not addressed by the SMS. There is uncertainty associated with the assumption that COPC concentrations in crab tissue would represent those of other mobile, higher-trophic-level benthic invertebrates in the LDW, which include sea stars and shrimp. Dungeness crabs are scavengers; their diet includes shrimp, mussels, small crabs, clams, and sea urchins. Thus, crabs are likely to be similarly exposed through their diet as sea stars and shrimp, which have comparable diets. Thus, there is relatively little uncertainty in using crabs as representatives of larger, more mobile benthic invertebrates.

The COI screening process is also uncertain because the available toxicity studies for decapods investigated only survival or growth endpoints. Toxicity studies using reproductive endpoints would potentially be more sensitive and thus could have identified additional COPCs had they been available.

Exposure Assessment

Because tissue was used to estimate crab exposure, all potential exposure pathways were integrated. However, because crabs' home ranges can include areas outside of the LDW, the concentrations of COPCs in crab tissue may not be fully reflective of LDW exposure.

The EPCs, which were estimated by combining the Dungeness and slender crab tissue data, are uncertain because the approach assumes similar whole-body concentrations for the two crab species. Combining crab tissue samples likely has little effect on the assessment of risk from zinc because zinc concentrations are similar between the two species. Whole-body concentrations for Dungeness crabs and slender crabs ranged from 22.7 to 33 mg/kg ww and from 33.6 to 35.9 mg/kg ww, respectively. In contrast, combining crab tissue samples may slightly underestimate the risk from PCBs because Dungeness crabs have slightly higher concentrations. Whole-body PCB concentrations for Dungeness crabs and slender crabs ranged from 420 to 1,900 µg/kg ww and from 250 to 838 µg/kg ww, respectively.

There is also uncertainty associated with the LDW crab whole-body tissue-residue data because whole-body concentrations were estimated based on chemical concentrations in edible meat and the hepatopancreas. It is unknown if whole-body concentrations estimated from these data result in an overestimate or underestimate of the actual tissue residue of COPCs in crabs.

Effects Assessment

The primary uncertainty in the crab effects assessment is the limited number of tissue-based TRVs available in the literature. Effects data for decapods were found for only 14 of the 54 COIs detected in LDW crab tissue. Furthermore, most of these toxicity studies investigated the survival endpoint rather than the potentially more sensitive sublethal endpoints. In some of the studies, tests were conducted only with adults, although juveniles or early life stages may be more sensitive than adults. Additional uncertainties with these studies were associated with exposure durations, exposure pathways (water exposure vs. dietary exposure), and test organism used (decapods other than crabs). The relative uncertainties in the selected TRVs for crabs and the potential effect on the risk estimates are summarized in Table A.6-10.

Table A.6-10. Level of uncertainty associated with crab TRVs

Chemical	Number of Toxicity Studies	Uncertainty in TRV
Zinc	1	High TRV uncertainty because of small dataset with only acute studies that assessed mortality; whole-body tissue concentrations were estimated.
Total PCBs	4	High TRV uncertainty because of small dataset with only acute studies that assessed mortality; whole-body tissue concentrations were estimated.

PCB – polychlorinated biphenyl

TRV – toxicity reference value

The selected LOAEL for zinc and total PCBs and the selected NOAEL for zinc were based on acute exposure studies that assessed survival, which is a less-sensitive endpoint than growth or reproductive endpoints; the NOAEL TRV for PCBs was derived from the LOAEL by dividing by an uncertainty factor of 10. No uncertainty factor was applied to the acute LOAELs to derive chronic LOAELs; therefore, risks from zinc and PCBs may be underestimated.

Arsenic was identified as a COPC for crabs but was not evaluated in the exposure and effects assessments because a LOAEL was not available. The NOAEL-based HQ was 8.6. At EPA's request, a LOAEL for arsenic was derived by multiplying the NOAEL by an uncertainty factor of 10. Using this highly uncertain approach,[71] the LOAEL-based HQ would be 0.86, indicating low risk of adverse effects to crabs.

[71] In contrast to the application of uncertainty factors to LOAELs to estimate NOAELs, no information is available to estimate a LOAEL from a single NOAEL.

Risk Characterization

Total DDTs and methoxychlor were identified as COPCs because their estimated maximum concentrations in crab tissue exceeded the selected NOAELs (Section A.2.5.1.3). Risks to crabs from total DDTs and methoxychlor were not included in the risk estimates because of high uncertainty in the JN-qualified organochlorine pesticide tissue data, which resulted in suspected false identifications of the presence of some organochlorine pesticides as well as overestimates in their concentrations, as discussed in Section A.2.4.2.2. The exposure and effect assessments as well as the risk estimates for these two organochlorine pesticides are discussed in this section.

Table A.6-11 presents whole-body tissue-residue concentrations for total DDTs and methoxychlor in Dungeness and slender crabs using the equation presented in Section A.3.3. Data for Dungeness and slender crabs were combined because insufficient numbers of Dungeness crabs were collected in some of the sampling areas.

Table A.6-11. Estimated total DDT and methoxychlor concentrations in whole-body crab tissue

COPC	Number of Samples	Minimum Concentration (µg/kg ww)	Maximum Concentration (µg/kg ww)	UCL (µg/kg ww)
Total DDTs	19	47.6	150	48
Methoxychlor	19	5.8	90	54

COPC – chemical of potential concern

UCL – upper confidence limit on the mean

ww – wet weight

Three studies assessed the effects of DDTs on decapods (Johnson et al. 1971; Nimmo et al. 1970; Leffler 1975) (Table A.6-12). A mortality rate of 30% was reported in pink shrimp at a whole-body DDT tissue concentration of 60 µg/kg ww after water exposure to DDT and its metabolites for 56 days (Nimmo et al. 1970). No mortality was reported in crayfish exposed to DDT in water for 3 days, resulting in a whole-body DDT tissue concentration of 46 µg/kg ww (Johnson et al. 1971). These two tissue concentrations were selected as the LOAEL and NOAEL TRVs, respectively.

Table A.6-12. Total DDT and methoxychlor critical tissue-residue toxicity studies for crabs and other decapods

Chemical	Test Species	Exposure Conditions	NOAEL (µg/kg ww)	LOAEL (µg/kg ww)	Effect	Source
Total DDTs	crayfish (*Orconectes nais*)	0.8 µg/L in water for 3 days	**46**	na	reduced survival	Johnson et al. (1971)
	pink shrimp (*Penaeus duorarum*)	0.05 µg/L in water for 56 days	na	**60**	reduced survival	Nimmo et al. (1970)
	juvenile blue crab (*Callinectes sapidus*)	dietary exposure for 5 weeks	26	200	increased metabolic rate	Leffler (1975)
Methoxychlor	juvenile Dungeness crab (*Cancer magister*)	0.04 µg/L in water for 18 days	**15**[a]	**150**	reduced survival	Armstrong et al. (1976)
	larval blue crab (*Callinectes sapidus*)	0.7 and 1.0 µg/L in water for a mean of 63 days	< 100	150	reduced survival	Bookhout et al. (1976)
	larval mud crab (*Rhithropanopeus harrisii*)	2.5 µg/L in water for a mean of 20 days	< 100	230	reduced survival	Bookhout et al. (1976)
	adult Dungeness crab (*Cancer magister*)	7.5 µg/L in water for up to 15 days	na	570	reduced survival	Armstrong et al. (1976)

[a] Calculated from LOAEL by dividing by 10.

na – not available

LOAEL – lowest-observed-adverse-effect level

NOAEL – no-observed-adverse-effect level

ww – wet weight

Bold identifies the NOAELs and LOAELs selected as the TRVs.

Two studies assessed the effects of methoxychlor on decapods (Armstrong et al. 1976; Bookhout et al. 1976) (Table A.6-12). Armstrong et al. (1976) assessed the effects of methoxychlor on juvenile and adult Dungeness crabs. After 36 days of exposure to 0.04 µg/L of methoxychlor in water, juvenile crabs had an approximate 20% increase in mortality over the controls. A similar increase in mortality over the controls was observed in juvenile crabs after an 80-day exposure to 0.04 µg/L of methoxychlor in water. Tissue concentrations were not analyzed in these experiments. However, after exposing juvenile crabs to 0.04 µg methoxychlor/L in water for 18 days, the whole-body concentration was 150 µg/kg. In a similar experiment, adult crabs exposed to 0.4 and 4.0 µg/L of methoxychlor in water for 85 days had survival rates of 100% and 0%, respectively. Concentrations of methoxychlor in tissue were not detected in adult crabs exposed to these concentrations. However, increased mortality was observed in adult crabs exposed to 7.5 µg/L methoxychlor in water for 15 days, and a concentration of 570 µg/kg ww was detected in crab tissue.

Bookhout et al. (1976) assessed the effects of methoxychlor on the larval development of mud and blue crabs. Significant mortality was reported in mud crabs exposed to methoxychlor through the five larval developmental stages. At the end of the last larval stage, an 18% increase in mortality over the controls was reported in organisms, with a whole-body concentration of 230 µg/kg ww. Similarly, significant mortality was reported for blue crabs exposed to methoxychlor through the nine larval developmental stages. High mortality was reported among the later larval stages in the controls. At the end of the last larval stage, a 27% increase in mortality over the controls was reported in organisms, with a whole-body concentration of 150 µg/kg ww. The lowest LOAEL of 150 µg/kg ww reported in the two studies was selected as the LOAEL TRV. Because none of the studies provided a usable NOAEL, a NOAEL of 15 µg/kg ww was calculated using an uncertainty factor of 10 (because the LOAEL was based on an acute endpoint). The NOAELs and LOAELs for total DDTs and methoxychlor were based on survival, which is a less sensitive endpoint than growth or reproduction.

HQs were calculated for total DDTs and methoxychlor based on the UCL concentrations of these COPCs in whole-body crab tissue relative to tissue-based NOAEL and LOAEL TRVs (Table A.6-13). Because of suspected false identifications of the presence of some organochlorine pesticides as well as overestimates in their concentrations, the tissue concentrations are highly uncertain and biased high (Section A.2.4.2.2).

Table A.6-13. Crab HQs using whole-body exposure and effects data

COPC	Whole-Body Tissue Concentration (µg/kg ww)			NOAEL-Based HQ	LOAEL-Based HQ
	Exposure Concentration	NOAEL TRV	LOAEL TRV		
Total DDTs	48	46	60	**1.0**	0.80
Methoxychlor	54	15[a]	150	**3.6**	0.36

[a] Calculated from LOAEL by dividing by 10.

COPC – chemical of potential concern

HQ – hazard quotient

LOAEL – lowest-observed-adverse-effect level

NOAEL – no-observed-adverse-effect level

UCL – upper confidence limit on the mean

ww – wet weight

Bold identifies NOAEL-based HQs greater than 1.0 and LOAEL-based HQs greater than or equal to 1.0.

The NOAEL-based HQ was greater than 1.0 for methoxychlor and equal to 1.0 for total DDTs. LOAEL-based HQs were less than 1.0 for both methoxychlor and total DDTs. The TRVs were based on survival following acute exposures, and the exposure estimates for these organochlorine pesticides were biased high. Therefore, risks to crabs from these pesticides are uncertain.

A.6.1.2.3 Risk conclusions

Risks to crabs from sediment-associated COPCs in the LDW appear to be low. While NOAEL-based HQs were 2.5 for zinc and 10 for total PCBs, the LOAEL-based HQs were less than or equal to 1.0 for the two COPCs (Table A.6-14). Because the true threshold for effects falls somewhere between the NOAEL and the LOAEL and because of the limited toxicity data, risks to crabs are possible but likely to be low because no LOAELs were exceeded.

Table A.6-14. Summary of risk characterization for crab

Chemical	NOAEL-Based HQ	LOAEL-Based HQ	Effect	Primary Uncertainty
Zinc	**2.5**	0.91	reduced survival	High TRV uncertainty because of small dataset with only acute studies; whole-body tissue concentrations were estimated.
Total PCBs	**10**	**1.0**	reduced survival	High TRV uncertainty because of small dataset with only acute studies; whole-body tissue concentrations were estimated.

HQ – hazard quotient

LOAEL – lowest-observed-adverse-effect level

NOAEL – no-observed-adverse-effect level

TRV – toxicity reference value

Bold identifies NOAEL-based HQs greater than 1.0 and LOAEL-based HQs greater than or equal to 1.0.

HQs for total DDTs and methoxychlor were greater than or equal to 1.0 based on the NOAEL TRV, but less than 1.0 based on the LOAEL TRV (0.80 and 0.36, respectively). Therefore, risks for these pesticides are uncertain. Arsenic had a NOAEL-based HQ of 8.6, but no LOAEL-based TRV was available. Without a LOAEL, effects from arsenic cannot be assessed.

A.6.1.3 Summary of risk conclusions for benthic invertebrates

In summary, results of the benthic invertebrate risk estimates and evaluation of associated uncertainties are as follows:

Benthic invertebrate community. The results of the sediment risk characterization, which used Thiessen polygons and the combination of chemistry and toxicity test results, indicated that:

- No adverse effects to benthic invertebrates living in intertidal and subtidal sediments are predicted for 75% of the LDW area (i.e., the area in which chemical concentrations were less than or equal to the SQS chemical criteria and where sediments were non-toxic according to SQS biological effects criteria).
- 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 criteria.
- The remaining 18% of the LDW area had chemical concentrations or biological effects between the SQS and CSL criteria, indicating that risks are less certain in these areas than in areas with chemical concentrations greater than one or more CSL values.

There is some uncertainty associated with these area estimates because areas were estimated by interpolating individual points at which sediments were sampled. The spatial extent of individual samples exceeding criteria is relevant to the assessment of overall risks to the benthic invertebrate community, both as an ROC and as a food resource. Uncertainty in the areal extent of effects increases as the size of the polygon increases. In some locations, groups of samples exceeded criteria and in other locations only an isolated sample exceeded criteria. In areas with isolated exceedances, the potential impact to the ecological function of the benthic invertebrate community is likely to be less than in areas with numerous locations with exceedances.

The potential for adverse effects is more uncertain at locations where no detected chemical concentrations were greater than SQS chemical criteria or DMMP guidelines but RLs were greater than criteria and guidelines. However, based on an analysis of these elevated RLs, the likelihood of risks from non-detected chemicals with RLs that exceeded their respective SQS chemical criteria is low.

Based on the porewater risk estimates and uncertainties, risks to benthic invertebrates from exposure to COPCs in LDW porewater are low and localized. Risks to benthic invertebrates from TBT were very low based on NOAEL-based HQs less than 1.0 and the absence of imposex in all gastropods, except one species of neogastropod with imposex characterized as Stage 2, a stage that is not expected to impact reproduction.

Crabs. Risks to crabs from zinc and total PCBs in the LDW were low based on the critical tissue-residue approach. Risks from arsenic are uncertain because a LOAEL was not available and exposure concentrations were greater than the NOAEL. Risks from pesticides are likely to be low.

A.6.2 RISK CHARACTERIZATION FOR FISH

This section presents a risk characterization and uncertainty analysis for each of the three fish ROCs. The assessment for each ROC estimates risk based on the exposure and effects assessments discussed in Sections A.4.1 and A.4.2, respectively. Following the risk estimates, a detailed evaluation of uncertainty associated with these calculations is presented. Risk conclusions are then presented for each ROC synthesizing risk estimates and uncertainties.

A.6.2.1 Juvenile chinook salmon

This section presents risk estimates, uncertainties, and risk conclusions for juvenile chinook salmon.

A.6.2.1.1 Risk estimates

This section presents the HQ calculations for juvenile chinook salmon. COPCs evaluated for juvenile chinook salmon included arsenic, cadmium, copper, and vanadium, all of which were evaluated using a dietary approach. Two lines of evidence were assessed for this dietary approach: comparison of concentrations in a single composite sample of stomach contents with the TRV, and comparison of estimated prey concentrations using chemistry data for benthic invertebrates (see Section A.4.1.2) with the TRV.

For cadmium, the LOAEL-based HQ was 1.0 based on consumption of benthic invertebrate prey, whereas the LOAEL-based HQ was less than 1.0 based on stomach contents data (Table A.6-15). LOAEL-based HQs were less than 1.0 for all other COPCs. NOAEL-based HQs were greater than 1.0 for arsenic, copper, and vanadium based on consumption of benthic invertebrate prey data but less than 1.0 based on stomach content data. [72]

[72] Vanadium was not analyzed in the composite sample of stomach contents.

Table A.6-15. HQ calculations for juvenile chinook salmon

Approach	COPC	Exposure Concentration (mg/kg dw)	NOAEL TRV (mg/kg dw)	LOAEL TRV (mg/kg dw)	NOAEL-Based HQ	LOAEL-Based HQ
dietary (stomach contents)	arsenic	3.9	20	30	0.20	0.13
	cadmium	0.46	0.1	0.5	**4.6**	0.92
	copper	42	50	100	0.83	0.42
dietary (benthic invertebrates)	arsenic	22	20	30	**1.1**	0.73
	cadmium	0.50	0.1	0.5	**5.0**	**1.0**
	copper	93	50	100	**1.9**	0.93
	vanadium	8.1	2.04	10.2	**4.0**	0.79

COPC – chemical of potential concern
dw – dry weight
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
NOAEL – no-observed-adverse-effect level
TRV – toxicity reference value
ww – wet weight
Bold identifies NOAEL-based HQs greater than 1.0 and LOAEL-based HQs greater than or equal to 1.0.

A.6.2.1.2 Uncertainty analysis

This section presents a discussion of the uncertainty associated with the problem formulation, the exposure and effects assessments, and the risk characterization for juvenile chinook salmon.

Problem Formulation

The primary uncertainties in the problem formulation for juvenile chinook salmon were associated with ROC selection, assessment endpoints, and the COPC screen.

ROC Selection

Uncertainties associated with the representation of other juvenile salmonids in the LDW by juvenile chinook salmon were addressed in the Phase 1 ERA (Windward 2003b). Juvenile chinook salmon are regarded as the most estuarine-dependent juvenile salmonid, so they are likely to have equal or greater exposure to sediment-associated chemicals than other estuarine-dependent salmonids such as juvenile chum salmon. The Phase 1 ERA (Windward 2003a) also concluded that although yearling juvenile chinook salmon may occur in the LDW in limited numbers and eat at a higher trophic level than subyearling chinook salmon (for which tissue data were collected and analyzed), subyearling juvenile chinook salmon are likely to have greater exposure to sediment-associated chemicals than yearling chinook salmon because of their greater residence time in the LDW (subyearling chinook salmon are more estuarine-dependent than are yearling chinook salmon) (Kerwin and Nelson 2000). Therefore, risk estimates for juvenile chinook salmon should be similar to or greater than those for other juvenile salmonids in the LDW.

Assessment Endpoints

Uncertainties associated with how well the endpoints selected to represent potential adverse effects on fish populations in the LDW were addressed in the Phase 1 ERA (Windward 2003b). In that analysis, it was concluded that biomarkers of effect such as cytochrome P450-dependent mono-oxygenase induction (e.g., CYP1a [van der Weiden et al. 1994]), DNA changes (e.g., DNA-xenobiotic adduct formation [Rice et al. 2000]), and sub-organismal immune dysfunction (e.g., antibody formation [Arkoosh et al. 1991]) may provide an early warning of adverse effects on fish at the individual and population levels (e.g., Schmitt et al. 2000). However, from an individual or population level, the overall significance of these changes is unknown. Thus, assessment endpoints were limited to survival and growth in this ERA.[73]

COPC Screen

Uncertainties associated with the COPC screen included a lack of fish tissue data or TRVs for some chemicals detected in sediment and mercury egg LOAELs that were lower than the selected mercury NOAEL TRV. Each of these uncertainties is addressed below.

Of the chemicals screened out as COIs for all fish ROCs using criteria presented in Section A.2.5.2, several chemicals were detected in at least 5% of baseline surface sediment samples but were not analyzed in tissue samples (Table A.6-16). None of these chemicals are defined as bioaccumulative chemicals (EPA 2000a); therefore, risks to fish from exposure to these chemicals are assumed to be very low.

Table A.6-16. Chemicals detected in ≥ 5% of baseline surface sediment samples that were not analyzed in LDW tissue samples

Chemicals	
Acetone	Magnesium
Aluminum	Manganese
Barium	Methyl ethyl ketone
Benzaldehyde	Potassium
Beryllium	Retene
Calcium	Sodium
Caprolactam	Tin
Carbon disulfide	Toluene
p-Cymene	Total petroleum hydrocarbons – gasoline, diesel and oil range
Iron	

[73] Effects on reproduction were not evaluated for juvenile chinook salmon because such effects would only be relevant for adult salmon, which acquire nearly all of their body burden of chemical contaminants outside the LDW.

Some chemicals have been detected in LDW sediment and in LDW tissue for which no toxicological data exist. TRVs were available for 47 of the 85 fish COIs. Twenty of these chemicals without toxicological data were PAHs or alkylated PAHs. As discussed in Section A.2.5.2, maximum dietary exposure concentrations for fish (the sum of all alkylated and non-alkylated PAHs) were lower than the available TRVs for benzo(a)pyrene or a PAH mixture. Insufficient effects data were available to evaluate risks to fish from the remaining 18 chemicals.

The NOAEL TRV for mercury used in the COPC screening process was the highest identified whole-body NOAEL below the lowest identified whole-body LOAEL; however, some studies reported adverse effects in fish associated with lower egg or early life stage tissue residues. The lowest whole-body LOAEL was 470 µg/kg ww for reduced survival of male mummichog exposed to dietary methyl mercury for 42 days (Matta et al. 2001). Birge et al. (1979) reported reduced survival of channel catfish embryos exposed to mercury in water associated with a 4-day post-hatch embryo tissue concentration LOAEL of 14 µg/kg ww. Birge et al. (1979) also reported reduced survival of rainbow trout alevins and embryos exposed to mercury at tissue LOAELs of 36 and 41 µg/kg ww, respectively.

Although these egg and embryo effects concentrations were lower than those reported by Matta et al. (2001) in more mature fish, egg/embryo and adult tissue-residue data are not directly comparable. Species-specific ratios relating mercury concentrations in whole bodies of maternal adults to concentrations in unfertilized eggs for yellow perch, smallmouth bass, white bass, white sucker, and rainbow trout ranged from 9.9 to 26 (Niimi 1983). Adjusting for a two-fold weight increase in concentration in fertilized eggs relative to unfertilized eggs (Niimi 1983) results in adult-to-fertilized egg ratios that ranged from 20 to 52 for these species. Therefore, mercury concentrations in whole bodies of adult female fish were likely to be 20 to 52 times higher than concentrations in fertilized eggs from those females. This range is uncertain because available data represent few species, with little or no replication. Based on this range of adult-to-egg mercury ratios, the estimated range of LOAELs in maternal adults associated with the lowest reported egg/embryo LOAEL (14 µg/kg ww) would have been 280 to 728 µg/kg ww. The maximum mercury whole-body fish tissue concentration in the LDW was 39 µg/kg ww; therefore, mercury would have screened out as a COPC for fish even if egg/embryo effects data had been included in the screen.

Exposure Assessment

Uncertainties in the exposure assessment for juvenile chinook salmon were associated with the following factors:

- Water pathway for metals and PAHs
- Dietary composition

- Foraging preferences
- Benthic invertebrate tissue data

Water Pathway for Metals and PAHs

A dietary approach was used in this Phase 2 ERA to evaluate risks to juvenile chinook salmon resulting from exposures to metals and PAHs. This approach does not incorporate exposure from water. To provide an assessment of the risks to fish from metals and PAHs from multiple exposure pathways, this section provides a brief summary of an analysis of water quality (WQA) conducted by King County (King County 1999d), in which risks to fish from estimated water column exposures were evaluated.[74] A more complete summary of the WQA approach was presented in Attachment A.2 of the Phase 1 ERA (Windward 2003b).

Chemical concentrations in surface water in the WQA were estimated based on results of a detailed three-dimensional fate and transport hydrodynamic model calibrated with field data. The model divided the LDW into 129 grid cells, each with 10 water layers and 1 sediment layer, resulting in 1,290 water column and 129 sediment cells. The model estimated chemical concentrations in water in each grid element every 15 minutes for 1 year.

The WQA identified 23 COPCs requiring risk analysis.[75] Risks related to these COPCs were evaluated using a tiered approach. Tier 1 consisted of a comparison of modeled water concentrations to acute and chronic TRVs.[76] For each COPC, the maximum monthly 1-hour moving average and maximum monthly 4-day moving average total recoverable[77] concentration in each cell in the study area for each month of the year were compared to acute and chronic TRVs, respectively. Eight metal and PAH COPCs had concentrations exceeding acute or chronic TRVs in at least one cell, as summarized in Table A.6-17. These COPCs were further evaluated for juvenile chinook salmon by comparing modeled water concentrations to salmonid-specific TRVs. COPCs were subsequently evaluated for all aquatic species in a second tier of the risk assessment.

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[74] This section discusses only results for the King County (1999d) WQA baseline scenario (includingCSO and stormwater inputs).

[75] Chemicals with the ability to cause cancer in humans were identified as COPCs. For non-carcinogenic chemicals detected more than 5% of the time, COPCs were identified by King County (1999d) based on a comparison of their 95th percentile concentrations in water and sediment to water quality criteria and sediment standards.

[76] TRVs were based on marine Washington State water quality standards or marine federal water quality criteria (WQC) (10 COPCs), toxicity studies from the literature (7 COPCs), and quantitative structure-activity relationships (QSARs) (6 individual PAHs). An uncertainty factor of 20 was applied to the literature- and QSAR-based TRVs by King County (1999d) for this analysis.

[77] Total recoverable concentrations were used as a conservative estimate of bioavailable metals (Prothro 1993).

Table A.6-17. Metals and PAHs with maximum total concentrations in water in any cell exceeding TRVs following the Tier 1 analysis

COPC	Number of Months in the Year in Which TRV was Exceeded	
	Acute TRV	Chronic TRV
Arsenic[a]	2	0
Copper[a]	12	12
Lead[a]	1	5
Nickel[a]	4	4
Zinc[a]	11	0
Benz(a)anthracene[b]	1	0
Benzo(g,h,i)perylene[b]	0	2
Fluoranthene[b]	1	0

[a] TRVs were based on state and federal marine AWQC for protection of aquatic life (state and federal criteria are the same for each of these chemicals).

[b] TRVs were based on QSARs divided by a safety factor of 20.

COPC – chemical of potential concern

PAH – polycyclic aromatic hydrocarbon

TRV – toxicity reference value

In the second tier of the WQA, with an approach equivalent to the Water Environment Research Foundation (WERF) Tier 3 risk assessment methodology (WERF 1996), risks associated with dissolved COPCs (rather than total recoverable concentrations)[78] were evaluated as the probability of affecting a given percentage of species. A logistic regression model was used to fit genus mean acute and chronic toxicity data (based on dissolved concentrations) for each COPC. This method estimates the percent of taxa affected at any given concentration. For each COPC, the percentage of species affected was estimated for each month based on the modeled maximum 1-hour moving average and 4-day moving average dissolved COPC concentrations in each model cell and layer.

Modeled concentrations of metals did not exceed salmonid-specific TRVs. Results of the second tier of the WQA showed that the percentages of representative species that might be subject to acute and chronic risks in the LDW[79] were less than or equal to 1% for all metals, except copper. Maximum monthly acute and chronic exposures to dissolved copper concentrations were estimated to affect 2 to 4% of aquatic species on the west side of Kellogg Island and less than or equal to 2% in all other locations. Based on an EPA-recommended level of protection of at least 95% of species to ensure overall community function (Stephan et al. 1985, as cited in the King County WQA),

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[78] Dissolved metal concentrations were used in the Tier 3 evaluations because they more closely represent the bioavailable fraction (Prothro 1993).

[79] Based on all species represented in toxicity studies included in AWQC plus toxicity studies identified in additional literature searches.

these results indicated low risks to the aquatic community from exposure to surface water concentrations of COPCs; 96% or more of the aquatic community were not expected to be affected by copper.

Risks from water exposure to PAHs were not evaluated in the Tier 3 assessment because insufficient toxicity data were available. Based on the Tier 1 assessment, most of the PAH compounds evaluated had acute and chronic HQs less than 1.0, indicating low risk. However, benzo(a)anthracene and fluoranthene had maximum acute HQs of 1.4 and 1.1, respectively. Acute HQs were greater than 1.0 for benzo(a)anthracene and fluoranthene in one month of the year in 0.08 and 0.4%, respectively, of the cells evaluated (i.e., 1 to 5 of the 1,290 cells evaluated had HQs slightly greater than 1.0). For all other months of the year, acute HQs were below 1.0. Chronic HQs were always below 1.0. For benzo(g,h,i)perylene, the highest chronic HQ was 1.4. In two of the modeled months, chronic HQs for this COPC were greater than 1.0 in 0.2 to 3.6% of the cells evaluated (i.e., 3 to 46 of the 1,290 cells evaluated had HQs slightly greater than 1.0). Therefore, overall aquatic life risks from estimated water column PAH exposures were low.

Methodology to combine water and dietary exposures for fish are not currently available. This topic is under study by EPA but remains an uncertainty at the time of this risk assessment.

Dietary Composition

COPCs in the diet of juvenile chinook salmon are a function of the types of prey consumed and their COPC concentrations. It was assumed that juvenile chinook salmon consume only benthic invertebrates, although they could also consume various zooplankton, larval fish, clam siphons, and terrestrial organisms that drift in the current, such as wasps and ants (Cordell et al. 1996; 1997; 1999). Because benthic invertebrates are more closely associated with sediments than other juvenile chinook salmon prey, dietary exposure concentrations calculated based on benthic invertebrate data are likely to be higher for dietary COPCs (which do not biomagnify). Prey that are less closely associated with sediment (e.g., drift organisms and zooplankton) would likely result in lower overall exposures, and therefore, lower risks from sediment-associated COPCs.

Foraging Preferences

In calculating dietary exposure concentrations, juvenile chinook salmon were assumed to forage in intertidal areas of the LDW. Therefore, intertidal benthic invertebrate and sediment data were used to calculate exposure. Although the literature indicates that juvenile chinook salmon primarily use shallow water areas, no studies have been conducted on the foraging depth of juvenile chinook salmon in the LDW. If juvenile chinook salmon were to forage throughout the LDW, their exposure concentrations would be the same as those calculated for English sole assuming 0% sediment ingestion (Table A.6-18). Different foraging assumptions did not substantially affect the HQs calculated for juvenile chinook salmon for arsenic, cadmium, or copper (Table A.6-18). For vanadium, the LOAEL-based HQ increased from 0.79 to 1.2.

Table A.6-18. Comparison of juvenile chinook salmon HQs assuming that they forage only in intertidal areas versus throughout the LDW

COPC	Intertidal Only		LDW-Wide	
	NOAEL-Based HQ	LOAEL-Based HQ	NOAEL-Based HQ	LOAEL-Based HQ
Arsenic	**1.1**	0.73	**1.2**	0.80
Cadmium	**5.0**	**1.0**	**6.0**	**1.2**
Copper	**1.9**	0.93	**1.8**	0.92
Vanadium	**4.0**	0.79	**5.9**	**1.2**

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.

The location of fish foraging and prey preference may change as a result of potential ecological improvements associated with future habitat restoration projects. This uncertainty was analyzed in the Phase 1 ERA (Windward 2003b). That analysis concluded that fish exposure to sediment-associated chemicals could either increase or decrease depending on changes in foraging behavior and on prey and sediment chemical concentrations at restored sites.

Benthic Invertebrate Tissue Data

There is uncertainty in the benthic invertebrate tissue data used to estimate dietary exposure of juvenile chinook salmon to some COPCs. Sampling of benthic invertebrate tissue was designed to include stations that represented the full range of concentrations of a subset of chemicals (arsenic, PCBs, and lead) in surface sediments (Windward 2004d). Sediment concentrations of other chemicals were not considered in the sampling design.

To evaluate whether the full range of chemical concentrations was sampled for other COPCs, COPC concentrations in sediment samples co-located with the benthic invertebrate tissue samples were compared to the cumulative distribution of that COPC in the full baseline surface sediment dataset. Results for all dietary COPCs are presented in Table A.6-19. The maximum sediment concentrations reported for the co-located sediment samples were greater than 80% of the site-wide concentrations of these COPCs. Therefore, the co-located sediment and tissue data for these COPCs are likely sufficient to assess the relationship between tissue and sediment concentrations for the site.

Table A.6-19. COPC concentrations in sediment samples co-located with benthic invertebrate tissue samples relative to the site-wide baseline surface sediment dataset for dietary COPCs

COPC	Maximum COPC Concentration in Sediment (mg/kg dw)		Percent of Concentrations in LDW-Wide Dataset Less Than or Equal to the Maximum Concentration in Co-Located Samples
	LDW-Wide Baseline Surface Sediment Dataset	Sediment Samples Co-Located with Benthic Invertebrate Tissue Samples	
Arsenic	1,100	725	99.8
Cadmium	120	1.67	93.7
Copper	12,000 J	495	98.8
Vanadium	150	72.6	83.6

COPC – chemical of potential concern

dw – dry weight

J – estimated concentration

LDW – Lower Duwamish Waterway

As discussed in Attachment 11, benthic invertebrate tissue concentrations were estimated from the linear regression of co-located sediment and benthic invertebrate chemical concentrations for chemicals where a significant sediment-tissue relationship existed. Note that because the regressions were not significant for cadmium, copper, and vanadium, the empirical data, rather than a regression, were used in exposure calculations for these COPCs. A significant sediment-tissue relationship existed for arsenic so benthic invertebrate tissue concentrations were estimated using the linear regression. The arsenic sediment concentrations used in the regression were highly skewed, and the highest values were identified as overly influential points using Cook's distance. However, because the arsenic sediment UCL on the mean concentrations in the fish exposure area were intermediate between the majority of the co-located sediment concentrations and the highly influential points, the influential points were retained in the regression model in order to avoid extrapolation outside the dataset. If the influential points had been excluded from the regression model, the regression would have been linear and would have estimated higher arsenic concentrations. Because none of the co-located sediment samples were similar in concentration to the sediment UCL, the regression is uncertain in this concentration range, and effects on the risk estimates are unknown.

If the exposure estimate had been calculated using empirical intertidal benthic invertebrate tissue data rather than the regression, the UCL would have been 14 mg/kg dw, while the UCL was estimated at 22 mg/kg dw using the regression statistics. Because benthic invertebrate tissue sample locations were selected to be representative of the range of sediment arsenic concentrations in the LDW (Table A.6-19), the regression-estimated UCL likely provides a protective estimate of arsenic concentrations in invertebrates representative of intertidal habitats in the LDW. Therefore, using the regression-estimated UCL to assess exposure likely would not have affected risk conclusions.

There is also uncertainty in the tissue data collected for benthic invertebrates because the tissue samples contained various benthic invertebrate species composited together in order to collect sufficient sample for analysis. Different benthic invertebrate species may bioaccumulate chemicals from sediment to different extents. However, without detailed dietary preference information for the fish ROCs and data for individual invertebrate species, the uncertainty introduced through compositing multiple species is unknown.

Effects Assessment

Uncertainties in the effects assessment for juvenile chinook salmon were associated with the following factors:

- Chemical mixtures
- Exclusion of field studies from TRV selection
- Estimation of NOAELs from LOAELs
- Salmonid-specific TRVs for cadmium
- COPCs without LOAEL TRVs
- Regional field studies

In addition, some of the selected TRVs are considered less certain than others if there were a small number of studies, if endpoints were subchronic, or if data quality was questionable. The relative uncertainties in the selected TRVs for each ROC/COPC pair are summarized below.

NOAEL and LOAEL TRVs for arsenic were selected based on review of six studies, several of which reported similar LOAELs. Because only a moderate number of studies were available, toxicity data may not represent the sensitivities of the various fish species in the LDW. The effect of this uncertainty on risk conclusions is unknown.

NOAEL and LOAEL TRVs for cadmium were selected based on review of nine studies. Although a moderate number of studies were available, TRV uncertainty is high because toxicity data varied greatly among studies, and the selected LOAEL TRV was two orders of magnitude lower than next lowest LOAEL identified. In addition, in the study reporting the lowest LOAEL (Kim et al. 2004; Kang et al. 2005), the growth effect reported was partially attributed to reduced feeding. Effects thresholds from salmonid-specific cadmium toxicity studies were higher. These uncertainties are discussed in greater detail below.

NOAEL and LOAEL TRVs for copper were selected based on a review of 15 studies, a reasonable number of studies. Sufficient data were available to show that the lowest LOAEL reported for growth of channel catfish (8 mg/kg dw) (Murai et al. 1981) was lower than NOAELs for channel catfish growth reported in two other studies (Gatlin and Wilson 1986; Erickson et al. 2003), so this study was rejected as the TRV.

NOAEL and LOAEL TRVs for vanadium were highly uncertain because they were based on only one study. The effect of this uncertainty on risk conclusions is unknown.

Chemical Mixtures

Effects from exposure to multiple chemicals that share the same mode of toxic action and other environmental stressors in the LDW that could result in additive, synergistic, or antagonistic effects were not factored into the effects assessment. This uncertainty was analyzed in the Phase 1 ERA (Windward 2003b). That analysis concluded that because the combined effects of complex chemical mixtures and other stressors occurring in the environment have not been sufficiently studied, effects of this uncertainty on risk predictions are unknown.

Exclusion of Field Studies from TRV Selection

TRVs were derived from laboratory studies in which fish were exposed to a single chemical or, in the case of PCBs, DDTs, and PAHs, to well-defined chemical mixtures under controlled conditions. By using such studies, specific chemical exposures associated with toxicity can be determined. By controlling for natural sources of variability, laboratory studies do not address potential implications associated with mixtures of contaminants or the interaction of chemical toxicity with other stressors that occur in the natural environment. A number of studies in which fish from contaminated sites were raised in the laboratory have been conducted to investigate potential adverse effects associated with sites contaminated with PCBs, DDTs, and PAHs (e.g., Arkoosh et al. 1991; Berlin et al. 1981; Hopkins et al. 1969; Mac et al. 1985). Other studies have exposed fish to field-collected contaminated sediments to investigate potential adverse effects associated with specific mixtures of site chemicals(e.g., Roberts et al. 1989). Such studies incorporate conditions and exposure scenarios that provide insight into risks associated with specific sites and the chemical mixtures present at those sites. However, these studies were not used to derive TRVs in this ERA because adverse effects observed in organisms from studies at other sites may be attributed to the presence of multiple chemicals not present in the LDW and/or to other uncontrolled environmental factors, rather than to a single test chemical.

In the problem formulation, PAHs were screened out as a COPC for fish because maximum dietary exposure concentrations were lower than the lowest dietary LOAELs identified from controlled laboratory studies in which fish were exposed to well-defined PAH mixtures. The analysis presented in the problem formulation did not consider studies with a field component in which fish were exposed to PAHs and other chemicals. Because of the uncertainties inherent in field studies described above, the field component of a dietary PAH exposure study conducted by Rice et al. (2000) and several other PAH sediment exposure studies presented in Table A.6-20 are discussed here to further evaluate uncertainties associated with PAH exposure and effects in fish.

Table A.6-20. Toxicity studies for total PAHs and fish, including studies with a field component

Chemical	Test Species	NOAEL (mg/kg dw)	LOAEL (mg/kg dw)	Exposure Duration	Effect	Source	Notes
Total PAHs (in the presence of a mixture of other chemicals in Eagle Harbor sediments)	English sole	11.3	na	28 days	reduced growth rate	Rice et al. (2000)	1
Benzo(a)pyrene	gizzard shad	1	na	22 days	reduced body weight	Kolok et al. (1996)	2
Total PAHs (in the presence of a mixture of other chemicals in Elizabeth River sediments)	spot	81	322	28 days	reduced survival, weight loss	Roberts et al. (1989)	3
Total PAHs (in the presence of a mixture of other chemicals in Eagle Harbor sediments)	English sole	2	4	175 days	reduced growth rate	Kubin (1997)	

na – not available, no LOAEL was identified.

Notes:

1. Dietary exposure to polychaete worms previously exposed to sediment from the Eagle Harbor, Washington Superfund site mixed with 99.9% reference site sediments. Adverse effects were observed in one of the two experimental trials.
2. Dietary exposure of field-collected shad to benzo(a)pyrene-spiked sediments augmented with trout chow.
3. Direct exposure to sediment from the Elizabeth River mixed with sediment from a reference site. Concentrations are those reported for day 0 of exposure. Concentrations at day 15 were approximately half those reported here.

Rice et al. (2000) exposed juvenile English sole to polychaete worms that had previously been exposed in the laboratory to 0.1% of sediments from the Eagle Harbor, Washington Superfund site mixed with 99.9% sediments from a reference site. PAH concentrations were reported for the sediment and worm tissue; other uncharacterized chemicals may also have been present but were not analyzed.[80] The major chemical contaminants present at Eagle Harbor were associated with creosote, which is used as a wood preservative. The major creosote-related chemicals generally associated with toxicity are PAHs, phenols, and cresols (ATSDR 2002). About 300 chemicals have been identified in coal tar creosote, but as many as 10,000 other chemicals may also be in this mixture (ATSDR 2002). Rice et al. (2000) showed that in one experimental trial, fish exposed to contaminated worms that contained a total PAH concentration of 11.3 µg/g dw had a lower daily growth rate than controls. In a second trial of the experiment, a similar trend was observed, but the effect was not

[80] Concentrations were reported as LPAHs and HPAHs; the specific individual PAHs quantitated were not reported.

statistically significant. This study was not selected to derive a LOAEL TRV because the worms used were exposed to field-collected sediments with uncharacterized chemicals, and the significance of the effects observed at this concentration was statistically ambiguous.

The literature was also searched to identify studies in which PAH risks to fish were evaluated based on contaminated sediment exposure. Three studies examined the effects of fish exposed to PAH-contaminated sediments. In the first study, fish were exposed to sediments spiked with benzo(a)pyrene in the laboratory. Kolok et al. (1996) exposed field-collected gizzard shad to sediments spiked with 1 mg/kg dw benzo(a)pyrene for 22 days. Gizzard shad are detritus feeders that incidentally ingest large amounts of sediment. The sediment was augmented with ground trout chow. No effects were reported on body mass between exposed and control fish. The UCL concentration of benzo(a)pyrene in LDW-wide sediment was 0.41 mg/kg dw and ranged from 0.26 to 0.72 mg/kg dw for the four Pacific staghorn sculpin areas. Thus, the UCL concentration of benzo(a)pyrene in LDW sediment is less than the NOAEL reported in Kolok et al. (1996).

In a study of sediment toxicity in the Elizabeth River, Virginia (Roberts et al. 1989), field-collected juvenile spot were exposed for 28 days to field-collected sediments from a clean site amended with 1, 3.2, 10, or 32% Elizabeth River sediments as well as to undiluted Elizabeth River sediment. Resulting sediment total PAH concentrations ranged from 4 (control) to 21,200 mg/kg dw. Fish exposed to 3.2% Elizabeth River sediment (total PAH concentration of 322 mg/kg dw) were reported to exhibit increased mortality and significant weight loss compared to controls.

In a study of sediment toxicity in Eagle Harbor, Washington, Kubin (1997) exposed field-collected juvenile English sole to sediment from a clean site amended with 0.8 and 1.6% Eagle Harbor sediment for 175 days. Sediment concentrations of total PAHs were approximately < 0.50 (control), 2.0, and 4.0 mg/kg dw. Growth was similar for all treatments for the first 3 months; growth was significantly lower in the high-exposure group (4.0 mg/kg dw) for the second 3 months. The percent change in weight was 0.35% per day for fish exposed to the highest concentration of PAHs compared to 0.43% in control fish. The fish exposed to sediments with 2.0 mg/kg dw total PAHs showed no significant decrease in growth rate relative to control fish. It is unknown whether these results are applicable to PAH exposure within the LDW because total PAHs and other chemicals associated with creosote waste may differ considerably from PAH mixtures in the LDW, both in terms of toxicity and bioavailability.

Estimation of NOAELs from LOAELs

There is also uncertainty if NOAELs were estimated from LOAELs because NOAELs were not available. This uncertainty was analyzed in the Phase 1 ERA (Windward 2003b). That analysis concluded that because NOAELs were generally found to be less than a factor of 5 different from LOAELs in studies reporting both, larger safety factors may overestimate the difference between LOAELs and NOAELs.

Selection of Salmon-Specific TRVs for Cadmium

As discussed in Section A.4.2.2.2, the selected LOAEL and NOAEL TRVs (0.5 and 1.0 mg/kg dw, respectively) were based on reduced growth of rockfish exposed to dietary cadmium; adverse effects in salmonids have been observed only at much higher dietary cadmium concentrations. Because juvenile chinook salmon as an ROC are representative solely of outmigrant juvenile salmonids in the LDW, the selected TRVs may result in the overprediction of risk for this ROC.

Salmonid-specific LOAELs for cadmium were reported in two studies, ranging from 1,395 mg/kg dw for mortality of rainbow trout fry exposed to dietary cadmium for 30 days (Szebedinsky et al. 2001) to 10,000 mg/kg dw, also for mortality of juvenile rainbow trout exposed to dietary cadmium for 28 days (Handy 1993) (Table A.6-21). Salmonid-specific dietary NOAELs were reported in five studies (Baldisserotto et al. 2005; Franklin et al. 2005; Lundebye et al. 1999; Mount et al. 1994; Szebedinsky et al. 2001), ranging from 55 mg/kg dw for growth of rainbow trout fry exposed to dietary and aqueous cadmium for 60 days (Mount et al. 1994) to 1,395 mg/kg dw, also for growth of rainbow trout fry exposed to dietary cadmium for 30 days (Szebedinsky et al. 2001) (Table A.6-21). Because a moderate number of the studies that were available evaluated dietary toxicity of cadmium to salmonids and these studies consistently showed that no adverse effects on growth or survival were observed below dietary concentrations of 55 mg/kg dw, it is unlikely that adverse effects would be observed in salmonids at lower concentrations. Assuming a NOAEL of 55 mg/kg dw, the NOAEL-based HQs would be less than 0.01 calculated using either exposure approach (diet or stomach content data). Based on this analysis, risk to juvenile chinook salmon from cadmium is likely to be very low.

Table A.6-21. Cadmium dietary toxicity studies for fish

Chemical	Test Species	NOAEL (mg/kg dw)	LOAEL (mg/kg dw)	Exposure Duration	Effect	Source	Notes
Cadmium chloride	rainbow trout fry	55	na	60 days	reduced body weight, length, or survival	Mount et al. (1994)	1, 2
Cadmium	Atlantic salmon	250	na	4 weeks	reduced growth rate (body weight)	Lundebye et al. (1999)	3, 4
Cadmium chloride	juvenile rainbow trout	294	na	15 – 30 days	reduced growth rate or survival	Baldisserotto et al. (2005)	4
Cadmium chloride	juvenile rainbow trout	471	na	28 days	reduced growth rate or survival	Franklin et al. (2005)	
Cadmium nitrate	juvenile rainbow trout	786	1,395	30 days	reduced survival	Szebedinsky et al. (2001)	4, 5
Cadmium nitrate	juvenile rainbow trout	1,395	2,265	30 days	reduced growth rate (weight)	Szebedinsky et al. (2001)	4
Cadmium sulfate	rainbow trout (130 g)	na	10,000	28 days	reduced survival	Handy (1993)	3, 4, 6

Notes:

1. Fish were exposed to copper, cadmium, lead, and zinc in water at 23.0, 0.97, 3.32, and 46.3 µg/L, respectively, at the same time as the dietary exposure to cadmium chloride. No effect on growth or survival was observed following exposure; therefore, a dietary NOAEL TRV was obtained from this study.
2. Fish fed live *Artemia* exposed to cadmium chloride in water. Dietary dose corrected for a theoretical 20% loss related to cadmium depuration from the *Artemia* food source.
3. Only the nominal concentration was reported.
4. Dietary dose was not reported as ww or dw and was assumed to be a dw concentration.
5. In a separate experiment reported in this study, 92% survival was reported for juvenile rainbow trout exposed to dietary cadmium concentrations of 1,419 mg/kg dw over a 39-day exposure period. Survival data were not statistically analyzed in either experiment.
6. Fish expelled food so the ingested dose is unknown.

na – not available

NOAEL – no-observed-adverse-effect level

LOAEL – lowest-observed-adverse-effect-level

COPCs Without LOAEL TRVs

As discussed in Section A.2.5.2, one juvenile chinook salmon COPC, chromium, had a maximum exposure concentration exceeding NOAEL TRVs and no LOAEL TRV was identified. Based on benthic invertebrate and juvenile chinook salmon stomach contents, NOAEL-based HQs for chromium were 0.18 and 2.1, respectively). Because no LOAEL toxicity data were available, the low HQs calculated using unbounded NOAEL TRVs are assumed to indicate low to very low risks to fish in the LDW from chromium.

Regional Field Studies

Three studies have been conducted with field-collected juvenile chinook salmon from the LDW. These studies are relevant because the studies are site-specific (i.e., fish collected from the LDW) (Table A.6-22). These studies evaluated effects on survival, growth, and immunocompetence in juvenile chinook salmon. Chemical-specific NOAELs and LOAELs cannot be determined from these studies because the fish were exposed to chemical mixtures.

Table A.6-22. LDW-specific studies of juvenile chinook salmon

Study	Exposure	Endpoint
Varanasi et al. (1993)	LDW field/Green River hatchery	survival, growth
Arkoosh et al. (1998)	LDW field/Green River hatchery	survival-disease susceptibility (*L. anguillarum*)
Casillas et al. (1995a; 1995b)	LDW field/Green River hatchery	growth, insulin-like growth factor

While survival and growth are widely recognized endpoints of importance in ERAs, immunocompetence of fish can also be an important factor in fish population demographics (Anderson and May 1979). The importance of immune function in a given fish population is influenced by several factors, including the quality of the environment, differential susceptibility of individual fish (as determined by genetics and physiological health), and the presence and virulence of infectious agents (Snieszko 1973).

Varanasi et al. (1993) collected juvenile chinook salmon from hatcheries and the respective estuaries of the Green/Duwamish and Nisqually rivers in 1989 and 1990 and assessed survival and growth of these fish under laboratory conditions. Casillas et al. (1995a; 1995b) reported that a growth experiment similar to that of Varanasi et al. (1993) was carried out in 1993, but the data have not been published. Arkoosh et al. (1998) collected juvenile chinook salmon from the LDW, the Nisqually River, and their respective upstream hatcheries (i.e., Green River and Kalama Creek, respectively) in the spring of 1993 and 1994, and assessed these fish for immunocompetence following acclimation to laboratory conditions.

Arkoosh et al. (1998) reported that LDW fish frequently had the highest cumulative mortality relative to Green River hatchery fish or Nisqually River estuary fish when exposed in the laboratory to the bacterium *Listonella anguillarum* (previously known as *Vibrio anguillarum*). Based on these results, the authors suggested that immunosuppression in juvenile chinook salmon from the LDW may lead to increased susceptibility to infection by a virulent marine bacterium. Because of the implied association between reduced immunocompetence in these fish and exposure of these fish to chemicals from the LDW, toxicity data were reviewed from laboratory studies investigating the immunocompetence of fish exposed to mixtures of either PCBs or PAHs to assess whether single chemical exposures would result in similar effects.

PAHs

The literature search identified two studies in which juvenile chinook salmon were exposed to dietary PAH mixtures and cumulative mortality was assessed following exposure to a pathogen (Palm et al. 2003; Bravo et al., undated draft).

Palm et al. (2003) investigated chinook salmon survival relative to controls following a dietary exposure of total PAHs up to 280 mg/kg dw and subsequent bacterial challenge under freshwater conditions.[81] No difference in mortality was observed between fish exposed to both dietary PAHs and *L. anguillarum* and fish exposed to *L. anguillarum* alone (controls). The highest dietary NOAEL was 280 mg/kg dw.

Bravo et al. (undated draft) exposed juvenile rainbow trout to either a mixture of non-alkylated PAHs (33 and 390 mg/kg dw), benzo(a)pyrene (BaP) (150 mg/kg dw), or benzo(e)pyrene (BeP) (150 mg/kg dw) in their diet for 50 days under freshwater conditions.[82] Following chemical exposure, fish were exposed to two serial concentrations of the pathogen *Aeromonas salmonicida*. At the lower bacterial concentration ($10^{5.5}$ cfu/mL), mortalities of fish fed the PAH mixture at both the 33 and 390 mg/kg dw concentrations were significantly higher than in controls. At the higher bacterial concentration ($10^{5.7}$ cfu/mL), mortality of fish exposed to 390 mg/kg dw was significantly higher than in *A. salmonicida*-exposed controls, and mortality of

[81] Converted from wet weight to dry weight based on reported 10.2% moisture content in food.
[82] Converted from wet weight to dry weight assuming 10% moisture content in food.

fish exposed to 33 mg/kg dw was not significantly higher than in controls. Mortality of BeP-exposed fish was significantly higher than in controls at the lower bacterial concentration but not at the higher bacterial concentration. Mortality of BaP-exposed fish was similar to controls at both bacterial concentrations. Bravo et al. concluded that the potential for reduced immunocompetence for juvenile chinook salmon exposed to dietary PAHs at both exposure levels was high.

In the LDW, the maximum concentration of total PAHs (including all alkylated and non-alkylated PAHs) in LDW benthic invertebrate tissue samples was 36.1 mg/kg dw and the UCL was 11 mg/kg dw (Table A-2.27). Because the UCL was less than the LOAEL of 33 mg/kg dw from Bravo et al. and well below the highest concentration in which no adverse effects were observed in juvenile chinook salmon (280 mg/kg dw from Palm et al. [2003]), risks of reduced immunocompetence for juvenile chinook salmon exposed to dietary PAHs appear to be low.

PCBs

The literature search identified three studies that evaluated the survival of fish following exposure to PCBs and subsequent immunological challenge (Powell et al. 2003; Mayer et al. 1985; Snarski 1982). Powell et al. (2003) investigated juvenile chinook salmon mortality relative to controls following a dietary exposure of up to 17 mg/kg ww Aroclor 1254 for 4 weeks and subsequent 14-day challenge with *Vibrio anguillarum* under freshwater conditions. Powell et al. (2003) reported no increase in juvenile chinook salmon mortality relative to controls. The tissue concentration of Aroclor 1254 in whole bodies of juvenile chinook salmon was 0.98 mg/kg ww at the highest dose.

Mayer et al. (1985) exposed 18-day-old rainbow trout for 90 days to an Aroclor 1254:1260 mixture in water. Fish were exposed to an LD50 concentration of *Yersinia ruckeri* via flush exposure at several times during the PCB exposure. Independent disease challenge trials were conducted at 45, 60, and 90 days of PCB exposure and again at 30 and 60 days following cessation of PCB exposure. In all exposures, time to 50% mortality was not significantly different between PCB-dosed fish and controls. The PCB concentration in tissue associated with the highest NOAEL was 120 mg/kg ww. The authors reported a trend toward higher resistance to disease in fish exposed to PCBs, which they suggested was a result of physical damage (reducing pathogen uptake) to the gills resulting from PCB exposure. In a separate experiment reported in this same paper, juvenile rainbow trout were exposed for 60 days to an Aroclor 1254:1260 mixture in water and then dosed once with *Yersinia ruckeri* via IP injection or flush exposure. No significant difference in survival between PCB-dosed fish and controls was reported for either IP injection or flush pathogen exposures.

Snarski (1982) reported that rainbow trout exposed to Aroclor 1254 in water at 14.7 µg/L for 30 days were less susceptible to the bacteria *Aeromonas hydrophila* than were controls. At this PCB dose, no mortality was observed in any fish following exposure to the pathogen. PCB concentrations in tissue were not reported.

Based on the available data, the tissue-residue NOAELs for survival following PCB exposure and subsequent immunological challenge ranged from 0.98 to 120 mg/kg ww. No LOAELs were identified. The maximum total PCB whole-body tissue concentration in juvenile chinook salmon from the LDW was 1.2 mg/kg ww. Therefore, total PCB tissue concentrations of juvenile chinook salmon within the LDW appear to be below concentrations at which no adverse immunosuppressive effects were observed.

Conclusion and Uncertainties

Based on the results of these laboratory tests, reduced immunocompetence in juvenile chinook salmon from exposure to PCBs or PAHs at concentrations in the LDW is not expected. However, there are a number of uncertainties associated with this conclusion. Uncertainty is associated with:

- The dietary approach for PAHs because water exposure and variation in exposures resulting from specific prey preferences could not be evaluated
- Application of laboratory data to fish in the field because the environmental conditions to which the juvenile salmon in the LDW are exposed, such as variable salinities, were not reflected by the laboratory studies, and the biological conditions of juvenile salmonids in the LDW, including smoltification, were not captured by the laboratory experiments
- Numerous differences between experimental and wild fish that may affect immunocompetence in juvenile chinook salmon such as differences in the lipid content of the fish in the laboratory relative to those in the field, as well as in lipids in their food items
- Interpretation of results from studies with field-exposed fish because field-exposure studies involve uncontrolled variables and include exposure to a mixture of chemicals and other stressors, complicating interpretation of the cause-and-effect relationship

Risk Characterization

Risks to fish from organochlorine pesticides were not included in the risk estimates because of high uncertainty in the JN-qualified tissue pesticide data, resulting in suspected false identifications of some pesticides as well as overestimates in their concentrations, as discussed in Section A.2.4.2. Exposure estimates, TRVs, and risk estimates are discussed in this section for endrin, which was identified as a COPC for juvenile chinook salmon in Section A.2.5.2.

TRVs were searched using the same techniques described in Section A.2.4.5.2. Five endrin toxicity studies were identified for five different species of fish (Table A.6-23). LOAELs ranged from 11.5 µg/kg ww for mortality of fingerling largemouth bass exposed to endrin in water for 120 days (Fabacher 1976) to 1,660 µg/kg ww for golden shiner exposed to endrin in water for 8 hours (Ludke 1976). The LOAEL of 11.5 µg/kg ww reported in Fabacher (1976) was selected as the LOAEL TRV. No NOAEL lower than the LOAEL was identified, so a NOAEL of 1.15 µg/kg ww was derived using a safety factor of 10 for an acute study because the study evaluated the survival endpoint. The selected endrin TRVs may underestimate toxicity because they are based on a survival endpoint. Although higher growth or reproduction LOAELS and NOAELs are reported for other fish species, it is not known if growth and reproductive endpoints would have resulted in lower TRVs for largemouth bass.

Table A.6-23. Critical tissue-residue toxicity studies of endrin in fish

Test Species	NOAEL (µg/kg ww)	LOAEL (µg/kg ww)	Exposure Route and Duration	Effect	Source	Notes
Largemouth bass fingerlings	**1.15**[a]	**11.5**	water for 120 days	reduced survival	Fabacher (1976)	static tanks treated every 5 days; residues from dead fish
Fathead minnow	na	240	food and water for 300 days	reduced survival	Jarvinen and Tyo (1978)	exposure during reproductive period
Channel catfish fingerling	307	na	food for 198 days	no effect on growth or survival	Argyle et al. (1973)	NOAEL is average residue from day 20 to day 198
Channel catfish fingerling	410	720	water for 55 days	reduced survival (40%)	Argyle et al. (1973)	LOAEL is concentration at 26 days when mortality began; NOAEL is mean concentration when residue levels were maximal from days 49 to 55
Sheepshead minnow	110	880	water for 4 weeks	reduced second generation juvenile survival	Hansen et al. (1977)	embryos spawned from field-collected adults; tissue residues in juvenile fish- effects based on mortality effects in fry
Sheepshead minnow	260	940	water for two generations	reduced female fertility	Hansen et al. (1977)	embryos spawned from field-collected adults
Golden shiner	na	1,660	water for 8 hours	100% mortality	Ludke et al. (1968)	LOAEL is average residue at time of death

[a] NOAEL estimated using uncertainty factor of 10 (acute LOAEL to chronic NOAEL).

LOAEL – lowest-observed-adverse-effect level

na – not available

NOAEL – no-observed-adverse-effect level

ww –wet weight

Bold identifies the NOAEL and LOAEL selected as TRVs.

The NOAEL- and LOAEL-based HQs for endrin and juvenile chinook salmon were 3.7 and 0.37, respectively (Table A.6-24). The selected endrin TRVs may underestimate the potential for sublethal effects because they are based on a survival endpoint. Although higher growth or reproduction LOAELs and NOAELs are reported for other fish species, it is not known if other endpoints would have resulted in lower TRVs for largemouth bass. The exposure concentration is probably an overestimate of actual exposure as a result of analytical interference from PCBs (Section A.2.4.2). Therefore, although the risks associated with this organochlorine pesticide are uncertain, they are likely lower than these HQs suggest.

Table A.6-24. HQs for juvenile chinook salmon and endrin based on a critical tissue-residue approach

UCL Concentration (µg/kg ww)	NOAEL TRV (µg/kg ww)	LOAEL TRV (µg/kg ww)	NOAEL-Based HQ	LOAEL-Based HQ
4.3	1.15	11.5	**3.7**	0.37

HQ – hazard quotient

LOAEL – lowest-observed-adverse-effect level

NOAEL – no-observed-adverse-effect level

TRV – toxicity reference value

UCL – upper confidence limit on the mean

ww – wet weight

Bold identifies NOAEL-based HQs greater than 1.0 and LOAEL-based HQs greater than or equal to 1.0.

Summary of Uncertainties

Uncertainties in the problem formulation, the effects and exposure assessments, and risk characterization for juvenile chinook salmon were evaluated, as summarized below:

- Uncertainties in ROC selection, assessment endpoints, dietary composition, water exposure to PAHs and metals, benthic invertebrate tissue data, and future habitat changes resulting from restoration are not expected to have an effect on risk conclusions.
- The field component of a dietary PAH exposure study conducted by Rice et al. (2000) and several other studies with field components were discussed to further evaluate uncertainties associated with PAH exposure and effects in fish. No TRVs were selected from these studies because the mixture of chemicals to which the fish were exposed was not fully characterized.
- Risks to juvenile chinook salmon from exposure to vanadium were slightly higher if juvenile chinook salmon were assumed to forage throughout the LDW rather than only in intertidal areas. However, it is not known if juvenile chinook salmon forage in deeper waters, such as the navigation channel, of the LDW.
- The use of safety factors of greater than 5 to estimate NOAELs from LOAELs may overestimate the difference between LOAELs and NOAELs.
- The effect of uncertainties associated with exposures of fish in the LDW to chemical mixtures is not known; it is possible that risks are over- or underestimated.
- Immunocompetence experiments conducted with juvenile chinook salmon collected from the LDW indicated that reduced immunocompetence may occur

in juvenile salmonids migrating through the LDW. However, PAH and PCB toxicity studies conducted in the laboratory did not result in reduced immunocompetence in juvenile chinook salmon from exposure to PAHs or PCBs at levels comparable to dietary concentrations of these chemicals that occur in the LDW. Uncertainties remain regarding the interpretation of the field studies as well as the application of the laboratory results to the field.

- Selection of non-salmonid TRVs for cadmium resulted in a LOAEL-based HQ equal to 1.0; however, salmonid-specific effects data show that risks are likely to be very low.
- Risks from chromium were low but uncertain because a LOAEL TRV was not identified, and the NOAEL-based HQ was 2.1 based on benthic invertebrate tissue data and 0.18 based on stomach contents data.
- An evaluation of risks to juvenile chinook salmon from exposure to endrin indicated a low risk. The selected TRVs are based on survival and may underestimate the potential for sublethal effects. However, because analytical interference from PCBs is likely to have overestimated actual organochlorine pesticide concentrations in LDW tissue samples, actual risks may be lower than the HQs suggest.

A.6.2.1.3 Risk conclusions

Juvenile chinook salmon were evaluated as an ROC to represent all migratory juvenile salmonids in the LDW and because they have been listed as threatened under the Endangered Species Act (Section A.2.2.1). Results of the risk characterization for juvenile chinook salmon are summarized in Table A.6-25.

Two lines of evidence were assessed for juvenile chinook salmon exposure to dietary COPCs: 1) dietary exposure based on a single composite sample of stomach contents (composited from 72 fish), and 2) dietary exposure based on consumption of benthic invertebrates. Dietary exposure based on benthic invertebrate tissue data resulted in higher HQs for all COPCs than did juvenile chinook salmon stomach content data.

No COPC had a LOAEL-based HQ exceeding 1.0. 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. There is uncertainty in these HQs because the cadmium TRVs were based on juvenile rockfish, and several toxicity studies using salmonids were available with TRVs 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 likely overestimate the sensitivity of juvenile chinook salmon to cadmium. NOAEL-based HQs for arsenic, copper, and vanadium were greater than 1.0 (1.1, 1.9, and 4.0, respectively), whereas LOAEL-based HQs for these COPCs were less than 1.0. Therefore, risks from these COPCs were low. Uncertainty in the TRV affects the risk conclusion for vanadium. The vanadium HQ is based on a NOAEL TRV extrapolated from an unbounded LOAEL TRV. No other dietary toxicity studies were available. Because of the high uncertainty in the effects data, risks from vanadium are uncertain and could be higher or lower than the HQs indicate.

Table A.6-25. Summary of risk characterization for juvenile chinook salmon

COPC	Highest NOAEL-Based HQ[a]	Highest LOAEL-Based HQ[a]	LOAEL Endpoint	Primary Uncertainty
Arsenic	**1.1**	0.73	juvenile growth	few toxicity studies available
Cadmium	**5.0**	**1.0**	juvenile growth	high TRV uncertainty[b]
Copper	**1.9**	0.93	juvenile growth	medium uncertainty[c]
Vanadium	**4.0**	0.79	juvenile growth	only one toxicity study available, foraging range exposure assumption

a Dietary exposure based on benthic invertebrate tissue data resulted in higher HQs for all COPCs than did juvenile chinook salmon stomach contents data. Thus, benthic invertebrate tissue HQs are shown in this table.

b Although a moderate number of toxicity studies were available, the six available dietary toxicity studies conducted with salmonids showed that salmonids in these studies were likely less sensitive to cadmium than suggested by the selected NOAEL and LOAEL TRVs.

c A large number of studies that presented a range of effects thresholds were available. Sufficient data from other studies were available to suggest that the lowest LOAEL was not supported by related research; therefore, this LOAEL was not selected. Furthermore, the available toxicity data suggest that the likely dietary threshold for toxic effects in salmonids is higher than the selected dietary LOAEL of 100 mg/kg dw.

COPC – chemical of potential concern

HQ – hazard quotient

LOAEL – lowest-observed-adverse-effect level

NOAEL – no-observed-adverse-effect level

TRV – toxicity reference value

Bold identifies NOAEL-based HQs greater than 1.0 and LOAEL-based HQs greater than or equal to 1.0.

A.6.2.2 English sole

This section presents risk estimates, uncertainties, and risk conclusions for English sole.

A.6.2.2.1 Risk estimates

COPCs evaluated for English sole included arsenic, cadmium, copper, and vanadium, which were evaluated using a dietary approach, and total PCBs, which were evaluated using a critical tissue-residue approach. Estimated exposure concentrations for English sole were greater than both LOAEL and NOAEL TRVs for cadmium and vanadium, each with a LOAEL-based HQ of 1.2 (Table A.6-26). Exposure concentrations of total PCBs were compared to the range of effect concentrations selected as TRVs. LOAEL-based HQs calculated with this range of LOAELs ranged from 0.98 to 5.0. NOAEL-based HQs ranged from 4.9 to 25 (Table A.6-26). Exposure concentrations of arsenic and copper exceeded NOAEL TRVs, with NOAEL-based HQs of 1.2 and 1.9, respectively; LOAEL-based HQs for arsenic and copper were less than 1.0.

Table A.6-26. HQ calculations for English sole

Approach	COPC	Exposure Concentration	Unit	NOAEL TRV	LOAEL TRV	NOAEL-Based HQ	LOAEL-Based HQ
Tissue residue	total PCBs	2,600	µg/kg ww	104 – 528	520 – 2,640	**4.9** – **25**[a]	0.98 – **5.0**[a]
Dietary	arsenic	24	mg/kg dw	20	30	**1.2**	0.80
	cadmium	0.61	mg/kg dw	0.1	0.5	**6.1**	**1.2**
	copper	93	mg/kg dw	50	100	**1.9**	0.93
	vanadium	12	mg/kg dw	2.04	10.2	**5.9**	**1.2**

[a] Because of the 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

dw – dry weight

HQ – hazard quotient

LOAEL – lowest-observed-adverse-effect level

NOAEL – no-observed-adverse-effect level

PCB – polychlorinated biphenyl

TRV – toxicity reference value

ww – wet weight

Bold identifies NOAEL-based HQs greater than 1.0 and LOAEL-based HQs greater than or equal to 1.0.

A.6.2.2.2 Uncertainty analysis

This section presents a discussion of the uncertainty associated with the problem formulation, the exposure and effects assessments, and the risk characterization for English sole.

Problem Formulation

Primary uncertainties in the problem formulation for English sole include ROC selection, assessment endpoints, and the COPC screen. Uncertainties associated with fish assessment endpoints are the same as those discussed in Section A.6.2.1.2 for juvenile chinook salmon. Uncertainties associated with ROC selection and the COPC screen are discussed below.

ROC Selection

English sole are benthic fish that live in close contact with sediments and thus have a high likelihood of exposure to sediment-associated chemicals through direct contact and through their diet. Other fish represented by English sole as an ROC have either similar exposure pathways (e.g., starry flounder), or less direct contact with sediments (e.g., shiner surfperch). As part of the Phase 2 RI, shiner surfperch were collected and analyzed to represent prey for various fish and wildlife ROCs. PCB concentrations in shiner surfperch tissue were also compared to TRVs to assess the suitability of English sole as a representative species. Shiner surfperch HQs are compared to English sole HQs for PCBs (the only critical tissue-residue COPC for English sole) in Table A.6-27.

Table A.6-27. HQ calculations for shiner surfperch and English sole for total PCBs

Species	Minimum PCB Conc. (µg/kg ww)	Maximum PCB Conc. (µg/kg ww)	Mean PCB Conc. (µg/kg ww)	UCL Conc. (µg/kg ww)	NOAEL TRV (µg/kg ww)	LOAEL TRV (µg/kg ww)	NOAEL-Based HQ	LOAEL-Based HQ
English sole	450	4,700	2,200	2,600	104 – 528	520 – 2,640	**4.9 – 25**	0.98 – **5.0**
Shiner surfperch	350	18,400	1,800	3,500	104 – 528	520 – 2,640	**6.6 – 34**	**1.3 – 6.7**

HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
NOAEL – no-observed-adverse-effect level
PCB – polychlorinated biphenyl
TRV – toxicity reference value
UCL –-upper confidence limit on the mean
ww – wet weight

Bold identifies NOAEL-based HQs greater than 1.0 and LOAEL-based HQs greater than or equal to 1.0.

The exposure concentration for shiner surfperch resulted in a LOAEL-based HQ range of 1.3 to 6.7, whereas the exposure concentration for English sole resulted in a LOAEL-based HQ range of 0.98 to 5.0. Because shiner surfperch and English sole HQs for PCBs were similar and the total PCBs LOAEL TRV is uncertain, shiner surfperch tissue data do not change risk estimates for fish represented by English sole as an ROC.

COPC Screen

Uncertainties associated with chemicals not analyzed in LDW tissue samples, chemicals for which toxicological data were not identified, and the mercury NOAEL TRV are the same as those discussed in Section A.6.2.1.2 for juvenile chinook salmon.

Risks to fish from exposures to BEHP, dimethyl phthalate, and di-n-butyl phthalate in the LDW are uncertain because although these chemicals were not detected in English sole whole-body tissue samples, reporting limits for these chemicals in tissue were greater than the selected NOAEL TRVs (Table A.2-21 in Section A.2.5.2). The reporting limits for BEHP were elevated because of analytical dilutions of the samples. Forty-nine tissue samples with BEHP RLs of 7,200 µg/kg ww were re-analyzed. BEHP was not detected in any of the re-analyzed tissue samples that had RLs from 66 to 130 µg/kg ww.

HQs calculated using minimum and maximum reporting limits relative to NOAEL TRVs for each chemical are presented in Table A.6-28. LOAEL TRVs were not available for any of these chemicals. HQs are also presented for shiner surfperch because BEHP was detected in 5/27 shiner surfperch tissue samples. Dimethyl phthalate and di-n-butyl phthalate detection frequencies were 0/27 and 1/27, respectively, in shiner surfperch.

Table A.6-28. HQs for BEHP, dimethyl phthalate, and di-n-butyl phthalate

Chemical	ROC	Reporting Limit (µg/kg ww)		TRV (µg/kg ww)		NOAEL-Based HQ			
		Min	Max	NOAEL	LOAEL	Min RL	Max RL		
BEHP	English sole	66	3,600	390	na	0.17	**9.2**		
	shiner surfperch	280[a]	2,100[b]	390	na	0.72	**5.4**		
Dimethyl phthalate	English sole	290	580	498	na	0.58	**1.2**		
	shiner surfperch	9.9	2,900	498	na	<0.1	**5.8**		
Di-n-butyl phthalate	English sole	290	1,200	1,170	na	0.25	1.0		
	shiner surfperch	40	2,300[b]	1,170	na	<0.1	**2.0**		

[a] Minimum detected concentration is presented.

[b] Maximum detected concentration is presented.

BEHP – bis(2-ethylhexyl) phthalate
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
na – no TRV available
nc – not calculated because no LOAEL TRV was available
Bold identifies NOAEL-based HQs greater than 1.0.

NOAEL – no-observed-adverse-effect level
RL – reporting limit
ROC – receptor of concern
ww – wet weight

All but one of the NOAEL-based HQs calculated using maximum reporting limits as exposure concentrations were greater than 1.0 (Table A.6-28). All NOAEL-based HQs calculated using minimum reporting limits were less than 1.0. Detected concentrations of BEHP and di-n-butyl phthalate in shiner surfperch resulted in HQs similar to English sole HQs based on reporting limits. Based on this analysis, and assuming that English sole and shiner surfperch have similar exposures to phthalates, phthalate risks to English sole are uncertain but appear to be low.

Exposure Assessment

Uncertainties in the exposure assessment for English sole were associated with the following factors:

- Water pathway for metals and PAHs
- Incidental sediment ingestion
- Foraging range
- Benthic invertebrate tissue data and linear regressions
- Future habitat changes resulting from restoration
- Use of starry flounder tissue in Area T4 as a surrogate for English sole

The uncertainties associated with exposure via the water pathway and future habitat changes are similar to those discussed for juvenile chinook salmon in Section A.6.2.1.2.

Incidental Sediment Ingestion

The exposure assessment for English sole assumed 1% of the diet was incidental sediment ingestion, as described in Section A.4.1.2. There is uncertainty in the percentage of sediment ingestion assumed because it is based on subjective observations by experienced fish biologists and not based on empirical data. Estimates ranged from 1% (Lange 2006) to as high as 10% (Johnson 2006). Therefore, uncertainty in dietary exposure estimates calculated using Equation 4-1, as described in Section A.4.1.2 were evaluated assuming 0, 1, and 10% sediment ingestion to bracket the 1% estimate (Table A.6-29).

Table A.6-29. English sole dietary exposure estimates based on three assumed incidental sediment ingestion scenarios

COPC	Dietary Exposure Concentrations Based on Different Incidental Sediment Ingestion Percentages (mg/kg dw)		
	0%	1%	10%
Arsenic	24	24	25
Cadmium	0.60	0.61	0.74
Copper	92	93	100
Vanadium	12	12	17

COPC – chemical of potential concern

dw – dry weight

An assumption of 10% sediment ingestion resulted in slightly higher exposure estimates for cadmium, copper, and vanadium, whereas an assumption of 0% sediment ingestion resulted in slightly lower exposure estimates for these COPCs. However, HQs did not change substantially for any COPC (Table A.6-30), and HQs did not change from less than 1.0 to greater than 1.0 or vice versa, except for copper, for which the LOAEL changed from 0.93 to 1.0.

Table A.6-30. Dietary HQs for English sole as a function of sediment consumption

COPC	0% Sediment consumption		1% Sediment Consumption		10% Sediment Consumption	
	NOAEL-Based HQ	LOAEL-Based HQ	NOAEL-Based HQ	LOAEL-Based HQ	NOAEL-Based HQ	LOAEL-Based HQ
Arsenic	**1.2**	0.80	**1.2**	0.80	**1.3**	0.83
Cadmium	**6.0**	**1.2**	**6.1**	**1.2**	**7.4**	**1.5**
Copper	**1.8**	0.92	**1.9**	0.93	**2.0**	**1.0**
Vanadium	**5.9**	**1.2**	**5.9**	**1.2**	**8.3**	**1.7**

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.

Foraging Range

English sole were assumed to forage exclusively in the LDW even though it is known that they migrate seasonally to spawn in Puget Sound. Uncertainty analysis in the Phase 1 ERA (Windward 2003b) concluded that the assumption of foraging throughout the LDW may potentially overestimate or underestimate exposure depending on the relative magnitude and extent of contamination in other foraging areas.

Benthic Invertebrate Tissue Data and Linear Regressions

The uncertainties associated with benthic invertebrate tissue data in the exposure assessment for English sole are the same as those discussed in Section A.6.2.1.2 for juvenile chinook salmon, with the following exception. More samples were used in the LDW-wide EPC calculations for English sole (n = 20 or 24 depending on the COPC) than for the intertidal EPC calculations for juvenile chinook salmon (n = 13 or 17, depending on the COPC), which improves confidence that the combined sample is representative of English sole exposures in the LDW.

If the exposure estimate had been calculated using empirical benthic invertebrate tissue data rather than the regression, the UCL would have been 37 mg/kg dw, in comparison to the UCL of 24 mg/kg dw estimated using the regression. Assuming an exposure concentration of 37 mg/kg dw, the LOAEL-based HQ would have increased from 0.8 to 1.2, still suggesting a relatively low risk from arsenic exposure.

Use of starry flounder tissue data from Area T4

There is uncertainty associated with the LDW-wide English sole tissue concentrations because three starry flounder composite tissue samples from Area T4 were included in the English sole dataset. These three samples were included because in the 2004 sampling event, it was difficult to catch a sufficient number of English sole to populate six composite tissue samples of English sole. Thus, in 2004, three starry flounder and three English sole composite tissue samples were analyzed. Total PCB concentrations in starry flounder tissue composite samples were lower than those in English sole composite tissue samples from Area T4 in 2004 (mean concentrations of 570 µg/kg ww vs. 1700 µg/kg ww). Therefore, inclusion of starry flounder data may have decreased the exposure estimate because concentrations of total PCBs in starry flounder were lower than those in English sole composite samples from the same area.

Effects Assessment

Uncertainties in the effects assessment for English sole were associated with the following factors:

- Effects from chemical mixtures and estimation of NOAELs from LOAELs
- Exclusion of field studies from TRV selection
- PCB TRV

- Cadmium TRV
- COPCs without LOAEL TRVs
- TEQ approach for PCBs
- Regional field studies
- Critical tissue residue approach

In addition, some of the selected TRVs are considered less certain than others if there were a small number of studies, if endpoints were subchronic, or if data quality was questionable. The relative uncertainties in the selected TRVs for each English sole COPC are the same as those discussed in Section A.6.2.1.2 for juvenile chinook salmon, except for the TRVs for cadmium and total PCBs, which are discussed below.

Effects of Chemical Mixtures and Estimation of NOAELs from LOAELs

Uncertainties associated with effects of chemical mixtures and estimation of NOAELs from LOAELs are the same as those discussed in Section A.6.2.1.2 for juvenile chinook salmon with the following additional uncertainty for PCBs, which were not evaluated as a COPC for juvenile chinook salmon. The laboratory effects studies included in the total PCBs assessment used unweathered Aroclor mixtures. PCBs present in fish tissue have undergone physico-chemical weathering and differential accumulation in the food web, resulting in PCB congener mixtures that are potentially more or less biologically active than the commercial Aroclor mixtures. Because laboratory toxicity tests evaluated for TRV selection generally were conducted with a single Aroclor mixture, the potency of PCB mixtures in LDW fish relative to those in fish from toxicity studies is uncertain.

Exclusion of Field Studies from TRV Selection

Uncertainties discussed in Section A.6.2.1.2 for juvenile chinook salmon associated with the exclusion of field studies from TRV selection are relevant to English sole. Rice et al. (2000) tested the toxicity of sediments contaminated with PCBs, PAHs, and other chemicals using English sole. This study could not be used to derive a PCB or a PAH TRV for English sole because the test sediments were taken from contaminated waterways (Eagle Harbor and Commencement Bay) that contained mixtures of chemicals that included more than PAHs or PCBs, respectively, confounding interpretation of results. The direct sediment exposure studies discussed in Section A.6.2.1.2 are also relevant to English sole, a benthic species.

PCB TRV

As discussed in Section A.4.2.2.1, there is uncertainty in selecting the TRV for total PCBs and fish because: 1) there is uncertainty in interpretation of several studies reporting the lowest concentrations associated with effects, and 2) some effects concentrations were expressed as egg/embryo-tissue concentrations rather than whole-body concentrations.

As noted in Section A.4.2.1.1, there are a number of uncertainties associated with the Hugla and Thome (1999) study reporting the lowest effects concentrations. Because of these uncertainties, rather than selecting a single LOAEL, the range of effects concentrations reported in this paper for the fecundity endpoint (520 and 2,640 µg/kg ww) and for the spawning and egg hatchability endpoints (2,640 µg/kg ww) were considered to represent the lowest exposure levels over which adverse reproductive effects may occur in these fish. This section presents a detailed evaluation of uncertainties associated with this study and implications for risk conclusions. Uncertainties discussed include those associated with the statistical analysis for the fecundity endpoint and the fact that this endpoint was not dose responsive, uncertainties related to test conditions, and uncertainties in the estimation of the whole-body concentration associated with effects.

The number of fish used to evaluate effects endpoints in Hugla and Thome (1999) is unclear. Hugla and Thome (1999) presented the following information:

- A sample size of six was used in the statistical analysis conducted to assess the significance of effects in exposed fish relative to control fish.
- Ten males and six females were exposed in the 2.5-µg/g PCB treatment.
- Six fish were analyzed for PCB concentrations after 50 days of exposure.
- Six ovaries were analyzed 1 year later.

Thus, from the information presented in the Hugla and Thome (1999), it would appear that whole-body concentrations in the fish analyzed at 50 days must be for males only (if any females were analyzed at 50 days, fewer than six would have been available for analysis 1 year later). However, in recent correspondence, the authors have stated that both male and female fish were included in whole-fish tissue analyses conducted at 50 days (Leroy 2007 [pers. comm.]). They also indicated that the total number of fish exposed may have been incorrectly reported in the paper.

Understanding the number of spawnings and the number of fish tested is critical to statistical analysis and interpretation of results. Hugla and Thome (1999) state that a one-way analysis of variance (ANOVA) was used to analyze the data, and ANOVA assumes independence of observations. In recent correspondence (Leroy 2007 [pers. comm.]), the authors reported that three fish were spawned two times each. Under this scenario, the six resulting data points for each female reproductive endpoint would not be independent. Statistical analyses conducted using a sample size of 3 (exposed fish) rather than a sample size of 6 (spawning events), would result in a concomitant reduction of statistical power and potentially different conclusions about the differences in fecundity between the control and the exposed fish.

Another element of uncertainty in the fecundity LOAEL is that observed effects on fecundity were not dose responsive after two spawning seasons, as noted in Section 4.2.1.1. While the PCB-treated fish produced half as many eggs as the control fish, and the number of eggs produced by the control fish was consistent with other studies for this species (Philippart et al. 1989), the average fecundity after two spawning seasons was similar between the high and low doses; variance in the fecundity endpoint was greater at the higher dose. Furthermore, no barbel PCB toxicity data are available from other studies to compare with the fecundity effect reported in Hugla and Thome (1999).

In addition, the fish holding and exposure conditions used by Hugla and Thome (1999) may have influenced the reproductive effects observed. In recent correspondence, the authors have provided more detailed information than was discussed in the 1999 publication. Fish were kept in artificially heated water and were spawned when 4 years old (Leroy 2007 [pers. comm.]). A separate study indicated that increased water temperatures of 20 to 24° C at the facility where this study was conducted were used to accelerate the sexual maturation of the experimental fish (Philippart et al. 1989). Philippart et al. showed that by manipulating the temperature and/or photoperiod under which fish are reared, barbel are spawned at an earlier age and smaller size than the typical minimum spawning age of 6 years. Because temperature was also used to affect the barbel reproductive cycle in this study, it is uncertain whether these manipulations may have also affected the sensitivity of barbel reproduction to PCBs. The fecundity of the control fish was similar to that found in another study with this species under similar conditions (Philippart et al. 1989); therefore, the effect of temperature, if any, is not known.

Therefore, because of the uncertainties in the statistical analysis, in the effects of elevated fish holding and exposure temperatures, and in the exposure-response relationship for fecundity, the fecundity LOAEL from the low dose is highly uncertain. There is greater confidence in the higher whole-body LOAEL as an effects level for these fish because this treatment resulted in complete reproductive failure following exposure in the first spawning year, and resulted in 96% mortality in offspring 1 year after exposure (relative to 52.4% in controls). Furthermore, increased egg/larval mortality was reported at higher PCB concentrations in eggs (i.e., the response appeared to be exposure-dependent). Given these uncertainties and the lack of confirming studies using the same species, the range of LOAEL TRVs provided by Hugla and Thome (1999) are considered to provide a conservative assessment of PCB risks to fish.

If the next higher LOAEL of 9,300 µg/kg ww for sheepshead minnow egg and larval survival (Hansen et al. 1974a) has been selected as the LOAEL TRV, the LOAEL-based HQ for English sole would have been 0.28. The highest NOAEL below this LOAEL was 1,900 µg/kg ww from the same study. If this NOAEL had been selected as the NOAEL TRV, the NOAEL-based HQ would have been 1.4. Uncertainties in the Hansen et al. (1974a) study include elevated PCB concentrations of 520 to 640 µg/kg ww in control fish and enhancement of egg production by injecting the fish with human chorionic gonadotrophic hormone. Potential confounding effects of hormonal injections on egg survival are unknown.

Four available studies presented PCB effects concentrations in egg and embryo tissues. Effects concentrations ranged from 857 to 77,900 µg/kg ww in egg and embryo tissues (Fisher et al. 1994; Freeman and Idler 1975; Hendricks et al. 1981; McCarthy et al. 2003). The lowest effects concentration was for reduced growth of Atlantic salmon fry held in PCB-free water for 176 days following egg exposure to aqueous PCB concentrations of 625 to 62,500 µg/L for 48 hours (Fisher et al. 1994). The highest effects concentration was for brook trout embryos exposed to 200 µg/L of PCBs in water for 21 days (Freeman and Idler 1975). No NOAELs were identified.

Although these egg and embryo effects concentrations were generally lower than effects concentrations in more mature fish, egg/embryo and adult tissue-residue data are not directly comparable. Species-specific ratios relating PCB concentrations in maternal adults to unfertilized eggs for yellow perch, smallmouth bass, white bass, white sucker, and rainbow trout ranged from 0.83 to 2.35 (Niimi 1983). Adjustment for a 2-fold weight increase in PCB concentration in fertilized eggs relative to unfertilized eggs (Niimi 1983) resulted in adult-to-fertilized egg ratios ranging from 1.7 to 4.7 for these species. Sheepshead minnow adult-to-fertilized-egg ratios were reported to range from 0.90 to 2.3 (Hansen et al. 1974a). Therefore, based on these studies, the ratio of PCB concentrations in fertilized eggs to those in maternal adults would likely range from 0.90 to 4.7. This range is uncertain because data represent only six fish species, with little to no replication.

Using this range of adult-to-egg PCB ratios, the maternal adult PCB concentrations associated with the reported egg LOAELs resulted in an estimated whole-body LOAEL range of 771 to 366,000 µg/kg ww, which is comparable to the range of measured whole-body LOAELs from studies with adults (520 to 429,000 µg/kg ww). Although there are additional uncertainties associated with these studies, based on the likely range of adult tissue concentrations extrapolated from these studies, use of egg LOAELs would not have resulted in risk conclusions different from those based on whole-body effects data.

Cadmium TRV

The lowest cadmium dietary LOAEL and NOAEL TRVs (0.5 and 0.1 mg/kg dw, respectively) for growth of juvenile rockfish were more than an order of magnitude lower than the other cadmium dietary TRVs and the observed effect was partially attributed to reduced food intake (see Section A.4.2.2.2). The next higher dietary LOAEL was 800 mg/kg dw (for growth of guppy) with a corresponding NOAEL of 500 mg/kg dw from the same study (Hatakeyama and Yasuno 1987). Both studies evaluated growth effects, although the fish species and form of cadmium differed. These results show that although few species have been investigated, there is wide variability in dietary toxicity data reported for cadmium.

HQs calculated using the next higher LOAEL and its associated NOAEL TRVs (Hatakeyama and Yasuno 1987) would have been less than 1.0 for English sole. Thus, risks from cadmium are uncertain for English sole and other fish represented by English sole (Section A.6.2.2.3).

COPCs Without LOAEL TRVs

As discussed in Section A.2.5.2, four chemicals had maximum exposure concentrations exceeding NOAEL TRVs but LOAEL TRVs were not available. One chemical (chromium) was evaluated using a dietary approach, and three chemicals (benzoic acid, dimethyl phthalate, and di-n-butyl phthalate) were evaluated using a critical tissue-residue approach. Risks to English sole from dimethyl phthalate and di-n-butyl phthalate were evaluated in the exposure assessment uncertainty section above because concentrations of these chemicals were generally below detection limits in fish tissue samples.

Dietary concentrations of chromium were calculated using Equation 4-1, as described in Section A.4.1.2. Benzoic acid critical tissue-residue concentrations were calculated as the UCL of the mean of all English sole tissue composite samples from throughout the LDW. NOAEL-based HQs were 1.1 for chromium and 1.5 for benzoic acid (Table A.6-31). Because no LOAEL toxicity data were available for these chemicals, the low HQs calculated using unbounded NOAEL TRVs are assumed to indicate low risks.

Table A.6-31. English sole NOAEL-based HQs for chemicals without LOAEL TRVs

COPC	Units	Exposure Concentration	NOAEL TRV	NOAEL-Based HQ
Chromium	mg/kg dw	10	9.4[a]	**1.1**
Benzoic acid	µg/kg ww	5,100	3,380[b]	**1.5**

a No effects on growth of 2-year-old gray mullet exposed to chromium III through diet and sediment for 8 weeks (Walsh et al. 1994).

b No effects on survival of mosquitofish exposed in a mesocosm for 24 hours (Lu and Metcalf 1975).

COPC – chemical of potential concern

dw – dry weight

HQ – hazard quotient

NOAEL – no-observed-adverse-effect level

TRV – toxicity reference value

ww – wet weight

Bold identifies NOAEL-based HQs greater than 1.0.

Toxic Equivalent Approach for PCBs

Risks from PCBs were evaluated in this ERA based on total PCB concentrations in fish tissue. Risks from PCBs can also be evaluated on a subset of PCB congeners that have dioxin-like properties, using a TEQ approach. English sole tissues were analyzed for dioxin-like PCB congeners and PCB Aroclors. PCB TEQ exposures calculated for

English sole were compared to the lowest LOAEL and NOAEL TRVs for 2,3,7,8-TCDD (2,3,7,8-TCDD toxicity studies are discussed in Attachment 8).

The lowest LOAEL for 2,3,7,8-TCDD was estimated using a study in which hatchery-reared adult female rainbow trout were exposed to dietary 2,3,7,8-TCDD for up to 320 days (Giesy et al. 2002; Jones et al. 2001). Dietary doses were 0, 1.8, 18, and 90 ng/kg ww. Giesy et al. (2002) reported significant effects on adult survival in fish fed 1.8 ng/kg ww.[83] Fillet tissue concentrations of 0.44 ng/kg ww were reported at day 200 in fish fed 1.8 ng/kg ww (Jones et al. 2001). The authors estimated that the fillet concentration reported in fish fed 1.8 ng/kg ww at day 200 was a 28-fold underestimation of the fillet concentration when mortality became significant (Giesy 2006). Therefore, the fillet concentration associated with adverse effects is assumed to be approximately 12 ng/kg ww. Whole-body data were not reported.

Because this study did not meet the TRV selection criteria presented in Section A.2.5.2,[84] but it was the only long-term 2,3,7,8-TCDD toxicity study available for fish, TEQ risks for fish are uncertain. Because a NOAEL lower than the LOAEL was not available, the NOAEL was calculated by dividing the selected LOAEL by 5, resulting in a NOAEL TRV of 2.4 ng/kg ww (for fillets).

TEQs were calculated for dioxin-like PCB congeners in eight English sole composite tissue samples.[85] WHO TEFs for fish reported in Van den Berg et al. (1998) were used to calculate fish TEQs. NOAEL- and LOAEL-based TEQ HQs for English sole were less than 1.0 (Table A.6-32). HQs calculated using a total PCB approach are presented in Table A.6-32 for comparison. HQs using the total PCB approach were higher than the PCB TEQ HQs, indicating that the total PCB approach should provide health-protective risk estimates for fish for PCBs. However, risk estimates for fish based on TEQs were calculated using only the dioxin-like PCB congeners because dioxin and furan tissue data were not available. Thus, risks associated with exposure to all dioxin-like chemicals were likely underestimated; the degree of underestimation is uncertain.

[83] There was 2.3 ng TCDD TEQ/kg ww of background PCB contamination in the diet, which was not accounted for in radiometric analyses of 2,3,7,8 TCDD.

[84] Neither whole-body nor egg tissue concentrations were presented for the time period when adverse effects were observed. Instead, fillet concentrations were estimated by the author from data collected at a different time.

[85] One starry flounder was included in the composite sample for Sampling Area 4, where few English sole were captured. Data are described in Section A.2.4.1.2.

Table A.6-32. HQs for English sole total PCBs and PCB TEQs

Basis	UCL Concentration	NOAEL TRV	LOAEL TRV	NOAEL-Based HQ	LOAEL-Based HQ
PCB TEQ	1.88 ng/kg ww	2.4 ng/kg ww	12 ng/kg ww	0.78	0.16
Total PCBs (sum of Aroclors)	2,600 µg/kg ww	104 – 528 µg/kg ww	520 – 2,640 µg/kg ww	**4.9 – 25**	0.98 – **5.0**

HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
NOAEL – no-observed-adverse-effect level
PCB – polychlorinated biphenyl
TEQ – toxic equivalent
TRV – toxicity reference value
UCL – upper confidence limit on the mean
ww – wet weight

Bold identifies NOAEL-based HQs greater than 1.0 and LOAEL-based HQs greater than or equal to 1.0.

Regional Field Studies

Several regionally relevant studies have been conducted with field-collected English sole from the LDW (Table A.6-33). Studies with LDW-exposed fish are discussed in detail in the Phase 1 ERA (Section A.4.3) and summarized in this section because of the relevance of the exposure route (i.e., field exposure in the LDW) and mixture of chemicals. However, chemical-specific NOAELs and LOAELs cannot be determined from these studies because the fish were exposed to chemical mixtures under uncontrolled conditions.

Table A.6-33. Growth, reproductive effects, and survival studies using English sole collected from the LDW

Study	Collection location	Effect
Johnson et al. (1988; 1997a; 1998; 1999)	Duwamish Waterway	inhibited gonadal development, altered plasma 17 beta-estradiol levels, reduced fecundity
Casillas et al. (1991a)	Duwamish Waterway	reduced spawning success
Rhodes et al. (1987)	two locations: Upper Duwamish Waterway and Lower Duwamish Waterway	reduced survival, increased lesion incidence
Johnson and Landahl (1994)	four or five locations in LDW	reduced survival, increased lesion incidence

LDW – Lower Duwamish Waterway

English sole collected from the LDW are more likely to have lesions than the sole collected from less-contaminated locations (Johnson and Landahl 1994; Rhodes et al. 1987). Hepatic neoplasms were found to be confined mainly to fish in urban areas (LDW, Commencement Bay [Hylebos Waterway], and the harbor area of Everett, Washington) (Malins et al. 1984).

Potential reproductive effects on English sole exposed to sediment-associated chemicals in the LDW and other sites throughout Puget Sound have been assessed using various endpoints. Concentrations of the female reproductive hormone 17 beta-estradiol in plasma have been analyzed (Johnson et al. 1997a; 1998; 1999), and ovarian development (Johnson et al. 1988; 1993), spawning success (Casillas et al. 1991a), and fecundity (Johnson et al. 1997a) have been assessed.

English sole from the LDW have been reported to exhibit inhibited gonadal development (Johnson et al. 1988), depressed plasma estradiol and reduced ovarian production *in vitro* (Johnson et al. 1988; 1993), and reduced spawning success (Casillas et al. 1991a). The reports suggest that these effects are a result of elevated concentrations of aromatic and chlorinated hydrocarbons present in LDW and Eagle Harbor sediments. These elevated concentrations have also been suggested to be significant risk factors for development of these reproductive abnormalities (Johnson et al. 1988; Casillas et al. 1991a).

These field studies are supported by laboratory experiments showing that pretreatment of gravid female English sole with extracts of contaminated sediment or crude oil (containing high levels of PAHs) decreased levels of endogenous estradiol (Johnson et al. 1995; Stein et al. 1991). Related experiments suggest that exposure to BaP- or PAH-contaminated sediment may suppress estradiol-induced vitellogenin production in fish, including English sole (Anulacion et al. 1997; Nicolas 1999).

Chemicals implicated as causal factors include PAHs and PCBs; however, linking the results of field studies to risks from specific chemicals is difficult considering, among other factors, the complex mixtures of chemicals in the field and the uncertainties in English sole home range. In addition, interpreting cause and effect of the adverse effects reported in field studies is complicated because of genetic variation, health, and seasonal variation in the spawning cycle. Therefore, although regional studies indicate an increased risk of adverse effects on English sole reproduction in the LDW, these effects cannot be conclusively associated with exposure to specific chemicals or chemical mixtures in the LDW.

Critical Tissue-Residue Approach

Uncertainties inherent in the use of chemical concentrations in tissue to estimate exposure from sediment-associated COPCs in the LDW include: 1) chemical concentrations detected in tissue may not accurately reflect the internal dose at target organs; 2) variability in chemical structure, species sensitivity, and biotransformation may affect critical tissue residues; and 3) the parent compound may be metabolized to more toxic chemicals. The influence of uptake or depuration kinetics on biological responses can also affect tissue residues. For example, short-term exposure of fish to relatively high chemical concentrations that elicited toxicity have been shown to result in lower tissue chemical concentrations than those reported in longer-term exposures to lower chemical concentrations that did not result in adverse effects (e.g., van Wezel et al. 1995).

Risk Characterization

Uncertainties in the risk characterization for English sole were associated with the organochlorine pesticides data.

Risks to fish from organochlorine pesticides were not included in the risk estimates because of high uncertainty in the tissue pesticide data resulting in suspected false identifications of some pesticides as well as overestimates in their concentrations, as discussed in Section A.2.4.2. Therefore, exposure estimates, TRVs, and risk estimates are presented in this uncertainty section for the three organochlorine pesticides (alpha-endosulfan, beta-endosulfan, and endrin) that were identified as COPCs in Section A.2.5.2.

TRVs for endosulfan and endrin were selected using the same techniques described in Section A.2.5.2. One endosulfan toxicity study reported that spot, pinfish, and mullet were exposed to endosulfan in water for 96 hours (Table A.6-34). The lowest LOAEL in the study (31 µg/kg ww) was reported for spot. No lower NOAEL was identified, so a NOAEL of 3.1 was derived using a safety factor of 10 because this was an acute study. As discussed in Section A.6.2.2.2, six endrin toxicity studies were identified for six different species of fish (Table A.6-2). LOAELs ranged from 11.5 µg/kg ww for mortality of fingerling largemouth bass exposed to endrin in water for 120 days (Fabacher 1976) to 1,660 µg/kg ww for golden shiner exposed to endrin in water for 8 hours (Ludke 1976). The LOAEL of 11.5 µg/kg ww reported in Fabacher (1976) was selected as the LOAEL TRV. No NOAEL lower than the LOAEL was identified, so a NOAEL of 1.2 µg/kg ww was derived for endrin using a safety factor of 10 for an acute study. The selected endrin TRVs may underestimate the potential for sublethal effects because they are based on a survival endpoint. Although higher growth or reproduction LOAELS and NOAELs are reported for other fish species, it is not known if other endpoints would have resulted in lower TRVs for largemouth bass.

Table A.6-34. Critical tissue-residue toxicity studies of endosulfan in fish

Test Species	NOAEL (µg/kg ww)	LOAEL (µg/kg ww)	Exposure Route and Duration	Effect	Source	Notes
Spot	**0.62**[a]	**31**	water for 96 hours	reduced survival (65%)	Schimmel et al. (1977a)	tissue residues of surviving fish
Pinfish	195	272	water for 96 hours	reduced survival (65%)	Schimmel et al. (1977a)	tissue residues of surviving fish
Mullet	na	360	water for 96 hours	reduced survival (60%)	Schimmel et al. (1977a)	tissue residues of surviving fish

[a] NOAEL estimated using uncertainty factor of 50 (acute LC50 LOAEL to chronic NOAEL).

LOAEL – lowest-observed-adverse-effect level

NOAEL – no-observed-adverse-effect level

na – not available

ww – wet weight

Bold identifies the NOAEL and LOAEL selected as TRVs.

HQs for endosulfan and endrin are presented in Table A.6-35. The LOAEL-based HQ for endrin was greater than 1.0. LOAEL-based HQs for alpha-endosulfan and beta-endosulfan were less than 1.0. NOAEL-based HQs were greater than 1.0 for all three COPCs. Endosulfan effects data are uncertain because only three studies were available, and all evaluated the survival endpoint; thus, risk of sublethal effects may be underestimated. However, because the exposure concentrations are probably overestimates of actual exposure (Section A.2.4.5), it is likely that risks associated with organochlorine pesticides are lower than HQs suggest.

Table A.6-35. HQs for English sole and endosulfan and endrin

COPC	UCL Concentration (µg/kg ww)	NOAEL TRV (µg/kg ww)	LOAEL TRV (µg/kg ww)	NOAEL-Based HQ	LOAEL-Based HQ
alpha-endosulfan	4.2	0.62	31	**6.8**	0.14
beta-endosulfan	18	0.62	31	**29**	0.58
endrin	14	1.2	11.5	**12**	**1.2**

COPC – chemical of potential concern
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
NOAEL – no-observed-adverse-effect level
TRV – toxicity reference value
UCL – upper confidence limit on the mean
ww – wet weight
Bold identifies NOAEL-based HQs greater than 1.0 and LOAEL-based HQs greater than or equal to 1.0.

Summary of Uncertainties

Uncertainties in the problem formulation, the effects and exposure assessments, and risk characterization for English sole were evaluated, as summarized below:

- Uncertainties in 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 are not expected to have an effect on risk conclusions.
- Three starry flounder composite tissue samples from Area T4 were included in the LDW-wide estimate of total PCB concentration in English sole because starry flounder served as a surrogate species in this area where English sole were more difficult to catch. Inclusion of starry flounder data may have decreased the exposure estimate because concentrations of total PCBs in starry flounder were lower than those in English sole composite tissue samples from the same area.
- A field study investigating effects on growth of English sole exposed to worms previously exposed in the laboratory to clean sediment amended with 0.1% of sediments from the Eagle Harbor, Washington, Superfund site was not selected as a PAH TRV because the worms used were exposed to field-collected sediments with uncharacterized chemicals, and the significance of the effects observed was statistically ambiguous.
- Effects data for cadmium are highly variable, with most studies showing lower sensitivity than the selected TRV. There is also uncertainty associated with the selected TRV because effects were partially attributed to reduced feeding. Thus, risks from cadmium are low but uncertain.

- Risks from exposure to chromium and benzoic acid were low but uncertain because LOAEL TRVs were not identified and NOAEL-based HQs were close to 1.0.
- Phthalates were not detected in English sole tissue samples. Based on an analysis of RLs and shiner surfperch tissue data, risks from phthalates are likely to be very low.
- Risks from exposure to chromium and benzoic acid, for which no LOAELs were available, are likely to be low because NOAEL-based HQs were just over 1.0.
- Risks to English sole from dioxin-like PCB congeners that were estimated using a TEQ approach were very low and would not change risk conclusions for PCBs.
- Because of numerous uncertainties in the study reporting the lowest PCB effects concentrations, risks to English sole from PCBs are uncertain. LOAEL-based HQs based on toxicity data reported in Hugla and Thome (1999) ranged from 0.98 to 5.0.
- Safety factors of greater than 5 used to estimate NOAELs from LOAELs may overestimate the difference between LOAELs and NOAELs.
- There is uncertainty in the critical tissue-residue approach because concentrations may not reflect the site of action, chemicals may be metabolized, and species- and chemical-specific factors may be important.
- An evaluation of risks from exposure to organochlorine pesticides indicated low risks from endrin and endosulfan. The selected TRVs may underestimate the potential for sublethal effects. However, because of analytical interference from PCBs, the risks are likely to be lower than the HQs suggest.
- Site-specific studies suggest the potential for adverse effects on English sole reproduction; however, these effects have not been conclusively associated with specific chemicals or chemical mixtures.

A.6.2.2.3 Risk conclusions

The English sole ROC was evaluated to represent all fish in the LDW not specifically covered by juvenile chinook salmon or Pacific staghorn sculpin. English sole are more highly exposed to sediment-associated chemicals based on their close sediment proximity and diet of benthic invertebrates. To provide a conservative estimate of risk from COPCs in the calculation of HQs, toxicological data from the most sensitive fish species tested for these COPCs were compared to exposure concentrations detected or estimated in English sole. Results of the risk characterization for English sole are summarized in Table A.6-36.

Table A.6-36. Summary of risk characterization for English sole

APPROACH	COPC	NOAEL-BASED HQ	LOAEL-BASED HQ	EFFECTS	PRIMARY UNCERTAINTY
Tissue residue	total PCBs	**4.9 – 25**[a]	0.98 – **5.0**[a]	reduced fecundity; lack of spawning in first reproductive season; egg and larval mortality[b]	high TRV uncertainty[c]
Dietary	arsenic	**1.2**	0.80	reduced body weight	moderate number of toxicity studies
	cadmium	**6.1**	**1.2**	reduced growth rate and condition factor	high TRV uncertainty[d]
	copper	**1.9**	0.93	reduced growth rate	medium TRV uncertainty[e]
	vanadium	**5.9**	**1.2**	reduced body weight	only one toxicity study available

[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.

[b] LOAEL-based HQ range was calculated based on two effects concentrations. The high end of the HQ range was based on a reduced fecundity endpoint; the low end of the HQ range was based on reduced fecundity, egg/larval mortality, and lack of spawning in the first reproductive season.

[c] Results from the studies reporting the lowest LOAELs were uncertain. The study reporting the lowest effects concentrations was uncertain because of uncertain statistical significance of the fecundity endpoint for the low dose, a lack of dose-response in the fecundity endpoint, uncertain number of fish used in the experiment, and uncertainties associated with fish handling and maintenance protocols.

[d] Selected lowest LOAEL and respective NOAEL TRV were two orders of magnitude lower than the next lowest TRVs, and effects were partially attributed to reduced feeding.

[e] A large number of studies that presented a range of effects thresholds were available; sufficient data were available to show that the lowest LOAEL reported for growth of channel catfish (8 mg/kg dw) (Murai et al. 1981) was inconsistent with NOAELs for channel catfish growth reported in other studies (Gatlin and Wilson 1986; Erickson et al. 2003), and thus that LOAEL was not selected.

COPC – chemical of potential concern

HQ – hazard quotient

LOAEL – lowest-observed-adverse-effect level

NOAEL – no-observed-adverse-effect level

TRV – toxicity reference value

Bold identifies NOAEL-based HQs greater than 1.0 and LOAEL-based HQs greater than or equal to 1.0.

Both the NOAEL- and LOAEL-based HQs were greater than 1.0 for cadmium (6.1 and 1.2, respectively) based on growth of juvenile fish. There is uncertainty in the cadmium risk because the toxicity data are highly variable, and the effects associated with the lowest LOAEL were partially attributed to reduced feeding. Use of the next higher LOAEL and NOAEL TRVs would result in HQs less than 1.0.

Both the NOAEL- and LOAEL-based HQs for vanadium were greater than 1.0 (5.9 and 1.2, respectively). The vanadium LOAEL-based HQ is 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 (Section A.4.2.2.4). Because of the very limited effects data, risk from vanadium is uncertain and could be higher or lower than HQs indicate.

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 Hugla and Thome (1999), 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 Hugla and Thome (1999) study (see Section 6.2.2.2). 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). PCB risks were also calculated using a PCB TEQ approach and were found to be low, although dioxins and furans were not included in the TEQ assessment. Therefore, the risk associated with exposure to all dioxin-like chemicals was underestimated by an unknown amount.

Of the chemicals evaluated solely in the uncertainty analysis, only the endrin risk analyses resulted in a LOAEL-based HQ greater than 1.0 (LOAEL-based HQ of 1.2). The selected endrin TRVs may underestimate risk because they are based on a survival endpoint. Although higher growth or reproduction LOAELs and NOAELs are reported for other fish species, it is not known if other endpoints would have resulted in lower TRVs for largemouth bass. However, because endrin exposures were likely overestimated because of interference from PCBs in the analysis of organochlorine pesticides in tissue samples (Section A.2.4.2), risks are likely to be low.

Results from several studies suggest that some reproductive functions in English sole collected in the LDW may be impaired relative to fish from reference sites (Johnson et al. 1988; 1993; 1997a; Casillas et al. 1991a) (Section A.6.2.2.3). Chemicals implicated as potential causal factors include PAHs and PCBs; however, linking the results of field studies to risks from specific chemicals is difficult considering, among other factors, the complex mixtures of chemicals in the field and the uncertainties in English sole home range. In addition, interpreting cause and effect of the adverse effects reported in field studies is complicated because of genetic variation, health, or seasonal variation in the spawning cycle. As summarized above for juvenile chinook salmon, PAH risks appear to be low based on dietary and water exposure pathways. However, no effects data were available to evaluate PAH effects on reproduction, and the selected dietary TRVs were based on growth endpoints. Effects data evaluated for PCBs included reproduction endpoints; however, because of high TRV uncertainty, risk estimates for PCBs are uncertain.

A.6.2.3 Pacific staghorn sculpin

This section presents risk estimates, uncertainties, and risk conclusions for Pacific staghorn sculpin.

A.6.2.3.1 Risk estimates

This section presents the HQ calculations for Pacific staghorn sculpin. COPCs were evaluated for two foraging range scenarios because the foraging range of Pacific staghorn sculpin 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 the modeling areas (M1, M2, M3, and M4) shown in Map A.2-2. COPCs evaluated for Pacific staghorn sculpin included arsenic, cadmium, copper, and vanadium, which were evaluated using a dietary approach, and total PCBs and TBT, which were evaluated using a critical tissue-residue approach (Section A.4.0). Dietary exposure concentrations of arsenic were less than both the NOAEL and LOAEL TRVs (maximum HQs of 0.80 and 0.53, respectively) (Table A.6-37).

Table A.6-37. HQ calculations for Pacific staghorn sculpin

Approach	COPC	Foraging Assumption	Exposure Concentration	Unit	NOAEL TRV	LOAEL TRV	NOAEL-Based HQ	LOAEL-Based HQ
Tissue residue	total PCBs	LDW-wide	1,100	µg/kg ww	104 – 528	520 – 2,640	**2** – **11**	0.42 – **2.1**
		M1	800	µg/kg ww	104 – 528	520 – 2,640	**1.5 – 7.7**	0.3 – **1.5**
		M2	920	µg/kg ww	104 – 528	520 – 2,640	**1.7 – 8.8**	0.35 – **1.8**
		M3	2,000	µg/kg ww	104 – 528	520 – 2,640	**3.8** – **19**	0.76 – **3.8**
		M4	940	µg/kg ww	104 – 528	520 – 2,640	**1.8 – 9.0**	0.36 – **1.8**
	TBT	LDW-wide	36	µg/kg ww	18	159	**2.0**	0.23
		M1	37	µg/kg ww	18	159	**2.1**	0.23
		M2	36	µg/kg ww	18	159	**2.0**	0.23
		M3	28	µg/kg ww	18	159	**1.6**	0.18
		M4	53	µg/kg ww	18	159	**2.9**	0.33
Dietary	arsenic	LDW-wide	15	mg/kg dw	20	30	0.75	0.50
		M1	15	mg/kg dw	20	30	0.75	0.50
		M2	16	mg/kg dw	20	30	0.80	0.53
		M3	15	mg/kg dw	20	30	0.75	0.50
		M4	12	mg/kg dw	20	30	0.60	0.40
	cadmium	LDW-wide	0.38	mg/kg dw	0.1	0.5	**3.8**	0.76
		M1	0.49	mg/kg dw	0.1	0.5	**4.9**	0.98
		M2	0.30	mg/kg dw	0.1	0.5	**3.0**	0.60
		M3	0.52	mg/kg dw	0.1	0.5	**5.2**	**1.0**
		M4	0.32	mg/kg dw	0.1	0.5	**3.2**	0.64
	copper	LDW-wide	56	mg/kg dw	50	100	**1.1**	0.56
		M1	65	mg/kg dw	50	100	**1.3**	0.65
		M2	77	mg/kg dw	50	100	**1.5**	0.77
		M3	45	mg/kg dw	50	100	0.90	0.45
		M4	48	mg/kg dw	50	100	**1.0**	0.48

Approach	COPC	Foraging Assumption	Exposure Concen-tration	Unit	NOAEL TRV	LOAEL TRV	NOAEL-Based HQ	LOAEL-Based HQ
Dietary, cont.	vanadium	LDW-wide	8.1	mg/kg dw	2.04	10.2	**4.0**	0.79
		M1	8.8	mg/kg dw	2.04	10.2	**4.3**	0.86
		M2	12	mg/kg dw	2.04	10.2	**5.9**	**1.2**
		M3	12	mg/kg dw	2.04	10.2	**5.9**	**1.2**
		M4	6.6	mg/kg dw	2.04	10.2	**3.2**	0.65

COPC – chemical of potential concern
dw – dry weight
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
NOAEL – no-observed-adverse-effect level
PCB – polychlorinated biphenyl
TBT – tributyltin
TRV – toxicity reference value
ww – wet weight

Bold identifies NOAEL-based HQs greater than 1.0 and LOAEL-based HQs greater than or equal to 1.0.

Exposure concentrations of total PCBs resulted in LOAEL-based HQs ranging from 0.30 to 3.8 and NOAEL-based HQs ranging from 1.5 to 19. Dietary exposure concentrations of cadmium and vanadium equaled or exceeded both the LOAEL and NOAEL TRVs in at least one modeling area (maximum LOAEL-based HQ of 1.0 and 1.2 for cadmium and vanadium, respectively). Exposure concentrations of TBT and copper exceeded their NOAEL TRVs (maximum NOAEL-based HQs of 2.9 and 1.5, respectively), but none exceeded their LOAEL TRVs.

With the exception of PCBs, cadmium, and vanadium, HQs were similar for all COPCs in all modeling areas and at the LDW-wide scale. The cadmium LOAEL-based HQ was 1.0 in modeling area M3 but less than 1.0 in all other modeling areas and LDW-wide. The vanadium LOAEL-based HQ was greater than 1.0 in modeling areas M2 and M3 but not in M1, M4, or LDW-wide. The PCB NOAEL- and LOAEL-based HQs for modeling area M3 were approximately two times higher than those for the other modeling areas.

A.6.2.3.2 Uncertainty analysis

This section presents a discussion of the uncertainty associated with the problem formulation, the exposure and effects assessments, and the risk characterization for Pacific staghorn sculpin.

Problem Formulation

Primary uncertainties in the problem formulation for Pacific staghorn sculpin include ROC selection, assessment endpoints, and the COPC screen. Uncertainties associated with assessment endpoints are the same as those discussed in Section A.6.2.1.2 for juvenile chinook salmon. Uncertainties associated with the COPC screen are the same as for English sole, as discussed in Section A.6.2.2.2, except that risks for chemicals with no detected concentrations in tissue are discussed below. Uncertainties associated with ROC selection are also discussed below.

COPC screen

Risks to Pacific staghorn sculpin from exposures to BEHP, dimethyl phthalate, and di-n-butyl phthalate in the LDW are uncertain because these chemicals were not detected in Pacific staghorn sculpin tissue samples. RLs for these chemicals were greater than the selected NOAEL TRVs (Table A.2-21). As discussed for English sole, re-analysis of tissue samples with high BEHP RLs, resulting from sample dilutions, showed that BEHP was not detected at much lower RLs. HQs calculated using minimum and maximum RLs relative to the NOAEL TRVs for each chemical are presented in Table A.6-38. LOAEL TRVs were not available for any of these chemicals.

Table A.6-38. Pacific staghorn sculpin HQs for BEHP, dimethyl phthalate, and di-n-butyl phthalate using reporting limits as exposure concentrations

Chemical	Reporting Limit (µg/kg ww)		TRV (µg/kg ww)		NOAEL-Based HQ			
	Min	Max	NOAEL	LOAEL	Min RL	Max RL		
BEHP	490	5,000	390	na	**1.3**	**13**		
Dimethyl phthalate	40	400	498	na	< 0.1	0.80		
Di-n-butyl phthalate	200	1,300[a]	1,170	na	0.17	**1.1**		

[a] Maximum detected concentration is presented.

BEHP – bis(2-ethylhexyl) phthalate

HQ – hazard quotient

LOAEL – lowest-observed-adverse-effect level

na – no TRV available

nc – not calculated because no LOAEL TRV was available

NOAEL – no-observed-adverse-effect level

ROC – receptor of concern

ww – wet weight

Bold identifies NOAEL-based HQs greater than 1.0.

NOAEL-based HQs were greater than 1.0 for BEHP and di-n-butylphthalate using maximum RLs and were either just above 1.0 or much lower than 1.0 using minimum RLs. NOAEL-based HQs were less than 1.0 for all dimethylphthalate RLs. Phthalate risks to Pacific staghorn sculpin were low with some uncertainty.

ROC Selection

In Section A.6.2.2.2, it was found that risk estimates for English sole as an ROC were health protective of risks to shiner surfperch for all critical tissue-residue COPCs, except TBT. To evaluate whether risk estimates for Pacific staghorn sculpin as an ROC are protective of risks from TBT to shiner surfperch, NOAEL- and LOAEL-based HQs calculated using both Pacific staghorn sculpin and shiner surfperch tissue data were compared in Table A.6-39. Pacific staghorn sculpin and shiner surfperch HQs for TBT were similar; therefore, shiner surfperch tissue data do not change risk estimates for fish represented by Pacific staghorn sculpin as an ROC.

Table A.6-39. Shiner surfperch and Pacific staghorn sculpin UCLs and HQs for TBT

Species	COPC	Exposure Concentration (µg/kg ww)	NOAEL TRV	LOAEL TRV	NOAEL-Based HQ	LOAEL-Based HQ
Pacific staghorn sculpin	TBT	36	18	159	**2.0**	0.23
Shiner surfperch	TBT	69	18	159	**3.8**	0.43

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.
TBT – tributyltin
TRV – toxicity reference value
UCL – upper confidence limit on the mean
ww – wet weight

Exposure Assessment

Uncertainties in the exposure assessment for Pacific staghorn sculpin were associated with the following factors:

- Water pathway for metals and PAHs
- Dietary composition
- Future habitat changes resulting from restoration
- Benthic invertebrate tissue data and linear regressions

The uncertainties associated with exposure via the water pathway and future habitat changes are the same as those discussed in Section A.6.2.1.2 for juvenile chinook salmon.

Dietary Composition

Concentrations of COPCs in the diet of Pacific staghorn sculpin are a function of the types of prey consumed and their COPC concentrations. Pacific staghorn sculpin were assumed to ingest fish and benthic invertebrates. According to regional studies, crabs and shrimp constitute 25 to 32% of Pacific staghorn sculpin diets in Puget Sound (Fresh et al. 1979; Miller et al. 1977c; Wingert et al. 1979). For the exposure calculations in Section A.4.1.2, it was assumed that dietary exposure from crabs and shrimp in the LDW was represented by benthic invertebrate tissue samples containing small (≤ 20 mm) crabs and shrimp in addition to other invertebrate species (Windward 2005b, f).[86] Pacific staghorn sculpin in the LDW are likely to prey on some crabs and shrimp > 20 mm. Tissue data for larger slender and Dungeness crabs (≥ 90 mm) collected from the LDW are also available.[87] Although these crabs are larger than prey

[86] Crabs or shrimp were present at 15 of 26 locations where taxonomic samples were collected and constituted from 0.04 to 2.5% of organisms collected at each location.
[87] Data are described in Section A.2.4.1.2.

that Pacific staghorn sculpin could possibly consume, these data provide another means to represent crabs and shrimp consumed by Pacific staghorn sculpin. For dietary COPCs, the UCL concentrations of the crab and benthic invertebrate tissues were generally within a factor of 2, except for vanadium (Table A.6-40). Vanadium concentrations in crab tissue were lower than those in benthic invertebrate tissue for all areas.

Table A.6-40. Comparison of crab and benthic invertebrate tissue concentrations for Pacific staghorn sculpin dietary COPCs

COPC	Foraging Assumption	UCL Concentration in Crab Tissue (mg/kg dw)	UCL Concentration in Benthic Invertebrate Tissue (mg/kg dw)
Arsenic	LDW-wide	25	24
	M1	36	23
	M2	17	25
	M3	17	23
	M4	27	18
Cadmium	LDW-wide	1.1	0.60
	M1	2.2	0.81
	M2	0.95	0.48
	M3	0.71	0.77
	M4	1.2	0.54
Copper	LDW-wide	90	92
	T1	110	110
	T2	75	130
	T3	63	65
	T4	52	81
Vanadium	LDW-wide	1.0	12
	M1	1.1	12
	M2	1.3	18
	M3	1.0	20
	M4	0.5[a]	10

[a] Maximum RL for single non-detected concentration.

COPC – chemical of potential concern

dw – dry weight

UCL – upper confidence limit on the mean

Risks for Pacific staghorn sculpin were evaluated using both crab and benthic invertebrate tissue data for arsenic, cadmium, copper, and vanadium for locations where exposure concentrations of these tissue types differed by >10% (Table A.6-41). When crab tissue data were included in arsenic risk calculations, risk predictions were similar to those predicted without crab data. For cadmium, the LOAEL-based HQ changed from < 1.0 to > 1.0 for LDW-wide and modeling areas M1 and M4 when crab data were included. The HQs for vanadium changed from > 1.0 to < 1.0 for M2 and

M3 when crab tissue data were included in exposure calculations. The HQs for copper changed from 1.0 to < 1.0 when crab data were included in exposure calculations in M4, but no change occurred in M2. Therefore, inclusion of crab tissue data would increase the Pacific staghorn sculpin risk estimates for cadmium for the LDW as a whole and for modeling areas M1 and M4, and would decrease the risk estimates for vanadium for modeling areas M2 and M3 and for copper for modeling area M4. The inclusion of crab tissue data would not affect risk conclusions for arsenic.

Table A.6-41. HQs for Pacific staghorn sculpin for arsenic, cadmium, copper, and vanadium with and without the use of larger crab tissue data

COPC	Foraging Assumption	Alternative Diet[a]		Diet Assumed in Exposure Assessment[b]	
		NOAEL-Based HQ	LOAEL-Based HQ	NOAEL-Based HQ	LOAEL-Based HQ
Arsenic	M1	0.75	0.50	0.75	0.50
	M2	0.82	0.55	0.80	0.53
	M3	0.73	0.49	0.75	0.50
	M4	0.59	0.39	0.60	0.40
Cadmium	LDW	**5.4**	**1.1**	**3.8**	0.76
	M1	**9.3**	**1.9**	**4.9**	0.98
	M2	**4.5**	0.90	**3.0**	0.60
	M4	**5.3**	**1.1**	**3.2**	0.64
Copper	M2	**1.2**	0.58	**1.5**	0.77
	M4	0.76	0.38	**1.0**	0.48
Vanadium	LDW	**2.3**	0.45	**4.0**	0.79
	M1	**2.6**	0.52	**4.3**	0.86
	M2	**3.3**	0.67	**5.9**	**1.2**
	M3	**3.1**	0.63	**5.9**	**1.2**
	M4	**1.8**	0.35	**3.2**	0.65

[a] Assuming a diet of 32% crabs, 23% benthic invertebrates, 44% fish, and 1% sediment.

[b] Assuming a diet of 55% benthic invertebrates, 44% fish, and 1% sediment.

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.

Benthic Invertebrate Tissue Data and Linear Regressions

The uncertainties associated with the representativeness of benthic invertebrate tissue data are the same as those discussed in Section A.6.2.1.2 for juvenile chinook salmon, with one difference. For Pacific staghorn sculpin, fewer benthic invertebrate tissue samples were available to calculate UCL concentrations for each modeling area (n = 10, 6, 4, and 4 for modeling areas M1, M2, M3, and M4, respectively) because benthic invertebrate sampling locations were selected to be representative of arsenic, lead, and PCB concentrations in sediment rather than an even distribution of sampling locations throughout the LDW. The relatively low number of benthic invertebrate tissue samples results in higher uncertainty in the UCL in exposure areas where there is high variability in the tissue concentrations. The ProUCL guidelines recommend using the UCL even if it is greater than the maximum concentration (EPA 2004a). In one instance, ProUCL calculated a UCL that was greater than the maximum concentration (for vanadium in modeling area M4); however, this difference was small, and risk conclusions would not have changed if maximum values had been used instead of the UCL.

If the exposure estimate had been calculated using all empirical benthic invertebrate tissue data rather than the regression, the benthic invertebrate UCL would have been 37 mg/kg dw, in comparison to the UCL of 24 mg/kg dw estimated using the regression. Assuming this value in the exposure calculation for benthic invertebrates and 4.2 mg/kg dw for shiner surfperch results in an exposure concentration of 22 mg/kg dw in comparison to 15 mg/kg dw calculated using the regression-estimated benthic invertebrate tissue concentration. Based on this exposure concentration, the LDW-wide NOAEL-based HQ would have increased from 0.75 to 1.1, and the LDW-wide LOAEL-based HQ would have increased from 0.50 to 0.73. Therefore, this uncertainty would not have substantially affected the risk conclusions for arsenic.

Effects Assessment

Uncertainties in the effects assessment for Pacific staghorn sculpin were associated with the following factors:

- Effects from chemical mixtures and estimation of NOAELs from LOAELs
- Exclusion of field studies from TRV selection
- Estimation of NOAELs from LOAELs
- PCB TRV
- Cadmium TRV
- COPCs without LOAEL TRVs
- TEQ approach for PCBs
- Critical tissue residue approach

All of the above uncertainties associated with the effects assessment for Pacific staghorn sculpin were assessed in Section A.6.2.2.2 for English sole, except that COPCs without LOAEL TRVs are assessed below.

In addition, some of the selected TRVs are considered less certain than others if there were a small number of studies, if TRVs were based on less-protective measures such as LC50s, or if data quality was questionable. The relative uncertainties in the selected TRVs for each COPC are the same as those discussed in Section A.6.2.2.2 for English sole, with the exception of TRVs for TBT. For TBT, only three studies were available. In the study reporting the lowest LOAEL, which was selected as the TRV, treatments were not replicated, and there was high mortality in all treatments and in the controls. The effect of these uncertainties on risk conclusions is unknown.

PCB TRV

Uncertainties associated with the PCB TRV for Pacific staghorn sculpin are the same as those discussed in Section A.6.2.2 for English sole. If the study with the next lowest LOAEL (9,300 µg/kg ww reported in Hansen et al. (1974a)) had been selected as the LOAEL TRV, the LDW-wide LOAEL-based HQ for Pacific staghorn sculpin would have changed from a range of 0.42 to 2.1 based on the selected LOAEL range to a LOAEL-based HQ of 0.12.

COPCs without LOAEL TRVs

As discussed in Section A.2.5.2, four chemicals had maximum exposure concentrations exceeding NOAEL TRVs but LOAEL TRVs were not available. One chemical (chromium) was evaluated using a dietary approach, and three chemicals (benzoic acid, dimethyl phthalate, and di-n-butyl phthalate) were evaluated using a critical tissue-residue approach. Risks to Pacific staghorn sculpin from dimethyl phthalate and di-n-butyl phthalate were evaluated in the exposure assessment uncertainty section above because concentrations of these chemicals were generally below detection limits in fish tissue samples.

Table A.6-42 presents chromium dietary concentrations for Pacific staghorn sculpin, which were calculated using Equation 4-1, as described in Section A.4.1.1.2. Benzoic acid critical tissue-residue concentrations were calculated as the UCL of the mean of all Pacific staghorn sculpin tissue composite samples from throughout the LDW. NOAEL-based HQs ranged from 0.18 to 2.1 for chromium and from 1.5 to 2.1 for benzoic acid (Table A.6-42). Because no LOAEL toxicity data were available for these chemicals, the low NOAEL-based HQs calculated using unbounded NOAEL TRVs are assumed to indicate low risks.

Table A.6-42. Pacific staghorn sculpin NOAEL-based HQs for chemicals for which no LOAEL TRVs were identified

COPC	Units	Dietary Exposure Concentration	NOAEL TRV	NOAEL-Based HQ
Chromium	mg/kg dw	8.0	9.4[a]	0.85
Benzoic acid	µg/kg ww	7,000	3,380[b]	**2.1**

[a] No effects on growth of 2-year-old gray mullet exposed to chromium III through diet and sediment for 8 weeks (Walsh et al. 1994).

[b] No effects on survival of mosquitofish exposed in a mesocosm for 24 hours (Lu and Metcalf 1975).

dw – dry weight
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
NOAEL – no-observed-adverse-effect level
TRV – toxicity reference value
ww – wet weight
Bold identifies NOAEL-based HQs greater than 1.0.

Risk Characterization

Risks to Pacific staghorn sculpin from organochlorine pesticides were not included in the risk estimates because of high uncertainty in the JN-qualified tissue pesticide data resulting in suspected false identifications of presence of some pesticides as well as overestimates in their concentrations, as discussed in Section A.2.4.2. Exposure estimates and risk estimates are discussed in this section for the three organochlorine pesticides (alpha-endosulfan, beta-endosulfan, and endrin) that were identified as COPCs in Section A.2.5.2. TRVs for these pesticides are the same as those discussed in Section A.6.2.2.2 for English sole.

HQs for endosulfan and endrin are presented in Table A.6-43. LOAEL-based HQs were greater than 1.0 for endrin and less than 1.0 for alpha-endosulfan and beta-endosulfan. NOAEL-based HQs were greater than 1.0 for all three of these COPCs. The selected endrin TRVs may underestimate risks because they are based on a survival endpoint. Although higher growth or reproduction LOAELs and NOAELs are reported for other fish species, it is not known if other endpoints would have resulted in lower TRVs for largemouth bass. Endosulfan effects data are uncertain because only three studies were available and all evaluated the survival endpoint; thus, risks from sublethal endpoints may be underestimated. However, because the exposure concentrations are likely overestimates of actual exposure (Section A.2.4.2), it is likely that risks associated with organochlorine pesticides are lower than these HQs suggest.

Table A.6-43. HQs for Pacific staghorn sculpin and endosulfan and endrin

COPC	UCL Concentration (µg/kg ww)	NOAEL TRV (µg/kg ww)	LOAEL TRV (µg/kg ww)	NOAEL-Based HQ	LOAEL-Based HQ
alpha-Endosulfan	1.4	0.62	31	**2.3**	<0.1
beta-Endosulfan	4.1	0.62	31	**6.6**	0.13
Endrin	36	1.2	11.5	**31**	**3.1**

COPC – chemical of potential concern
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
NOAEL – no-observed-adverse-effect level
TRV – toxicity reference value
UCL – upper confidence limit on the mean
ww – wet weight
Bold identifies NOAEL-based HQs greater than 1.0 and LOAEL-based HQs greater than or equal to 1.0.

Summary of Uncertainties

Uncertainties in the problem formulation, the effects and exposure assessments, and risk characterization for Pacific staghorn sculpin were evaluated, as summarized below:

- Uncertainties in the COPC screen, ROC selection, water exposure to PAHs and metals, PAH field studies, PCB analysis approach (total PCBs versus PCB TEQ), and future habitat changes resulting from restoration are not expected to have an effect on risk conclusions.

- Effects data for cadmium are highly variable with most studies showing lower sensitivity than the selected TRV. Because the relative sensitivities of Pacific staghorn sculpin, fish represented by Pacific staghorn sculpin, and fish used in the toxicity studies is unknown, it is not known if the conservative TRV selected overestimates risk.
- When the small crabs and shrimps in the invertebrate portion of Pacific staghorn sculpin diets were represented by Phase 2 crab tissue data rather than benthic invertebrate tissue data (which include crabs and shrimps), the LDW-wide, modeling area M1, and modeling area M4 LOAEL-based HQs increased from slightly less than 1.0 to slightly greater than 1.0 for cadmium; decreased from slightly greater than 1.0 to slightly less than 1.0 for modeling areas M2 and M3 for vanadium; and the NOAEL-based HQ decreased from 1.0 to slightly less than 1.0 for modeling area M4 for copper. For arsenic, no HQs changed from less than 1.0 to greater than 1.0 or vice versa.
- The use of safety factors of greater than 5 to estimate NOAELs from LOAELs may overestimate the difference between LOAELs and NOAELs.
- Because of numerous uncertainties in the study reporting the lowest PCB effects concentrations, the estimated risk to Pacific staghorn sculpin from PCBs is uncertain. LOAEL-based HQs based on toxicity data reported in Hugla and Thome (1999) and sculpin exposures in specific areas in the LDW ranged from 0.30 to 3.8.
- Risks from exposure to chromium and benzoic acid, for which no LOAELs were available, are likely to be very low for chromium (NOAEL-based HQ < 1.0) and low for benzoic acid.
- There is uncertainty in the critical tissue-residue approach because concentrations may not reflect the site of action, chemicals may be metabolized, and species- and chemical-specific factors may be important.
- An evaluation of risks from exposure to organochlorine pesticides indicated a low risk from endosulfan and the potential for adverse effects from endrin (LOAEL-based HQ of 3.1). The selected TRVs are based on a survival endpoint and may underestimate the potential for sublethal effects. However, because of analytical interference from PCBs, the risks are likely to be lower than the HQs suggest.

A.6.2.3.3 Risk conclusions

Pacific staghorn sculpin, a benthic omnivorous fish, was selected to represent upper-trophic-level fish in the LDW. This distinction was made because upper-trophic-level fish may have higher body burdens of biomagnifying chemicals than fish such as English sole that consume primarily small invertebrates at a lower trophic level. Two exposure scenarios were evaluated for Pacific staghorn sculpin because their foraging range is uncertain. The scenarios were: 1) foraging throughout the LDW, and 2) foraging in smaller segments of the LDW corresponding to the modeling areas (M1, M2, M3, and M4) shown in Map A.2-2. Results of the risk characterization for Pacific staghorn sculpin are summarized in Table A.6-44.

Table A.6-44. Summary of risk characterization for Pacific staghorn sculpin

Approach	COPC	Foraging Assumption	NOAEL-Based HQ	LOAEL-Based HQ	LOAEL Endpoint	Primary Uncertainty
Tissue residue	total PCBs	LDW-wide	**2 – 11**[a]	0.42 – **2.1**[a]	reduced fecundity; lack of spawning in first reproductive season; egg and larval mortality[b]	high TRV uncertainty[c]
		M1	**1.5 – 7.7**[a]	0.30 – **1.5**[a]		
		M2	**1.7- 8.8**[a]	0.35 – **1.8**[a]		
		M3	**3.8 – 19**[a]	0.76 – **3.8**[a]		
		M4	**1.8 – 9.0**[a]	0.36 – **1.8**[a]		
	TBT	LDW-wide	**2.0**	0.23	reduced juvenile body weight	few toxicity studies available
		M1	**2.1**	0.23		
		M2	**2.0**	0.23		
		M3	**1.6**	0.18		
		M4	**2.9**	0.33		
Dietary	arsenic	LDW-wide	0.75	0.50	reduced juvenile body weight	moderate number of toxicity studies
		M1	0.75	0.50		
		M2	0.80	0.53		
		M3	0.75	0.50		
		M4	0.60	0.40		
	cadmium	LDW-wide	**3.8**	0.76	reduced juvenile body weight and length growth rate; and condition factor	high TRV uncertainty,[d] inclusion of crab data in exposure calculations[e]
		M1	**4.9**	0.98		
		M2	**3.0**	0.60		
		M3	**5.2**	**1.0**		
		M4	**3.2**	0.64		
	copper	LDW-wide	**1.1**	0.56	reduced juvenile body weight growth rate	medium TRV uncertainty[f]
		M1	**1.3**	0.65		
		M2	**1.5**	0.77		
		M3	0.90	0.45		
		M4	**1.0**	0.48		

Approach	COPC	Foraging Assumption	NOAEL-Based HQ	LOAEL-Based HQ	LOAEL Endpoint	Primary Uncertainty
Dietary, cont.	vanadium	LDW-wide	**4.0**	0.79	reduced juvenile body weight	only one toxicity study available
		M1	**4.3**	0.86		
		M2	**5.9**	**1.2**		
		M3	**5.9**	**1.2**		
		M4	**3.2**	0.65		

[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.

[b] LOAEL-based HQ range was calculated based on two effects concentrations. The high end of the HQ range was based on a reduced fecundity endpoint; the low end of the HQ range was based on reduced fecundity, egg/larval mortality, and lack of spawning in the first reproductive season.

[c] Results from the studies reporting the lowest LOAELs were uncertain. The study reporting the lowest effects concentrations was uncertain because of uncertain statistical significance of the fecundity endpoint for the low dose, a lack of dose-response in the fecundity endpoint, uncertain number of fish used in the experiment, and uncertainties associated with fish handling and maintenance protocols.

[d] Selected LOAEL and respective NOAEL TRV were two orders of magnitude lower than the next lowest TRVs, and effects were partially attributed to reduced feeding.

[e] When the small crabs and shrimps in the invertebrate portion of Pacific staghorn sculpin diets were represented by Phase 2 crab tissue data rather than benthic invertebrate tissue data (which include crabs and shrimps), the cadmium LOAEL-based HQ increased from slightly less than 1.0 to slightly greater than 1.0 for the LDW-wide exposure area and modeling area M1. Vanadium LOAEL-based HQs decreased from slightly greater than 1.0 to slightly less than 1.0 for modeling areas M2 and M3 when crab data were included.

[f] A large number of studies that presented a range of effects thresholds were available, and sufficient data were available to suggest that the lowest LOAEL was not supported by data from other studies.

COPC – chemical of potential concern

HQ – hazard quotient

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 estimate because the study with the lowest effects concentrations was highly uncertain (see Section A.6.2.2.2). If the study reporting the next higher LOAEL TRV had been selected, exposures would have been below the effects threshold (LOAEL-based HQ of 0.28).

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). 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 very limited effects data, risk from vanadium is uncertain and could be higher or lower than HQs indicate.

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.

NOAEL-based HQs were greater than 1.0 but LOAEL-based HQs were less than or equal to 1.0 for TBT and copper in at least one modeling area. Uncertainty in data used to represent crabs in sculpin dietary exposure calculations could result in a cadmium LOAEL-based HQ of up to 1.9. Of the chemicals evaluated in the uncertainty analysis, only the endrin risk analysis resulted in a LOAEL-based HQ greater than 1.0 (LOAEL-based HQ of 3.1). The selected endrin TRVs may underestimate risk because they are based on a survival endpoint. Higher growth or reproduction LOAELs and NOAELs are reported for other fish species; it is not known if other endpoints would have resulted in lower TRVs for largemouth bass. However, endrin exposures are likely overestimated because of interference from PCBs in the analysis of organochlorine pesticides in tissue samples (Section A.2.4.2), so risks are likely to be low.

A.6.2.4 Summary of risk conclusions for fish

In summary, results of the risk estimates and evaluation of associated uncertainties for fish are as follows:

- Exposure concentrations of cadmium in the English sole and Pacific staghorn sculpin diets in modeling area M3 were equal to or greater than those associated with adverse effects, although there is uncertainty in selected TRVs. Low risks were estimated for cadmium and Pacific staghorn sculpin and juvenile chinook salmon LDW-wide and in all other modeling areas.
- Exposure concentrations of vanadium in the English sole and Pacific staghorn sculpin diets in modeling areas M2 and M3 were greater than those associated with adverse effects. Risks were very low (NOAEL-based HQs < 1.0) for juvenile chinook salmon and for Pacific staghorn sculpin in modeling areas M1, M4, and LDW-wide. A paucity of effects studies make these risks somewhat uncertain.
- There is high uncertainty in the PCB TRVs used to evaluate risks to English sole and Pacific staghorn sculpin, and thus a range of LOAEL TRVs was selected from the study reporting the lowest effects concentrations. Exposure concentrations of PCBs in English sole and Pacific staghorn sculpin tissue were in between the concentrations selected as LOAELs from this study. Therefore, there is a potential for adverse effects from PCBs, but risk estimates are uncertain. Risks to juvenile chinook salmon from PCBs were estimated to be low. Risks to all three fish ROCs from dioxin-like PCB congeners using the TEQ approach were also low.
- No quantitative ecological risk estimates were calculated for dioxins and furans and thus the level of ecological risk from dioxins and furans is unknown.

- Exposure concentrations of TBT were less than those associated with adverse effects for Pacific staghorn sculpin. TBT was not identified as a COPC for English sole and juvenile chinook salmon in the problem formulation.
- Exposure concentrations of arsenic and copper were less than those associated with adverse effects for all fish ROCs.
- Risks from chromium, organochlorine pesticides, benzoic acid, PAHs, and some phthalates were evaluated in the uncertainty analysis because of uncertainties in exposure or effects data. Considering the uncertainty, risk estimates for these COPCs appear to be low to very low.
- Various exposure assumptions were evaluated in the uncertainty analysis. None of the alternative assumptions had a large effect on risk conclusions.
- Site-specific studies suggest the potential for adverse effects on English sole reproduction; however, these effects have not been conclusively associated with specific chemicals or chemical mixtures.
- Site-specific studies investigating growth or immunocompetence of juvenile chinook salmon are uncertain and also have not been definitively associated with specific chemicals or chemical mixtures.

A.6.3 RISK CHARACTERIZATION FOR WILDLIFE

This section presents a risk characterization and uncertainty analysis for each of the five wildlife ROCs. The assessment for each ROC estimates risk by calculating HQs using estimated ingested doses of COPCs, as described in Section A.5.1, and TRVs, as presented in Section A.5.2. Uncertainties in the exposure and effects data that may result in overestimates or underestimates of risk for each of the COPCs are discussed. Risk conclusions are presented for each ROC that integrate risk estimates with associated uncertainties.

A.6.3.1 Spotted sandpiper

This section presents risk estimates, uncertainties, and risk conclusions for spotted sandpiper.

A.6.3.1.1 Risk estimates

This section presents the HQ calculations for spotted sandpiper. Ingested doses, NOAEL and LOAEL TRVs, and HQs are presented in Table A.6-45.

Table A.6-45. HQ calculations for spotted sandpiper in the six exposure scenarios evaluated

COPC	Exposure Scenario[a]	Ingested Dose (mg/kg bw/day)	Contribution to Ingested Dose (%)[b, c]		TRVs (mg/kg bw/day)		NOAEL-Based HQ	LOAEL-Based HQ
			Prey	Sediment	NOAEL	LOAEL		
Arsenic	Area 1/high	4.6	78	22	10	40	0.46	0.12
	Area 1/high and poor	3.8	87	13	10	40	0.38	0.10
	Area 2/high	5.3	74	26	10	40	0.53	0.13
	Area 2/high and poor	3.8	87	13	10	40	0.38	0.10
	Area 3/high	3.4	84	16	10	40	0.34	<0.1
	Area 3/high and poor	5.8	71	29	10	40	0.58	0.15
Cadmium	Area 1/high	0.10	87	13	1.5	4.0	< 0.1	< 0.1
	Area 1/high and poor	0.10	83	17	1.5	4.0	< 0.1	< 0.1
	Area 2/high	0.090	69	31	1.5	4.0	< 0.1	< 0.1
	Area 2/high and poor	0.10	75	25	1.5	4.0	< 0.1	< 0.1
	Area 3/high	0.12	85	15	1.5	4.0	< 0.1	< 0.1
	Area 3/high and poor	0.32	31	69	1.5	4.0	0.21	< 0.1
Chromium	Area 1/high	1.4	34	66	1.0	5.0	**1.4**	0.28
	Area 1/high and poor	1.3	36	64	1.0	5.0	**1.3**	0.26
	Area 2/high	4.0	70	30	1.0	5.0	**4.0**	0.80
	Area 2/high and poor	8.8	90	10	1.0	5.0	**8.8**	**1.8**
	Area 3/high	1.6	47	53	1.0	5.0	**1.6**	0.32
	Area 3/high and poor	4.1	19	81	1.0	5.0	**4.1**	0.82
Cobalt	Area 1/high	0.53	53	47	2.3	23.1	0.23	<0.1
	Area 1/high and poor	0.50	56	44	2.3	23.1	0.22	< 0.1
	Area 2/high	0.58	54	46	2.3	23.1	0.25	< 0.1
	Area 2/high and poor	0.57	61	39	2.3	23.1	0.25	< 0.1
	Area 3/high	0.57	61	39	2.3	23.1	0.25	< 0.1
	Area 3/high and poor	0.62	55	45	2.3	23.1	0.27	< 0.1

COPC	Exposure Scenario[a]	Ingested Dose (mg/kg bw/day)	Contribution to Ingested Dose (%)[b, c]		TRVs (mg/kg bw/day)		NOAEL-Based HQ	LOAEL-Based HQ
			Prey	Sediment	NOAEL	LOAEL		
Copper	Area 1/high	21	88	12	21	29	1.0	0.72
	Area 1/high and poor	21	89	11	21	29	1.0	0.72
	Area 2/high	25	87	13	21	29	**1.2**	0.86
	Area 2/high and poor	24	91	9	21	29	**1.1**	0.83
	Area 3/high	13	86	14	21	29	0.62	0.45
	Area 3/high and poor	31	35	65	21	29	**1.5**	**1.1**
Lead	Area 1/high	7.4	66	34	5.82	20	**1.3**	0.37
	Area 1/high and poor	7.4	66	34	5.82	20	**1.3**	0.37
	Area 2/high	6.8	35	65	5.82	20	**1.2**	0.34
	Area 2/high and poor	110	98	2	5.82	20	**19**	**5.5**
	Area 3/high	3.4	26	74	5.82	20	0.58	0.17
	Area 3/high and poor	29	3	97	5.82	20	**5.0**	**1.5**
Mercury	Area 1/high	0.020	77	23	0.018	0.091	**1.1**	0.22
	Area 1/high and poor	0.020	78	22	0.018	0.091	**1.1**	0.22
	Area 2/high	0.022	47	53	0.018	0.091	**1.2**	0.24
	Area 2/high and poor	0.019	58	42	0.018	0.091	**1.1**	0.21
	Area 3/high	0.095	83	17	0.018	0.091	**5.3**	**1.0**
	Area 3/high and poor	0.090	88	12	0.018	0.091	**5.0**	0.99
Nickel	Area 1/high	1.1	48	52	17	33	< 0.1	< 0.1
	Area 1/high and poor	1.4	39	61	17	33	< 0.1	< 0.1
	Area 2/high	1.5	60	40	17	33	< 0.1	< 0.1
	Area 2/high and poor	1.4	60	40	17	33	< 0.1	< 0.1
	Area 3/high	1.8	65	35	17	33	0.11	< 0.1
	Area 3/high and poor	3.7	31	69	17	33	0.22	0.11

COPC	Exposure Scenario[a]	Ingested Dose (mg/kg bw/day)	Contribution to Ingested Dose (%)[b, c]		TRVs (mg/kg bw/day)		NOAEL-Based HQ	LOAEL-Based HQ
			Prey	Sediment	NOAEL	LOAEL		
Selenium	Area 1/high	0.32	95	5	0.50	0.82	0.64	0.39
	Area 1/high and poor	0.31	95	5	0.50	0.82	0.62	0.38
	Area 2/high	0.45	56	44	0.50	0.82	0.90	0.55
	Area 2/high and poor	0.37	77	23	0.50	0.82	0.74	0.45
	Area 3/high	0.41	53	47	0.50	0.82	0.82	0.50
	Area 3/high and poor	0.40	55	45	0.50	0.82	0.80	0.49
Vanadium	Area 1/high	2.5	40	60	1.2	2.3	**2.1**	**1.1**
	Area 1/high and poor	2.4	41	59	1.2	2.3	**2.0**	**1.0**
	Area 2/high	3.2	49	51	1.2	2.3	**2.7**	**1.4**
	Area 2/high and poor	3.1	50	50	1.2	2.3	**2.6**	**1.3**
	Area 3/high	3.1	48	52	1.2	2.3	**2.6**	**1.3**
	Area 3/high and poor	3.1	48	52	1.2	2.3	**2.6**	**1.3**
Zinc	Area 1/high	35	85	15	82	124	0.43	0.28
	Area 1/high and poor	34	86	14	82	124	0.41	0.27
	Area 2/high	53	87	13	82	124	0.65	0.43
	Area 2/high and poor	64	92	8	82	124	0.78	0.52
	Area 3/high	46	91	9	82	124	0.56	0.37
	Area 3/high and poor	62	68	32	82	124	0.76	0.50
Total PCBs	Area 1/high	0.25	96	4	0.49	1.4	0.51	0.18
	Area 1/high and poor	0.25	96	4	0.49	1.4	0.51	0.18
	Area 2/high	1.0	93	7	0.49	1.4	**2.0**	0.71
	Area 2/high and poor	0.64	94	6	0.49	1.4	**1.3**	0.46
	Area 3/high	0.45	96	4	0.49	1.4	0.92	0.32
	Area 3/high and poor	0.57	95	5	0.49	1.4	**1.2**	0.41

COPC	Exposure Scenario[a]	Ingested Dose (mg/kg bw/day)	Contribution to Ingested Dose (%)[b, c]		TRVs (mg/kg bw/day)		NOAEL-Based HQ	LOAEL-Based HQ
			Prey	Sediment	NOAEL	LOAEL		
PCB TEQs	Area 1/high	2.6×10^{-5}	96	4	1.4×10^{-5}	1.43×10^{-4}	**1.9**	0.18
	Area 1/high and poor	2.6×10^{-5}	96	4	1.4×10^{-5}	1.43×10^{-4}	**1.9**	0.18
	Area 2/high	2.1×10^{-4}	43	57	1.4×10^{-5}	1.43×10^{-4}	**15**	**1.5**
	Area 2/high and poor	1.4×10^{-4}	64	36	1.4×10^{-5}	1.43×10^{-4}	**10**	0.98
	Area 3/high	6.4×10^{-5}	96	4	1.4×10^{-5}	1.43×10^{-4}	**4.6**	0.45
	Area 3/high and poor	1.4×10^{-4}	43	57	1.4×10^{-5}	1.43×10^{-4}	**10**	0.98

[a] Six exposure scenarios were evaluated; in each of three exposure areas, foraging in high-quality habitat only and foraging in both high- and poor-quality habitat were evaluated. These exposure scenarios are described in detail in Section A.5.1.2.1.

[b] The percent contribution of ingested dose from water is very low.

[c] The percent contribution of ingested dose from prey tissue and sediment is based on both chemical concentration and intake rate.

HQ – hazard quotient

LOAEL – lowest-observed-adverse-effect level

NOAEL – no-observed-adverse-effect level

PCB – polychlorinated biphenyl

TEQ – toxic equivalent

TRV – toxicity reference value

Bold identifies NOAEL-based HQs greater than 1.0 and LOAEL-based HQs greater than or equal to 1.0.

Twelve COPCs were evaluated for spotted sandpiper: eleven metals and PCBs. Six metals (i.e., arsenic, cadmium, cobalt, nickel, selenium, and zinc) had both NOAEL- and LOAEL-based HQs that were less than 1.0 in all exposure areas. The NOAEL-based HQs ranged from < 0.1 to 0.90 for these six metals.

PCBs were evaluated both as total PCBs and as PCB TEQs. HQs for PCB TEQs were greater than those for total PCBs. For total PCBs, NOAEL-based HQs were greater than 1.0 in some areas, but none of the LOAEL-based HQs were greater than 1.0. The NOAEL-based HQs for total PCBs that were greater than 1.0 ranged from 1.2 to 2.0 in Areas 2 and 3. For PCB TEQs, all NOAEL-based HQs were greater than 1.0, ranging from 1.9 in Area 1 to 15 in Area 2. Only one LOAEL-based HQ exceeded 1.0; this HQ was 1.5 in Area 2.

The remaining five COPCs (chromium, copper, lead, mercury, and vanadium) had LOAEL-based HQs greater than or equal to 1.0 for spotted sandpiper. LOAEL-based HQs ranged from 1.0 to 5.5 for these COPCs in at least one area in the LDW, indicating risks to spotted sandpiper in those areas. LOAEL-based HQs were greater than or equal to 1.0 in Area 2 for chromium (1.8) and lead (5.5), in Area 3 for copper (1.1) and mercury (1.0), and in all three exposure areas for vanadium (1.0 to 1.4). As shown in Table A.6-45, the highest LOAEL-based HQs (1.8 for chromium and 5.5 for lead) were based primarily on elevated concentrations in prey (benthic invertebrates).

A.6.3.1.2 Uncertainties

This section presents a discussion of uncertainties associated with the problem formulation, the exposure and effects assessments, and the risk characterization for spotted sandpiper.

Problem Formulation

The primary uncertainties in the problem formulation for spotted sandpiper are associated with ROC selection and the COPC screen.

ROC Selection

Uncertainties related to how well spotted sandpiper represents other benthivorous birds in the LDW were addressed in the Phase 1 ERA (Windward 2003b). In that analysis, it was concluded that spotted sandpiper is expected to have an exposure that is similar or higher than those of other benthivorous LDW bird species because its diet consists primarily of benthic invertebrates, it has a high sediment ingestion rate and a high body-weight-normalized FIR, and its home range is within the LDW during the nesting season. Thus, risk estimates for spotted sandpiper in the LDW should be higher than would be calculated for other species in the LDW with different diets, lower sediment and FIRs, and less frequent site use.

COPC Screen

Of the chemicals screened out in the first step of the COPC screen (Section A.2.5.3), 20 chemicals were detected in sediment but were not analyzed in tissue samples (Table A.6-16). None of these chemicals are defined as bioaccumulative chemicals (EPA 2000a), so exposure to these chemicals is assumed to be very low.

Eighty-six chemicals were identified as COIs for birds. Effects data for birds were not available for 40 of the COIs, including 20 individual PAHs. Risks to birds from PAHs were evaluated using TRVs for total PAHs and benzo(a)pyrene. The remaining COIs for which there were no TRVs for birds included antimony, silver, three organotin compounds, eleven SVOCs, and four organochlorine pesticides. Risks to birds from exposure to these COIs could not be evaluated.

Two bioaccumulative chemicals, chromium VI and methylmercury, were detected in more than 5% of the surface sediment samples in which they were analyzed;[88] they were not analyzed in tissue. Therefore, these chemicals were evaluated as components of total chromium and total mercury, respectively. The exposure concentrations of total mercury were compared to TRVs for methylmercury for birds and mammals, so risks from methylmercury exposure may have been overestimated. The only avian or mammalian toxicity studies for chromium VI were based on subchronic or drinking water exposures, so the selected TRVs were for chromium III. Therefore, risk could be underestimated for total chromium if chromium VI is the dominant oxidation state of chromium in LDW benthic invertebrates or fish.

Exposure Assessment

Uncertainties in the exposure assessment for spotted sandpiper were associated with the following factors:

- Direct sediment contact
- Incidental sediment ingestion rate
- COPC bioavailability
- Dietary composition
- Site use
- Benthic invertebrate tissue data and linear regressions
- TEQ approach

These uncertainties are discussed in detail below.

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[88] Chromium VI was analyzed in seven sediment samples in the vicinity of Harbor Island, and methylmercury was analyzed in 20 sediment samples in the vicinity of Duwamish/Diagonal and Norfolk.

Direct Sediment Contact

Risks to wildlife from direct contact with sediment are considered insignificant relative to risks from incidental sediment ingestion (EPA 2000b). However, the exclusion of this pathway adds a small amount of uncertainty to the risk estimate for spotted sandpiper.

Incidental Sediment Ingestion Rate

Uncertainty in the incidental SIR for spotted sandpiper was discussed in the Phase 1 ERA (Windward 2003b). Using an incidental SIR that is 30% of the FIR would increase the spotted sandpiper HQs by an average of less than 0.1. Three LOAEL-based HQs that were less than 1.0 assuming a SIR that is 18% of the FIR would slightly exceed 1.0 if the SIR were assumed to be 30% of the FIR. In Area 3 high- and poor-quality habitat, the chromium HQ would increase from 0.83 to 1.3, and the mercury HQ would increase from 0.99 to 1.1; in Area 2 high- and poor-quality habitat, the PCB TEQ HQ would increase from 0.97 to 1.2. This conservative analysis indicates that risks could be slightly underestimated for chromium, mercury, and PCBs if the incidental SIR is as high as 30% of the FIR.

COPC Bioavailability

Metals may be less bioavailable in ingested sediment than in ingested prey. In calculating the ingested doses, it was assumed that metals were 100% bioavailable, which may overestimate risk if the primary source of the dose is sediment. Table A.6-45 shows the relative contributions of sediment and prey to the total ingested doses. Up to 97% of the ingested dose is from sediment exposure in some areas, indicating that risks may be overestimated in those areas. Contributions from water are very low.

Dietary Composition

The possibility that spotted sandpipers could consume fish, crabs, or mussels in their diet was considered. To address this uncertainty, HQs were calculated assuming that 25% of the spotted sandpiper diet consisted of the alternative prey type (i.e., fish, crabs, or mussels) with the highest UCL concentration. This assumption resulted in slight changes in HQs, but did not cause a change in risk conclusions.

Site Use

Phase 1 uncertainties regarding site-specific habitat for spotted sandpipers were addressed by the survey of spotted sandpiper presence and habitat conducted in the LDW in 2004 (Windward 2004h). Results of the survey were used to develop various exposure scenarios, which reduced uncertainty in the Phase 2 risk conclusions.

Benthic Invertebrate Tissue Data and Linear Regressions

There is uncertainty in the benthic invertebrate tissue data used to estimate dietary exposure of spotted sandpipers to some COPCs because of the relatively small numbers of samples in some exposure areas. The uncertainty is relatively low for PCBs and arsenic because a significant linear relationship exists between sediment and benthic invertebrate tissue concentrations (see Attachment 11), and this relationship was used to estimate tissue UCL concentrations from a relatively larger sediment dataset for each exposure area. However, uncertainty is greater for other COPCs that did not have a significant relationship between sediment and tissue concentrations. For these COPCs, the UCL concentration in tissue is more uncertain because it was based on a small tissue dataset.

The lack of a sediment/tissue relationship for a particular COPC may be because a relatively small range of sediment concentrations was sampled for that COPC or because a correlation does not exist. The sampling of benthic invertebrate tissue was designed to include stations that represented the full range of concentrations of arsenic, PCBs, and lead in surface sediments (Windward 2004d). To evaluate whether the full range of chemical concentrations was sampled for other COPCs, each COPC concentration in co-located sediment was plotted against the cumulative distribution of that COPC in the full baseline surface sediment dataset. For example, the results for copper indicate that the smaller co-located dataset is closely representative of the entire LDW, with the highest concentration in the co-located dataset at the 99th percentile for the entire LDW dataset (Figure A.6-1). In contrast, Figure A.6-2 shows a poorer representation for nickel, with the highest concentration in the co-located dataset at the 64th percentile for the entire LDW dataset.

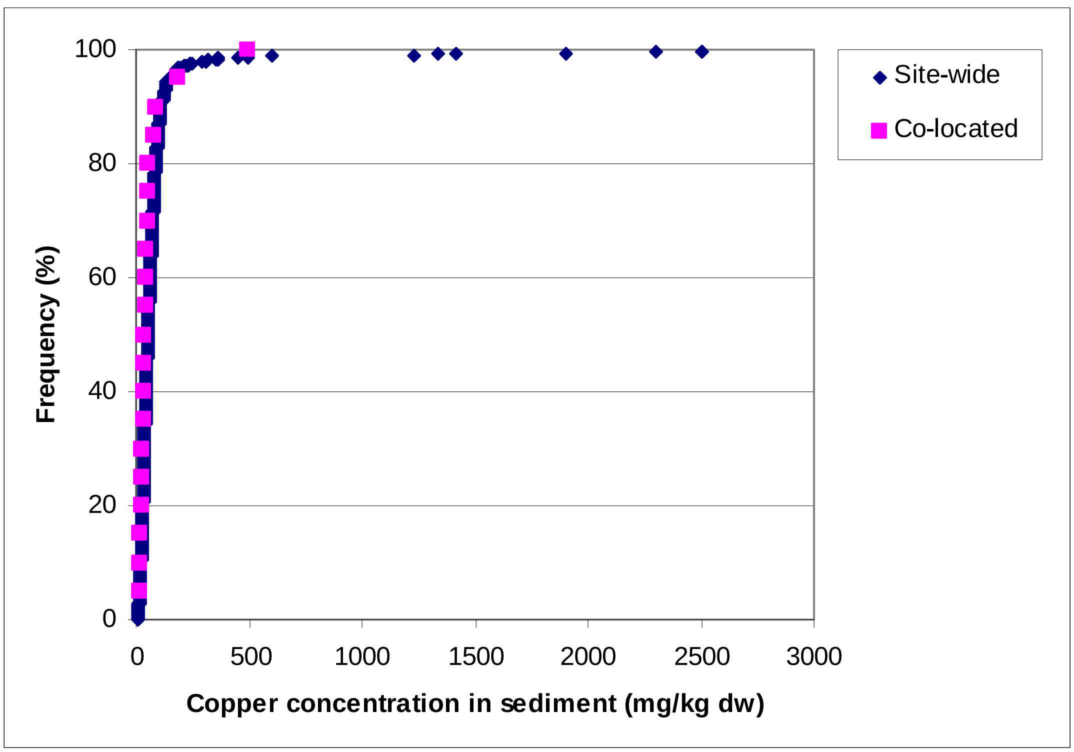

Figure A.6-1. Copper concentrations in sediment samples co-located with benthic invertebrate tissue samples relative to the LDW-wide baseline surface sediment dataset

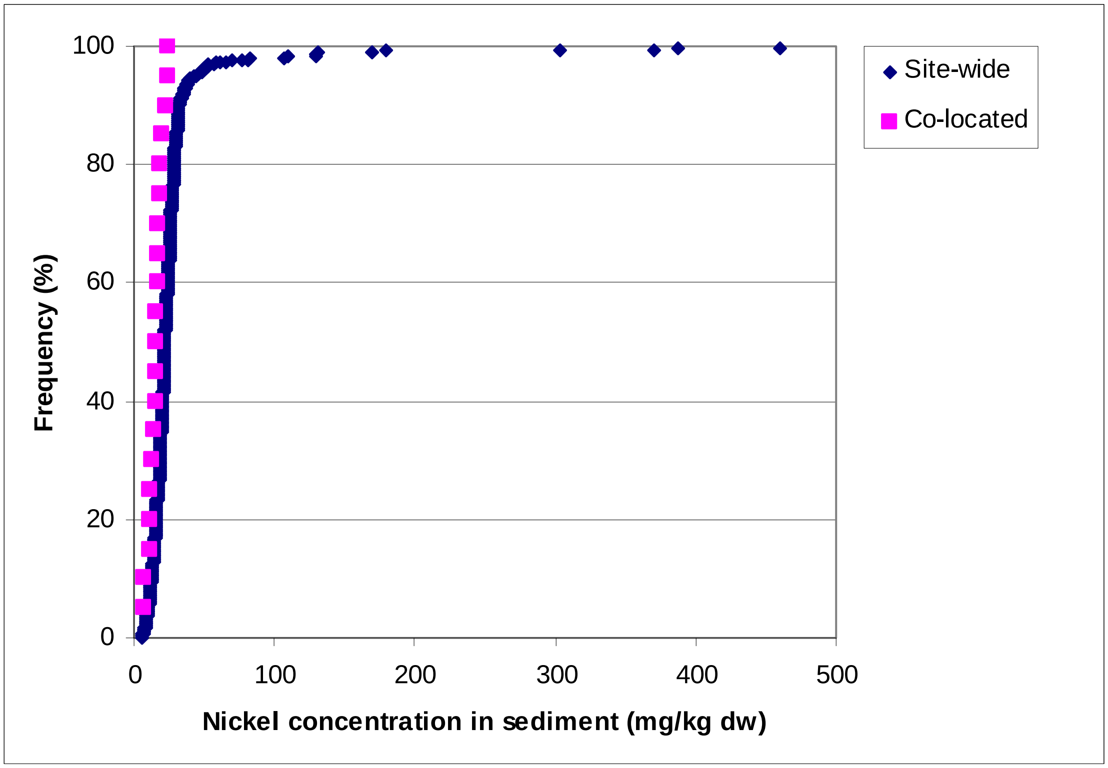

Figure A.6-2. Nickel concentrations in surface sediment samples co-located with benthic invertebrate tissue samples relative to the LDW-wide baseline surface sediment dataset

Results for all COPCs without a significant sediment/tissue relationship are presented in Table A.6-46. The uncertainty in the relationship between sediment and tissue concentrations for cadmium, chromium, cobalt, copper, mercury, vanadium, and zinc is low because benthic invertebrate tissue samples were analyzed over a relatively wide range of sediment concentrations. For the remaining two COPCs (i.e., nickel and selenium), the range of sediment concentrations in the co-located dataset did not cover the range found in the entire LDW dataset. It is possible that a significant sediment/tissue relationship was not found because the range of sediment concentrations was not large enough to capture a relationship, if one existed.

Table A.6-46. COPC concentrations in sediment samples co-located with benthic invertebrate tissue samples relative to the site-wide baseline surface sediment dataset for COPCs without a significant sediment/tissue relationship

COPC	Maximum COPC Concentration in Sediment (mg/kg dw)		Percent of Concentrations in LDW-Wide Dataset Less Than or Equal to the Maximum Concentration in Co-Located Samples
	LDW-Wide Baseline Surface Sediment Dataset	Sediment Samples Co-Located with Benthic Invertebrate Tissue Samples	
Cadmium	120	1.67	94
Chromium	1,100 J	42.5	84
Cobalt	140	31.5	99
Copper	12,000 J	495	99
Mercury	4.6 J	0.528	97
Nickel	910	24.8	64
Selenium	28	1.4	14
Vanadium	150	72.6	84
Zinc	9,700	2,080 J	99

COPC – chemical of potential concern

dw – dry weight

J – estimated concentration

LDW – Lower Duwamish Waterway

The relatively low number of benthic invertebrate tissue samples results in higher uncertainty in the UCL in exposure areas where there is high variability in the tissue concentrations. In four instances, ProUCL calculated a UCL that was greater than the maximum concentration. The ProUCL guidelines recommend using the UCL even if it is greater than the maximum concentration (EPA 2004a). In three of these instances, differences between the maximum concentration and the UCL were small. In one instance (lead in Area 2 high- and poor-quality habitats), the UCL tissue concentration was three times higher than the maximum concentration. Nevertheless, in all instances, risk conclusions would not have changed if maximum values had been used instead of the UCL.

Toxic Equivalent Approach

For the calculation of PCB risks using the TEQ approach, fish, benthic invertebrate, and sediment samples were analyzed for PCB congeners, and TEFs were used to account for toxicity relative to 2,3,7,8-TCDD. The TEFs used to calculate TEQs for dioxin-like PCB congeners were WHO consensus values for birds and mammals from Van den Berg et al. (1998; 2006); these TEFs are presented in Attachment 3. The rationale for the use of TEFs is based on evidence that there is a common mechanism of toxicity for certain dioxins, furans, and PCB congeners, which involves binding to the aryl hydrocarbon (Ah) receptor as an initial step. Data on the relative binding affinity of particular PCB congeners compared to 2,3,7,8-TCDD are available from *in* *vivo* and *in vitro* studies. These data have been used to derive TEFs for PCB congeners that show structural similarity to dioxins, bind to the Ah receptor, and elicit dioxin-specific biochemical and toxic responses.

An uncertainty in the TEQ approach is related to the derivation of consensus TEF values. Limitations in the underlying data used to derive TEFs, such as the relevance of the endpoints in the studies and the lack of information on interspecies variability, contribute to the uncertainty. Although these uncertainties have been identified by Van den Berg (1998), it was decided at a 1997 WHO expert meeting that an additive TEQ method is the most appropriate risk assessment method for complex mixtures of dioxin-like PCB congeners (EPA 2003b). According to the EPA, the TEQ method is technically appropriate for evaluating risks to birds and mammals, and uncertainties associated with the method are not greater than other sources of uncertainty in the ERA process (EPA 2003b).

The four most potent Ah receptor agonists in birds among PCB congeners are the non-ortho PCBs 77, 81, 126, and 169. The variability in the TEFs appears high for PCB congeners that have been tested on multiple species (Van den Berg et al. 1998). For PCB 77, five studies have been conducted, resulting in a TEF range of < 0.0003 to 0.15 for the various bird species tested for ethoxyresorufin-O-deethylase (EROD) induction or *in ovo* effects. For PCB 81, two identified studies tested several species for EROD induction, with TEFs highly variable, ranging from 0.001 to 0.5. For PCB 126 and 169, data are available from only one study (*in ovo* with chickens). These TEFs derived by EROD induction or *in ovo* studies are most accurate for the assessment of effects based on concentrations in whole embryos (EPA 2003b). Thus, the relevance of applying the bird TEFs to dietary exposure is uncertain. Egg concentrations were not evaluated in this risk assessment because concentrations of dioxin-like compounds in bird eggs were not available, and reliable models for predicting egg concentrations from concentrations in the diet were not available. The absence of data or reliable predictions of dioxin-like compounds in eggs of birds using the LDW results in additional uncertainty in risk estimates for birds. It is not known if the uncertainties discussed above would overestimate or underestimate risks.

It should also be noted that the TEQs calculated in this ERA were used only for assessing the toxicity of the dioxin-like PCB congeners and do not account for TEQ contributions from dioxin and furan congeners; dioxin and furan concentrations in tissue were not available for the LDW. Risk estimates for wildlife based on TEQs calculated considering only dioxin-like PCB congeners are likely lower than the actual risk resulting from the cumulative exposure of wildlife to dioxins, furans, and dioxin-like PCB congeners in the LDW.

Effects Assessment

Uncertainty associated with available toxicity benchmarks for birds may affect risk estimates. These uncertainties were discussed in detail in the Phase 1 ERA (Windward 2003b). The primary uncertainties include:

- None of the laboratory toxicological studies used to derive TRVs were conducted using ROC species.
- The laboratory studies on which TRVs are based were conducted in controlled settings using single-contaminant exposures. Effects associated with multiple-chemical exposure and other environmental stressors present at the site (e.g., habitat loss) were not factored into these studies. It is unknown if these factors would result in additive, synergistic, antagonistic, or neutral effects on overall risk conclusions.
- NOAELs were not available for some COPCs, so they were estimated from LOAELs.

In addition, TRVs are considered less certain if there were a small number of studies, if endpoints were subchronic, or if data quality was questionable. The relative uncertainties in the selected TRVs for birds and the potential effect on the risk estimates are summarized in Table A.6-47.

Table A.6-47. Level of uncertainty associated with TRVs for birds

COPC	Number of TRV Studies	Uncertainty in TRV[a]
Arsenic	3	high; reproductive studies with arsenite (the most toxic form of arsenic) were not available
Cadmium	8	high; the only effect endpoints were eggshell thinning (with no effect on reproductive success) and growth in chicks after subchronic exposure
Chromium	3	high; only one study reported effects, but this study was unpublished and could not be obtained for review of data quality; selected TRV was based on a survival endpoint, so the resulting HQ may underestimate the potential for sublethal effects
Cobalt	1	high; only one toxicity study, which evaluated growth after a 2-week exposure period, was available
Copper	7	medium; selected TRVs were based on subchronic growth and survival endpoints
Lead	4	medium; selected TRVs were based on a chronic reproductive endpoint
Mercury	7	medium; selected TRVs were based on a chronic growth endpoint
Nickel	3	medium; selected TRVs were based on a subchronic growth endpoint
Selenium	5	medium; selected TRVs were based on chronic reproduction and survival endpoints
Vanadium	2	high; only one toxicity study, which evaluated growth after a 4-week exposure period, was available
Zinc	6	high; selected TRVs were based on a subchronic growth endpoint
Total PCBs	13	medium; selected TRVs were based on a chronic reproduction endpoint
PCB TEQs	2	high; no dietary studies were available; selected TRV was based on a study with acute high-level weekly exposure via IP injection

[a] Level of uncertainty key:

Low = large dataset including chronic studies

Medium = moderately sized dataset including chronic studies

High = small dataset with only subchronic studies, unbounded NOAELs/LOAELs, or data with questionable data quality

COPC – chemical of potential concern
HQ – hazard quotient
IP – intraperitoneal
PCB – polychlorinated biphenyl
TEQ – toxic equivalent
TRV – toxicity reference value

In summary, uncertainty associated with the chromium TRV is high because only one study that reported effects was available; this study is unpublished and could not be reviewed. Uncertainty is also high for the PCB TEQ TRV because the selected TRV was based on exposure from acute high-level weekly IP injections. Risks for cadmium, cobalt, nickel, vanadium, and zinc could be under- or overestimated because the endpoints were based on subchronic growth effects and no reproductive endpoints were available. Risks are uncertain for chromium because the TRV was based on a single unpublished study with a survival endpoint. The effect of uncertainties in toxicity data for copper, lead, mercury, selenium and total PCBs on the risk estimates for these COPCs is unknown.

Risk Characterization

Risks to spotted sandpiper from total DDTs were not included in the risk estimates because of high uncertainty in the tissue pesticide data. Probable analytical interference from PCBs in benthic invertebrate, fish, and crab tissue samples collected and analyzed in 2004 resulted in suspected false identifications of presence of some organochlorine pesticides as well as overestimates in their concentrations (JN-qualified). The uncertainty and high bias of these results are discussed in more detail in Section A.2.4.2.2. Ingested dose estimates, TRVs, and risk estimates are presented in this section for total DDTs, which was identified as a spotted sandpiper COPC in Section A.2.5.3.

Methods used to calculate the ingested dose of total DDTs are the same as those used for other COPCs, as described in Section A.5.1.1. The tissue and sediment exposure concentrations are presented in Table A.6-48, and resulting ingested doses are presented in Table A.6-49. Exposure to DDTs via water was not estimated because the contribution of total DDTs via water is expected to be very low based on the low solubility of DDTs and results from PCB exposure calculations, which showed the contribution from water was less than 0.01 percent of the contribution from food and sediment exposure. Tables containing all exposure assumptions and data used in the ingested dose calculations are presented in Attachment 12.

Table A.6-48. Exposure concentrations of total DDTs in benthic invertebrate tissue and sediment for spotted sandpiper

EXPOSURE AREA[a]	EXPOSURE CONCENTRATION IN BENTHIC INVERTEBRATE TISSUE (mg/kg dw)	EXPOSURE CONCENTRATION IN SEDIMENT (mg/kg dw)
Area 1 – high	0.29	0.017
Area 1 – high and poor	0.29	0.018
Area 2 – high	1.1	1.0
Area 2 – high and poor	0.96	0.52

Exposure Area[a]	Exposure Concentration in Benthic Invertebrate Tissue (mg/kg dw)	Exposure Concentration in Sediment (mg/kg dw)
Area 3 – high	1.8	0.024
Area 3 – high and poor	1.8	0.020

[a] Six exposure scenarios were evaluated for spotted sandpiper. In each of three exposure areas, foraging only in high-quality habitat and foraging in both high- and poor-quality habitat were evaluated. These exposure scenarios are described in detail in Section A.5.1.2.1.

dw – dry weight

Table A.6-49. Ingested doses of total DDTs for spotted sandpiper

Exposure Area	Ingested Dose (mg/kg bw/day)
Area 1 – high	0.046
Area 1 – high and poor	0.046
Area 2 – high	0.20
Area 2 – high and poor	0.17
Area 3 – high	0.28
Area 3 – high and poor	0.28

bw – body weight

Toxicity studies with any form of DDT, including DDD and DDE, were evaluated to select the TRV for total DDTs. The evaluation identified numerous studies that analyzed the dietary toxicity of DDT, DDD, and DDE to birds. Table A.6-50 presents the results from 10 studies with the lowest reported effect concentrations. All of the reviewed studies evaluated reproductive endpoints; eight of the studies reported increased eggshell thinning. A dose of 0.15 mg/kg bw/day resulted in eggshell thinning in Japanese quail exposed to p,p′-DDT in the diet for 194 days. Although eggshell thinning was significantly different in the treated group than in controls, hatchability of the eggs was not affected, therefore, it was concluded that the degree to which eggshell thinning was observed would not affect reproductive success. The next lowest dose of 0.32 mg/kg bw/day resulted in eggshell thinning, eggshell breakage, and nestling mortality in barn owls exposed to dietary p,p′-DDE for 2 years. This dose was selected as the LOAEL TRV because a clear impairment to reproductive success was observed. The highest NOAEL below the LOAEL was a dose of 0.19 mg/kg bw/day, which did not cause eggshell thinning in mallards exposed to DDT for 11 months (Davison and Sell 1974). Because other reproductive endpoints were not assessed in this study, and it is unknown whether the no effect level for eggshell thinning would be the same as the no effect level for direct measures (e.g., hatchability, viability of offspring) of reproductive success, this NOAEL was not selected. Instead, the NOAEL TRV was estimated from the selected LOAEL TRV using an uncertainty factor of 5, resulting in a NOAEL TRV of 0.065 mg/kg bw/day.

Table A.6-50. DDT, DDD, and DDE dietary toxicity studies for birds

Chemical	Test Species	NOAEL (mg/kg bw/day)	LOAEL (mg/kg bw/day)	Exposure Duration	Effect	No-Effect Conc.[a]	Effect Conc.[a]	Body Weight (kg)	Food Consumption Rate[b]	Source	Notes
p,p'-DDT	quail	na	0.15	194 days	increased eggshell thinning	na	2.5 mg/kg ww	0.09 (Dunning 1993)	0.0048 kg dw/day galliformes (Nagy 2001)	Stickel and Rhodes (1970)	1
p,p'-DDT	mallard	0.19	na	11 months	no effect on eggshell thinning	2 mg/kg ww	na	1.19	0.115 kg ww/day	Davison and Sell (1974)	2
DDE	barn owl	na	**0.32**	2 years	increased eggshell thinning, eggshell breakage, and nestling mortality	na	2.83 mg/kg ww	0.5235 (Dunning 1993)	0.0539 kg dw/day carnivores (Nagy 2001)	Mendenhall et al. (1983)	1
DDE	America n kestrel	na	0.35	14 days	increased eggshell thinning	na	3 mg/kg ww	0.13 (Pattee 1984)	0.0136 kg dw/day Eurasian kestrel (Nagy 2001)	Peakall et al. (1973)	1
Technical DDD	mallard	na	0.90	2 years	decreased hatchling survival and production	na	10 mg/kg dw	1.082 (Dunning 1993)	0.1082 kg ww/day (Heinz et al. 1987)	Heath et al. (1969)	1
p,p'-DDE	mallard	na	0.90	2 years	increased eggshell thinning and number of cracked eggs; reduced hatchling survival and production	na	10 mg/kg dw	1.082 (Dunning 1993)	0.1082 kg ww/day (Heinz et al. 1987)	Heath et al. (1969)	1
p,p-DDE	black duck	na	1.0	7 months	increased eggshell thinning and duckling mortality; reduced hatchability	na	10 mg/kg ww	1.25 (Dunning 1993)	0.125 kg ww/day (Heinz et al. 1987)	Longcore and Samson (1973)	
DDE	mallard	na	1.0	30 days	increased eggshell thinning	na	10 mg/kg ww	1.082 (Dunning 1993)	0.1082 kg ww/day (Heinz et al. 1987)	Kolaja (1977)	1

Chemical	Test Species	NOAEL (mg/kg bw/day)	LOAEL (mg/kg bw/day)	Exposure Duration	Effect	No-Effect Conc.[a]	Effect Conc.[a]	Body Weight (kg)	Food Consumption Rate[b]	Source	Notes
DDT	mallard	na	1.0	30 days	increased eggshell thinning	na	10 mg/kg ww	1.082 (Dunning 1993)	0.1082 kg ww/day (Heinz et al. 1987)	Kolaja (1977)	1
p,p' DDE	America n kestrel	na	1.0	1 year (two clutches)	increased eggshell thinning	na	2.8 mg/kg ww	0.13 (Pattee 1984)	0.0136 kg dw/day Eurasian kestrel (Nagy 2001)	Wiemeyer and Porter (1970); Porter and Wiemeyer (1970)	1

Notes:

1. Effect concentration converted into dry weight assuming 10% moisture in prepared diet
2. Body weight and food consumption rate reported in study

[a] No-effect and effect concentrations are presented in the units given in the studies reviewed. Table notes indicate how units were converted to wet weight or dry weight to correspond to the food consumption rate units for calculating NOAELs and LOAELs.

[b] Consumption rates are from equations for bird groups presented in Nagy (2001), from data presented for individual bird species (Nagy 2001), or from other sources as noted.

dw – dry weight

LOAEL – lowest-observed-adverse-effect level

na – not available or not applicable

NOAEL – no-observed-adverse-effect level

ww – wet weight

Bold identifies the NOAEL and LOAEL selected as the TRVs. A NOAEL TRV was not available for DDE from the study in which the chronic LOAEL of 0.32 mg/kg bw/day was reported, so it was estimated using an uncertainty factor of 5. The resulting NOAEL TRV is 0.065 mg/kg bw/day.

The estimated ingested dose for spotted sandpiper was compared to the NOAEL and LOAEL TRVs, as presented in Table A.6-51. No LOAEL-based HQs were greater than 1.0. In Area 2, the NOAEL-based HQs were 3.1 and 2.6; and in Area 3, the NOAEL-based HQs were both 4.3. The NOAEL-based HQs were less than 1.0 in Area 1.

Table A.6-51. DDT HQs for spotted sandpiper

Receptor	Ingested Dose (mg/kg bw/day)	TRVs (mg/kg bw/day)		NOAEL-Based HQ	LOAEL-Based HQ
		NOAEL	LOAEL		
Area 1 – high	0.046	0.065	0.32	0.71	0.14
Area 1 – high and poor	0.046	0.065	0.32	0.71	0.14
Area 2 – high	0.20	0.065	0.32	**3.1**	0.63
Area 2 – high and poor	0.17	0.065	0.32	**2.6**	0.53
Area 3 – high	0.28	0.065	0.32	**4.3**	0.88
Area 3 – high and poor	0.28	0.065	0.32	**4.3**	0.88

HQ – hazard quotient

LOAEL – lowest-observed-adverse-effect level

NOAEL – no-observed-adverse-effect level

TRV – toxicity reference value

Bold identifies NOAEL-based HQs greater than 1.0.

The high bias for DDTs was confirmed by reanalyzing sediment samples co-located with benthic invertebrate tissue samples and fish and crab tissue samples that had high PCB and DDT concentrations, as described in Section A.2.4.2.2. The presence of DDTs was confirmed in six of the eight tissue samples at concentrations ranging from 5 to 34% of the original results and in sediment samples at concentrations ranging from 4 to 60% of the original results. Because of these uncertainties, it is likely that the NOAEL-based HQs for DDTs would have been lower if there had been no analytical interferences.

Summary of Uncertainties

Uncertainties in the problem formulation, the effects and exposure assessments, and risk characterization for spotted sandpiper were evaluated, with the following conclusions:

- Uncertainties in FIR, direct sediment contact, incidental sediment ingestion, dietary composition, and site use are expected to have minimal or no effect on risk conclusions.
- Uncertainties in COPC selection are not expected to have an effect on risk conclusions.
- For COPCs other than PCBs and arsenic, there is some uncertainty associated with risk estimates because of the relatively small number of benthic invertebrate tissue samples in each sandpiper foraging area.

- For COPCs whose sediment concentrations contributed a large portion of the ingested dose, risks could be overestimated if these COPCs are not 100% bioavailable in sediments.
- The PCB TEQs were calculated using TEFs for individual PCB congeners; the derivation of some of these TEFs is uncertain. It is not known if this uncertainty would overestimate or underestimate risk. Risk estimates for wildlife based on TEQs calculated considering only dioxin-like PCB congeners are likely lower than the actual risk resulting from the cumulative exposure of wildlife to dioxins, furans, and dioxin-like PCB congeners in the LDW.
- Uncertainties in toxicity data may affect risk estimates. Risks could be underestimated for arsenic because of the lack of arsenite toxicity data. Risks are uncertain for cadmium, cobalt, nickel, vanadium, and zinc because TRVs were based on subchronic growth endpoints. Risks are uncertain for chromium because the TRV was based on a single unpublished study with a survival endpoint. Risks are also uncertain for PCB TEQs because the TRVs were based on a study using an acute weekly dose via IP injection and TEFs were applied to dietary exposure estimates. Effects of uncertainties in toxicity data on risk estimates for copper, lead, mercury, selenium, and PCBs (based on a total PCB approach) are unknown.
- The LOAEL-based HQs for spotted sandpiper and DDTs were less than 1.0, although NOAEL-based HQs were greater than 1.0 in Areas 2 and 3. It is likely that NOAEL-based HQs would have been closer to 1.0 if there had been no analytical interferences from PCBs because the interference is known to bias the DDT concentrations high.

A.6.3.1.3 Risk conclusions

Spotted sandpiper, a bird which has been observed nesting along the LDW, was selected to represent benthivorous birds such as dunlin, dowitcher, western sandpiper, and dabbling ducks. The risk characterization for sandpiper should be protective of other benthivorous birds because of the spotted sandpiper's high exposure to COPCs through the ingestion of benthic invertebrates and the incidental ingestion of sediment.

Results of the risk characterization for spotted sandpiper are summarized in Table A.6-52.

Table A.6-52. Summary of risk characterization for spotted sandpiper

COPC	Range of NOAEL-Based HQs[a]	Range of LOAEL-Based HQs[a]	Effect	Primary Uncertainty[b, c]
Arsenic	0.34 – 0.58	< 0.1 – 0.15	delayed egg laying; depressed egg weight, production, and shell thinning; decreased offspring body weight[d]	high uncertainty in TRV because reproductive studies using arsenite were not available; risk could be underestimated
Cadmium	< 0.1 – 0.21	<0.1	reduced body weight (subchronic exposure)	high uncertainty in TRV; risk could be under- or overestimated because of endpoint
Chromium	**1.3** – **8.8**	0.26 – **1.8**	reduced survival	high uncertainty in TRV because study was unpublished; risk could be overestimated because of high sediment contribution to ingested dose and 100% bioavailability assumption; risk may be underestimated because TRV was based on survival rather than a sublethal endpoint
Cobalt	0.22 – 0.27	<0.1	reduced body weight (chronic exposure)	high uncertainty in TRV; risk could be under- or overestimated because of endpoint
Copper	0.62 – **1.5**	0.45 – **1.1**	reduced growth and survival (subchronic exposure)	medium uncertainty in TRV; risk could be under- or overestimated because of endpoint
Lead	0.58 – **19**	0.17 – **5.5**	reduced egg hatchability	medium uncertainty in TRV; risk could be overestimated because of high sediment contribution to ingested dose and 100% bioavailability assumption
Mercury	**1.1** – **5.3**	0.21 – **1.0**	reduced growth (chronic exposure)	medium uncertainty in TRV
Nickel	< 0.1 – 0.22	< 0.1– 0.11	reduced body weight (subchronic exposure)	medium uncertainty in TRV; risk could be over- or underestimated because of endpoint; uncertainty in tissue dataset
Selenium	0.62 – 0.90	0.38 – 0.55	reduced offspring growth and survival	medium uncertainty in TRV; uncertainty in tissue dataset
Vanadium	**2.0** – **2.7**	**1.0** – **1.4**	reduced body weight (subchronic exposure)	high uncertainty in TRV; risk could be under- or overestimated because of endpoint
Zinc	0.41 – 0.78	0.27 – 0.52	reduced growth (subchronic exposure)	high uncertainty in TRV; risk could be under- or overestimated because of endpoint
Total PCBs	0.51 – **2.0**	0.18 – 0.71	reduced hatching success	medium uncertainty in TRV

COPC	Range of NOAEL-Based HQs[a]	Range of LOAEL-Based HQs[a]	Effect	Primary Uncertainty[b, c]
PCB TEQs	**1.9** – **10**	0.18 – **1.5**	reduced body weight, egg production, and survival of adults and embryos	high uncertainty in TRV because of high-level weekly exposures and IP injection; risk could be overestimated; derivation of TEFs is uncertain

a Range of HQs calculated for the six spotted sandpiper exposure scenarios.

b Level of uncertainty key:

Low = large dataset including chronic studies with species taxonomically similar to the ROC

Medium = moderately sized dataset including chronic studies

High = small dataset with only subchronic studies, unbounded NOAELs/LOAELs, or data with questionable data quality

c There may also be some uncertainty associated with risk estimates for COPCs other than arsenic and total PCBs because of the relatively small number of benthic invertebrate tissue samples.

d Effects were from exposure to sodium arsenate.

COPC – chemical of potential concern

HQ – hazard quotient

IP – intraperitoneal

LOAEL – lowest-observed-adverse-effect level

NOAEL – no-observed-adverse-effect level

PCB – polychlorinated biphenyl

TEF – toxic equivalency factor

TEQ – toxic equivalent

TRV – toxicity reference value

Bold identifies NOAEL-based HQs greater than 1.0 and LOAEL-based HQs greater than or equal to 1.0.

The following COPCs had LOAEL-based HQs greater than or equal to 1.0 for spotted sandpiper: chromium, copper, lead, mercury, and vanadium. LOAEL-based HQs ranged from 1.0 to 5.5 for these COPCs. LOAEL-based HQs were greater than or equal to 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). Risks to spotted sandpiper from chromium are uncertain because the TRV was based on a single unpublished study with a survival endpoint. 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 is primarily from sediment for copper and lead in Area 3 and for vanadium in all areas. Bioavailability of metals in sediment is not likely 100%, so the LOAEL-based HQs (ranging from 1.0 to 1.5) for copper, lead, and vanadium may be overestimated. Overall, these findings indicate risk for 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.

For total PCBs, 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 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 Area 2 under the high-quality habitat scenario, in which the LOAEL-based HQ was 1.5. Uncertainty is high for PCB TEQ risk estimates because the TRVs were based on a study using an acute weekly dose via IP injection, and TEFs derived in studies of toxicity to eggs were applied to dietary exposure estimates. Therefore, risks to spotted sandpiper from PCBs are low in most areas of the LDW, with some uncertainty.

The remaining COPCs (arsenic, cadmium, cobalt, nickel, selenium, and zinc) had both NOAEL- and LOAEL-based HQs that were less than 1.0 in all sandpiper exposure areas. Exposure of spotted sandpipers to these COPCs is not expected to result in adverse effects.

For total DDT, 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.6.3.2 Great blue heron

This section present risk characterization, uncertainties, and risk conclusions for great blue heron.

A.6.3.2.1 Risk estimates

Four COPCs were evaluated for great blue heron: chromium, lead, mercury, and PCBs (using both a total PCB and a PCB TEQ approach). As shown in Table A.6-53, all of the NOAEL- and LOAEL-based HQs were less than 1.0.

Table A.6-53. HQ calculations for great blue heron

ROC	COPC	Ingested Dose (mg/kg bw/day)	TRVs (mg/kg bw/day)		NOAEL-Based HQ	LOAEL-Based HQ
			NOAEL	LOAEL		
Great blue heron	chromium	0.067	1.0	5.0	< 0.1	< 0.1
	lead	0.27	5.82	20	< 0.1	< 0.1
	mercury	0.0031	0.018	0.091	0.17	< 0.1
	total PCBs	0.17	0.49	1.4	0.35	0.12
	PCB TEQs	9.3×10^{-6}	1.4×10^{-5}	1.4×10^{-4}	0.66	< 0.1

[a] No PCB congener data were available for juvenile chinook salmon. When calculating risk estimates, the portion of prey ingestion that had been assigned to juvenile chinook salmon (24%) was divided proportionally among the remaining prey categories.

bw – body weight

COPC – chemical of potential concern

HQ – hazard quotient

LOAEL – lowest-observed-adverse-effect level

NOAEL – no-observed-adverse-effect level

PCB – polychlorinated biphenyl

ROC – receptor of concern

TEQ – toxic equivalent

TRV – toxicity reference value

A.6.3.2.2 Uncertainties

This section presents a discussion of the uncertainty associated with the problem formulation, the exposure and effects assessments, and the risk characterization for great blue heron.

Problem Formulation

Primary uncertainties in the problem formulation for great blue heron are associated with ROC selection and the COPC screen. Uncertainties in the COPC screen for bird ROCs in general were discussed in Section A.6.3.1.2 for spotted sandpiper.

Uncertainties related to the representation of other bird species by great blue heron were addressed in the Phase 1 ERA (Windward 2003b). In that analysis, it was concluded that the great blue heron may have a lower daily FIR relative to its body weight than some other piscivorous species (i.e., common murre, pigeon guillemot, and Caspian tern), but these other species forage in the LDW infrequently. As a result, the overall ingestion of prey from the LDW is likely to be higher for great blue heron than for other piscivorous species.

Exposure Assessment

Uncertainties in the exposure assessment for great blue heron were associated with the following factors:

- Direct sediment contact
- Incidental sediment ingestion rate
- Dietary composition
- Site use
- TEQ approach

The uncertainties associated with direct sediment contact and the TEQ approach are the same as those discussed in Section A.6.3.1.2 for spotted sandpiper.

To address uncertainties in the amount of sediment incidentally ingested by great blue heron while foraging, ingested doses of COPCs were calculated assuming the SIR was 10% of the FIR versus 2% assumed in Section 5.1.2.2. This conservative assumption would result in an increase of HQs by an average of less than 0.1 and would not change risk conclusions.

Dietary Composition

The assessment for great blue heron is assumed to be protective of omnivorous bird species such as bufflehead, Barrow's goldeneye, and surf scoter, which may consume primarily mussels, clams, or crabs. To address the uncertainty that invertebrate species could be ingested, HQs for each COPC were calculated using the conservative assumption that the heron's diet consisted of the invertebrate species (i.e., mussels, clams, or crabs) with the maximum concentration of that COPC. This conservative assumption did not result in any NOAEL- or LOAEL-based HQs greater than 1.0 for great blue heron.

There are some uncertainties in the proportions of different fish species in the great blue heron's diet. Different preferences for fish could result in greater exposure of great blue heron. To address this uncertainty, HQs were calculated assuming that 75% of the great blue heron diet was composed of the fish species with the highest concentration for each COPC, with the remainder of the diet composed of equal proportions of the other two species.[89] This assumption resulted in changes in the HQs but did not result in a change of any HQ from less than 1.0 to greater than 1.0.

Site Use

There are uncertainties in the site-use factor for great blue herons because of the lack of site-specific information on the foraging of great blue herons from the Black River and Kiwanis Ravine colonies. A site-use factor of 0.5 is likely to be conservative; but if a higher site-use factor of 0.75 had been used, the HQs for great blue heron would still have remained below 1.0.

Effects Assessment

Uncertainties associated with available toxicity benchmarks for birds are discussed in Section A.6.3.1.2.

Risk Characterization

Risk estimates for organochlorine pesticides were not discussed in Section A.6.3.2.1 because the JN-qualified pesticide tissue data are biased high as a result of analytical interference from PCBs, as discussed in Section A.6.3.1.2 for spotted sandpiper. The same methods and TRVs used for estimating risks to spotted sandpiper from total DDTs were used for great blue heron. The tissue and sediment exposure concentrations are presented in Table A.6-54.

Table A.6-54. Exposure concentrations of total DDTs

Ingested Media	Exposure Concentration (mg/kg dw)
Shiner surfperch	1.0
English sole/starry flounder	0.76
Juvenile chinook salmon	0.21
Pacific staghorn sculpin	0.50
Crab	0.61
Intertidal sediment	0.29

dw – dry weight

[89] In the exposure assessment, it is assumed that heron ingest 34% shiner surfperch, 21% English sole, and 45% juvenile chinook salmon.

The calculated ingested dose of total DDTs for great blue heron, based on the exposure concentrations presented in Table A.6-54, was 0.012 mg/kg bw/day. This concentration is lower than the selected NOAEL TRV of 0.056 mg/kg bw/day, indicating that there is very low risk to great blue heron from exposure to total DDTs.

Summary of Uncertainties

Uncertainties in the problem formulation, the effects and exposure assessments, and risk characterization for great blue heron are summarized as follows:

- Uncertainties in incidental sediment ingestion, dietary composition, and site use are expected to have minimal or no effect on risk conclusions.
- The PCB TEQs were calculated using TEFs for individual PCB congeners; the derivation of some of these TEFs is uncertain, with unknown effects on risk conclusions. Risk estimates for wildlife based on TEQs calculated considering only dioxin-like PCB congeners are likely lower than the actual risk resulting from the cumulative exposure of wildlife to dioxins, furans, and dioxin-like PCB congeners in the LDW.
- Gaps in toxicity data result in uncertainty in risk estimates. Risks are uncertain for chromium because the TRV was based on a single unpublished study with a survival endpoint. Risks are uncertain for PCB TEQs because the TRVs were based on a study using an acute weekly dose via IP injection, and TEFs were applied to dietary exposure estimates. Effects of gaps in toxicity data on risk estimates for lead, mercury, and PCBs (based on a total PCB approach) are unknown.
- The NOAEL-based HQ for great blue heron and total DDTs was less than 1.0, indicating very low risk.

A.6.3.2.3 Risk conclusions

Great blue heron, a commonly observed bird along the LDW, was selected as an ROC to represent primarily piscivorous birds in the LDW, such as loons, western grebe, mergansers, double-crested cormorant, pigeon guillemot, Caspian tern, and common murre. Results for the risk characterization for great blue heron are summarized in Table A.6-55.

Table A.6-55. Summary of risk characterization for great blue heron

COPC	NOAEL-Based HQ	LOAEL-Based HQ	Effect	Primary Uncertainty[a]
Chromium	< 0.1	< 0.1	reduced survival	high uncertainty in TRV because study was unpublished; risk may be underestimated because TRV was based on survival rather than a sublethal endpoint
Lead	< 0.1	< 0.1	reduced egg hatchability	medium uncertainty in TRV
Mercury	0.17	< 0.1	reduced growth (chronic exposure)	medium uncertainty in TRV
Total PCBs	0.35	0.12	reduced hatching success	medium uncertainty in TRV
PCB TEQs	0.66	< 0.1	reduced body weight, egg production, and survival of adults and embryos	high uncertainty in TRV because of high-level weekly exposures and IP injection; risk could be overestimated; derivation of TEFs is uncertain

[a] Level of uncertainty key:

Low = large dataset including chronic studies with species taxonomically similar to the ROC

Medium = moderately sized dataset including chronic studies

High = small dataset with only subchronic studies, unbounded NOAELs/LOAELs, or data with questionable data quality

COPC – chemical of potential concern

HQ – hazard quotient

IP – intraperitoneal

LOAEL – lowest-observed-adverse-effect level

NOAEL – no-observed-adverse-effect level

PCB – polychlorinated biphenyl

TEF – toxic equivalency factor

TEQ – toxic equivalent

The NOAEL- and LOAEL-based HQs for great blue heron are all less than 1.0, indicating that very low risks to great blue herons are expected from exposure to chromium, lead, mercury, and PCBs in the LDW. For chromium and PCB TEQs, there is high uncertainty in the TRVs, but the LOAEL TRVs would need to be lower by a factor of 14 for the LOAEL-based HQs to equal 1.0. Uncertainties in the TEFs used to calculate the PCB TEQ concentrations could also result in overestimates or underestimates of risk.

A.6.3.3 Osprey

This section presents risk estimates, uncertainties, and risk conclusions for osprey.

A.6.3.3.1 Risk estimates

Four COPCs were evaluated for osprey: chromium, lead, mercury, and PCBs (using both a total PCB and a PCB TEQ approach). The only HQ for osprey greater than 1.0 was the NOAEL-based HQ of 1.6 for PCB TEQs (Table A.6-56). All other HQs, both NOAEL- and LOAEL-based HQs, were less than 1.0.

Table A.6-56. HQ calculations for osprey

ROC	COPC	Ingested Dose (mg/kg bw/day)	TRVs (mg/kg bw/day)		NOAEL-Based HQ	LOAEL-Based HQ
			NOAEL	LOAEL		
Osprey	chromium	0.11	1.0	5.0	0.11	< 0.1
	lead	0.26	5.82	20	< 0.1	< 0.1
	mercury	0.0051	0.018	0.091	0.28	< 0.1
	total PCBs	0.32	0.49	1.4	0.65	0.23
	PCB TEQs	2.3×10^{-5}	1.4×10^{-5}	1.4×10^{-4}	**1.6**	0.16

[a] No PCB congener data were available for juvenile chinook salmon. When calculating risk estimates, the portion of prey ingestion that had been assigned to juvenile chinook salmon (45%) was divided proportionally among the remaining prey categories.

bw – body weight
COPC – chemical of potential concern
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
NOAEL – no-observed-adverse-effect level
PCB – polychlorinated biphenyl
ROC – receptor of concern
TEQ – toxic equivalent
TRV – toxicity reference value

Bold identifies NOAEL-based HQs greater than 1.0.

A.6.3.3.2 Uncertainties

This section presents a discussion of the uncertainty associated with the problem formulation, the exposure and effects assessments, and the risk characterization for osprey.

Problem Formulation

Primary uncertainties in the problem formulation for osprey are associated with ROC selection and the COPC screen. Uncertainties in the COPC screen for birds were discussed in Section A.6.3.1.2.

Osprey were selected to represent piscivorous birds in addition to great blue heron, as well as carnivorous birds in the LDW. Osprey may have lower daily food consumption rates relative to their body weight than some other piscivorous species (i.e., common murre, pigeon guillemot, and Caspian tern), but these other species forage in the LDW infrequently. As a result, the overall consumption of prey from the LDW is likely to be higher for osprey than for other piscivorous species.

Some carnivorous birds, such as bald eagles, consume fish as well as other birds, such as grebes, gulls, and other waterfowl. Birds used as prey by the bald eagle may contain higher concentrations of bioaccumulative COPCs in their tissue than fish if the birds consumed are consumers of fish. Thus, if bald eagles forage primarily in the LDW, they may be more exposed to bioaccumulative COPCs than osprey because of the eagles' consumption of birds. Data on chemical concentrations in tissues of birds using the LDW are not available. The potential effect of the inclusion of birds in the bald eagle diet on the risk to bald eagles estimated from the risks to osprey is not known because of uncertainties in the proportion of birds in the eagle diet, the amount of fish those birds consume from the LDW, and the extent of biomagnification from fish to birds.

Exposure Assessment

Uncertainties in the exposure assessment for osprey were associated with the following factors:

- Direct sediment contact
- Incidental sediment ingestion rate
- Dietary composition
- TEQ approach

The uncertainties associated with direct sediment contact and the TEQ approach are the same as those discussed in Section A.6.3.1.2 for spotted sandpiper.

To address uncertainties in the amount of sediment incidentally ingested by osprey while foraging, ingested doses of COPCs were calculated assuming the SIR was 10% of the FIR versus 1% assumed in Section 5.1.2.3. This conservative assumption would result in an increase of HQs by an average of less than 0.1 and would not change risk conclusions.

There are some uncertainties in the proportions of different types of fish species in osprey's diet, as discussed in Section A.5.1.2.3, which could have an effect on exposure. To address this uncertainty, HQs for each COPC were calculated assuming that 75% of the osprey's diet was composed of the fish species with the highest concentration for that COPC, with the remainder of the diet composed of equal proportions of the other fish species.[90] This assumption resulted in changes in the HQs but did not result in a change of any HQ from less than 1.0 to greater than 1.0, or vice versa.

Effects Assessment

Uncertainties associated with available toxicity benchmarks for birds are discussed in Section A.6.3.1.2.

Risk Characterization

This section estimates risks using two types of data that are uncertain: 1) DDT data, which were affected by analytical interference from PCBs, and 2) PCB data for osprey eggs, which were estimated from biomagnification factors (BMFs) derived from the literature.

[90] In the exposure assessment, it is assumed that heron ingest 24% shiner surfperch, 24% English sole, 24% juvenile chinook salmon, and 24% Pacific staghorn sculpin.

Risk Estimates for DDTs

The same methods and TRVs used for estimating risks to spotted sandpiper from total DDTs (Section A.6.3.1.2) were used to estimate risks to osprey. The tissue and sediment exposure concentrations of total DDTs are presented in Table A.6-57.

Table A.6-57. Exposure concentrations of total DDTs

Ingested Media	Exposure Concentration (mg/kg dw)
Shiner surfperch	1.0
English sole	0.76
Juvenile chinook salmon	0.21
Intertidal sediment	0.29

dw – dry weight

The calculated ingested dose of total DDTs for osprey, based on the exposure concentrations presented in Table A.6-57, is 0.022 mg/kg bw/day. This concentration is lower than the selected NOAEL TRV of 0.056 mg/kg bw/day, indicating that there is very low risk to osprey from exposure to total DDTs.

Risk Estimates Using COPC Concentrations in Osprey Eggs

Risks to birds were estimated in this baseline ERA by calculating the daily ingested dose for each COPC individually and comparing exposure estimates for each bird ROC to TRVs. Alternatively, if chemical concentration data were available for bird eggs, these data could have been used to estimate risks to birds using TRVs derived from toxicity studies based on bird egg concentrations. No egg data were available for use in this ERA.[91] Therefore, PCB concentrations in osprey eggs[92] were estimated using a BMF, and risks were estimated. The PCB concentration in LDW fish was multiplied by this BMF[93] to estimate the PCB concentration in LDW osprey eggs.

Two studies provide BMFs for PCBs, relating PCB concentrations in fish to concentrations in bird eggs. One of the studies calculated BMFs for herring gulls feeding from Lake Ontario (Braune and Norstrom 1989), and the other calculated BMFs for osprey feeding from the Willamette River in Oregon (Henny et al. 2003). The data from the osprey study were used in this evaluation because osprey is one of the ROCs in this risk assessment. To estimate BMFs for osprey, Henny et al. (2003)

[91] Quality assurance/quality control information for existing data on PCB concentrations in great blue heron eggs collected from the LDW (Krausmann 2002a) was insufficient, precluding the use of these data in this baseline ERA. In addition, although PCBs have been analyzed in osprey eggs collected from nests along the LDW, these data have not yet been made available from USGS.

[92] This analysis was conducted only for osprey, and not for other bird ROCs, because BMFs for osprey from data collected in the Willamette River were available.

[93] The PCB BMF used in this section is the PCB concentration in bird eggs divided by the PCB concentration in fish assumed to be preyed upon by the birds.

collected 10 osprey eggs (one per nest) from various sites along more than 100 miles of the Willamette River. Twenty-five whole-body composite fish samples of three species were collected at 11 locations along the Willamette River. These species were collected because they comprised 91% of the biomass of the osprey's diet, as observed in prey remains collected beneath the osprey nests. Fish and eggs were analyzed for PCBs. The PCB BMF was calculated by Henny et al. (2003) using the geometric mean of total PCB concentrations in eggs and fish tissue.

The calculated PCB BMF relating concentrations in fish and osprey eggs was 11 on a wet-weight basis (Henny et al. 2003). The estimated concentration of PCBs in the fish in the diet of osprey feeding from the LDW was calculated using Equation A.5-2 and the data presented in Table A.6-58. A BMF of 11 and a PCB concentration of 2.0 mg/kg ww in the osprey's diet resulted in an estimated PCB concentration in osprey eggs of 22 mg/kg ww.

Table A.6-58. Estimated PCB concentration in osprey eggs

PCB Concentration in Prey (mg/kg ww)[a]			Fraction of Prey in Diet[b]			PCB Concentration in Diet (mg/kg ww)	BMF	Estimated PCB Concentration in Osprey Eggs (mg/kg ww)
Perch	Sole	Salmon	Perch	Sole	Salmon			
3.5	2.6	0.70	0.34	0.21	0.45	2.0	11	22

a Concentrations are UCLs calculated using methods presented in Attachment 11.

b Fraction of prey in osprey diet, as discussed in Section A.5.1.2.3.

BMF – biomagnification factor

PCB – polychlorinated biphenyl

ww – wet weight

Five studies that related PCB concentrations in bird eggs to adverse effects were available (Table A.6-59.[94] These studies exposed adult birds to individual PCB Aroclors or Aroclor mixtures in their diet and evaluated a range of reproductive effects. Three studies reported adverse effects on reproduction at PCB concentrations in eggs ranging from 5.6 mg/kg ww (resulting in reduced eggshell thickness in American kestrel) to 34.1 mg/kg ww (resulting in reduced reproductive success in American kestrel). The lowest LOAEL of 5.6 mg/kg ww was not selected as a TRV because this effect was not at a level at which shell breakage would be expected, as discussed in more detail in Section A.5.2.1.1. The next lowest LOAEL (16 mg/kg ww [Aroclor 1254]), which resulted in reduced hatching success in the second generation of ringed turtle-doves, was selected as the LOAEL TRV (Peakall et al. 1972; Peakall and Peakall 1973). The only NOAEL that was lower than the LOAEL was a concentration of 7.1 mg/kg ww (Aroclor 1248), which did not result in any reported

[94] Reproductive studies with chickens were not considered, as discussed in Section A.5.2.1.1.

reproductive effects in the screech owl (McLane and Hughes 1980). This egg concentration was selected as the NOAEL TRV.

Table A.6-59. Toxicity studies for PCBs in bird eggs

Chemical	Test Species	NOAEL (mg/kg ww egg)	LOAEL (mg/kg ww egg)	Exposure Duration	Effect	Source
Aroclor 1248	screech owl	**7.1**	na	two generations	no effect on eggshell thickness, egg production, hatching success, or fledging success	McLane and Hughes (1980)
Aroclor 1248	American kestrel	na	5.6	5.5 months	reduced eggshell weight and thickness	Lowe and Stendell (1991)
Aroclor 1254	ringed turtle-dove	na	**16**	two generations	reduced hatchability and embryo survival	Peakall et al. (1972); Peakall and Peakall (1973)
Aroclor 1254	mallard	23	na	~ 1 month	no effect on number of laying hens, time to first hatch, clutch size, egg fertility, egg hatchability, or duckling survival to 3 weeks	Custer and Heinz (1980)
Mixture of Aroclors 1248, 1254, and 1260	American kestrel	na	34.1	100 days until eggs hatched	reduced reproductive success of parents exposed *in ovo*	Fernie et al. (2000; 2001)

LOAEL – lowest-observed-adverse-effect level

na – not available

NOAEL – no-observed-adverse-effect level

ww – wet weight

Bold identifies the NOAEL and LOAEL selected as the TRVs.

The NOAEL- and LOAEL-based HQs were calculated using the estimated egg concentration and the NOAEL and LOAEL TRVs, resulting in HQs of 3.1 and 1.4, respectively (Table A.6-60). These HQs are higher than NOAEL- and LOAEL-based HQs of 0.65 and 0.23, respectively, for total PCBs calculated based on ingested doses (Section A.6.3.1.1).

Table A.6-60. HQs for PCBs estimated in osprey eggs

Estimated Egg Concentration (mg/kg ww)	NOAEL TRV (mg/kg ww egg)	LOAEL TRV (mg/kg ww egg)	NOAEL-Based HQ	LOAEL-Based HQ
22	7.1	16	**3.1**	**1.4**

HQ – hazard quotient

LOAEL – lowest-observed-adverse-effect level

NOAEL – no-observed-adverse-effect level

TRV – toxicity reference value

ww – wet weight

Bold identifies NOAEL-based HQs greater than 1.0 and LOAEL-based HQs greater than or equal to 1.0.

There are several uncertainties associated with the BMF approach. The BMF used in this assessment was calculated using a relatively small dataset (i.e., 10 eggs) from a large area in the Willamette River. In addition, as noted by Henny et al. (2003), some osprey may capture different fish species than those used to calculate the BMF because they are opportunistic feeders, and some may forage in nearby ponds or lakes nearby in addition to the Willamette River. There are also uncertainties in applying site-specific data from the Willamette River to the LDW, which contains different fish species and different PCB mixtures. It is not known if these uncertainties would underestimate or overestimate risk to osprey.

Summary of Uncertainties

Uncertainties in the problem formulation, the effects and exposure assessments, and risk characterization for osprey are summarized as follows:

- Uncertainties in incidental sediment ingestion and dietary composition are expected to have minimal or no effect on risk conclusions.
- The PCB TEQs were calculated using TEFs for individual PCB congeners; the derivation of some of these TEFs is uncertain, with unknown effects on risk conclusions. Risk estimates for wildlife based on TEQs calculated considering only dioxin-like PCB congeners are likely lower than the actual risk resulting from the cumulative exposure of wildlife to dioxins, furans, and dioxin-like PCB congeners in the LDW.
- Gaps in toxicity data resulted in uncertainty in risk estimates. Risks to spotted sandpiper from chromium are uncertain because the TRV was based on a single unpublished study with a survival endpoint. Risks are uncertain for PCB TEQs because the TRVs were based on a study using an acute weekly dose via IP injection, and TEFs were applied to dietary exposure estimates. Effects of gaps in toxicity data on risk estimates for lead, mercury, and PCBs (based on a total PCB approach) are unknown.
- The NOAEL-based HQ for osprey and total DDTs was less than 1.0, indicating very low risk.
- Estimates of risk to osprey from total PCBs using egg concentrations resulted in NOAEL- and LOAEL-based HQs greater than 1.0, whereas NOAEL-based HQs for total PCBs were less than 1.0 using an ingested dose approach. Risk estimates from the estimated PCB egg concentrations are considered highly uncertain.

A.6.3.3.3 Risk conclusions

Osprey was chosen as an ROC because of its known site use and diet and to represent other piscivorous birds. The results of the risk characterization for osprey are summarized in Table A.6-61.

Table A.6-61. Summary of risk characterization for osprey

COPC	NOAEL-Based HQ	LOAEL-Based HQ	Effect	Primary Uncertainty[a]
Chromium	0.11	< 0.1	reduced survival	high uncertainty in TRV because study was unpublished; risk may be underestimated because TRV was based on survival rather than a sublethal endpoint
Lead	< 0.1	< 0.1	reduced egg hatchability	medium uncertainty in TRV
Mercury	0.28	< 0.1	reduced growth (chronic exposure)	medium uncertainty
Total PCBs	0.65	0.23	reduced hatching success	medium uncertainty in TRV
PCB TEQs	**1.6**	0.16	reduced body weight, egg production, and survival of adults and embryos	high uncertainty in TRV because of high-level weekly exposures and IP injection; risk could be overestimated; derivation of TEFs is uncertain

[a] Level of uncertainty key:

Low = large dataset including chronic studies with species taxonomically similar to the ROC

Medium = moderately sized dataset including chronic studies

High = small dataset with only subchronic studies, unbounded NOAELs/LOAELs, or data with questionable data quality

COPC – chemical of potential concern

HQ – hazard quotient

IP – intraperitoneal

LOAEL – lowest-observed-adverse-effect level

NOAEL – no-observed-adverse-effect level

PCB – polychlorinated biphenyl

TEF – toxic equivalency factor

TEQ – toxic equivalent

TRV – toxicity reference value

Bold identifies NOAEL-based HQs greater than 1.0.

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. There is high uncertainty in the chromium TRV, but the LOAEL TRV would need to be lower by a factor of 50 for the LOAEL-based HQ to equal 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 high 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 those of juvenile chinook salmon, so is it possible that PCB TEQ HQs would have been lower if PCB congener data had been available for salmon. There is also high uncertainty in the PCB TEQ TRV because birds in the study were dosed via IP injection rather than via the diet and were a high-level weekly dose, which may result in an overestimate of risk.

Comparison of the BMF-estimated PCB concentration in osprey eggs, to the LOAEL TRV resulted in an HQ of 1.4. The risk based on this approach is highly uncertain, as discussed in Section A.6.3.3.2, resulting in overestimates or underestimates of risk. Thus, for PCBs, the ingested dose approach with its associated uncertainties indicated that risks to osprey were very low, while risks estimated using estimated concentrations in eggs indicated that risks were low with a high level of uncertainty.

A.6.3.4 River otter

This section present risk estimates, uncertainties, and risk conclusions for river otter.

A.6.3.4.1 Risk estimates

Five COPCs were evaluated for river otter: arsenic, cobalt, mercury, selenium, and PCBs (using both a total PCB and PCB TEQ approach). The only COPC with a LOAEL-based HQ greater than 1.0 was total PCBs, with a LOAEL-based HQ of 2.9. Mercury and PCB TEQs had NOAEL-based HQs of 2.8 and 4.5, respectively; the LOAEL-based HQs for these COPCs were less than 1.0 (Table A.6-62).

Table A.6-62. HQ calculations for river otter

ROC	COPC	Ingested Dose (mg/kg bw/day)	TRVs (mg/kg bw/day)		NOAEL-Based HQ	LOAEL-Based HQ
			NOAEL	LOAEL		
River otter	arsenic	0.32	2.6	5.4	0.12	< 0.1
	cobalt	0.016	0.10	1.0	0.16	< 0.1
	mercury	0.0048	0.0017	0.0084	**2.8**	0.57
	selenium	0.032	0.055	0.080	0.58	0.40
	Total PCBs	0.26	0.045	0.089	**5.8**	**2.9**
	PCB TEQs[a]	2.9×10^{-6}	6.5×10^{-7}	4.9×10^{-6}	**4.5**	0.59

[a] No PCB congener data were available for juvenile chinook salmon or mussels. When calculating risk estimates, the portion of prey ingestion that had been assigned to juvenile chinook salmon (22%) was divided proportionally among the remaining fish species, and mussel portion was added to the clam portion.

bw – body weight

COPC – chemical of potential concern

HQ – hazard quotient

LOAEL – lowest-observed-adverse-effect level

NOAEL – no-observed-adverse-effect level

PCB – polychlorinated biphenyl

ROC – receptor of concern

TEQ – toxic equivalent

TRV – toxicity reference value

Bold identifies NOAEL-based HQs greater than 1.0 and LOAEL-based HQs greater than or equal to 1.0.

Table A.6-63 presents the relative contribution of each of the prey categories to the ingested PCB dose for river otter. For both the total PCB and PCB TEQ approaches, the largest contributions were from shiner surfperch and English sole.

Table A.6-63. Contribution of each prey category to the ingested PCB dose for river otter

Prey Species	Contribution to Ingested Dose (%)	
	Total PCBs	PCB TEQs
Shiner surfperch	39.6	53.5
English sole	28.3	27.4
Juvenile chinook salmon	9.6	0[a]
Pacific staghorn sculpin	14.2	12.7
Crab	7.7	5.7
Clam	0.5	0.6
Mussel	0.03	0[a]

[a] No PCB congener data were available for juvenile chinook salmon or mussels. The portion of prey ingestion that had been assigned to juvenile chinook salmon was divided proportionally among the other fish species, and the mussels portion was added to the clam portion.

PCB – polychlorinated biphenyl

TEQ – toxic equivalent

A.6.3.4.2 Uncertainties

This section presents a discussion of uncertainty associated with the problem formulation, the exposure and effects assessments, and the risk characterization for river otter.

Problem Formulation

Primary uncertainties in the problem formulation for river otter are associated with ROC selection and the COPC screen, as discussed below.

ROC Selection

Uncertainties related to the representation of other mammals in the LDW by river otter were addressed in the Phase 1 ERA (Windward 2003b). In that analysis, it was concluded that river otters are likely to be more highly exposed to COPCs from the LDW than muskrats or raccoons. Muskrats feed on plants, which are not abundant in the LDW, and raccoons feed on a greater proportion of terrestrial prey. Therefore, risk estimates for river otter should provide conservative risk estimates for other mammal species (i.e., muskrat and raccoon).

COPC Screen

Of the chemicals screened out as COIs for river otter in the first step of the COPC screen (Section A.2.5.3), 29 chemicals were detected in sediments but were not analyzed in tissue samples (Table A.6-16). None of these chemicals are defined as bioaccumulative chemicals (EPA 2000a), so risks to mammals from exposure to these chemicals are assumed to be very low.

Eighty-six chemicals were identified in the problem formulation as COIs for mammals. Effects data for mammals were not available for 27 of the COIs. Of these COIs with no information on toxicity, 17 were individual PAHs. Risks to mammals from PAHs were evaluated using TRVs for four individual PAH compounds (1-methylnaphthalene, 2-methylnapthalene, benzo(a)pyrene, and naphthalene). The remaining COIs for which there were no TRVs for mammals included silver, two organotin compounds, four SVOCs, and three organochlorine pesticides. Risks to mammals from exposure to these COIs could not be evaluated.

Exposure Assessment

Uncertainties in the exposure assessment for river otter were associated with the following factors:

- Direct sediment contact
- Food and incidental sediment ingestion rates
- Dietary composition
- TEQ approach

Direct Sediment Contact

Risks to wildlife from direct contact with sediment are considered insignificant relative to risks from incidental sediment ingestion (EPA 2000b). However, the exclusion of this pathway adds a small amount of uncertainty to the risk estimate for river otter.

Food and Incidental Sediment Ingestion Rates

As discussed in the Phase 1 ERA (Windward 2003b), there is some uncertainty in the FIR for the river otter because it was calculated from an allometric equation for non-herbivorous placental mammals, rather than based on species-specific information. The FIR rate would need to increase by at least a factor of 2 to increase the NOAEL-based HQs that are less than 1.0 (i.e., arsenic, cobalt, and selenium) to greater than 1.0. This scenario is unlikely, so the uncertainty in the river otter's FIR is not expected to have an effect on risk conclusions.

To address uncertainties in the amount of sediment incidentally ingested by river otters while foraging, ingested doses of COPCs were calculated assuming the SIR was 10% of the FIR versus 2% assumed in Section 5.1.2.4. This conservative assumption would result in a slight increase of HQs; however, it would not change risk conclusions.

Dietary Composition

There are some uncertainties in the proportion of different types of fish species in the river otter's diet, which could affect exposure. To address this uncertainty, HQs for each COPC were calculated assuming that 75% of the river otter diet was composed of the fish species with the highest concentration of that COPC, with the remainder of the diet composed of equal proportions of the other fish species.[95] This assumption resulted in changes in HQs but did not result in a change of any HQ from less than 1.0 to greater than 1.0, or vice versa.

Toxic Equivalent Approach

WHO consensus TEF values from Van den Berg et al. (2006) were used to calculate TEQs for dioxin-like PCB congeners for mammals; these TEFs are presented in Attachment 3.

The TEFs for mammals were derived from a large number of studies, with priority given to *in vivo* toxicity data over *in vitro* data. Despite the numerous biological variables such as species, strain, sex, and age included in these studies, the TEF values for a given congener generally fall within a range of about an order of magnitude for mammals (Sanderson and Van den Berg 1999). It is not known if the uncertainties in these TEFs would overestimate or underestimate risk. It should also be noted that the TEQs calculated in this ERA were used only for assessing the toxicity of the dioxin-like PCB congeners and do not account for TEQ contributions from dioxin and furan congeners; dioxin and furan concentrations in tissue were not available for the LDW. Risk estimates for wildlife based on TEQs calculated considering only dioxin-like PCB congeners are likely lower than the actual risk resulting from the cumulative exposure of wildlife to dioxins, furans, and dioxin-like PCB congeners in the LDW.

Effects Assessment

Uncertainty associated with available toxicity benchmarks for mammals may affect risk estimates. The general uncertainties associated with toxicity studies are the same as those discussed in Section A.6.3.1.2 for spotted sandpiper. Specific uncertainties associated with toxicity studies for the mammalian COPCs are presented in Table A.6-64.

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[95] In the exposure assessment, it is assumed that river otter ingest 22% shiner surfperch, 22% English sole, 22% juvenile chinook salmon, and 22% Pacific staghorn sculpin.

Table A.6-64. Level of uncertainty associated with TRVs for mammals

COPC	Number of TRV Studies	Uncertainty in TRV[a]
Arsenic	1	medium; selected TRVs were based on a chronic growth study[b]
Cobalt	3	medium; selected TRVs were based on a subchronic growth endpoint
Mercury	3	medium; selected TRVs were based on a chronic growth endpoint
Selenium	4	medium; selected TRVs were based on a subchronic growth endpoint
Total PCBs	10	medium
PCB TEQs	7	medium; selected TRVs were based on a subchronic growth endpoint

[a] Level of uncertainty key:

Low = large dataset including chronic studies with species (e.g., mink) taxonomically similar to the ROC

Medium = moderately sized dataset including chronic studies

High = small dataset with only subchronic studies, unbounded NOAELs/LOAELs, or data with questionable data quality

[b] Although only one study was available, it was a 2-year study with good data quality; the level of uncertainty is considered medium.

COPC – chemical of potential concern

PCB – polychlorinated biphenyl

TRV – toxicity reference value

There are uncertainties associated with the arsenic, cobalt, selenium, and PCB TEQ TRVs because they were based on subchronic or chronic growth endpoints, which could under- or overestimate risk. Uncertainty in the mercury total PCB TRVs could result in either an overestimate or underestimate of risk.

Summary of Uncertainties

Uncertainties in the problem formulation, the effects and exposure assessments, and risk characterization are summarized as follows:

- Uncertainties in food and incidental sediment ingestion rates and dietary composition are expected to have minimal or no effect on risk conclusions.
- The PCB TEQs were calculated using TEFs for individual PCB congeners, and the derivation of some of these TEFs is uncertain, with unknown effects on risk conclusions. Risk estimates for wildlife based on TEQs calculated considering only dioxin-like PCB congeners are likely lower than the actual risk resulting from the cumulative exposure of wildlife to dioxins, furans, and dioxin-like PCB congeners in the LDW.
- Gaps in toxicity data resulted in uncertainty in risk estimates. Risks may be under- or overestimated for arsenic, cobalt, mercury, and selenium, and overestimated for PCB TEQs; the effect of uncertainties associated with the total PCB TRVs is not known.

A.6.3.4.3 Risk conclusions

Risks were characterized for river otter as the most highly exposed semi-aquatic mammal using the LDW. Results from the risk characterization for river otter are summarized in Table A.6-65.

Table A.6-65. Summary of risk characterization for river otter

COPC	NOAEL-Based HQ	LOAEL-Based HQ	Effect	Primary Uncertainty[a]
Arsenic	0.12	< 0.1	reduced body weight (chronic exposure)	medium uncertainty in TRV[b]
Cobalt	0.16	< 0.1	reduced body weight (subchronic exposure)	medium uncertainty in TRV risk could be under- or overestimated because of endpoint
Mercury	**2.8**	0.57	reduced growth (chronic exposure)	medium uncertainty in TRV
Selenium	0.58	0.40	reduced body weight (subchronic exposure)	medium uncertainty in TRV; risk could be under- or overestimated because of endpoint
Total PCBs	**5.8**	**2.9**	reduced offspring growth	medium uncertainty in TRV
PCB TEQs	**4.5**	0.59	reduced body weight (subchronic exposure)	medium uncertainty in TRV; risk could be under- or overestimated because of endpoint; derivation of TEFs

[a] Level of uncertainty key:

Low = large dataset including chronic studies with species taxonomically similar to the ROC (i.e., mink)

Medium = moderately sized dataset including chronic studies

High = small dataset with only subchronic studies, unbounded NOAELs/LOAELs, or data with questionable data quality

[b] Although only one study was available, it was a 2-year study with good data quality, so the level of uncertainty is considered medium

COPC – chemical of potential concern

HQ – hazard quotient

LOAEL – lowest-observed-adverse-effect level

NOAEL – no-observed-adverse-effect level

PCB – polychlorinated biphenyl

TEF – toxic equivalency factor

TEQ – toxic equivalent

TRV – toxicity reference value

Bold identifies NOAEL-based HQs greater than 1.0 and LOAEL-based HQs greater than or equal to 1.0.

The LOAEL- and NOAEL-based HQs for PCBs of 2.9 and 5.8, respectively, using the total PCB approach, indicate risk to river otters from PCB exposure. The PCB TRVs are based on an 18-month study with mink in which reduced growth of offspring was reported. The NOAEL-based HQ for PCBs of 4.5, using the PCB TEQ approach, indicates that there could be some risk to river otters from PCB exposure, although the LOAEL-based HQ was < 1.0 using this approach. The PCB TEQ TRVs are based on a 90-day study with guinea pigs in which reduced body weights were reported. There are some uncertainties associated with the PCB TRVs and with the TEF approach, but it is not known whether these uncertainties would underestimate or overestimate risk. The next highest NOAEL and LOAEL values for total PCBs were from a chronic study with mink, which resulted in no effects on offspring kit survival at a dose of 2.6×10^{-6}

mg/kg bw/day but decreased survival in kits at 3 weeks at a dose of 9.1×10^{-6} mg/kg bw/day (Hochstein et al. 2001). If the NOAEL and LOAEL from this study had been used to estimate risk, the LOAEL-based HQs would have been less than 1.0, and the NOAEL-based HQ would be 2.5. Based on this analysis, risks to river otter from PCBs are low, with some uncertainty.

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 is some uncertainty in the TRV, which was based on a chronic study of rats that reported effects on growth because the NOAEL was estimated from the LOAEL using an uncertainty factor of 5.0. The next highest NOAEL and LOAEL values were from a 93-day study with mink, which resulted in no effects on growth and survival at a dose of 0.16 mg/kg bw/day and effects on growth and survival at a dose of 0.25 mg/kg bw/day (Wobeser et al. 1976). If the NOAEL and LOAEL from this study had been used to estimate risk, the NOAEL- and LOAEL-based HQs would have been substantially less than 1.0. Based on this analysis, risks to river otter from mercury are low, with some uncertainty.

The NOAEL- and LOAEL-based HQs for river otter for arsenic, cobalt, and selenium were less than 1.0, indicating that there are very low risks to river otters from exposure to these COPCs in the LDW.

A.6.3.5 Harbor seal

This section present risk estimates, uncertainties, and risk conclusions for harbor seal.

A.6.3.5.1 Risk estimates

The COPCs evaluated for harbor seal were mercury and PCBs (using both a total PCB and PCB TEQ approach). The NOAEL- and LOAEL-based HQs for both COPCs were less than 1.0 (Table A.6-66).

Table A.6-66. HQ calculations for harbor seal

ROC	COPC	Ingested Dose (mg/kg bw/day)	TRVs (mg/kg bw/day) NOAEL	TRVs (mg/kg bw/day) LOAEL	NOAEL-Based HQ	LOAEL-Based HQ
Harbor seal	mercury	0.00033	0.0017	0.0084	0.19	< 0.1
	total PCBs	0.020	0.045	0.089	0.44	0.22
	PCB TEQs	2.3×10^{-7}	6.5×10^{-7}	4.9×10^{-6}	0.35	< 0.1

[a] No PCB congener data were available for juvenile chinook salmon. When calculating risk estimates, the portion of prey ingestion that had been assigned to juvenile chinook salmon (25%) was divided proportionally among the remaining prey categories.

bw – body weight
COPC – chemical of potential concern
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
NOAEL – no-observed-adverse-effect level
PCB – polychlorinated biphenyl
ROC – receptor of concern
TEQ – toxic equivalent
TRV – toxicity reference value

A.6.3.5.2 Uncertainties

This section presents a discussion of uncertainty associated with the problem formulation, the exposure and effects assessments, and the risk characterization for harbor seal.

Problem Formulation

Primary uncertainties in the problem formulation for harbor seal are associated with ROC selection and the COPC screen. Uncertainties in the COPC screen for mammals were discussed in Section A.6.3.4.2 for river otter.

Uncertainties related to the representation of other marine mammals in the LDW (i.e., sea lions or harbor porpoises) by harbor seal were addressed in the Phase 1 ERA (Windward 2003b). In that analysis, it was concluded that all three species are opportunistic feeders, primarily feeding on fish and some invertebrates, so harbor seals should have an exposure similar to that of sea lions and harbor porpoises. Furthermore, harbor seals and sea lions are expected to have similar exposure to LDW prey because LWG site usage for these two species is similar. LDW site use by harbor porpoise is unknown because data are not available.

Exposure Assessment

Uncertainties in the exposure assessment for harbor seal were associated with the following factors:

- Direct sediment contact
- Incidental sediment ingestion rate
- Dietary composition
- TEQ approach

The uncertainties associated with direct sediment contact and the TEQ approach are the same as those discussed in Section A.6.3.4.2 for river otter.

Incidental Sediment Ingestion Rate

To address uncertainties in the amount of sediment incidentally ingested by harbor seal while foraging, ingested doses of COPCs were calculated assuming the SIR was 10% of the FIR versus 2% assumed in Section 5.1.2.5. This conservative assumption would result in a slight increase of HQs (by an average of less than 0.1) but would not change risk conclusions.

Dietary Composition

There are some uncertainties in the proportions of different types of fish species in the harbor seal's diet, which could affect exposure. To address this uncertainty, HQs for each COPC were calculated assuming that 75% of the harbor seal diet was composed of the fish species with the highest concentration of that COPC, with the remainder of the diet composed of equal proportions of the other fish species.[96] This assumption resulted in changes in HQs but did not result in a change of any HQ from less than 1.0 to greater than 1.0, or vice versa.

Effects Assessment

Uncertainties associated with mammalian TRVs are summarized in Table A.6-64.

Summary of Uncertainties

Uncertainties in the problem formulation, the exposure and effects assessments, and the risk characterization are summarized as follows:

- Uncertainties in incidental sediment ingestion and dietary composition are expected to have minimal or no effect on risk conclusions.
- The PCB TEQs were calculated using TEFs for individual PCB congeners, and the derivation of some of these TEFs is uncertain, with unknown effects on risk estimates. Risk estimates for wildlife based on TEQs calculated considering only dioxin-like PCB congeners are likely lower than the actual risk resulting from the cumulative exposure of wildlife to dioxins, furans, and dioxin-like PCB congeners in the LDW.
- Gaps in toxicity data resulted in uncertainty in risk estimates. The effects of uncertainties in the TRVs are not known.

A.6.3.5.3 Risk conclusions

The harbor seal was chosen to represent marine mammals in the LDW. Results of the harbor seal risk characterization are summarized in Table A.6-67.

Table A.6-67. Summary of risk characterization for harbor seal

COPC	NOAEL-Based HQ	LOAEL-Based HQ	Effect	Primary Uncertainty[a]
Mercury	0.19	< 0.1	reduced growth (chronic exposure)	medium uncertainty in TRV
Total PCBs	0.44	0.22	reduced offspring growth	medium uncertainty in TRV
PCB TEQs	0.35	< 0.1	reduced body weight (subchronic exposure)	medium uncertainty in TRV; risk could be under- or overestimated because of endpoint; derivation of TEFs

[a] Level of uncertainty key:

Low = large dataset including chronic studies with species taxonomically similar to the ROC

Medium = moderately sized dataset including chronic studies

High = small dataset with only subchronic studies, unbounded NOAELs/LOAELs, or data with questionable data quality

[96] In the exposure assessment, it is assumed that harbor seal ingest 25% shiner surfperch, 25% English sole, 25% juvenile chinook salmon, and 25% Pacific staghorn sculpin.

COPC – chemical of potential concern
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
NOAEL – no-observed-adverse-effect level
PCB – polychlorinated biphenyl
TEF – toxic equivalency factor
TEQ – toxic equivalent
TRV – toxicity reference value

The NOAEL- and LOAEL-based HQs for harbor seal for mercury and PCBs were less than 1.0, indicating that there is very low risk to harbor seals from exposure to these COPCs in the LDW. The primary uncertainties associated with this conclusion are the general limitations in derivation of TRVs from laboratory toxicity tests.

A.6.3.6 Summary of risk conclusions for wildlife

In summary, results of the wildlife risk estimates and evaluation of associated uncertainties are as follows:

- **Spotted sandpiper.** Exposure estimates were greater than doses associated with adverse effects for chromium and lead in Area 2, copper and mercury in Area 3, and for vanadium in all three exposure areas. Risks from PCBs and DDT were low to very low, with some uncertainty. Risks from arsenic, cadmium, cobalt, nickel, selenium, and zinc were very low.
- **Great blue heron.** Risks to great blue heron from exposure to COPCs in the LDW were very low.
- **Osprey.** PCB risks to osprey were low, with high uncertainty associated with the TEQ approach and BMF calculations. Risks to osprey from other COPCs in the LDW were very low.
- **River otter.** PCBs pose a risk to otters in the LDW with exposure concentrations greater than those associated with adverse effects. Risk to river otter from exposure to mercury was low. There were very low risks to river otter from other COPCs.
- **Harbor seal.** Risks to harbor seals from exposure to COPCs in the LDW were very low.

A.7.0 Selection of Primary Ecological Risk Drivers/Indicator Hazardous Substances

This section presents the rationale for the identification of chemicals as risk drivers (CERCLA terminology) or indicator hazardous substances (MTCA terminology) based on estimated risks to ecological receptors.[97] The risk drivers from both this ERA and the HHRA will be the focus of remedial analyses in the FS.

In this ERA, ecological risks from chemicals were assessed consistent with CERCLA (EPA 1998) and SMS guidance (WAC 173-204). As a result, chemicals detected in sediment and tissue samples collected from the LDW were grouped as follows: 1) chemicals for which there is no cause for concern, 2) COPCs, and 3) COPCs with LOAEL-based HQs ≥ 1.0 or that exceed the SQS of the SMS. This section describes additional considerations and documents decisions about which of the chemicals in the third group are considered risk drivers for ecological receptors (Table A.7-1).

Table A.7-1. COPCs, COPCs that exceed relevant risk thresholds, and risk drivers

Designation	Chemical	
	Benthic Invertebrates	Crab, Fish, or Wildlife Receptors
COPCs	**SMS chemicals** – 41 COPCs including metals, PAHs, PCBs, phthalates, and other SVOCs based on detected exceedance of SQS at one or more locations **non-SMS chemicals** – TBT, nickel, total DDTs, total chlordane, cis-1,2-dichloroethene	**Crabs** – zinc and PCBs **Fish** – arsenic, cadmium, copper, vanadium, PCBs, TBT **Birds** – arsenic, cadmium, chromium, cobalt, copper, lead, mercury, nickel, selenium, vanadium, zinc, PCBs **Mammals** – arsenic, cobalt, mercury, selenium, PCBs
COPCs above risk thresholds[a]	**SMS chemicals** – 41 COPCs **non-SMS** chemicals – nickel, total DDTs, total chlordane	**Crabs** – PCBs **Fish** – cadmium, vanadium, PCBs **Birds** –chromium, copper, lead, mercury, vanadium, PCBs **Mammals** – PCBs
Risk drivers	**SMS chemicals** – 41 COPCs	**Mammals** – PCBs

[a] LOAEL-based HQ ≥ 1.0 or exceedance of the SQS of the SMS.

CERCLA – Comprehensive Environmental Response, Compensation, and Liability Act

COPCs – chemicals of potential concern

MTCA – Model Toxics Control Act

PAHs – polycyclic aromatic hydrocarbons

PCBs – polychlorinated biphenyls

SMS – Washington State Sediment Management Standards

SQS – sediment quality standard

SVOCs – semivolatile organic compounds

TBT – tributyltin

[97] For simplicity, these chemicals will be referred to as risk drivers in the remainder of this section.

A.7.1 CONSIDERATIONS FOR IDENTIFICATION OF RISK DRIVERS

In the problem formulation of this ERA, all analyzed chemicals in the LDW were screened to identify COPCs by comparing maximum exposure concentrations either to the SQS or to NOAEL TRVs (Section A.2.5). The resulting COPCs were further evaluated to derive risk estimates based on more realistic exposure assumptions and a range of toxicity data (i.e., from the highest NOAEL TRV to the lowest LOAEL TRV). Uncertainties in risk estimates were discussed in the risk characterization. In this section, the following additional factors are considered to identify which COPCs that exceeded effects thresholds are risk drivers. Considerations include:

- Uncertainty in risk estimates based on quantity and quality of exposure data
- Uncertainty in risk estimates based on quantity and quality of effects data
- Comparison of concentrations in LDW sediment to regional natural background concentrations in sediment
- The likely magnitude of residual risks following planned sediment remediation within early action areas in the LDW

Risk drivers will be the focus of detailed analyses presented in the FS for all remedial alternatives. COPCs that exceeded risk thresholds, also referred to as chemicals of concern (COCs), that were not selected as risk drivers may also be further evaluated, in consultation with EPA and Ecology. This evaluation may include:

- Assessment of the likely reductions in sediment concentrations or residual risks from these COCs following implementation of the preferred alternative selected in the FS
- Review of any new effects or exposure information as part of the 5-year review
- Inclusion of COCs in any post-remedial monitoring program

A.7.2 IDENTIFICATION OF RISK DRIVER CHEMICALS

Based on the four considerations outlined above, and in consultation with EPA and Ecology, 41 chemicals were selected as risk drivers for benthic invertebrates. PCBs were the only chemical identified as a risk driver for ecological receptors other than benthic invertebrates. The rationale for the selection of these risk drivers is summarized in Table A.7-2 and described briefly below.

Table A.7-2. Rationale for risk driver designation

COPC	ROC	Maximum NOAEL-Based HQ	Maximum LOAEL-Based HQ	Additional Considerations	Risk Driver?
Total PCBs	crabs	**10**	**1.0**	Uncertainty in exposure data: whole-body concentrations were estimated Uncertainty in effects data: LOAEL-based HQ was based on a study with Aroclor 1016 and grass shrimp, and NOAEL was estimated using an uncertainty factor; selection of next higher TRV would result in LOAEL-based HQ < 1.0	no
	river otter	**5.8**	**2.9**	Uncertainty in exposure data: low uncertainty in diet assumptions and home range Uncertainty in effects data: low uncertainty in TRV (growth endpoint in kits)	yes
	English sole	**4.9 – 25**[a]	0.98 – **5.0**[a]	Uncertainty in exposure data: low uncertainty in tissue concentrations Uncertainty in effects data: high uncertainty in lowest LOAEL TRV because of uncertain statistical significance of the fecundity endpoint for the low dose, a lack of dose-response in the fecundity endpoint, uncertain number of fish used in the experiment, and uncertainties associated with fish handling and maintenance protocols	no
	Pacific staghorn sculpin	**3.8 – 19**[a]	0.76 - **3.8**[a]	Same considerations as listed above for English sole	no
PCB TEQ[b]	spotted sandpiper – Area 2 (high-quality foraging habitat)	**15**	**1.5**	Uncertainty in exposure data: low uncertainty in diet assumptions and home range Uncertainty in effects data: high uncertainty in TRV, which was based on study of reproduction with weekly IP injection; high uncertainty in TEFs; effects data for total PCBs are less uncertain than for PCB TEQs and the LOAEL-based HQ for total PCBs was < 1.0	no
Cadmium	juvenile chinook salmon	**5.0**	**1.0**	Uncertainty in exposure data: LOAEL-based HQ < 1.0 if empirical juvenile chinook salmon stomach contents data from the LDW are used to estimate exposure, instead of estimating exposure based on ingestion of benthic invertebrates Uncertainty in effects data: high uncertainty in the lowest TRV because selection of next higher TRV would result in LOAEL-based HQ < 1.0, all salmonid-specific studies for cadmium with NOAELs result in NOAEL-based HQs less than 0.01	no
	English sole	**6.1**	**1.2**	Uncertainty in exposure data: low uncertainty (LDW-collected benthic invertebrate tissue samples) Uncertainty in effects data: high uncertainty in the lowest TRV because selection of next higher TRV would result in LOAEL-based HQ < 1.0; all other NOAELs and LOAELs were orders of magnitude higher than the selected LOAEL	no

Table A-7.2. Rationale for risk driver designation, cont.

COPC	ROC	Maximum NOAEL-Based HQ	Maximum LOAEL-Based HQ	Additional Considerations	Risk Driver?
	Pacific staghorn sculpin	**5.2**	**1.0**	Uncertainty in exposure data: low uncertainty (LDW-collected shiner surfperch and benthic invertebrate tissue samples) Uncertainty in effects data: high uncertainty in the lowest TRV because selection of next higher TRV would result in LOAEL-based HQ < 1.0; all other NOAELs and LOAELs were orders of magnitude higher than the selected LOAEL	no
Chromium	spotted sandpiper –Area 2 (high- and poor-quality foraging habitat)	**8.8**	**1.8**	Uncertainty in exposure data: high uncertainty because LOAEL-based HQ would be less than 1.0 if the single anomalously high benthic invertebrate tissue sample from RM 3.0 west was excluded; chromium concentrations in sediment were low in this area Uncertainty in effects data: high uncertainty; only one study with reported effects, and study was unpublished and could not be obtained for review	no
Copper	spotted sandpiper –Area 3 (high- and poor-quality foraging habitat)	**1.5**	**1.1**	Uncertainty in exposure data: low uncertainty Comparison to natural background: concentration in sediment (SWAC of 57 mg/kg dw) from Area 3 (high- and poor-quality foraging habitat) similar to PSAMP rural Puget Sound concentrations (50 mg/kg dw [90th percentile]) Residual risk: following planned sediment remediation within early action areas, LOAEL-based HQ would be < 1.0	no
Lead	spotted sandpiper –Area 2 (high- and poor-quality foraging habitat)	**19**	**5.5**	Uncertainty in exposure data: high uncertainty because LOAEL-based HQ would be less than 1.0 if the single anomalously high benthic invertebrate tissue sample from RM 3.0 west was excluded; lead concentrations in sediment were low in this area Uncertainty in effects data: low uncertainty (reproductive endpoint)	no
	spotted sandpiper –Area 3 (high- and poor-quality foraging habitat)	**5.0**	**1.5**	Uncertainty in exposure data: low uncertainty Uncertainty in effects data: low uncertainty (reproductive endpoint) Residual risk: following planned sediment remediation within early action area, LOAEL-based HQ would be < 1.0	
Mercury	spotted sandpiper –Area 3 (high- quality foraging habitat)	**5.3**	**1.0**	Uncertainty in exposure data: low uncertainty Uncertainty in effects data: low uncertainty (TRV was based on a growth endpoint) Residual risk: following planned sediment remediation within early action area, LOAEL-based HQ would be < 1.0	no

Table A-7.2. Rationale for risk driver designation, cont.

COPC	ROC	Maximum NOAEL-Based HQ	Maximum LOAEL-Based HQ	Additional Considerations	Risk Driver?
Vanadium	English sole	**5.9**	**1.2**	Uncertainty in exposure data: low uncertainty Uncertainty in effects data: high uncertainty in TRV because only one study was available Comparison to natural background: exposure concentration in LDW sediment (SWAC of 58 mg/kg dw) was less than PSAMP rural Puget Sound concentration (64 mg/kg dw [90th percentile])	no
	Pacific staghorn sculpin	**3.2 – 5.9**	0.65 **– 1.2**	Same considerations as listed for English sole above	no
	spotted sandpiper – all exposure areas	**2.0 – 2.7**	**1.0 – 1.4**	Uncertainty in exposure data: low uncertainty Uncertainty in effects data: TRV was based on a 4-week growth endpoint, with uncertainty (two available studies: one with reduced body weight in chickens after 4 weeks and the other with no effect on body weight in mallards after 10 weeks) Comparison to natural background: mean exposure concentrations in sandpiper exposure areas ranged from 49 to 57 mg/kg dw, compared to PSAMP rural Puget Sound background concentration of 64 mg/kg dw (90th percentile)	no
41 SMS chemicals[c]	benthic invertebrates	range of values	range of values	Each of these 41 chemicals had at least one detected exceedance of SQS in baseline surface sediment dataset.	yes
Nickel	benthic invertebrates	**6.6**	**2.5**	Uncertainty in exposure data: low uncertainty Uncertainty in effects data: medium uncertainty in the TRV (i.e., the ML) because only no-effects data (amphipod mortality and community abundance AETs) were available; no information was available regarding concentrations associated with adverse effects Residual risk: ML was exceeded at four locations in LDW – all within early action areas with planned sediment remediation	no
Total DDTs	benthic invertebrates	**5.1**	**2.7**	Uncertainty in exposure data: medium uncertainty (i.e., likely interference in pesticide analyses from PCBs) Uncertainty in effects data: medium uncertainty; based on a single study with spiked sediment Residual risk: LOAEL was exceeded at only one location in LDW, location is within early action area with planned sediment remediation	no

Table A-7.2. Rationale for risk driver designation, cont.

COPC	ROC	Maximum NOAEL-Based HQ	Maximum LOAEL-Based HQ	Additional Considerations	Risk Driver?
Total chlordane	benthic invertebrates	**82**	**48**	Uncertainty in exposure data: highly uncertain because all total chlordane concentrations in samples from Phase 2 locations were JN-qualified as a result of probable PCB interference; except one location at RM 2.2, all locations with detected total chlordane concentrations co-occurred with elevated PCB concentrations Uncertainty in effects data: TRV is highly uncertain because it was based on a general Canadian sediment guideline (PEL); this guideline is based mainly on field-collected data with complex mixtures of chemicals Residual risk: LOAEL was exceeded at 14 locations in LDW; all but one of these locations are associated with an early action area with planned sediment remediation	no

Note: HQs for fish are the highest HQs in cases where more than one approach was used.

[a] 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.

[b] Risk estimates based on TEQs were calculated using only tissue data for dioxin-like PCB congeners because dioxin and furan tissue data were not available. Thus, risks associated with exposure to all dioxin-like chemicals were likely underestimated; the degree of underestimation is uncertain.

[c] Arsenic, 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, phenol, total PCBs.

dw – dry weight
ERA – ecological risk assessment
HQ – hazard quotient
IP – intraperitoneal
LDW – Lower Duwamish Waterway
LOAEL – lowest-observed-adverse-effect level
ML – maximum level
NOAEL – no-observed-adverse-effect level
PCB – polychlorinated biphenyl
PEL – probable effects level
RM – river mile
ROC – receptor of concern
SQS – sediment quality standard
SWAC – spatially weighted average concentration
TEQ – toxic equivalent
TEF – toxic equivalency factor
TRV – toxicity reference value

Bold identifies NOAEL-based HQ greater than 1.0 and LOAEL-based HQ greater than or equal to 1.0.

The 41 chemicals identified as risk drivers for benthic invertebrates were selected because the detected concentration of those chemicals in the LDW baseline surface sediment dataset exceeded the SQS of the SMS at one or more locations, and SMS is a key regulation governing sediment remediation in the State of Washington. Chemicals were selected as risk drivers regardless of site-specific toxicity testing results because of the uncertainty in the cause-and-effect relationship on a chemical-specific basis.

Five chemicals (hexachlorobutadiene, 2-methylphenol, acenaphthylene, di-n-octyl phthalate, and diethyl phthalate) had RLs that were greater than the SQS but did not have detected concentrations greater than the SQS. These five chemicals were not identified as risk drivers for the reasons discussed below.

Hexachlorobutadiene was not selected as a risk driver because although some of the RLs for hexachlorobutadiene were above the SQS, hexachlorobutadiene was never detected in the 782 samples that were analyzed for this compound (Table A.7-3). The improved analytical techniques used to develop the RI/FS dataset, relative to earlier studies, reduced the frequency of RLs above the SQS from 23% (non-LDWG data) to 8% (LDWG data) with no increase in detection frequency. The chemical 2-methylphenol was rarely detected (0.4% detection frequency). The frequency of RLs for 2-methylphenol above the SQS was reduced from 17% (non-LDWG data) to 8.3% (LDWG data), with no detected concentrations above the SQS. The RLs above the SQS for these compounds appear to reflect difficulties routinely encountered with the analysis of these compounds, and thus these chemicals were not selected as risk drivers.

Table A.7-3. Chemicals with RLs greater than SQS that were not identified as risk drivers

Chemical	Number of Samples Analyzed	% Detected	% RLs Above SQS
Hexachlorobutadiene	782	0%	18% (148 samples)
2-Methyl phenol	785	0.4%	15% (117 samples)
Acenaphthylene	782	16%	0.4% (3 samples)
Di-n-octyl phthalate	796	6%	1.0% (8 samples)
Diethyl phthalate	796	0.12%	0.7% (6 samples)

RL – reporting limits

SQS – sediment quality standards (SMS)

Acenaphthylene, di-n-octyl phthalate, and diethyl phthalate were detected in 16%, 6%, and 0.12%, respectively, of the samples analyzed for these compounds (Table A.7-3); none of the detected concentrations were greater than the SQS. The frequencies of RLs above the SQS for acenaphthylene, di-n-octyl phthalate, and diethyl phthalate were all very low (Table A.7-3). These chemicals were not selected as risk drivers because they were never detected above the SQS, and the frequencies of RLs above the SQS in the LDWG data were very low. The RLs above the SQS were generally associated with issues of sample dilution.

Nickel, total DDTs, and total chlordane do not have SQS values but did have LOAEL-based HQs for benthic invertebrates greater than 1.0 based on other guidelines or TRVs. However, these three chemicals were not selected as risk drivers for benthic invertebrates primarily because of uncertainties in their TRVs and because areas with concentrations greater than those TRVs were all within planned sediment remediation areas, except for one location with elevated total chlordane at RM 0.85 in the navigation channel. This sample was JN-qualified, indicating that the chlordane identification was uncertain and the concentration was likely biased high because of analytical interference with PCBs.

PCBs were selected as a risk driver for river otter because the LOAEL-based HQ exceeded 1.0 (HQ of 2.9), and the uncertainties associated with the exposure and effects data were relatively low.

COPCs not selected as risk drivers for ecological receptors other than benthic invertebrates but with LOAEL-based HQs greater than or equal to 1.0, included PCBs (crabs, English sole, Pacific staghorn sculpin, and spotted sandpiper), cadmium (juvenile chinook salmon, English sole, and Pacific staghorn sculpin), chromium (spotted sandpiper), copper (spotted sandpiper), lead (spotted sandpiper), mercury (spotted sandpiper), and vanadium (juvenile chinook salmon, English sole, Pacific staghorn sculpin, and spotted sandpiper).

For crabs, PCBs were not selected as a risk driver primarily because of the uncertainty in the effects data and the low risk estimate (LOAEL-based HQ equal to 1.0). The LOAEL-based HQ was based on a study with Aroclor 1016 and grass shrimp and the NOAEL was estimated using an uncertainty factor. The LOAEL-based HQ would have been less than 1.0 if the next highest TRV (based on Aroclor 1254 and grass shrimp) had been used instead.

For English sole and Pacific staghorn sculpin, PCBs were not selected as a risk driver. The risk estimates for PCBs in these two fish species are uncertain both because the exposure concentrations were in between the concentrations selected as the LOAEL range and because of uncertainty in the study that served as the basis for the LOAEL range itself.

For spotted sandpiper, the LOAEL-based HQ for PCB TEQ was 1.5. However, PCBs were not selected as a risk driver for spotted sandpiper primarily because LOAEL-based HQs for total PCBs were less than 1.0 and risk estimates for total PCBs are more certain than risk estimates for PCB TEQs.

Cadmium was not selected as a risk driver for juvenile chinook salmon, English sole, or for Pacific staghorn sculpin primarily because the selected TRV was highly uncertain (i.e., the selected LOAEL was orders of magnitude lower than NOAELs or LOAELs from all other studies and the observed effects were partially attributed to reduced food intake), and risk estimates were low (LOAEL-based HQs of 1.2 for both juvenile chinook salmon and English sole and up to 1.0 for Pacific staghorn sculpin).

Chromium and lead were not selected as risk drivers for spotted sandpiper because elevated risks within Area 2 were driven by elevated concentrations in a single benthic invertebrate tissue sample collected near RM 3.0 on the west side of the LDW. Dry weight concentrations of chromium and lead were much higher in this sample than in the rest of the benthic invertebrate tissue dataset, whereas concentrations of chromium and lead were not elevated in the sediment sample co-located with this benthic invertebrate composite tissue sample.[98]

Copper was not selected as a risk driver for spotted sandpiper primarily because sediment concentrations of copper in Area 3 (57 mg/kg dw) are similar to the 90th percentile copper concentration (50 mg/kg dw) in PSAMP rural Puget Sound sediment and because LOAEL-based HQs will be less than 1.0 following planned sediment remediation within early action areas. Lead (Area 3) and mercury also were not selected for spotted sandpiper primarily because LOAEL-based HQs will be less than 1.0 following planned sediment remediation within early action areas.

Vanadium was not selected as a risk driver for either fish or spotted sandpiper because of high uncertainty in effects data (few toxicity studies) and because sediment concentrations of vanadium in exposure areas (ranging from 49 to 58 mg/kg dw) are less than the 90th percentile vanadium concentration (64 mg/kg dw) in PSAMP rural Puget Sound sediment. Background information is important because the CERCLA program generally does not require clean up to concentrations below natural or anthropogenic background levels (EPA 2002).

No quantitative ecological risk estimates were calculated for dioxins and furans within the LDW and thus the level of ecological risk from dioxins and furans is not known. Risks were not calculated for several reasons. Primarily, human health risks from dioxins and furans through seafood consumption were assumed to be unacceptable; and therefore, neither tissues from the LDW nor from background areas were analyzed for dioxins and furans. Dioxins and furans were determined to be a risk driver based on human health risks from both seafood consumption and direct contact pathways. Risk management decisions to address dioxin and furan contamination in LDW sediment will be based on MTCA and CERCLA regulations and guidance. Remedial decisions to address dioxin and furan contamination in sediment will be made by EPA and Ecology as part of the FS process and will be documented in the Record of Decision. Additional detail on dioxins and furans is provided in Section B.5.5.2 of the HHRA.

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[98] Of the eight benthic invertebrate tissue samples analyzed in Area 2, the concentration of chromium in sample B7a was 58 mg/kg dw, compared to an average concentration of 8.7 mg/kg dw in the seven other samples. The concentration of lead in sample B7a was 220 mg/kg dw, compared to an average concentration of 11 mg/kg dw in the seven other samples. The chromium and lead concentrations in the co-located sediment sample were 22.9 mg/kg dw and 21.4 mg/kdw dw, respectively (the 90th percentile chromium and lead concentrations in PSAMP rural Puget Sound sediment were 58.8 mg/kg dw and 19.5 mg/kg dw, respectively).

A.8.0 Conclusions

Baseline risks were calculated in this ERA based on chemical concentrations in sediment, water, and tissue samples collected from the LDW to estimate the chemical exposure of benthic invertebrates, crabs, fish, birds, and mammals that may reside or forage in the LDW for at least a portion of their lives. Site-specific sediment toxicity tests and a site-specific investigation of imposex in gastropods were also conducted. Several conservative assumptions were employed in the risk assessment to compensate for a variety of uncertainties. Based on risk estimates and associated uncertainties, the main conclusions regarding risks to ecological receptors in the LDW from COPCs are as follows:

- **Benthic Invertebrate Community —** Sediment chemistry and site-specific toxicity test results indicate that no adverse effects to benthic invertebrates living in intertidal and subtidal sediments are predicted for 75% of the LDW area (i.e., the area in which chemical concentrations were less than or equal to SQS chemical criteria and where sediments were nontoxic according to SQS biological effects criteria). 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, 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. Some uncertainty is associated with these area estimates because areas were estimated by interpolating from individual points at which sediments were sampled. The SQS and CSL were exceeded by 39 chemicals; two additional chemicals exceeded only the SQS.[99] Risks to the benthic invertebrate community from all VOCs detected in sediment porewater were very low, except for cis-1,2-dichloroethene, which had concentrations 21 times the no-effects concentration in a small area at RM 2.4; all concentrations of cis-1,2-dichloroethene were less than the concentration associated with adverse effects. Therefore, there is uncertainty whether exposure to cis-1,2-dichloroethene within the LDW is sufficiently high to result in adverse effects in this small area. Risks to benthic invertebrates from TBT were very low based on NOAEL-based HQs less than 1.0 and the absence of imposex in all gastropods, except one species of neogastropod with imposex characterized as Stage 2, a stage that is not expected to impact reproduction.
- **Crabs—**Exposure concentrations of total PCBs in tissue were equal to concentrations associated with adverse effects in crabs, indicating the potential

[99] Total DDTs, nickel, and total chlordane also exceeded their DMMP guidelines or literature-based TRV at one or more locations.

for adverse effects. Exposure concentrations of zinc in tissue were greater than concentrations associated with no effects but less than those associated with adverse effects, indicating there is uncertainty whether exposure within the LDW is sufficiently high to result in adverse effects.

- **Fish —** Risks were evaluated for three fish ROCs (English sole, Pacific staghorn sculpin, and juvenile chinook salmon), which serve as representative surrogates for all other fish species in the LDW. Exposure concentrations for three of the six COPCs for fish (PCBs, cadmium, and vanadium) were greater than concentrations associated with adverse effects for English sole. LOAEL-based HQs for both metals were 1.2, and LOAEL-based HQs for PCBs ranged from 0.98 to 5.0 based on effects concentrations in the study reporting the lowest TRVs. Therefore, there is a potential for adverse effects to fish from PCBs, but risk estimates are uncertain because the exposure concentrations were in between the concentrations selected as the LOAEL range and because the study that served as the basis for the LOAEL range was uncertain. Estimated exposures of English sole to two additional COPCs (arsenic and copper) were greater than their respective no-effects levels but were lower than the adverse effect levels associated with survival, growth, or reproduction, indicating that there is uncertainty whether exposure within the LDW is sufficiently high to result in adverse effects. Site-specific studies of English sole indicate the potential for reproductive effects that correlate with exposure to chemical mixtures in the field. Exposure concentrations of PCBs, cadmium, and vanadium for Pacific staghorn sculpin were greater than or equal to the concentrations associated with adverse effects in at least one area within the LDW. LOAEL-based HQs of up to 1.0 and 1.2 were estimated for cadmium and vanadium, respectively, indicating a potential for adverse effects. LOAEL-based HQs for PCBs ranged from 0.30 to 3.8 based on effects concentrations in the study reporting the lowest TRVs. Therefore, there is a potential for adverse effects from PCBs, but risk estimates are uncertain because the exposure concentrations were in between the concentrations selected as the LOAEL range. The exposure concentrations of TBT and copper were greater than their respective no-effects concentrations for Pacific staghorn sculpin in at least one area within the LDW but less than those associated with adverse effects. Thus, the potential for adverse effects is uncertain. For juvenile chinook salmon, exposure concentrations of cadmium were greater than concentration associated with adverse effects in any fish species but lower than concentrations associated with adverse effects in salmonids. Exposure concentrations of arsenic, copper, and vanadium in the diet of for juvenile chinook salmon were greater than their respective no-effect concentrations but less than concentrations associated with adverse effects.

- **Birds** — Risks were evaluated for three bird ROCs (spotted sandpiper, great blue heron, and osprey), which serve as representative surrogates for all other bird species that may be exposed in the LDW. Estimated exposures of spotted sandpiper to six of the 12 COPCs (copper, chromium, lead, mercury, PCB TEQ, and vanadium) for spotted sandpiper were greater than the dietary doses associated with adverse effects on survival, growth, or reproduction in at least one area within the LDW (LOAEL-based HQs of up to 1.1, 1.8, 5.5, 1.0, 1.5, and 1.4, respectively). Therefore, there is a potential for adverse effects from these COPCs. Estimated doses to great blue heron of all four COPCs (chromium, lead, mercury, and total PCBs) were less than no-effects levels, indicating very low risk. For osprey, estimated doses of PCBs were greater than no-effect levels using a TEQ approach but less than those levels using a total PCBs approach; the latter risk estimate is less uncertain. Therefore, the potential for adverse effects from PCBs is uncertain for osprey. Estimated doses of the remaining three COPCs to osprey (chromium, lead, and mercury) were less than the doses associated with no-effects, indicating very low risk.

- **Mammals —** Risks were evaluated for two mammalian ROCs (river otters and harbor seals), which serve as representative surrogates for all other mammal species that may be exposed in the LDW. Estimated dietary doses of total PCBs were greater than doses associated with adverse effects for river otters, with a LOAEL-based HQ of 2.9. Estimated exposure of river otters to mercury was greater than a no-effects level but was less than adverse effects levels associated with survival, growth, or reproduction, indicating that the potential for effects is uncertain. Exposures of otter to the remaining three COPCs (arsenic, cobalt, and selenium) and exposures of harbor seals to both COPCs (mercury and total PCBs) were less than their respective no-effects levels, indicating very low risk.

Table A.8-1 provides a summary of COPCs for crabs, fish, or wildlife for which either the NOAEL-based HQ was > 1.0 or the LOAEL-based HQ was ≥ 1.0. The 44 COPCs for benthic invertebrates that exceeded SMS criteria, DMMP guidelines, or TRVs are listed in Table A.6-4. In summary, risk estimates for PCBs indicated a potential for adverse effects to the benthic invertebrate community, crabs, spotted sandpiper, and river otter. There is also a potential for adverse effects to English sole, Pacific staghorn sculpin, and osprey from PCBs, but risk estimates for these ROCs are more uncertain because exposures were greater than no-effect levels but less than levels associated with adverse effects. 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 shown in Table A.6-4.

Table A.8-1. COPCs and ROCs with HQs ≥ 1.0

COPC	ROC	NOAEL-Based HQ	LOAEL-Based HQ
COPCs with LOAEL-Based HQs ≥ 1.0[a]			
Total PCBs	crabs	**10**	**1.0**
	river otter	**5.8**	**2.9**
	English sole	**4.9** – **25**	0.98 – **5.0**
	Pacific staghorn sculpin	**1.5** – **19**	0.30 – **3.8**
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[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
	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

COPC	ROC	NOAEL-Based HQ	LOAEL-Based HQ
Zinc	crabs	**2.5**	0.91

Note: HQs 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 pesticide analyses, resulting in uncertainties in pesticide identification and a high bias in pesticide concentrations.

[b] The NOAEL-based HQs were > 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 crab s(3.6). These calculations were discussed in the uncertainty analysis because of analytical interferences from PCB Aroclors in the 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.

Based on the risk estimates, uncertainties discussed in this ERA, natural background concentrations, and residual risks following planned early actions in the LDW, chemicals were identified as risk drivers for ecological receptors in accordance with EPA (1998) and MTCA (WAC 173-340-703) guidance. The risk drivers from both this 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 benthic invertebrates because concentrations of these 41 chemicals exceed 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.

No quantitative ecological risk estimates were calculated for dioxins and furans within the LDW, and thus the level of ecological risk from dioxins and furans is unknown. Ecological risks associated with exposure to dioxins and furans within the LDW were not assessed for several reasons. Human health risks from these chemicals were assumed to be unacceptable,[100] and tissues were not analyzed for dioxins and furans. Tissues were not analyzed because of difficulties associated with assessing site-specific

[100] Dioxins and furans are a primary risk driver for human health.

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. Remedial decisions to address dioxins/furan contamination in LDW sediment will be based on MTCA and CERCLA regulations, including those specifically related to background (Ecology 2001; EPA 2002). Remedial decisions to address dioxin and furan contamination in sediment will be made by EPA and Ecology as part of the FS process.

This ERA is based on the baseline surface sediment dataset, which includes 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 storm drain in the Norfolk area). Therefore, the risks discussed in this ERA may represent an overestimate of current risks for areas where remediation has already occurred.

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