EAST WATERWAY OPERABLE UNIT
SUPPLEMENTAL REMEDIAL INVESTIGATION/ FEASIBILITY STUDY
APPENDIX A: BASELINE ECOLOGICAL RISK ASSESSMENT
FINAL

For submittal to:

The US Environmental Protection Agency
Region 10
Seattle, WA

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Acronym	Definition
ACR	acute-to-chronic ratio
AET	apparent effects threshold
Ah	aryl hydrocarbon
aRPD	apparent redox potential discontinuity
AWQC	ambient water quality criteria
BCA	bias-corrected accelerated
BCMOE	British Columbia Ministry of Environment
BHC	benzene hexachloride
bw	body weight
CERCLA	Comprehensive Environmental Response, Compensation, and Liability Act
COC	contaminant of concern
COI	chemical of interest
COPC	contaminant of potential concern
CPUE	catch-per-unit-effort
CSM	conceptual site model
CSL	cleanup screening level
CSO	combined sewer overflow
CYP1A	cytochrome P450 1A
DDD	dichlorodiphenyldichloroethane
DDE	dichlorodiphenyldichloroethylene
DDT	dichlorodiphenyltrichloroethane
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
EF	exceedance factor
EISR	existing information summary report
EPA	US Environmental Protection Agency
EPC	exposure point concentration
ERA	ecological risk assessment
ERED	Environmental Residue Effects Database
EROD	ethoxyresorufin-O-deethylase
ESA	Endangered Species Act
ESL	ecological screening level
EW	East Waterway

EWG	East Waterway Group
FAV	final mean acute value
FCV	final chronic value
FMR	free-living metabolic rate
FS	feasibility study
GMAV	genus mean acute value
ha	hectare
HHRA	human health risk assessment
HPAH	high-molecular-weight polycyclic aromatic hydrocarbon
HQ	hazard quotient
ID	identification
IP	intraperitoneal
IRI	index of relative importance
IRIS	Integrated Risk Information System
J-qualifier	estimated concentration
kcal	kilocalories
KM	Kaplan-Meier
LAET	lowest apparent effects threshold
2LAET	second lowest apparent effects threshold
LC50	concentration that is lethal to 50% of an exposed population
LDW	Lower Duwamish Waterway
LOAEL	lowest-observed-adverse-effect level
LOEC	lowest-observed-effect concentration
LPAH	low-molecular-weight polycyclic aromatic hydrocarbon
ME	metabolizable energy
MIS	multi-increment sampling
ML	maximum level
MLLW	mean lower low water
N-qualifier	tentative identification
NCMA	normalized combined mortality and abnormality
NMFS	National Marine Fisheries Service
NOAA	National Oceanic and Atmospheric Administration
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
PDM	post-dredge monitoring

PEF	potency equivalency factor
ppt	parts per thousand
PSDDA	Puget Sound Dredged Disposal Analysis
PSEP	Puget Sound Estuary Program
QAPP	quality assurance project plan
QC	quality control
RCM	recontamination monitoring
RI	remedial investigation
RL	reporting limit
ROC	receptor of concern
SEM	simultaneously extracted metals
SL	screening level
SMAV	species mean acute value
SMS	Washington State Sediment Management Standards
SPI	sediment profile imaging
SQS	sediment quality standards
SRI	supplemental remedial investigation
SVOC	semivolatile organic compound
T-5	Terminal 5
T-18	Terminal 18
T-30	Terminal 30
T-104	Terminal 104
T-105	Terminal 105
T-107	Terminal 107
T&E	threatened and endangered
TBT	tributyltin
TCDD	tetrachlorodibenzo-*p*-dioxin
TEF	toxic equivalency factor
TEQ	toxic equivalent
TOC	total organic carbon
TRV	toxicity reference value
U-qualifier	not detected at given concentration
UCL	upper confidence limit on the mean
USACE	US Army Corps of Engineers
USCG	US Coast Guard
USFWS	US Fish and Wildlife Service
USGS	US Geological Survey
VOC	volatile organic compound
WAC	Washington Administrative Code

ww	wet weight
WW	West Waterway
WDFW	Washington State Department of Fish and Wildlife
WHO	World Health Organization
WQA	water quality assessment
WQC	water quality criteria
WQG	water quality guidelines
WQS	water quality standard

Executive Summary

This document presents the baseline ecological risk assessment (ERA) for the East Waterway (EW), as outlined in the supplemental remedial investigation/feasibility study (SRI/FS) final work plan for the EW site (Anchor and Windward 2007). Baseline risk assessments, as defined in US Environmental Protection Agency (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 baseline ERA presents risk estimates for benthic invertebrate, fish, and wildlife species that may be exposed to contaminants of potential concern (COPCs) present in sediment, water, and aquatic biota in the EW. To the extent possible, this ERA is consistent with the approach and methods that were approved by EPA for use in the ERA for the Lower Duwamish Waterway (LDW) (Windward 2007d), which is a Superfund site that is located upstream of and contiguous with the EW and has many physical and functional characteristics similar to those of the EW. In addition, this ERA is consistent with the ERA technical memorandum, which was approved by EPA (Windward 2010g).

The dataset for the baseline ERA consisted primarily of tissue, sediment, and surface water chemistry data collected from the EW as part of the EW SRI/FS sampling efforts, along with available historical data collected since 1994. 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 ERA problem formulation establishes the overall scope of the assessment. Because it is impractical to evaluate risks to every potentially exposed species, it is standard ERA practice 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 osprey would be assumed to be protective of all piscivorous birds because of the higher exposure potential of osprey than that of other piscivorous birds. In addition, risks to some species were analyzed because those species are highly valued by society, such as juvenile Chinook salmon, which is listed as a threatened species under the Endangered Species Act. Representative receptors of concern (ROCs) selected for this ERA were the benthic invertebrate community, crab, three fish species (juvenile Chinook salmon, English sole, and brown rockfish), and four wildlife species (pigeon guillemot, osprey, river otter, and harbor seal).

The problem formulation discusses the data available for conducting the ERA and the suitability of the data for risk assessment purposes and conducts a risk-based screening evaluation that allows the risk assessment to focus on COPCs and eliminate chemicals that do not pose risks to the ROCs.

Data used in the ERA consisted largely of:

- Surface sediment (uppermost 10 cm) chemistry data
- Site-specific sediment toxicity test data
- Surface water chemistry data
- Sediment porewater chemistry data
- Tissue chemistry data for benthic invertebrates (including benthic infauna and epifauna, crabs, shrimp, clams, and mussels), English sole, brown rockfish, shiner surfperch, and juvenile Chinook salmon

For each ROC selected, COPCs were identified through a conservative risk-based screening process. COPCs identified included:

- **Benthic invertebrate community –** 4 metals (arsenic, cadmium, mercury, zinc); 1 organometal (tributyltin [TBT]); 16 polycyclic aromatic hydrocarbons (PAHs); total polychlorinated biphenyls (PCBs); 6 other semi-volatile organic compounds; 1 volatile organic compound (VOC) (naphthalene); and 1 pesticide (total dichlorodiphenyltrichloroethanes [DDTs])
- **Crab –** arsenic, cadmium, copper, mercury, zinc, TBT, and total PCBs
- **Fish –** arsenic, cadmium, chromium, copper, mercury, vanadium, TBT, benzo(a)pyrene, beta-endosulfan, and total PCBs, (each chemical was identified as a COPC for at least one but not necessarily all fish ROCs)
- **Wildlife –** mercury, selenium, total PCBs, and PCB TEQ[1] (each chemical was identified as a COPC for at least one but not necessarily all wildlife ROCs)

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 from the EW in aquatic biota, surface sediment, sediment porewater or surface water. The pathways evaluated in the ERA included sediment contact, sediment ingestion, water contact, water ingestion, and prey ingestion.

Finally, the problem formulation identifies assessment and measurement endpoints. Survival, growth, and reproduction were the key endpoints evaluated for the ROCs in this assessment. The representative ROCs, COPCs, exposure pathways, and endpoints formed the scope for the ERA.

[1] PCB TEQ is calculated using toxic equivalency factors, which relate the toxicity of the co-planar PCB congeners (i.e., those with dioxin-like properties) to the toxicity of 2,3,7,8-TCDD.

ES.2 EXPOSURE ASSESSMENT

The exposure assessment estimates the potential exposure of each ROC to the COPCs identified in the problem formulation. The exposure of the benthic invertebrate community (such as amphipods, bivalves and polychaetes) to COPCs was assessed by evaluating concentrations of COPCs in surface sediment, benthic invertebrate tissue, and surface water. In addition, risks to the benthic invertebrate community from exposure to VOCs were assessed using sediment porewater data. The exposure of crab, a wider-ranging, higher trophic level macroinvertebrate, was assessed using COPC concentrations in crab tissue and surface water.

Exposure of fish to COPCs was characterized based on either COPC concentrations in fish tissue or COPC concentrations in fish prey as well as an evaluation of COPCs in surface water. For wildlife ROCs, the exposure assessment identified equations and parameters to quantify the ingested dose of COPCs from aquatic prey, surface water and sediment. 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

The effects assessment presents effect threshold levels for COPCs based on criteria, guidelines, or toxicity data from the literature. For the exposure of the benthic invertebrate community to sediment, Washington State Sediment Management Standards (SMS) biological (i.e. toxicity test) and chemical criteria were used to determine the potential for adverse effects at specific locations. According to SMS, for locations where toxicity test data were available, the toxicity test results overruled sediment chemistry results in determining whether there was the potential for adverse effects. For locations without toxicity test results, chemical criteria provided by the SMS were presented as adverse effect levels. For COPCs without SMS criteria (i.e., total DDTs), guidelines from the Washington State Dredged Material Management Program (DMMP) were used.[2]

[2] DMMP guidelines for TBT were not used because they are based on TBT concentrations in interstitial water rather than whole sediment, since available evidence indicates that sediment concentrations are not as useful in predicting environmental effects (USACE et al. 2008). Therefore, risk to benthic invertebrates from exposure to TBT was addressed using tissue residues rather than whole sediment concentrations. For dioxins/furans, the DMMP recently developed new interim guidelines for sediment (WDNR et al. 2010). However, these guidelines are based on Puget Sound non-urban background concentrations after consideration of human health risk thresholds and, since they are not toxicity-based, are not appropriate for use as ecological risk thresholds for benthic organisms. Although the lack of toxicity information precluded the evaluation of risks to benthic organisms from dioxin/furan exposure, risks from dioxin/furans were addressed for fish and wildlife using available toxicity data for those ecological receptors.

For a number of ROCs (i.e., the benthic invertebrate community, crab, and fish), state or federal water quality criteria (WQC) and the information used to derive those criteria were considered in the identification of adverse effect levels for surface water.

For the remaining ROCs and pathways, toxicity data from the literature were used to identify threshold levels for potential adverse effects (i.e., reduced survival, reduced growth, or impaired reproduction) because state or federal criteria have not been established. For each ROC and COPC, a search was conducted to identify studies in the scientific literature that documented the effects of those COPCs on the ROCs or similar species, and then a detailed evaluation of these studies was performed. This literature review identified COPC concentrations in the exposure media, the receptor tissue, or the ingested dose associated with no effects (i.e., concentrations or doses at which no adverse effects were observed), as well as concentrations or doses at which adverse effects have been observed. Both sets (i.e., no-observed-adverse-effect level [NOAEL] or no-observed-effect-concentration and lowest-observed-adverse-effect level [LOAEL] or lowest-observed-effect-concentration) of toxicity reference values (TRVs) are summarized in tables, and the rationale for TRV selection is provided.

ES.4 RISK CHARACTERIZATION

The risk characterization compares the exposure and effects data to evaluate the potential for COPCs to cause adverse effects on the ROCs. COPCs were selected as contaminants of concern (COCs) if the risk conclusions indicated a potential for adverse effects. The findings of the risk characterization including the identification of COCs are presented below.

ES.4.1 Benthic invertebrate community

Risks to the benthic invertebrate community were evaluated through four different approaches: sediment, tissue-residue, surface water and porewater. Sediment chemistry and site-specific toxicity test results indicate that no adverse effects on benthic invertebrates living in intertidal and subtidal sediments are predicted for approximately 40% of the EW area (i.e., the area in which chemical concentrations were less than or equal to sediment quality standards (SQS) chemical criteria and/or sediments were non-toxic according to SQS biological effects criteria). There is a higher likelihood for adverse effects in approximately 21% of the EW area, which had chemical concentrations or biological effects in excess of the cleanup screening level (CSL) values. The remaining 39% of the EW area had chemical concentrations or biological effects between the SQS and CSL values, indicating the potential for minor adverse effects to benthic invertebrate communities. Some uncertainty is associated with these area estimates because areas were calculated by interpolating from the individual locations at which sediments were sampled. Twenty-nine chemicals or groups of chemicals had at least one concentration that exceeded its respective SQS or SL and were therefore identified as COCs for the benthic invertebrate community. These chemicals include

4 metals, 16 individual PAHs or group of PAHs, 3 phthalates, 4 other SVOCs, total PCBs, and total DDTs.

TBT was identified as a COC because the tissue-residue LOAEL TRV was exceeded in benthic tissue samples from two of the twelve sampling areas evaluated for TBT. For total PCBs, risk was predicted to be low and uncertain because tissue concentrations were below LOAEL TRVs but greater than NOAEL TRVs in ten of the thirteen sampling areas evaluated for total PCBs.

TBT and naphthalene were identified as COCs based on the surface water and porewater evaluations, respectively. VOCs in sediment porewater are unlikely to pose a risk to the benthic invertebrate community, except for naphthalene, which had a concentration that exceeded the LOAEL TRV at only one location. One detected TBT concentration in surface water exceeded the marine chronic WQC for TBT. However, reporting levels associated with the undetected results also exceeded the WQC. Therefore, it was concluded that risks are low and very uncertain for the exposure of the benthic invertebrate community to TBT in surface water.

ES.4.2 Crab

Risks to crabs were evaluated through two different approaches: a tissue-residue evaluation and a surface water evaluation. Cadmium, copper and zinc were identified as COCs because concentrations of these COPCs in crab tissue were greater than their LOAEL TRVs, indicating potential risks to crabs. Arsenic and total PCBs, the two remaining COPCs, had concentrations in crab tissue below LOAEL TRVs but above NOAEL TRVs indicating low but uncertain risks to crab. These two COPCs were not identified as COCs because concentrations were below LOAEL TRVs. Based on the surface water evaluation, no COCs were identified for crab.

ES.4.3 Fish

Risks to the fish ROCs were evaluated through tissue-residue or dietary evaluations (depending on the chemical) as well as a surface water evaluation. Five chemicals (cadmium, copper, vanadium, TBT, and total PCBs) were identified as COCs for fish, based on the tissue-residue or dietary evaluations, indicating a potential for risks; no COCs were identified through surface water evaluation. Cadmium was identified as a COC for all three fish ROCs while copper and vanadium were identified as COCs only for English sole based on the dietary evaluation. Total PCBs was identified as a COC for English sole and brown rockfish and TBT for rockfish based on the tissue-residue evaluation. Risks were considered low and uncertain or unlikely for the remaining COPCs.

ES.4.4 Wildlife

Risks to wildlife ROCs were evaluated based on ingested doses of aquatic prey, surface water and sediment. Risks were evaluated for two bird ROCs (pigeon guillemot and osprey), which serve as representative surrogates for all other bird species that may be exposed in the EW. No COCs were identified for bird ROCs. Exposures were below NOAELs for all COPCs and therefore risks to birds from exposures to chemicals in the EW are considered to be unlikely. Risks were evaluated for two mammal ROCs (river otters and harbor seals), which serve as representative surrogates for all other mammal species that may be exposed in the EW. No COCs were identified for the mammal ROCs. Exposures were below NOAELs for all COPCs for harbor seals and therefore risks from exposure to chemicals in EW are considered to be unlikely. For river otter, adverse effects associated with all COPCs except total PCBs are unlikely. The potential for adverse effects was considered low and uncertain for river otters exposed total PCBs because the NOAEL TRV was exceeded, but the LOAEL TRV was not exceeded.

ES.4.5 Summary of risk drivers

COCs were identified as risk drivers for ecological receptors based on the risk estimates, uncertainties discussed in this ERA, and background concentrations in accordance with EPA guidance (1992, 1997a, b, 1998) and consistent with the LDW ERA (Windward 2007c). The risk drivers from both this ERA and the HHRA will be the focus of remedial analyses in the FS. COCs not selected as risk drivers in the EW ERA will be evaluated qualitatively in the EW FS.

COCs that were identified as risk drivers are noted in Table ES-1. Twenty-eight COCs were selected as risk drivers in sediment for the benthic invertebrate community because the concentrations of these 28 chemicals exceeded SMS in one or more locations. TBT was identified as a risk driver for the benthic invertebrate community because TBT concentrations in benthic invertebrate composite tissue samples collected from 2 out of 12 areas with EW exceeded the LOAEL TRV. Total PCBs was identified as a risk driver for English sole and brown rockfish because tissue concentrations for these fish were greater than the higher LOAEL TRV by a factor of 1.6, and uncertainties in these risk estimates were relatively low. Other COCs were not selected as risk drivers because of uncertainties in exposure or effects data or consideration of sediment concentrations in EW relative to regional background data.

Table ES-1. COCs and risk drivers identified for ERA receptors

Receptor	Evaluation Type	COPCs	COCs	Risk Driver
Benthic Invertebrate Community	sediment	29 chemicals, including metals, PAHs, total PCBs, phthalates, other SVOCs and total DDTs	29 COPCs[a]	28 SMS chemicals
	tissue residue	TBT, total PCBs	TBT	TBT
	surface water	cadmium, mercury, TBT	TBT	none
	porewater	naphthalene	naphthalene	none
Crab	tissue residue	arsenic, cadmium, copper, zinc, and total PCBs	cadmium, copper, zinc	none
	surface water	cadmium, mercury, TBT	none	none
Fish	dietary	arsenic, cadmium, chromium, copper, vanadium, benzo(a)pyrene	cadmium, copper, vanadium	none
	tissue residue	beta-endosulfan, total PCBs, mercury, TBT	total PCBs, TBT	total PCBs
	surface water	cadmium, mercury, TBT	none	none
Birds	dietary dose	mercury, total PCBs, PCB TEQ, total TEQ	none	none
Mammals	dietary dose	mercury, selenium, total PCBs, PCB TEQ, total TEQ	none	none

[a] Arsenic, cadmium, mercury, zinc, acenaphthene, benzo(a)anthracene, benzo(a)pyrene, benzo(g,h,i)perylene, chrysene, dibenzo (a,h)anthracene, fluoranthene, fluorene, indeno(1,2,3,-c,d)pyrene, phenanthrene, pyrene, total benzofluoranthenes, HPAH, LPAH, bis(2-ethylhexyl) phthalate, butyl benzyl phthalate, di-n-butyl phthalate, 1,4-dichlorobenzene, 2-methylnaphthalene, 2,4-dimethylphenol, dibenzofuran, n- nitrosodiphenylamine, phenol, and total PCBs and total DDTs. All COCs had exceedances of SMS chemical criteria except total DDTs, which was based on exceedances of DMMP guideline

COC – chemical of concern
COPC – chemical of potential concern
DDT – dichlorodiphenyltrichloroethane
DMMP – Dredge Material Management Program
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
SMS – Washington State Sediment Management Standards
SVOC – semivolatile organic compound
TBT – tributyltin
TEQ – toxic equivalent

A.1 Introduction

This document presents the baseline ecological risk assessment (ERA) that has been completed as part of the supplemental remedial investigation and feasibility study (SRI/FS) for the East Waterway (EW). The EW is an operable unit of the Harbor Island Superfund site, which 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, in 1983. The key parties involved in the EW SRI/FS are the City of Seattle, King County, and the Port of Seattle, which work together as the East Waterway Group (EWG). Oversight of the EW SRI/FS is being provided by EPA as agreed in the Administrative Settlement and Order on Consent (EPA 2006a) signed by the Port of Seattle and EPA in October 2006.

As described in EPA's Superfund regulations (1988), EPA requires that an RI/FS be conducted for each site listed on the NPL. An RI evaluates the nature and extent of chemical contamination, estimates baseline risks to human health and the environment, 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. This baseline ERA is an appendix to the EW SRI. The SRI is supplemental to the Harbor Island RI (Weston 1993), which included the EW as an operable unit.

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." The baseline ERA presents risk estimates for ecological receptors of concern (ROCs) that may come in contact with sediment–associated chemicals of potential concern (COPCs) through exposure to or ingestion of sediment, surface water, fish, and invertebrates (e.g., clams, crabs, polychaete worms) in the EW. Ecological ROCs are those organisms or communities of organisms that may be exposed to site contaminants and are the focus of this ERA.

As outlined in the SRI/FS final work plan for the EW site (Anchor and Windward 2007), this draft baseline ERA is based on an earlier technical memorandum (Windward 2010g), which provided much of the approach and technical basis of the ERA for the EW and was developed in consultation with EPA. The baseline ERA is based on historical data that was previously summarized in the existing information summary report (EISR) (Anchor and Windward 2008) and more recent data collected as part of the SRI in 2008 and 2009. It was developed in accordance with both national and regional EPA guidance (1992, 1997a, b, 1998). To the extent possible, this ERA is consistent with the approach and methods that were approved by EPA for use in the ERA for the Lower Duwamish Waterway (LDW) (Windward 2007d), which is upstream of and contiguous with the EW.

This baseline ERA includes the following sections:

- Section A.2 – Problem Formulation
- Section A.3 – Exposure and Effects Assessment: Benthic Invertebrates
- Section A.4 – Exposure and Effects Assessment: Fish
- Section A.5 – Exposure and Effects Assessment: Wildlife
- Section A.6 – Risk Characterization and Uncertainty Analysis
- Section A.7 – Selection of Ecological Risk Drivers
- Section A.8 – Conclusions
- Section A.9 – References

A.2 Problem Formulation

This section presents the problem formulation for the baseline ERA and includes information regarding the environmental setting, the ecological resources that use the EW, the selection of ROCs, a summary of relevant site-specific data and the conceptual site model (CSM) for the EW. Through the use of a risk-based screening approach, the problem formulation also establishes which ROCs and which COPCs for those ROCs (i.e., ROC-COPC pairs) are further evaluated in the exposure and effects assessment, the risk characterization, and the uncertainty analysis. Together, these elements establish the scope for this baseline ERA, which is consistent with the overall management goals for the site, which include:

- Limit/reduce the exposure of the benthic invertebrate community to sediment-associated contaminants to concentrations below which unacceptable risks to the benthic invertebrate community do not occur.
- Limit/reduce the exposure of crabs, fishes, birds, and mammals to sediment-associated contaminants to concentrations below which unacceptable risks to those populations do not occur.
- Limit/reduce exposure of migratory juvenile salmonids to sediment-associated contaminants to concentrations below which no adverse effects on individuals occur.

A.2.1 Environmental Setting

This section describes the physical features of the site, as well as features associated with available habitat for benthic invertebrates, fish, and wildlife.

A.2.1.1 Site description

The EW is an industrial waterway located approximately 1 mile southwest of downtown Seattle, in King County, Washington and is used primarily for container loading and transport (Map A.1-1). The EW and West Waterway (WW) are dredged navigation channels that flow around either side of the man-made Harbor Island and together form the mouth of the Duwamish River, which discharges to Elliott Bay. The LDW, also a Superfund site, is located immediately upstream of the EW. Water from the LDW flows into both the EW and WW, with the greatest quantity flowing into the WW.

The EW is a straight channel that is approximately 1.5 mi long. The EW is approximately 750 ft wide across most of the channel, narrowing at the south end to a width of approximately 330 ft at the Spokane Street corridor and to approximately 170 ft just north of the junction with the LDW (Map A.1-2). The federally maintained shipping channel is -52 ft mean lower low water (MLLW) throughout much of the EW, with the exception of the southern end of the EW, with the exception of the southern end of EW. In this area, the EW is shallower than -50 ft MLLW with an authorized navigation depth of -35 ft MLLW. The southern end of EW has not been dredged in the past 20 years. The minimum sediment elevation of -6 to -12 ft MLLW occurs under the West Seattle and Spokane Street Bridges (Map A-1.2). This shallow area does not support shipping activity.

In addition to the primarily commercial uses of the EW, commercial netfishing operations are conducted in the EW by the Muckleshoot Tribe. The EW is part of the Suquamish and Muckleshoot Tribe's Usual and Accustomed (U&A) fishing grounds; consequently, they are permitted by federal law to harvest salmon in commercial quantities from this area and use the waterway for ceremonial and subsistence fishery. Tribal seafood harvesting practices are currently ongoing and will continue to occur in the future.

A.2.1.2 Habitat features

Dredging and development have substantially altered nearshore environments in Elliott Bay and the Duwamish River. Prior to the channelization and industrialization of the Duwamish River, the habitat associated with the river's mouth was predominantly an intertidal/shallow subtidal estuarine mudflat. Since the creation of Harbor Island, all of the original habitat in the area that is now the EW has been either filled or dredged.

The aquatic habitats in the EW include the water column and intertidal and subtidal substrates (typically mud, sand, gravel, cobble, or riprap). The shoreline of the EW is approximately 16,000 linear ft (excluding Slip 27 and Slip 36). Most of the shoreline (61%) is covered by wharves with engineered riprap slopes, roughly a third of the shoreline (30%) is covered with armored riprap with no wharf structures, and the remaining shoreline (9%) is predominantly characterized as bulkhead. The shoreline within Slip 27 and Slip 36 is predominantly armored riprap with limited wharf structures, although the southern shore of Slip 27 has an adjacent intertidal bench that was constructed during re-armoring of the Port of Seattle property. A limited number of small intertidal beaches are present above the riprap slopes in locations along the eastern shoreline of the waterway, including the head of Slip 27.

The standard concrete wharves in the EW are 100 ft wide from the outer edge to the inner bulkhead at +9 ft MLLW. Vertical bulkheads are usually present above +9 ft MLLW because the Washington State Department of Fish and Wildlife (WDFW) requirements limit their intertidal range. Areas below the bulkheads are typically engineered riprap slopes to approximately -50 ft MLLW (with some areas to -40 ft MLLW).

Shoreline armoring is usually present in the upper intertidal zone, but a few areas of sloping mud and sand flats and gravel/cobble exist in the lower intertidal zone. These lower intertidal flats are isolated from each other because of the shoreline armoring. In addition, overwater structures, which are common throughout the EW, shade shallow water and intertidal habitats, and inhibit the growth of plant communities (Battelle et al.

2001). Intertidal sediment is limited to small areas along the western shore of East Waterway. Gravel and cobble are the dominant matrices in the exposed intertidal areas.

The EW is part of the Green/Duwamish River watershed, with peak freshwater flows entering the Duwamish River from November through February, and minimum flows in August. The EW also receives discharges from 3 combined sewer overflows (CSOs) and 39 storm drains, but these inputs are small relative to freshwater inputs from the Green/Duwamish River. The outflow of freshwater from the Green/Duwamish River along with the marine tidal waters entering from Elliott Bay result in estuarine conditions in the EW with a characteristic increase in salinity with water depth and a thin layer of slightly lower salinity at the surface. For example, during the SRI surface water sampling events, salinity in the EW typically ranged from approximately 15 to 28 parts per thousand (ppt), with higher salinity measured in the bottom of the water column as compared with those at the top (Windward 2009b). The flows are characterized by outflow to Elliott Bay in the surface layer and inflow to the EW near the bottom of the waterway during the flood tide. The EW is tidally influenced, with an approximate average tidal range of 11.36 ft, as measured at the nearby Seattle waterfront National Oceanic and Atmospheric Administration (NOAA) station.

Subtidal surface sediment within the EW has been extensively reworked as a consequence of dredging shoreline development and prop wash from ships. Sediment grain size in subtidal surface sediment (0 to 10 cm) ranges from 20 to 80% fines (i.e., silt plus clay), but is primarily fine grained (median of 53% fines). Subtidal sediment in the northern portion of the EW tends to be coarser (20 to 60% fines) than that in the remainder of the waterway (40 to 80% fines); however, the middle area of the waterway has been influenced by dredging and capping activities and often has a relatively thin layer of sand on top of silt and clay. The total organic carbon (TOC) content of the surface sediment layer (0 to 10 cm) is less than 2% over nearly all of the EW, with small patches greater than 2% over the remainder, including much of Slip 27.

A.2.2 RESOURCES POTENTIALLY AT RISK

This section provides an overview of the ecological resources that use the EW, including threatened, endangered, and sensitive species. These resources, which include species that could be directly or indirectly exposed to contaminated sediment, include the benthic invertebrate community, fish, and aquatic-dependent birds and mammals. Reptiles and amphibians are not likely to be exposed to sediment contamination in the EW because habitat for these species is limited (i.e., no freshwater habitat exists in the EW), and their presence has not been reported in any wildlife surveys conducted in the LDW, upstream of the EW (Canning et al. 1979; Cordell et al. 1996; 1997; 1999). Therefore, reptiles and amphibians are not included as ecological resources within the EW. In addition, risks to vascular plants will not be evaluated because of the limited plant communities in the EW. The limited exposed shallow water habitat and the presence of engineered riprap slopes throughout the EW are physical constraints that limit the vascular plant communities.

A.2.2.1 State and federal threatened, endangered, and sensitive aquatic and aquatic-dependent species in the vicinity of EW

Sixteen aquatic and aquatic-dependent species reported in the vicinity of Elliott Bay area are listed under either the Endangered Species Act (ESA) or by the WDFW as candidate species, threatened species, endangered species, or species of concern (Table A.2-1).

Table A.2-1. Aquatic and aquatic-dependent species in the vicinity of EW that are listed under ESA or by WDFW

Common Name	Scientific Name	Status[a]
Fish		
Bull trout	*Salvelinus confluentes*	FT, SC
Chinook salmon	*Oncorhynchus tshawytscha*	FT, SC
Coho salmon	*Oncorhynchus kisutch*	FC
Pacific cod	*Gadus macrocephalus*	FCo, SC
Pacific herring	*Clupea pallasi*	FCo, SC
River lamprey	*Lampetra ayresi*	FCo, SC
Brown rockfish	*Sebastes auriculatus*	FCo, SC
Steelhead salmon	*Oncorhynchus mykiss*	FT
Walleye pollock	*Theragra chalcogramma*	FCo, SC
Birds		
Bald eagle	*Haliaeetus leucocephalus*	FCo, SS[b]
Common loon	*Gavia immer*	SS
Common murre	*Uria aalge*	SC
Merlin	*Falco columbarius*	SC
Peregrine falcon	*Falco peregrinus*	FCo, SS[c]
Western grebe	*Aechmophorus occidentalis*	SC
Marine Mammals		
Killer whale	*Orcinus orca*	FE, SE

Source – WDFW (2010)

[a] Status abbreviations are as follows: FC = federal candidate species, FCo = federal species of concern, FE = federal endangered species, FT = federal threatened species, SC = state candidate species, SE = state endangered species, and SS = state sensitive species.

[b] Downlisted from federal and state endangered to FCo and state sensitive in 2007.

[c] Downlisted from state endangered to state sensitive in April 2002.

ESA – Endangered Species Act

EW – East Waterway

WDFW – Washington State Department of Fish and Wildlife

Of the species listed in Table A.2-1, Chinook salmon, coho salmon, steelhead salmon, brown rockfish, bald eagle, western grebe, and Pacific herring are commonly observed in the EW. Orcas are occasionally found within Elliott Bay; however, there have been no specific reports of orcas within the EW. Peregrine falcon have been known to nest along the LDW (Anderson 2006), but no nests have been reported along the EW. Common murre, loons, merlin, river lamprey, and walleye pollock have been reported as rare in the Duwamish River and waterways in the lower portion of the river (Canning et al. 1979). Pacific cod have never been collected during any fish sampling events or surveys conducted in the EW. Bull trout were rarely collected during extensive seining in the LDW in 1994 (Warner and Fritz 1995) and have never been collected during any fish sampling events or surveys in the EW.

A.2.2.2 Benthic invertebrate community

Benthic invertebrate assemblages in Puget Sound marine environments comprise a variety of species from diverse phyla (e.g., Mollusca, Arthropoda, Annelida, and Echinodermata). Benthic invertebrates can be classified as infaunal (living within the sediment) or epifaunal (living on the surface of sediment or other substrates) and, by definition, are in direct contact with the sediment during part or all of their lives. Most benthic invertebrates tend to be sessile (i.e., stay in place) or have limited mobility as adults. Benthic invertebrates have numerous types of feeding modes that expose them to sediment. These include filtering suspended sediment, plankton, and detritus from the water column; gathering detritus or sediment grains coated with organic material from the sediment surface or near-bottom nepheloid layer; engulfing subsurface sediment to process the associated organic material; parasitizing other sediment-dwelling organisms; and preying on other invertebrates.

Benthic invertebrates are an important contributor to aquatic ecosystems, and their diversity and abundance are indicators of ecosystem health. Benthic invertebrates that process sediment or detritus support essential functions, such as nutrient cycling and sediment oxygenation. Benthic invertebrates are an important food source for other invertebrates and fish; larger invertebrates are also a major part of the diet of some birds and mammals.

In general, key physical factors that may influence the distribution and abundance of benthic invertebrates are salinity, tidal elevation (affecting the duration of exposure to air or heat), water depth, substrate composition, sediment organic carbon (OC) content, sediment stability, wave and current magnitude, and frequency of disturbance (e.g., flooding, prop wash, and anchor drag).

Limited characterization of the benthic invertebrate community in the EW has been conducted. Targeted invertebrate groups have been sampled as part of four studies: a 1999 epifaunal survey assessing salmonid prey (Taylor et al. 1999) and three tissue collection efforts (small benthic invertebrates and larger clams, mussels, shrimp, and crab) conducted as part of this SRI (Windward 2009a, 2010b, c) (Table A.2-2). No quantitative sampling of benthic community structure has been conducted; however, a sediment profile imaging (SPI) survey of the EW that examined the successional stage of the benthic community was conducted in October 2009 (Windward 2009a).

Table A.2-2. Summary of studies that assessed benthic invertebrates in the EW

Study	Year Completed	EW Location	Sampling Period	Sampling Method	Targeted Organisms
EW benthic invertebrate tissue and sediment sampling (Windward 2009a)	2009	throughout the EW	2008–2009	grab samples	subtidal benthic invertebrates
EW SPI survey (Germano & Associates 2009)	2008	throughout the EW	October 2008	SPI camera	infaunal invertebrates
EW clam survey and tissue sampling (Windward 2010b)	2008	selected areas with clam habitat	summer and fall 2008	hand-collection; hydraulic extraction for geoduck	clams
EW fish and crab tissue sampling (Windward 2010c)	2008	throughout the EW	summer and fall 2008	trawls and traps; hand-collection for mussels	fish and crabs
Epibenthic species assessment (Taylor et al. 1999)	1999	Slip 27	unknown	epibenthic suction pump	epibenthic invertebrates

EW – East Waterway
SPI – sediment profile imaging

Information from surveys conducted in the LDW downstream of Kellogg Island (i.e., north of the island and in the reach immediately upstream of the EW) and in nearshore Elliott Bay provide an indication of the invertebrates that could be present in the EW because these areas are adjacent to the EW, and Elliott Bay acts as a source of plankton that may be transported into the EW in the incoming marine layer and settle out in the EW. A summary of existing EW epifaunal invertebrate data is presented in Section A.2.2.2.1, and relevant supporting infaunal assemblage information from the northern portion of the LDW and nearshore Elliott Bay is presented in Section A.2.2.2.2.

Crab represent some of the larger benthic invertebrates that inhabit the EW. Crab species that are known to be present in the EW include Dungeness crab (*Cancer magister*), red rock crab (*Cancer productus*), graceful crab (*Cancer gracilis*), kelp crab (*Pugettia producta)*, decorator crab (*Loxorhynchus crispatus)*, pygmy rock crab (*Cancer oregonensis*). Dungeness crab is the largest crab species present in the EW, and red rock crab was the most abundant species collected during the SRI sampling events (Windward 2010c). Mating typically takes place in deeper, offshore locations but may occasionally occur in estuaries (Pauley et al. 1988). Gravid females migrate to shallow estuarine habitats or other protected areas until their eggs hatch; planktonic larvae tend to settle in vegetated estuaries, which also serve as nurseries for juvenile crab. The highest densities of juvenile crab are usually associated with eelgrass or other kinds of aquatic vegetation that are not present in the EW.

Although crab are primarily carnivores and scavengers, crab diets are characteristic of their life stage and size (Pauley et al. 1986). Planktonic larval crab ingest both zooplankton and phytoplankton. Following metamorphosis, the diet of juvenile crab consists largely of very small fish, molluscs, and crustaceans. Adult crab primarily prey on clams, crustaceans, and fish. Juvenile and adult crab may incidentally ingest sediment when preying on clams and benthic fish, but the rate of ingestion is likely to be low because many prey species (e.g., mussels and barnacles on pilings, and shrimp) do not dwell on the sediment surface. Crab prey size changes with age; crab tend to eat clams in their first year, shrimp in their second year, and small fish in their third year. Planktonic crab larvae (megalopae) are preyed upon by many fish, including juvenile salmon. Juvenile crab are eaten by various demersal fish in the nearshore area. Flatfish, such as starry flounder (*Platichthys stellatus*) and English sole (*Parophrys vetulus*), are the most important predators of crab in Puget Sound. Adult and juvenile crab are preyed upon by river otters, fish, aquatic birds (e.g., pigeon guillemots), and octopuses. Cannibalism is also common among crab.

A.2.2.2.1 Existing East Waterway benthic invertebrate data

Taylor et al. (1999) conducted a survey of epibenthic invertebrates as part of a juvenile salmonid prey assessment in several intertidal areas of the lower 2 mi (3.2 km) of the Duwamish River (including EW) and the northern shore of Elliott Bay in support of disposal site selection for an EW navigation project. Sampling was conducted at one location in the EW, Slip 27 (at the head and at the entrance of the slip); epifaunal samples (primarily crustaceans) were collected at 0 and 2 ft (0.6 m) MLLW using a suction pump. Most of the 110 invertebrate taxa collected at this location and the two other locations sampled outside of EW were potential fish prey species. The most diverse taxonomic groups were harpacticoid copepods, with 62 taxa, and gammarid amphipods, with 18 taxa. The dominant species at Slip 27 were harpacticoid copepods, including *Harpacticus uniremis*, *Tisbe* spp., and *Dactylopusia* sp. Other abundant crustaceans were gammarid amphipods such as *Paracalliopeilla pratti* (Table A.2-3). The highest epibenthic invertebrate density reported in this study was at Slip 27.

Table A.2-3. Benthic invertebrate taxa collected in the EW in 1999

Benthic Taxa		
Cnidaria		
Anthozoa		
Platyhelminthes		
Turbellaria		
Nematoda		
Annelida		
Polychaeta		
Mollusca		
Gastropoda juveniles	Nudibranchia	Bivalvia juveniles
Acarina		
Halacaridae		
Calanoida		
Pseudodiaptomus marinum	*Stephos* spp.	
Harpacticoida		
Ameira spp.	*Harpacticus compressus*	*Normanella* sp.
Ancorabolidae	*Harpacticus obscurus* group	*Parastenhelia spinosa*
Leimia vaga	*Harpacticus spinulosus*	Peltidiidae
Mesochra spp.	*Harpacticus uniremis*	*Tachidius discipes*
Cletodidae spp.	*Harpacticus* sp. A	*Tachidius traingularis*
Acrenhydrosoma sp.	*Harpacticus* sp.	Tegastidae
Enhydrosoma spp.	*Harpacticus* copepodids	Thalestridae spp.
Amonardia perturbata	*Zaus* spp.	*Dactylopusia* sp.
Amonardia normani	*Huntemannia jadensis*	*Dactylopusia crassipes*
Disaccus sp.	Laophontidae spp.	*Datrylopusia tisboides*
Diosaccus spinatus	*Echinolaophonte* sp.	*Dactylopusia paratisboides*
Amphiascopis cinctus	*Heterolaophonte discophora*	*Dactylopusia glacialis*
Amphiascus spp.	*Heterolaophonte longisetigera*	*Dactylopusia vulgaris*
Stenhelia spp.	*Heterolaophonte harmondi*	*Diarthrodes* spp.
Typhlamphiascus sp.	*Laophonte cornuta*	*Idomene* sp.
Amphiascoides spp.	*Laophonte elongate*	*Paradactylopodia* spp.
Amphiascoides sp. A	*Paralaophonte* sp.	*Parathalestris* spp.
Bulbamphiascus sp.	*Paralaophonte pacifica*	*Rhynchothalestris helgolandica*
Robertsonia cf. *knoxi*	*Paralaophonte perplexa*	*Tisbe* spp.
Ectinosomatidae	*Pseudonychocamptus* spp.	*Scutellidium* spp.
Harpacticus arcticus	*Longipedia* sp.	

Table A.2-3. Benthic invertebrate taxa collected in the EW in 1999 (cont.)

Benthic Taxa		
Copepoda		
Hemicyclops sp.	Ergasilidae	Cyclopoida
Ostracoda		
Podocopa		
Thoracica		
Unidentified nauplii	Unidentified cyprids	
Cumacea		
Diastylis santamariensis	*Nippoleucon hinumensis*	*Cumella vulgaris*
Isopoda		
Gnorimosphaeroma oregonense	*Munnogonium* sp.	*Leptochelia savignyi*
Munna sp.		
Tanaidacea		
Tanaidacea		
Gammaridae		
Anisogammaridae juveniles	*Corophium* spp.	*Photis* sp.
Eogammarus confervicolus	*Hyalella* sp.	Pleustidae spp.
Ampithoe sp.	*Gammaropsis* sp.	*Pleusirus secorrus*
Aoroides sp.	*Ischyrocerus* sp.	*Sympleustes* sp.
Capelliopius sp.	Melitidae	*Paramoera* sp.
Paracallipiella pratti	Oedicerotidae	*Pontogeneia* cf. *rostrata*
Caprellidae		
Caprella sp.		
Decapoda		
Unidentified larvae	Caridea	*Upogebia pugettensis*
Insecta		
Chironomidae	Unidentified larvae	

Source: (Taylor et al. 1999)
EW – East Waterway

Surveys that documented larger benthic invertebrates were conducted in 2008, during which clam, crab, shrimp, and mussel tissue data were collected for this SRI (Windward 2009a, 2010b, c). Clam surveys were conducted at 11 beach areas (Windward 2010b); five of these areas were located in the southern narrow portion of the EW, three were located in and near Slip 27, and three were located along the shoreline of Slip 36. During this survey, Macoma clams (*Macoma* spp.) were the most frequently observed species, followed by Japanese littleneck clams (*Venerupis philippinarum)* and butter clams (*Saxidomus gigantean)*. Cockles (*Clinocardium nuttali*) and eastern soft-shell clams (*Mya arenaria*) were observed only in the southernmost portion of the EW.

Crab and shrimp were collected using 12 crab traps and 16 shrimp traps dispersed throughout the EW in August 2008, as well as during 10 trawls conducted using a high-rise otter trawl in September 2008. The crab and shrimp species collected during these sampling events are listed in Table A.2-4; anemones, sea stars, and sea urchins were also identified during these sampling events, although they were not targeted species. Only 26 individual shrimp were collected (enough for one composite sample) in the shrimp traps. However, during the brown rockfish sampling event, scuba divers also noted the presence of numerous small shrimp that were too small to be collected in the shrimp traps. During the mussel survey conducted in July 2008, mussels were present wherever there was a suitable substrate, which was primarily on pilings and sheetpile walls.

Table A.2-4. Invertebrate species collected in the EW in 2008

Common Name	Scientific Name
Crab	
Decorator crab	*Loxorhynchus crispatus*
Dungeness crab	*Cancer magister*
Kelp crab	*Pugettia producta*
Pygmy rock crab	*Cancer oregonensis*
Red rock crab	*Cancer productus*
Graceful crab	*Cancer gracilis*
Bivalves	
Blue mussel	*Mytilus* spp.
Butter clam	*Saxidomus gigantea*
Cockle	*Clinocardium nuttali*
Eastern soft-shell clam	*Mya arenaria*
Geoduck	*Panopea generosa*
Japanese littleneck clam	*Venerupis (= Tapes) philippinarum (= japonica)*
Macoma clam	*Macoma* spp.
Native littleneck clam	*Leukoma (= Protothaca) staminea*
Other Invertebrates	
Coonstripe or dock shrimp	*Pandalus danae*
Plumose anemone	*Metridium senile*
Sea star	*Evasterias* sp.
Sea star	*Luidia* sp.
Solaster star	*Solaster stimpsoni*
Sunflower sea star	*Pycnopodia helianthoides*
Sea urchin	*Stronglyocentrotus* sp.

Source: Windward (2009a, 2010b, c)
EW – East Waterway

A.2.2.2.2 Other relevant benthic invertebrate information

Benthic invertebrate data from the northern portion of the LDW and nearshore Elliott Bay provide useful information regarding infaunal assemblages that may be present in the EW. Several studies are summarized below; additional detail is provided in the EISR (Anchor and Windward 2008).

The benthic assemblages in the northern portion of the LDW near and downstream of Kellogg Island were generally dominated by annelids, crustaceans, and molluscs (Windward 2005; Cordell et al. 2001; Williams 1990; Leon 1980). The dominant intertidal bivalve was the eastern soft-shell clam (*Mya arenaria)*. Common intertidal annelids included subsurface deposit feeders from the *Capitella capitata* complex, the filter feeder *Manayunkia aestuarina*, the surface detrital feeder *Pygospio elegans*, and oligochaetes. Common intertidal crustaceans included *Americorophium* and *Grandidierella japonica*, which feed on detrital material on the sediment surface or in the water column (Windward 2005). Very small invertebrates (meiofauna) in intertidal habitats were generally dominated by nematodes and epibenthic harpacticoid copepods (Cordell et al. 2001). Molluscs were not common in intertidal habitats.

The predominant species in the subtidal zone in the LDW included annelids, such as the deposit feeder *Aphelochaeta cf glandaria*, the deposit feeder *Lumbrineris californiensis* (which may also ingest tiny organisms that are present in the sediment), the surface deposit/detrital feeders *Scoletoma luti* and *Prionospio steenstrupi*, and oligochaetes. The amphipod *Anisogammarus* sp. was among the crustaceans common in subtidal habitats (Leon 1980). The subtidal epibenthos was dominated by nematodes, oligochaetes, small harpacticoids, and cumaceans (Williams 1990). Bivalves common in subtidal habitats included the surface deposit feeders *Axinopsida serricata*, *Parvilucina tenuisculpta*, and *Macoma* sp. (Windward 2005). The most common gastropod was *Alvania compacta*.

Benthic community sampling was conducted by the Washington State Department of Ecology (Ecology) in the LDW in 2006 (Ecology 2007) to assess the feasibility of using SPI technology to predict chemical impacts on benthic communities in lieu of performing more direct toxicity testing. Community information from stations downstream of Kellogg Island was evaluated to provide an indication of benthic invertebrate assemblages that may be present in the EW. Benthic organisms included polychaetes, molluscs, and crustaceans, in order of abundance. Dominant taxa were similar to those reported in previous studies in the LDW and included the polychaete *Aphelochaeta glandaria*; the molluscs *Axinopsida serricata*, *Macoma carlottensis*, *Nutricola lordi*, and *Parvilucina tenuisculpta*; and the crustacean *Euphilomedes carcharodonta*.

Numerous benthic invertebrate species have been found in Elliott Bay, including polychaetes, crustaceans, molluscs, echinoderms, nemerteans, and cnidarians. A large survey conducted in Puget Sound documented the benthic invertebrates present in both the outer bay and along the shoreline of Elliott Bay (NOAA and Ecology 2000). Data from locations with similar water depths, substrates, and salinities may be relevant to the EW, with the caveat that overall, EW habitats may be subject to more physical disturbance than are those of Elliott Bay. The benthic assemblages in Elliott Bay in habitats comparable to those of the EW tended to exhibit relatively high diversity and included species such as the polychaetes *Lumbrineris californiensis*, *Scoletoma luti*, *Prionospio steenstrupi*; the clams *Axinopsida serricata, Parvilucina tenuisculpta, Nutricola lordi*; and the gastropod *Alvania compacta*. Larger predatory and scavenging crustaceans present in Elliott Bay included Dungeness crab (*C. magister*), rock crab (*Cancer* sp.), sidestripe shrimp (*Pandalopsis dispar*), spot shrimp (*Pandalus platyceros*), humpback shrimp (*Pandalus goniurus*), and pink shrimp (*Pandalus* sp.) (Dinnel et al. 1986). The majority of these species would be expected to be present in the EW, particularly at the mouth, because of similar habitat characteristics and a shared pelagic larval pool, although the densities may be lower than those in Elliott Bay because of physical disturbance from ship traffic.

A.2.2.2.3 Qualitative information on the EW benthic community

A visual assessment of benthic habitats and the benthic invertebrate assemblages in EW was conducted in October 2008 using SPI. This involved the use of a specialized platform-mounted camera that was extended into the surface sediment to take photographs of the sediment surface (plan view) and sediment column (profile view). These images were then analyzed to characterize surface roughness, evidence of physical disturbance, apparent sediment grain size, stratification or layering within the sediment, depth of biological activity and oxygenated zone within the sediment, density of burrows, feeding voids or tubes, and presence of wood waste or other debris. This information was used qualitatively to assess the successional stage[3] of the community and determine whether or not the benthic community was responding to some type of perturbation (physical, chemical, or biological). Three replicate images of approximately the top 20 cm of sediment were collected from 63 stations. The SPI data are presented in Germano & Associates (2009). Selected metrics that describe the benthic habitat condition and the community successional stage are defined and the results presented below.

- **Boundary roughness (physical feature) –** Boundary roughness is the distance (or amplitude) between the highest and lowest mudline elevations within an image. The maximum roughness of the sediment was about 5.1 cm and averaged 1.1 cm. A higher roughness (> ~3 cm) is indicative of physical disturbance, such as current-induced ripples on the sediment bed; a lower roughness (generally < 3 cm) is usually indicative of a biological disturbance of the sediment bed. This information can be used to help characterize the physical regime to which the benthic communities are exposed and the likely community successional stage that may occur in response to the physical environment.

[3] Stage 1 assemblages are early colonizers following some perturbation; Stage 3 organisms represent mature, relatively stable communities; and Stage 2 organisms are transitioning from Stage 1 to Stage 3.

- **Oxygenated layer depth –** The depth of the oxygenated layer (i.e., apparent redox potential discontinuity [aRPD]) is a good indication of where Stage 1 and the majority of Stage 2 organisms may be found in the sediment column. The depth of the oxygenated layer ranged from 0.2 to 5.1 cm below the mudline and averaged 2.1 cm over the entire waterway.
- **Feeding voids –** Feeding voids are oxygenated spaces created by larger, longer-lived organisms that ingest sediment below the sediment surface. These feeding activities are the major contributor to the process known as bioturbation. Evidence of feeding voids was found in about 20% of the locations evaluated (representing 24 sampling locations). The average feeding void depth based on the bottom interval measured was 9.7 cm in fine-grained sediment.
- **Community successional stages –** An examination of successional stage classifications showed that the majority of the locations evaluated were mature (Stage 3) benthic communities, 28% were indicative of transitional communities (Stage 2), and < 1% were indicative of early colonizing communities (Stage 1); Map A.2-1 shows the typical (based on two or more replicates) successional stage at each sampling location.

The results from the SPI survey in the EW show that the benthic community is predominately composed of Stage 3 organisms that represent mature, stable communities. Earlier-stage organisms are present but usually in conjunction with a later-stage community, indicating ongoing recruitment or colonization following small-scale physical disturbances of the surface.

There was no evidence of subsurface methane (indicative of excess organic enrichment) or low dissolved oxygen concentrations in the overlying water at any of the locations sampled. Although the profile images show evidence of historical disturbance and depositional events, the benthic community appears to be representative of a stable and relatively mature community. Overall, with the exception of the locations near the Spokane Street Bridge at the southern end of the waterway, the benthic habitat in the majority of the waterway appears healthy given the habitat constraints and ongoing disturbance that typifies an industrial waterway.

A.2.2.2.4 Biologically active zone

According to Ecology guidance for characterizing surface sediment under the Washington State Sediment Management Standards (SMS), the exposure potential and sediment unit of concern is the "biologically active zone" (often the top 10 cm). Previous studies in Puget Sound have demonstrated that the majority of benthic macroinvertebrates are generally found within the uppermost 10 cm of sediment (Ecology 2008). Although some species may be present at lower depths below the sediment surface, 10 cm is generally assumed to represent a reasonable estimate of the sediment column where most benthic organisms can be exposed to sediment contaminants. SPI data can be used to provide site-specific information on the vertical distribution of benthic macroinvertebrates or the depth to anoxic sediment. Results from the recent SPI survey (Windward 2009a) indicate that the top 10 cm is a reasonable estimate of the biologically active zone in the EW and therefore the vertical extent of benthic invertebrate exposure to contaminants in the sediment of the EW. The aRPD depth (an SPI metric that represents the well-mixed, oxygenated sediment layer) ranged from < 1.0 to 5.1 cm, with an average of 2.0 cm. Individual worm tubes or feeding voids extended below the aRPD in a number of cases to an average depth of 9.0 cm. The maximum depth of any biological activity was recorded as 18 cm; however, feeding voids > 10 cm occurred in less than 15% of the cases.

A.2.2.3 Fish

Fish in the EW can be classified as demersal (living on or near the sediment and feeding on benthic organisms), benthopelagic (living and feeding near the sediment as well as in the water column), or pelagic (living and feeding in open water) (FishBase 2007). Demersal fish are, by definition, in direct contact with sediment during part or all of their lives, whereas benthopelagic and pelagic fish have less direct contact with sediment.

Fish species present in the EW are generally mobile predators and are exposed to chemicals through the ingestion of contaminated prey, incidental ingestion of sediment during prey capture, and uptake of chemicals in surface water through the gills during respiration. Fish are an important food source for other fish, some larger invertebrates, birds, and mammals. Fish from the EW also provide important recreational value and are a source of food for people, including tribal members.

Data on the fish species present in the EW are available from six studies that investigated site use by fish (Table A.2-5). From all 6 studies, 42 fish species have been identified in the EW (Table A.2-6). Taken together, these studies provide a fairly comprehensive characterization of the EW fish community because the range of available habitats was sampled and both active and passive collection methods were used. In addition, sampling using beach seines has been carried out in all seasons, except fall, thus providing an indication of seasonal differences. Furthermore, because trawl, trap, and scuba sampling was conducted during summer, these methods also characterize the season with the greatest productivity and diversity in the fish community in the Duwamish River estuary (Miller et al. 1977a; Dexter et al. 1981; Shannon 2006).

Table A.2-5. Summary of studies assessing the fish community in the EW

Study	Year Completed	EW Location	Sampling Period	Equipment Type	No. of Locations Sampled
EW juvenile Chinook salmon tissue collection data report (Windward 2010d)	2010	Slip 27	June 2009	beach seine	1
EW fish and shellfish tissue collection data report (Windward 2010c)	2008	throughout the EW	August to October 2008	otter trawl, scuba, and crab and shrimp traps	10 trawls, 13 scuba dives, 12 crab traps, 16 shrimp traps
EW fish tissue sampling (Windward 2006b)	2005	throughout the EW	July 20, 2005	otter trawl	9
EW Phase 1 Removal Action Chinook salmon and bull trout monitoring (Taylor Associates 2005)	2004	Slip 27	February 15 to March 1, 2004	beach seine	2
EW channel deepening project, juvenile salmonid and epibenthic prey assessment (Shannon 2006)	2003	Slip 27	April to August 1998, 2000, and 2003 (biweekly)	beach seine	2
EW juvenile Chinook salmon tissue chemistry results (Windward 2002c)	2002	Slip 27	June 2002	beach seine	1

EW – East Waterway

Table A.2-6. Fish species collected in the EW

Common Name	Scientific Name	Environment	Habitat	Source
American shad	*Alosa sapidissima*	marine and freshwater	bays, estuaries, freshwater	Gilbert and Williams (2002)
Bay goby	*Lepidogobius lepidus*	marine	demersal (mostly on mud bottom)	Eschmeyer et al. (1983)
Bay pipefish	*Syngnathus grisiolineatum*	marine	demersal (associated with eel grass in the intertidal areas)	Dawson (1985)
Brown rockfish	*Sebastes auriculatus*	marine	shallow, low-profile, rocky reefs	Gilbert and Williams (2002)
Chinook salmon	*Oncorhynchus tshawytscha*	marine and freshwater	benthopelagic	Groot and Margolis (1998)
Chum salmon	*Oncorhynchus keta*	marine and freshwater	benthopelagic	Groot and Margolis (1998)
Coho salmon	*Oncorhynchus kisutch*	marine and freshwater	benthopelagic	Groot and Margolis (1998)

Table A.2-6. Fish species collected in the EW (cont.)

Common Name	Scientific Name	Environment	Habitat	Source
Crescent gunnel	*Pholis laeta*	marine (estuary)	demersal (intertidal areas, under rocks)	Eschmeyer et al. (1983)
Cutthroat trout	*Oncorhynchus clarki*	marine and freshwater	benthopelagic	Morrow (1980)
Decorated warbonnet	*Chirolophis decoratus*	marine	demersal	Eschmeyer et al. (1983)
English sole	*Parophrys vetulus*	marine (estuary)	benthic (sand and mud bottoms)	Clemens and Wilbey (1961)
Flathead sole	*Hippoglossoides elassodon*	marine	benthic (soft mud bottom, adults below 180 m)	Eschmeyer et al. (1983)
Great sculpin	*Myoxocephalus polyacanthocephalus*	marine	benthic (sand and mud bottoms, often near shore)	Eschmeyer et al. (1983)
Kelp perch	*Brachyistius frenatus*	marine	among fronds in kelp beds from near surface to depths of about 30 m	Gilbert and Williams (2002)
Longfin smelt	*Spirinchus thaleichthys*	marine and freshwater	in sea, usually inshore	Eschmeyer et al. (1983)
Pacific herring	*Clupea pallasi*	marine	benthopelagic (coastal, first year in bays)	Hart (1973)
Pacific sand dab	*Citharichthys sordidus*	marine	over soft sand bottoms	Eschmeyer et al. (1983)
Pacific sandlance	*Ammodytes hexapterus*	marine (brackish)	benthopelagic (surface or burrowed in sand)	Eschmeyer et al. (1983)
Pacific staghorn sculpin	*Leptocottus armatus*	marine (lower estuary, offshore)	benthic (sandy bottom)	Eschmeyer et al. (1983)
Pacific tomcod	*Microgadus proximus*	marine (brackish)	benthic (over sand)	Cohen et al. (1990)
Penpoint gunnel	*Apodichthys flavidus*	marine (estuary)	demersal (intertidal tide pools)	Eschmeyer et al. (1983)
Pink salmon	*Oncorhynchus gorbuscha*	marine and freshwater	benthopelagic	Groot and Margolis (1998)
Plainfin midshipman	*Porichthys notatus*	marine	demersal	Eschmeyer et al. (1983)
River lamprey	*Lampetra ayresi*	marine and freshwater	demersal	Hart (1973)
Rock sole	*Lepidopsetta bilineata*	marine (estuary)	benthic (more pebbly bottom than most other flatfish)	Eschmeyer et al. (1983)
Rockfish	*Sebastes* spp.	marine	demersal (near structure)	Lamb and Edgell (1986)
Saddleback gunnel	*Pholis ornata*	marine (estuary)	demersal (sandy bottom)	Eschmeyer et al. (1983)
Sailfin sculpin	*Nautichthys oculofasciatus*	marine	over rocks from inshore to depths of 110 m, often with algae	Gilbert and Williams (2002)
Sand sole	*Psettichthys melanostictus*	marine, estuary	benthic (sandy bottom)	Hart (1973)
Shiner surfperch	*Cymatogaster aggregata*	marine (estuary)	demersal (in shallow water; around eelgrass beds, piers, and piles; commonly in bays and quiet back waters)	Eschmeyer et al. (1983)

Table A.2-6. Fish species collected in the EW (cont.)

Common Name	Scientific Name	Environment	Habitat	Source
Skate	*Rajidae* sp.	marine	demersal	Eschmeyer et al. (1983)
Slender sole	*Lyopsetta exilis*	marine	benthic (greater than 200 m in depth)	Eschmeyer et al. (1983)
Snake prickleback	*Lumpenus saggita*	marine	benthopelagic (shallow bays and offshore waters)	Eschmeyer et al. (1983)
Speckled sanddab	*Citharichthys stigmaeus*	marine	benthic (sand bottom near shore)	Eschmeyer et al. (1983)
Spiny dogfish	*Squalus acanthias*	marine	benthopelagic	Cox and Francis (1997)
Spotted ratfish	*Hydrolagus colliei*	marine	demersal	Eschmeyer et al. (1983)
Starry flounder	*Platichthys stellatus*	marine (estuary, brackish)	benthic	Morrow (1980)
Steelhead	*Oncorhynchus mykiss*	marine and freshwater	benthopelagic	Gall and Crandell (1992)
Striped seaperch	*Embiotoca lateralis*	marine	demersal	Eschmeyer et al. (1983)
Surf smelt	*Hypomesus pretiosus*	marine (brackish)	benthopelagic	Morrow (1980)
Three-spine stickleback	*Gasterosteus aculeatus*	marine and freshwater	benthopelagic (in/near vegetation)	Page and Burr (1991)
Whitespotted greenling	*Hexagrammos stelleri*	marine (intertidal)	demersal (nearshore near rocks, piles, and eelgrass beds)	Cohen et al. (1990)

EW – East Waterway

The most extensive surveys of fish populations in the EW have been conducted for the Port of Seattle by Taylor et al. (1999) using beach seines, which tend to capture small fish in nearshore habitats. Taylor et al. collected fish at the head and mouth of Slip 27 in 1998, 2000, 2002, and 2003. Sampling was conducted in April through August 1998, April through October 2000 and 2002, and February through April 2003. Additional sampling was conducted February 15 through March 2, 2004, at Slip 27 and nearby locations (Taylor Associates 2005). Twenty-two species of fish were captured during these studies. The top three numerically dominant species at the Slip 27 sampling location were juvenile chum salmon (*Oncorhynchus keta*), juvenile Chinook salmon (*Oncorhynchus tshawytscha*), and shiner surfperch (*Cymatogaster aggregata*). Together, these species represented 98% of the total catch at Slip 27. Additional species commonly captured in beach seines included juvenile coho salmon (*Oncorhynchus kisutch*), Pacific staghorn sculpin (*Leptocottus armatus*), Pacific herring (*Clupea pallasi*), surf smelt (*Hypomesus pretiosus*), and three-spine stickleback (*Gasterosteus aculeatus*).

Trawling throughout the EW was conducted one day each in July 2005 and September 2008 to capture fish for tissue sampling (Windward 2006b, 2010c). Trawling captured moderately slow-moving benthic species of fish from intertidal to deep subtidal depths. In 2005, 17 fish species were captured in 9 trawls, and in 2008, 23 fish species were captured in 10 trawls. English sole (*Parophrys vetulus)* was the most abundant species in both efforts and constituted more than 50% of the total catch in 2005 and more than 40% of the total catch in 2008. Pacific tomcod (*Microgadus proximus*), rock sole (*Lepidopsetta bilineata*), sand sole (*Psettichthys melanostictus*), and shiner surfperch were also abundant in both trawling events, with a catch-per-unit-effort (CPUE) greater than or equal to three individuals per trawl. Sanddab (*Citharichthys species*), Pacific staghorn sculpin, starry flounder, and Pacific herring were also common in both trawling events, with a CPUE greater than one individual per trawl. In 2005 surf smelt were also common, whereas in 2008 plainfin midshipman, bay goby, rat fish, and speckled sanddab were common, with a CPUE greater than one individual per trawl.

A few fish (Pacific staghorn sculpin and brown rockfish) were collected in traps set for collecting crabs and shrimp on August 26 and 27, 2008 (Windward 2010c). Scuba divers collected brown rockfish over 3 days in August and during 1 day in October 2008 at 13 locations throughout the EW. Brown rockfish were the only rockfish species encountered, and the divers noted that they were common in riprap habitat.

In the LDW, 53 resident and non-resident fish species were captured during LDW RI sampling events (Windward 2004a, 2005, 2006a). In earlier studies, Warner and Fritz (1995) recorded 33 resident and seasonal fish species, Miller et al. (1975; 1977a) observed a total of 29 species, and Matsuda et al. (1968) recorded a total of 28 species. Dominant species were similar to those observed in the EW, with shiner surfperch, snake prickleback (*Lumpenus saggita*), Pacific sandlance (*Ammodytes hexapterus*), Pacific staghorn sculpin, longfin smelt (*Spirinchus thaleichthys*), English sole, and starry flounder being particularly abundant, as were juvenile Chinook, chum, and coho salmon. Fish numerical abundance reached its maximum in late summer to early fall and was generally lowest in winter (Miller et al. 1977a; 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. 1977a). The following subsections detail the dominant species likely to be encountered in the EW.

A.2.2.3.1 Anadromous salmonids – Pacific salmon

Five species of juvenile salmon (Chinook, chum, coho, pink [*Oncorhynchus gorbuscha*], and steelhead) have been documented in the EW. Juvenile chum and Chinook salmon were the most abundant salmonid species captured in Slip 27 (Taylor Associates 2004; Shannon 2006; Windward 2010d). Sockeye salmon (*Oncorhynchus nerka*) have been found in the EW or LDW (Kerwin and Nelson 2000).

Salmon use the Duwamish River for rearing and as a migration corridor for adults and juveniles. Adult salmon found in the LDW and EW spawn mainly in the middle reaches of the Green River and its tributaries (Grette and Salo 1986). Among the beneficial uses identified for the Duwamish Waterway (including the LDW and EW), habitat for outmigrating juvenile salmonids is one of the most important (Harper-Owes 1983). The peak timing of outmigration for juveniles of all salmon species generally corresponds with March-to-June high flows. Peak outmigration usually lasts from mid-July through early August for most species (Warner and Fritz 1995; Nelson et al. 2004). In the EW, juvenile salmon were caught in seine nets from April through September, with peak numbers in April through July (Shannon 2006). During that time, juveniles complete their physiological adaptation to higher salinity, and they use the estuary to feed on epibenthic and neritic food sources (Salo 1991). As the juveniles move into estuaries and inhabit deeper water, their dietary preference appears to shift toward water column organisms such as larval and juvenile fish (Healey 1991). No specific information is available on their residence time in the EW.

A.2.2.3.2 Non-salmonid fish

Of non-salmonid fish, English sole, Pacific tomcod, rock sole, sand sole, shiner surfperch, sanddab species, Pacific staghorn sculpin, starry flounder, surf smelt, three-spine stickleback, and Pacific herring are at least seasonally abundant in the EW. Pacific herring, Pacific sandlance, surf smelt, and longfin smelt were encountered infrequently in recent beach seine and trawl samples in the EW but occasionally were present in large numbers (Shannon 2006; Windward 2006b, 2010c). Three-spine stickleback were abundant in monthly beach seine samples collected from both Slip 27 and Kellogg Island sampling locations (Shannon 2006). Longfin smelt abundance was highest in the summer, fall, and early winter based on historical otter trawl data from the LDW (Miller et al. 1977a). Miller et al. (1977a) suggested that the fall-winter peak abundance period (with 80- to 115-mm-long fish) may have represented part of a spawning run and that the late summer peak (with 30- to 50-mm-long fish) may have represented downstream migrant young-of-the-year individuals. Pacific herring were reported in purse seine samples from the LDW in May, June, July, November, and December (Shannon 2006; Weitkamp and Campbell 1980) and were present in trawl samples in the EW in July and September (Windward 2006b, 2010c). In Puget Sound, three-spine stickleback and surf smelt feed on both epibenthic and pelagic invertebrates; epibenthic invertebrates constitute a slight majority of their diet (Miller et al. 1977b; 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. 1977b; Fresh et al. 1979). Pacific tomcod is a demersal species that is associated with sandy bottoms (Cohen et al. 1990); they primarily feed on amphipods and shrimp (Fresh et al. 1979).

In the LDW, shiner surfperch abundance peaks in summer during the bearing of young (Miller et al. 1975). Taylor Associates recorded abundant shiner surfperch in the EW and LDW in May through October, with peak abundance in July (Shannon 2006). Shiner surfperch are opportunistic omnivores, feeding primarily on benthic invertebrates, including polychaetes, molluscs, and other benthic organisms (Fresh et al. 1979; Wingert et al. 1979; Miller et al. 1977b). Shiner surfperch are also noted to feed on zooplankton, small crustaceans, algae, and detritus (Gordon 1965; Bane and Robinson 1970).

English sole were the most abundant fish captured in recent trawl sampling of the EW, constituting approximately 50% of the total catch (Windward 2006b, 2010c). In Puget Sound, adult English sole are typically found on soft sand or mud bottoms at depths of 80 to 150 ft (25 to 50 m) (Smith 1936). English sole may exist in discrete populations with some site fidelity. Day (1976) conducted a tagging study in Puget Sound, the results of which 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 (Day 1976).

English sole migrate 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 1999) reported the offshore 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 1999). Data from Malins et al. (1982) indicated that during the winter and spring, more than 50% of the English sole in the LDW were juveniles (less than 150 mm standard length). Juvenile English sole (those less than 110 mm long) ingest annelids (Smith 1936), copepods, amphipods, and molluscs (Holland 1954). Adult English sole studied in Puget Sound ingest clams, clam siphons, small molluscs, marine worms, small crabs, and small shrimp (Fresh et al. 1979; Wingert et al. 1979).

Rock sole was among the most common species of fish captured in recent trawl sampling in the EW, while starry flounder was somewhat less common (Windward 2006b, 2010c). Similar to English sole, starry flounder and rock sole are also noted to migrate from shallow water and estuaries during the summer to deeper water in the winter (Morrow 1980; NOAA 2008). Young and adult starry flounder are tolerant of freshwater (Morrow 1980). Rock sole tend to be found on rocky or gravel substrates but are also found on sand and mud bottoms (NOAA 2008). Because they have larger mouths, starry flounder and rock sole are capable of consuming somewhat larger organisms than those consumed by English sole, although their diets greatly overlap. Starry flounder and rock sole in Puget Sound were reported to consume primarily benthic invertebrates, with bivalves, amphipods, and shrimp serving as important prey items for starry flounder, and polychaetes, amphipods, and bivalves being the primary prey for rock sole (Fresh et al. 1979).

Other flatfish that were common in recent EW trawl sampling included the Pacific sanddab and sand sole (Windward 2006b, 2010c). Pacific sanddab are found on sand and mud bottoms and consume a mixture of benthic invertebrate and pelagic invertebrate prey (Fresh et al. 1979). Sand sole are found over sandy bottoms and consume primarily fish from the water column, such as shiner surfperch (Love 1996).

The highest trophic-level fish species identified in the EW included brown rockfish (*Sebastes auriculatus*), Pacific staghorn sculpin, Pacific tomcod, spotted ratfish (*Hydrolagus colliei*), spiny dogfish (*Squalus acanthias*), sand sole, great sculpin (*Myoxocephalus polyacanthocephalus*), and starry flounder (Windward 2006b, 2010c). Dietary studies from Puget Sound showed that fish constitute a large fraction of the diets of sand sole, brown rockfish, spiny dogfish, and great sculpin, whereas the other species consume primarily invertebrates and are at a lower trophic level (Miller et al. 1977b; Wingert et al. 1979; Fresh et al. 1979). Spiny dogfish are expected to have home ranges that extend well beyond the EW. Great sculpin are rare in the EW (Windward 2006b, 2010c). Wingert et al. (1979) reported that brown rockfish from central Puget Sound primarily consume caridiean shrimp and fish. Tagging studies showed that brown rockfish demonstrated limited movement with home ranges on the order of 30 to 1,500 m^2 (Matthews 1990b). Brown rockfish are associated with structures such as riprap, piers, or submerged debris (Matthews 1990b; Love et al. 2002). During scuba sampling in the EW, brown rockfish were found to be common under piers in riprap habitats (Windward 2010c).

A.2.2.4 Birds

There is relatively little EW-specific information on bird populations. Surveys of the bird community have been conducted primarily upstream of the EW in the LDW, where there is a greater diversity of bird habitat. Formal studies, field observations, and anecdotal reports indicate that up to 87 species of birds use the LDW during at least part of the year to feed, rest, or reproduce (Table A.2-7). The relatively large home ranges associated with many bird species make the LDW data relevant to the EW, although the number of species dependent upon riparian, intertidal, and shallow water habitat are likely fewer because those habitats are limited in the EW. Instead, birds that feed in the pelagic zone or dive in deeper waters to feed on benthic fish and invertebrates are more likely to frequent the EW. The Puget Sound area is within the Pacific flyway, a major route of travel for migratory birds in the Americas that extends from Alaska in the north to Patagonia in the south. Water bodies and wetlands are important to birds for feeding and resting during their migratory travels (Page et al. 1992).

Table A.2-7. Bird species that use 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*
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	*Accipiter 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*

Table A.2-7. Bird species that use the LDW (cont.)

Common Name	Scientific Name	Common Name	Scientific Name
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*

Source: Windward (2007c)
LDW – Lower Duwamish Waterway

This section provides a general description of birds expected to use the EW based on formal surveys or other types of observations conducted in the LDW upstream of the EW, or based on informal observations of birds in the EW. No studies of bird populations have been conducted in the EW. Formal surveys conducted in the LDW include a year-round survey conducted of the entire waterway in 1977–1978 (Canning et al. 1979) and a monitoring study conducted over 14 seasons at 3 general areas of the LDW (Terminal 105 [T-105], Kellogg Island, and the Upper Turning Basin) between 1995 and 2000 (Cordell et al. 2001). Passerine/upland birds, raptors, shorebirds/waders, waterfowl, and seabirds are described in the subsections that follow.

A.2.2.4.1 Passerine/upland birds

Passerine and upland bird species that have been observed during surveys of the LDW are generally associated with terrestrial habitats, although they may occasionally forage in exposed mudflats or freshwater habitats (Canning et al. 1979). Therefore, these species are not expected to frequent the EW. Passerine and upland species that have been observed along the EW include northwestern crow (*Corvus corrinus*), rock pigeon (*Columba livia*), European starling (*Sturnus vulgaris*), English (house) sparrow (*Passer domesticus*), and belted kingfisher (*Ceryle alcyon*).

A.2.2.4.2 Raptors

Osprey (*Pandion haliaetus*) and bald eagle (*Haliaeetus leucocephalus*) have been observed along or in the vicinity of the EW. Two osprey nest boxes have been observed along the EW at Terminal 104 (T-104) and Terminal 18 (T-18) (Blomberg 2007). In 2006, WDFW reported 10 osprey nest sites located along the LDW, in addition to the nests along the EW (Thompson 2006). Osprey feed almost exclusively on fish captured from the water surface by hunting over open water (Poole et al. 2002). Overwintering migrant eagles have been routinely observed in the vicinity of the LDW from the beginning of October through late March (King County 1999). Five bald eagle nests within 8 km of the EW were occupied in 1999 (King County 1999). The closest nest is located in West Seattle, within 1.6 km of the EW. Bald eagles feed primarily on fish but may also feed on waterfowl during winter months (Buehler 2000). Other raptors in the LDW (e.g., merlin [*Falco columbarius*] and several species of hawks) feed primarily on upland birds or rodents and are not substantially exposed to aquatic species from the EW. Peregrine falcons have been known to nest along the LDW (Anderson 2006). Peregrine falcons in western Washington feed primarily on rock pigeons and European starlings, although they may also ingest some waterfowl (Anderson 2006).

A.2.2.4.3 Shorebirds/waders

Of the nine species of shorebirds and wading birds that have been documented in the LDW during monitoring studies conducted by Cordell et al. (2001), great blue heron (*Ardea herodias*) was the most abundant species recorded; great blue heron have also been observed using the EW. The closest great blue heron colonies are located about 14 km south of the EW in Renton, Washington, and 10 km northwest near Salmon Bay. A colony of up to 37 active great blue heron nests was located in West Seattle a few hundred meters from Kellogg Island until 1999, but the nests were abandoned in 2000 (Norman 2002a). Great blue heron feed in shallow water, primarily on fish, but they may also consume benthic invertebrates (Butler 1992). Other common shorebirds observed in the LDW were spotted sandpiper (*Actitis macularia*) and killdeer (*Charadrius vociferous*). Sandpipers probe in the sediment while feeding on benthic invertebrates in intertidal areas, resulting in exposure to sediment contamination through incidental sediment ingestion. The small amount of shallow water and intertidal habitat in the EW is likely to limit the use of the EW by shorebirds and waders, as discussed in more detail in Section A.2.3.3.

A.2.2.4.4 Waterfowl

Waterfowl species commonly observed in the EW include common and red breasted merganser (*Mergus merganser* and *Mergus serrater*, respectively), Barrow's goldeneye (*Bucephala islandica*), Canada goose (*Branta canadensis minima*), and bufflehead (*Bucephala albeola*). Cordell et al. (2001) and Canning et al. (1979) observed 20 waterfowl species during monitoring studies conducted in the nearby LDW. In general, the waterfowl species observed in the EW and along the LDW overwinter in the Puget Sound area (and farther south) and migrate north in the summer, although there are some non-migratory populations. The Puget Sound area is within the Pacific flyway, a major route of travel for migratory birds in the Americas that extends from Alaska in the north to Patagonia in the south. Water bodies and wetlands are important to birds for feeding and resting during their migratory travels (Page et al. 1992). Bufflehead, Barrow's goldeneye, and common and red-breasted merganser are species that eat benthic invertebrates and fish and dive deeper than other ducks; these species are more likely to use the EW for foraging than are other duck species. Dabbling ducks feed primarily on aquatic plants, seeds, and grasses, and Canada geese feed on grass and terrestrial vegetation; habitat types for these waterfowl species are generally lacking in the EW.

A.2.2.4.5 Seabirds

Seabirds observed using the EW include pelagic and double-crested cormorants (*Phalacrocoranx pelagicus* and *Phalacrocorax auritus*, respectively), pigeon guillemot (*Cepphus columba*), grebes (especially Western grebe [*Aechmophorus occidentalis*]), and gulls (especially glaucous-winged gull [*Larus glaucescens*]). Sixteen species of seabirds were documented in the nearby LDW by Cordell et al. (2001) and Canning et al. (1979).

Pigeon guillemot nests have been observed under the T-18 piers (Hotchkiss 2007), and the birds have been observed feeding in the EW (Musgrove 2010a). Pigeon guillemot are present in the Puget Sound region year-round (Seattle Audubon Society 2008). 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). Grebes arrive from October to November and depart by early May. Several species of gulls use the LDW and EW; glaucous-winged and mew gulls (*Larus canus*) are the only species reported to use the area in large numbers.

Pelagic cormorants and pigeon guillemot are both deep divers and feed primarily on bottom-dwelling fish, but may also consume some benthic invertebrates (Ewins 1993; Hobson 1997). Double-crested cormorants feed primarily on fish in shallower waters. Western grebe feed primarily on fish (Storer and Nuechterlein 1992). Gulls are omnivorous scavengers, consuming a wide variety of fish and shellfish.

A.2.2.5 Mammals

There is very little information on mammal populations in the vicinity of the EW or the LDW. The relatively large home ranges associated with many mammal species make the LDW data relevant to the EW.

Three marine mammal species enter the EW and LDW from Elliott Bay: harbor seal (*Phoca vitulina*), California sea lion (*Zalophus californianus*), and harbor porpoise (*Phocoena phocoena*) (Dexter et al. 1981). Harbor seals and California sea lions have been observed in the EW (Walker 1999). Recent information on harbor porpoise use was not available, although it has been noted that they occasionally enter the LDW (Dexter et al. 1981).

A survey was conducted to monitor for the presence of California sea lions and harbor seals in the EW on 30 individual days between December 1998 to June 1999 (Walker 1999). California sea lions were observed on 8 days, and harbor seals were observed on 1 day. California sea lions, harbor seals, and harbor porpoise are opportunistic feeders, consuming various fish species depending on availability (Marine Mammal Center 2002; Pitcher 1980; Pitcher and Calkins 1979; Schaffer 1989). Harbor seals may also feed on invertebrates such as squid, and California sea lions and harbor porpoises may also feed on squid and octopus.

Three species of aquatic-dependent terrestrial mammals use the LDW: raccoon (*Procyon lotor*), muskrat (*Ondatra zibethicus*), and river otter (*Lutra canadensis*). Raccoons are reported to be common along the forested ridge slopes to the west of the LDW, but information is not available regarding their presence in the EW. Raccoons are scavengers that feed on carrion and occasionally on fish and invertebrates. Muskrat populations have been reported to exist in the LDW at Terminal 107 (T-107) (near Kellogg Island) and at the Upper Turning Basin (approximately 5 miles upstream from Harbor Island (Canning et al. 1979). Muskrats are herbivores, feeding primarily on aquatic and semi-aquatic plants. The EW has limited aquatic and semi-aquatic plant populations because of limited shallow water habitat, so muskrats are less likely to use the EW habitat. Anecdotal information indicates that a river otter family lives year-round on Kellogg Island in the LDW, and a mother and her young have been observed feeding among the pilings in the WW (Musgrove 2010b). Local river otters feed primarily on fish but will also feed on crabs, mussels, and clams (Strand 1999).

A.2.3 Receptor of Concern Selection

This section presents the ROCs selected to represent benthic invertebrate, fish, bird, and mammal species evaluated in the ERA based on a set of key considerations. It would not be practical to evaluate risks to all species in the EW individually because of the large number of species present. Therefore, representative species were chosen as ROCs using a systematic process 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 used in Superfund ERAs.

Key considerations in the selection of ROCs included:

- Potential for direct or indirect exposure to sediment-associated chemicals
- Human and ecological significance
- Site use
- Sensitivity to chemicals at the site
- Susceptibility to biomagnification of chemicals (i.e., higher-trophic-level species)

This section provides the rationale for each of the ROCs selected based on these key considerations. To ensure that ROCs were selected to represent all important exposure pathways for sediment-associated chemicals, key direct and indirect exposure routes from sediment were identified (e.g., direct exposure to sediment or ingestion of prey associated with sediment). Groups of organisms that may be exposed via these pathways were then identified, and representative species expected to be most exposed were selected from these groups in order to represent the greatest potential for exposure. Next, human or ecological significance was considered (i.e., species valued by society, species with a special regulatory status [e.g., threatened or endangered], or species that serve a unique ecological function). ROCs that were selected for the LDW

ERA were preferred for the EW ERA, unless the EW habitat did not support those species.

Site use and sensitivity to chemicals often detected at the site were also evaluated to determine the final list of ROCs. Site use is an important consideration because it determines the exposure of a species; species that occupy the EW during a significant part of the year or during sensitive periods, such as gestation and rearing of young, were preferred. Sensitivity to chemicals was evaluated based on available toxicological data; although in many cases, the availability of toxicological data specific to species residing in the EW was limited. Therefore, where necessary, toxicological information from surrogate species, or a wide range of species, was used because species-specific data were not available.

Finally, susceptibility to biomagnification because of trophic status, which results in higher exposure to chemicals that biomagnify (e.g., polychlorinated biphenyls [PCBs]), was considered in selecting ROCs based on an understanding of the trophic relationships among the animals living in or feeding from the EW. Organisms at higher trophic levels are likely to have a higher exposure to bioaccumulative chemicals than are receptors lower in the food web because bioaccumulative chemicals increase in concentration in higher trophic-level prey. In marine and estuarine food webs, the lowest trophic-level organisms are primary producers (i.e., those that rely on photosynthesis for energy), such as phytoplankton, algae, and aquatic plants (Figure A.2-1). Primary consumers, such as some amphipods, clams and mussels, some of the diving ducks, and muskrats, are herbivores and feed almost exclusively on plants. Organisms that feed at the highest trophic level in marine and estuarine food webs prey primarily on fish; examples of these organisms in the EW include brown rockfish, sand sole, osprey, eagle, river otter, and marine mammals (Figure A.2-1). Many organisms present in marine and estuarine environments and in the EW are omnivores and feed at multiple trophic levels or change trophic status with life stage or size.

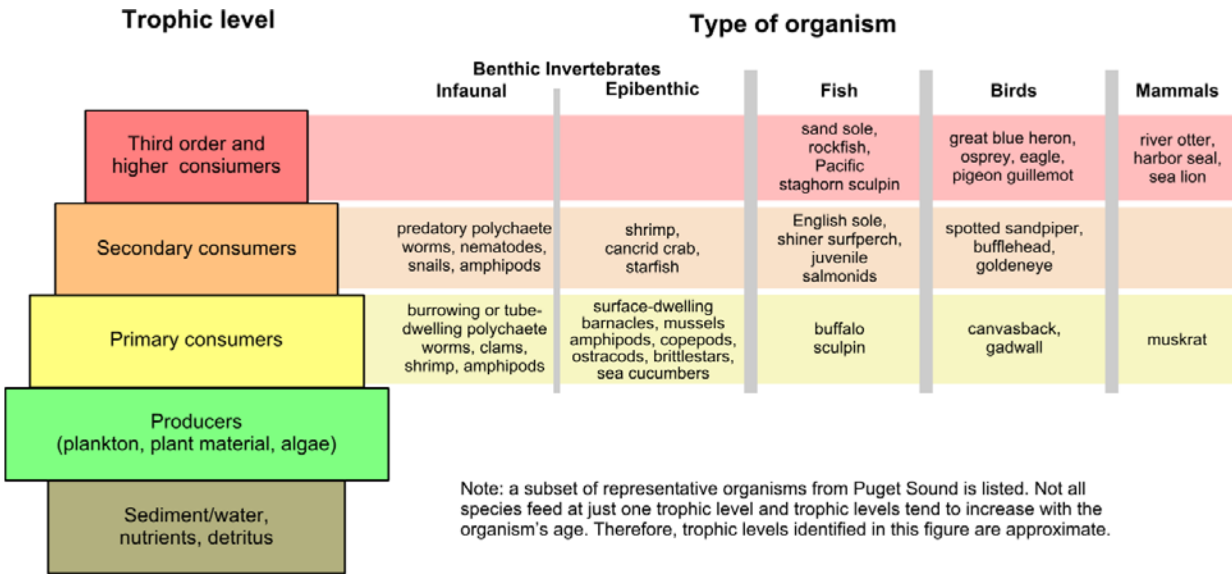

Figure A.2-1. Trophic levels of select organisms in Puget Sound estuaries

A.2.3.1 Benthic organisms

The benthic invertebrate community, as a whole, and cancrid crab were selected as benthic ROCs. Sessile benthic invertebrate assemblages are directly exposed to sediment and integrate both long- and short-term exposures. Numerous benthic species are known to be sensitive to the effects of a number of chemicals, including metals and polycyclic aromatic hydrocarbons (PAHs). In addition, sampling protocols and analytical techniques to assess the health of benthic assemblages are well established. Benthic invertebrates are present throughout the EW and tend to be representative of a mature benthic community (Windward 2010b). Benthic invertebrates are important prey items for ROCs, including fish, birds, and mammals.

Cancrid crabs were selected because they are ecologically and recreationally important and have a higher trophic level than do many other benthic invertebrates. Although no individual crab is likely to be a long-term resident of the EW (the species often exhibits seasonal use of shallow habitats as adults and select protected environments as juveniles), cancrid crab are present in the EW. Red rock crab were the most abundant crab collected during the 2008 sampling effort in the EW, but graceful crab and Dungeness crab were also collected (Windward 2010c). Cancrid crab of any species collected in the EW were considered an ROC because they all have similar exposure regimes.

A.2.3.2 Fish

For the purpose of ROC selection, the fish community was grouped into the following four broad categories, based on habitat use, life stage, and trophic level, to represent their potential sediment exposure at the site:

- **Anadromous juvenile salmonids** – representing juvenile salmon that commonly feed on plankton and epibenthic organisms in shallow, nearshore areas of estuaries during their outmigration; includes juvenile Chinook, chum, and coho salmon. Juvenile Chinook salmon are also listed as a threatened species under the ESA.
- **Planktivorous fish** – representing fish that live primarily in the water column and feed primarily on water column or encrusting organisms; includes Pacific herring, pile perch, surf smelt, longfin smelt, and three-spine stickleback.
- **Benthivorous fish** – representing benthic fish that live on or near the sediment and primarily feed on infaunal and epifaunal benthic invertebrates; includes English sole, rock sole, starry flounder, and shiner surfperch. Fish in this category have a greater potential for exposure to sediment-associated chemicals than fish such as Pacific herring and pile perch, which prey on lower-trophic-level water column and encrusting organisms.
- **Upper-trophic-level fish** – representing higher-trophic-level fish that have multiple exposure pathways and a higher potential to be exposed to bioaccumulative chemicals; these include brown rockfish, sand sole, and Pacific staghorn sculpin.

Based on the key considerations outlined in Section A.2.3, the following three fish species were selected as ROCs to represent the four broad categories of fish in the EW:

- Juvenile Chinook salmon
- English sole (representative of benthivorous fish and protective of planktivorous fish)
- Brown rockfish (representative of upper-trophic-level fish)

The subsections that follow discuss the rationale for selecting each fish ROC and how these species serve as surrogates for the protection of other similar and important species in the EW.

A.2.3.2.1 Juvenile Chinook salmon

Juvenile Chinook salmon were selected primarily because the Puget Sound evolutionary significant unit of Chinook salmon (to which the Green River, LDW, and EW belong) is a federally threatened species under the ESA. In addition, juvenile Chinook salmon serve as a surrogate for other juvenile anadromous salmon. Based on beach seine data, juvenile Chinook salmon are also among the most abundant fish in the EW during their spring outmigration (Shannon 2006) and are an important prey item for birds, piscivorous fish (Davis 2007; Warner and Fritz 1995), and possibly marine mammals. Residence times of all species of juvenile salmonids in the EW 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 salmon were also selected as an ROC for the LDW ERA (Windward 2007c).

Juvenile Chinook salmon 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. Whole-body and stomach contents tissue chemistry data are available to characterize their exposure within the EW and just upstream in the LDW. 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 the selection of juvenile Chinook salmon as an ROC over other juvenile salmonids. Chinook salmon are also culturally and economically important in the Pacific Northwest. Adult Chinook salmon have been used for centuries by indigenous people as a primary food source and are an economic resource for the region as a commercial fishery species.

A.2.3.2.2 English sole

English sole were selected to represent benthivorous fish and to be protective of planktivorous fish in the EW. English sole live in close proximity to sediment and thus have a high potential for direct exposure to sediment-associated chemicals. In addition, English sole feed extensively on infaunal and epifaunal invertebrates and thus are exposed to sediment-associated chemicals through their diet. Based on trawl data, English sole are one of the most abundant fish in the EW (Windward 2006b, 2010c). English sole were also selected as an ROC for the LDW ERA (Windward 2007c).

As discussed in Section A.2.2.3.2, English sole may exist in discrete populations with some site fidelity (Day 1976); however, home ranges of English sole in the EW likely extend beyond the boundaries of the EW. A few home range estimates have been developed for English sole using best professional judgment; these include a 9-km^2 home range, as reported in the Puget Sound Dredged Disposal Analysis (PSDDA) report (PSDDA 1988) and a 2-km^2 home range based on a literature review (Stern et al. 2003).

English sole whole-body tissue chemistry data are available to characterize their exposure within the EW. A number of studies (e.g., Johnson et al. 1997) have examined the potential effects of sediment-associated chemicals (e.g., PAHs) on flatfish in the LDW, particularly English sole. Several toxicology studies have used data from English sole collected in the LDW upstream of the EW, near Kellogg Island (Casillas et al. 1991; Johnson and Landahl 1994; Johnson et al. 1988; 1997; 1998; 1999; Kubin 1997; Malins et al. 1984; 1985a; 1985b; Schiewe et al. 1989). National Marine Fisheries Service (NMFS) data suggest that English sole are as sensitive to the effects of PAHs as other flatfish species tested (Myers et al. 1998). Available toxicity data are not sufficient to suggest that English sole are more or less sensitive than other EW species represented by English sole. Therefore, except for regionally specific studies conducted with English sole, no preference has been given to toxicological data for fish closely related to English sole. English sole are caught recreationally in the EW and have some value as a commercial fishery species in northern Puget Sound. The south Puget Sound fishery was closed in 1989 as a result of declining abundance (Pallson 2001); however, multi-agency efforts to restore Puget Sound are expected to result in increasing abundances of many declining species. Puget Sound-wide restoration efforts could increase English sole abundance to a point where a viable south Puget Sound fishery is possible.

English sole is a surrogate for other benthopelagic, pelagic, and demersal fish species. In general, benthic organisms preyed on by other fish in the EW are similar to those preyed on by English sole, so its primary exposure route to sediment-associated chemicals is similar to those of other fish with similar diets. English sole likely has higher exposure to sediment-associated chemicals than other demersal fish because it prefers to live on fine-grained sediment, which tends to accumulate chemical contaminants more readily than does coarse-grained sediment. English sole also likely has a relatively smaller home range than other demersal benthivores at a similar trophic level, resulting in relatively greater site-specific exposure. Therefore, exposure of English sole to sediment-associated chemicals is assumed to be greater than or similar to that of fish with similar habitat and prey preferences (e.g., starry flounder, rock sole, and sanddab species).

Other EW fish, such as pile perch, ingest organisms that encrust pilings and other hard structures. However, because these prey organisms do not have direct contact with sediment, this exposure route is not likely to result in a greater exposure to sediment-associated chemicals than would the ingestion of benthic invertebrates. Similarly, other EW fish species at a trophic level similar to that of English sole, such as herring, surf smelt, longfin smelt, and three-spine stickleback, which ingest significant quantities of pelagic prey, are likely to have less exposure to sediment-associated chemicals than do English sole, which consume benthic prey exclusively. English sole are also present in the EW year-round, except during spawning migrations; therefore, these other fish are not likely to have a higher residence time in the EW than English sole. The available information thus indicates that the assessment of risks associated with exposure to sediment-associated chemicals for English sole will be protective of fish with benthopelagic, demersal, and benthic habitat preferences.

A.2.3.2.3 Brown rockfish

Brown rockfish were selected to represent upper-trophic-level fish in the EW. Brown rockfish are long-lived demersal fish that feed on more fish and larger invertebrates than do English sole, thus increasing their potential exposure to bioaccumulative and biomagnifying chemicals, such as mercury and PCBs. Upper-trophic-level fish may have higher body burdens of biomagnifying chemicals than do lower-trophic-level fish, such as English sole, which ingest primarily invertebrates. In addition, because brown rockfish are long-lived compared to some other upper-trophic level fish in the EW, they can be exposed for longer periods of time, and thus have greater potential to bioaccumulate chemicals over time.

Brown rockfish are noted to be relatively sedentary, with home ranges that vary from 30 m^2 or less on artificial and high-relief reefs to 90 to 1,500 m^2 on low-relief reefs where bull kelp is present (Matthews 1990a). Their home range in the EW is uncertain because the availability of such habitats in the EW is uncertain. Based on reported habitat preferences (Love 1996; Matthews 1990a), brown rockfish in the EW are likely to be associated with pier structures, riprap, or other debris (e.g., old tires), and they have been observed under piers in the EW (Windward 2010b).

Other piscivorous fish, including quillback rockfish, copper rockfish, Pacific staghorn sculpin, and sand sole, are also upper-trophic-level species that have been observed in the EW or close by. It is believed that sand sole foraging ranges likely extend beyond the EW and that brown rockfish feed at a higher trophic level than do Pacific staghorn sculpin. Based on habitat preferences, diver observations, and trawl data, brown rockfish are more abundant than copper or quillback rockfish in the EW, so brown rockfish better represent exposure in the EW than do these other piscivores.

A.2.3.3 Wildlife

Potential aquatic-dependent wildlife ROCs were considered for selection from the following four categories:

- **Piscivorous birds** – representing birds that consume primarily fish; includes osprey, great blue heron, cormorants, western grebe, and bald eagle.
- **Piscivorous/benthivorous birds** – representing birds that consume both fish and benthic invertebrates; includes pigeon guillemot and merganser.
- **Benthivorous birds** – representing birds that consume primarily benthic invertebrates; includes spotted sandpiper, bufflehead, and goldeneye.
- **Piscivorous mammals** – representing mammals that consume primarily fish; includes river otter, harbor seal, and sea lion.

These categories were considered because representative species are expected to have higher dietary exposures to chemicals than do other bird and mammal species that may be present in the EW because prey items of representative species have higher trophic statuses or are more closely associated with sediment.

Other bird and mammal species, such as herbivorous birds (e.g., geese), passerine birds (e.g., pigeons), or omnivorous mammals (e.g., raccoon), are assumed to be less exposed to chemicals in the EW than are those listed above because of their foraging behavior and diet. Birds that are primarily herbivorous, such as geese and some dabbling ducks, may also consume a small amount of benthic invertebrates and may incidentally ingest sediment while foraging, but to a lesser extent than benthivorous birds that feed primarily on benthic invertebrates. Most passerine birds are likely to experience limited exposure to contaminated sediment in the EW because they forage primarily in upland habitats. Other mammals, such as raccoons, are expected to have less exposure to sediment-associated chemicals because their food is largely terrestrial in origin, especially as compared with the food of primarily piscivorous mammals, such as river otter and harbor seal.

Based on the key considerations outlined in Section A.2.3, which are discussed more thoroughly below, the following four wildlife species were selected as ROCs in the EW:

- Osprey – piscivorous birds
- Pigeon guillemot – piscivorous/benthivorous birds
- River otter – piscivorous semi-aquatic mammals
- Harbor seal – piscivorous marine mammals

An ROC was not selected to represent exclusively benthivorous birds because there is very little intertidal and shallow habitat in the EW to support the birds in this category (spotted sandpiper, bufflehead, and goldeneye). Pigeon guillemot are expected to have higher exposure than benthivorous birds because a higher proportion of its diet is likely to be obtained from the EW, as discussed in more detail in Section A.2.3.3.2.

A.2.3.3.1 Osprey

Osprey (*Pandion haliaetus*) was selected to represent piscivorous birds; osprey was also selected as an ROC for the LDW ERA. Osprey are generally present in Washington from late March or early April to August or September. Osprey are known to nest along the EW, with one nest located at T-104 and one at T-18 (Blomberg 2007). Osprey also nest along the LDW just south of the EW. Osprey prefer to feed close to their nests during fledgling development. During a survey by the US Fish and Wildlife Service (USFWS) and US Geological Survey (USGS) in April and May 2006, osprey nesting at T-104 and T-18 were observed to capture 67 and 15%, respectively, of their prey from the LDW and the remainder from Elliott Bay or Lake Washington (Davis 2007). Osprey are particularly sensitive to pesticides, which cause eggshell thinning. Exposure to pesticides through the mid-1970s resulted in the drastic reduction of osprey populations. In the 1970s, following the reduced use of many pesticides, most populations increased rapidly. Because osprey nest along the EW, they are exposed to EW chemicals during the sensitive reproductive period and are thus more susceptible to adverse effects than are other piscivorous birds that winter in the area but migrate elsewhere for breeding. Osprey generate high human interest and are protected under the Migratory Bird Treaty Act.

The osprey was selected as an ROC rather than the bald eagle, which also breeds in the area, is susceptible to eggshell thinning, and is listed as a threatened species. Both the osprey and bald eagle are exposed during sensitive reproductive stages, but osprey were selected because of their higher incidence in the EW, their smaller foraging ranges, and their higher ingestion rates normalized for body weight. Thus, risk estimates for osprey should be similar to or higher than those for bald eagle. In addition, information is available on the feeding preferences of osprey nesting at T-104 and T-18 as part of the USFWS and USGS study (Davis 2007). Although bald eagle are listed under ESA as a federally threatened species, all raptors tend to have high human interest and ecological significance.

Other piscivorous birds are less exposed than osprey either because they are not present year-round or because of limited habitat. For example, western grebe are common in the LDW during winter months, but they breed in inland areas east of the Cascades (Canning et al. 1979). Great blue heron forage while wading in shallow water and were selected as an ROC for the LDW ERA (Windward 2007c), but there are very few shallow water habitats in the EW. Thus, osprey are expected to be more exposed to EW chemicals than are other piscivorous birds, such as western grebe and great blue heron, because a higher proportion of the osprey's diet is likely to be obtained from the EW, particularly during the sensitive reproductive period.

A.2.3.3.2 Pigeon guillemot

The pigeon guillemot (*Cepphus columba*) was selected to represent birds that consume both fish and benthic invertebrates or mostly invertebrates. Pigeon guillemot was selected primarily because it is present year-round and breeds along the EW,[4] exposing females during egg development and the young during their most sensitive life stage. In addition, pigeon guillemot dive for prey in deep waters and are, therefore, not limited by the deeper water habitat of the EW. Pigeon guillemot are valued by society as a wildlife species and are protected under the Migratory Bird Treaty Act, as are the other potential ROCs that consume both fish and benthic invertebrates (i.e., merganser) or primarily benthic invertebrates (i.e., spotted sandpiper, bufflehead, and goldeneye). Species-specific toxicity data are not available to indicate whether one potential ROC is more sensitive to chemical exposures than another. Thus, the primary consideration for selecting the pigeon guillemot was the potential for higher exposure because of feeding habits and site use. The remainder of this section describes the rationale for selecting the pigeon guillemot rather than merganser, spotted sandpiper, bufflehead, or goldeneye.

The pigeon guillemot was chosen rather than a merganser species primarily because mergansers use the site less frequently; they are present for only a portion of the year and prefer shallower foraging habitats. However, the trophic positions of pigeon guillemot and merganser are similar, with mostly small fish and some crustaceans being consumed. Exposure to EW chemicals is expected to be less for merganser than for pigeon guillemot for the following two reasons:

[4] Pigeon guillemot are present in the Puget Sound region year-round (Seattle Audubon Society 2008), and their nests have been observed under the T-18 piers (Hotchkiss 2007).

- Common and red-breasted merganser are not known to breed along the LDW or the EW, and hooded merganser may overwinter but have not been reported to nest along the LDW or the EW, whereas pigeon guillemot are present year-round in Puget Sound and have nests along the EW.
- The deeper waters of the EW are likely to provide more foraging habitat for pigeon guillemot, which have an optimal diving and foraging efficiency in water 10 to 20 m deep (Ewins 1993), whereas merganser prefer shallower water (Mallory and Metz 1999; Dugger et al. 1994; Titman 1999), and shallow water habitat is limited in EW.

Pigeon guillemot are expected to be more exposed than the benthivorous bufflehead and goldeneye, because these diving ducks forage primarily in water depths of less than 5 m (Eadie et al. 2000; Gauthier 1993). Most of the EW is deeper than 5 m, so it is not likely that these diving ducks obtain much of their prey from the EW. Bufflehead and goldeneye are lower-trophic-level consumers than pigeon guillemot, consuming primarily invertebrates such as mussels, snails, shrimp, and small crabs; thus, they are less exposed to bioaccumulative chemicals. In addition, bufflehead and goldeneye do not breed in the vicinity of the EW,[5] so they are not exposed during their most sensitive life stages.

There are more uncertainties regarding the relative exposure of spotted sandpiper than merganser, bufflehead, and goldeneye compared with pigeon guillemot, because spotted sandpiper may breed in the vicinity of the EW. During a spotted sandpiper site use survey of the LDW (Windward 2004b), the closest potential sandpiper nesting habitat to the EW was observed at the T-105 restoration area along the LDW just south of the EW, although actual nests were not observed. It is unlikely that there are any areas closer to the EW that would be conducive to nesting. Nesting areas are characterized by the presence of shrubs, broad vegetation, slight gradient, and limited human activity. Assuming that spotted sandpipers could forage in intertidal areas up to 1 mi (1.6 km) from a potential nest at T-105 (Norman 2002b), there are approximately 16 hectares (ha) of foraging area within the LDW, compared to approximately 1 ha in the EW. Thus, it can be roughly estimated that spotted sandpipers nesting closest to the EW might obtain only 6% of their diet (1 ha in the EW of the total 17 ha in both the EW and LDW) from intertidal areas within the EW.

Limited data are available to estimate the foraging range of pigeon guillemot; Ewins (1993) cited other reports that document home ranges varying from 0.2 to 7 km from the nest, and Litzow and Piatt (2003) observed that radio-tagged pigeon guillemot foraged only in the areas in which they nested, although the size of those areas were not defined. Based on this limited information, the percentage of the pigeon guillemot diet obtained from the EW could be less than 10% or as much as 100%. Even with these

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[5] Bufflehead and goldeneye may be present in the EW during the winter; their breeding grounds are in Canada and Alaska (Eadie et al. 1995; Eadie et al. 2000; Gauthier 1993).

uncertainties, it is likely that pigeon guillemot consume a higher percentage of prey from the EW in their diet than do spotted sandpiper.

Other factors that would affect the relative comparison of the dietary exposures of spotted sandpipers and pigeon guillemot are: 1) prey preferences, 2) food ingestion rates normalized for differences in body weight, and 3) sediment ingestion rates as percentages of food ingestion rates. Data are not available to compare chemical concentrations in benthic invertebrates from intertidal areas (i.e., sandpiper prey) with those in benthic fish and invertebrates from both intertidal and subtidal areas (i.e., pigeon guillemot prey). Spotted sandpipers have higher food and sediment ingestion rates than do pigeon guillemot. However, based on the paucity of foraging habitat for spotted sandpipers, the spotted sandpiper was not selected as an ROC.

A.2.3.3.3 River otter

The river otter (*Lutra canadensis*) was chosen from the three aquatic-dependent mammals that use the EW (i.e., river otters, raccoons, and muskrats), because river otters are suspected to be year-round residents that may reproduce and feed in and around the EW. The river otter is susceptible to the biomagnification of chemicals because of its high trophic position and feeding habits, and because it is more likely to feed on fish or other prey from the EW than are raccoons or muskrats. River otters are in the same family as mink, which are known to be highly sensitive to PCBs and other chlorinated organic compounds, and relevant toxicological data are available for mink. River otters also attract a high level of societal interest. In addition, river otter was evaluated as an ROC in the LDW ERA (Windward 2007c).

A.2.3.3.4 Harbor seal

The harbor seal (*Phoca vituluna*) was chosen from the three marine mammals that may use the EW (i.e., harbor seals, sea lions, and harbor porpoises) to represent piscivorous mammals. All three of these marine mammals are susceptible to the biomagnification of chemicals because of their trophic positions and feeding habits. All are suspected to be sensitive to PCBs and other chlorinated organic compounds (Calambokidis et al. 1985; Tanabe et al. 1994), and toxicological data are available for mammals in general, although these data are not specific to marine mammals. In addition, all three species are protected under the Marine Mammal Protection Act and attract a high level of societal interest. There is more anecdotal evidence of the presence of harbor seals in the EW than of the other marine mammals, so it is assumed harbor seals feed there more often. In addition, the harbor seal was evaluated as an ROC in the LDW ERA (Windward 2007c). Therefore, based on the likelihood of higher use of the EW by harbor seals and consistency with the LDW ERA, the harbor seal was selected as an ROC for the EW. It is assumed that the harbor seal will act as a surrogate species for other marine mammals, such as sea lions or harbor porpoises. Orcas are not known to inhabit the EW, although they are observed infrequently in Elliott Bay (Traxler 2006). Orcas feed on salmon, which spend a small part of their lives in the LDW or EW and therefore would be expected to have a low exposure to sediment-associated chemicals in the EW.

A.2.3.4 Summary

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

- Benthic invertebrate community
- Cancrid crab species – higher-trophic-level benthic invertebrate
- Juvenile Chinook salmon – anadromous juvenile salmon
- English sole – benthivorous fish
- Brown rockfish – upper-trophic-level fish
- Osprey – piscivorous bird
- Pigeon guillemot – piscivorous/benthivorous bird
- River otter – piscivorous semi-aquatic mammal
- Harbor seal – piscivorous marine mammal

The key considerations used in the selection of each of the above receptors are summarized in Table A.2-8.

Table A.2-8. Receptors of concern selected for the EW and summary of rationale for selection

ROC	Exposure Route	Ecological Significance	Societal Significance	Site Use	Exposure Data Availability	Sensitivity
Benthic invertebrate community	direct contact with sediment; direct or incidental sediment ingestion; ingestion of contaminated prey; direct contact with water	food source for other invertebrates, fish, birds, and mammals; nutrient cycling; sediment oxygenation	surrogate for the protection of aquatic communities	present year-round; multiple life stages, diverse phyla	abundant surface sediment data available	range of chemical sensitivities represented
Cancrid crab	direct contact with sediment; incidental sediment ingestion; ingestion of contaminated prey; direct contact with water	higher-trophic-level benthic invertebrate; food for other invertebrates, fish, birds, and mammals	recreational and commercial value	present seasonally; multiple life stages (gravid females, juveniles)	site-specific tissue data available	effects data available for decapods; sensitivity relative to other decapods unknown
Brown rockfish	incidental sediment ingestion; ingestion of contaminated prey; direct contact with water	higher-trophic-level fish; important prey item for fish, birds, and mammals	some commercial (though not in EW) and recreational value	adults and juveniles present year-round; may spawn in the EW	site-specific tissue data and prey tissue data available	effects data available for other fish species; relative sensitivity of brown rockfish unknown; potential for elevated exposure via bioaccumulation because of trophic position; long-lived
English sole	direct contact with sediment; incidental sediment ingestion; ingestion of contaminated prey; direct contact with water	important prey item for fish, birds and mammals; key benthic invertebrate predator	some commercial and recreational value (though not in EW)	juveniles present year-round; adults present except during spawning migrations to Puget Sound	site-specific tissue data available	NMFS data suggest that they are as sensitive as other flatfish species
Juvenile Chinook salmon	ingestion of contaminated prey; direct contact with water	important prey item for fish, birds and mammals; seasonally one of the most abundant juvenile salmonids in the EW	T&E species; returning adults important to tribal, commercial, and sport fisheries	generally present April to July; most estuary-dependent juvenile salmonid	site-specific tissue data available	sensitive to a wide range of chemicals
Osprey	ingestion of contaminated prey and water; incidental sediment ingestion	high trophic level	highly valued and well-studied bird of prey; protected under the Migratory Bird Treaty Act	nests along the EW and likely forages in the EW	site-specific prey tissue data available	effects data available for other bird species; relative sensitivity of osprey unknown; potential for elevated exposure via bioaccumulation because of trophic position

Table A.2-8. Receptors of concern selected for the East Waterway and summary of rationale for selection (cont.)

ROC	Exposure Route	Ecological Significance	Societal Significance	Site Use	Exposure Data Availability	Sensitivity
Pigeon guillemot	ingestion of contaminated prey and water; incidental sediment ingestion	high trophic level	valued in general as wildlife species; protected under the Migratory Bird Treaty Act	nests observed along the EW	site-specific prey tissue data available	effects data available for other bird species; relative sensitivity of pigeon guillemot unknown; potential for elevated exposure via bioaccumulation because of trophic position
River otter	ingestion of contaminated prey and water; incidental sediment ingestion	high trophic level	highly valued by society	limited data, although anecdotal information indicates year-round presence of a river otter family on Kellogg Island	site-specific prey tissue data available	mink are sensitive to some chemicals, such as PCBs, although the relative sensitivity of river otter is unknown; potential for elevated exposure via bioaccumulation because of trophic position
Harbor seal	ingestion of contaminated prey and water; incidental sediment ingestion	high trophic level	protected under Marine Mammal Protection Act	occasional use based on a survey in the EW	site-specific prey tissue data available	pinnipeds suspected to be sensitive to some chemicals, such as PCBs, although the relative sensitivity of harbor seal is unknown; potential for elevated exposure via bioaccumulation because of trophic position

EW – East Waterway
NMFS – National Marine Fisheries Service
PCB – polychlorinated biphenyl
ROC – receptor of concern
T&E – threatened and endangered

A.2.4 DATA SELECTION, REDUCTION, AND SUITABILITY

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

A.2.4.1 Data summary

Environmental investigations conducted in the EW have included the collection of chemistry data from samples of surface sediment, tissue, or water. This section describes the datasets selected for use in the ERA for surface sediment, tissue, surface water, porewater, and sediment toxicity.

A.2.4.1.1 Surface sediment chemistry

The surface sediment dataset used for the ERA consisted of data collected after 1994 (Table A.2-9). These data included the most recent comprehensive sampling event conducted for the SRI in 2009. Most of the historical data (i.e., all but three samples) were collected between 2000 and 2009. The following considerations were made in selecting existing surface sediment data for the ERA dataset:

- **Depth of sample –** Only sediment samples collected from the uppermost 10 cm were included in the dataset because those data were most representative of the biologically active zone. Samples that were not collected to a depth of at least 5 cm were not included (e.g., samples collected from 0 to 2 cm were excluded).
- **Sampling date –** Only data collected after 1994 were included to ensure that the ERA would be based only on the most recent information collected in EW.
- **Dredging activities –** Only data collected from locations that were not subsequently dredged were included because the dredged sediment no long represents existing conditions.
- **Sample type –** The majority of the surface sediment data represented grab samples collected from discrete locations. One exception to this is the data for dioxins/furans and PCB congeners, which were collected as composite samples in 2009 for the SRI. In addition, three intertidal multi-increment sampling (MIS)[6] sediment samples were included for the characterization of intertidal sediment throughout EW.
- **Data quality –** Only data that were considered acceptable based on data validation results were included. Historical data quality for data collected prior to the SRI was determined based on a review of historical chemistry datasets

[6] The MIS approach integrates a large number of grab samples into a smaller number of composite samples to minimize the uncertainty associated with the estimate of an area-wide average; the approach used for EW surface sediment sampling is described in detail in the MIS sediment sampling QAPP (Windward 2009c).

(Anchor and Windward 2008). All surface sediment data collected under the SRI quality assurance project plan (QAPP) (Windward 2009d) met the data quality requirements presented in the QAPP.

- **Resampled locations** – Older data were excluded if a sediment sampling location was resampled at a later date within 10 ft of the original location to ensure that the ERA was based on only the most recent information in EW (see Attachment 1).

Table A.2-9. Summary of surface sediment data used in the EW

Year of Sample Collection	Sampling Event	Number of Samples	Analytes	Source
2009	EW Surface Sed Comps	16[a]	PCB congeners, dioxins/furans	Windward (2010e)
2009	EW Surface Sed	105	metals, organometals, SVOCs, Aroclors, pesticides, grain size, conventionals	Windward (2010b)
2009	EW T-30 PDM 2009	17	metals, SVOCs, Aroclors, grain size, conventionals	Windward (2010e)
2008	EW RCM 2008	12	metals, SVOCs, Aroclors, grain size, conventionals	Windward (2008b)
2007	EW – Slip 27	7	metals, organometals, SVOCs, Aroclors, pesticides, grain size, conventionals	Windward (2007a)
2007	EW – Recontamination Monitoring 2007	24	metals, SVOCs, Aroclors, pesticides, grain size, conventionals	Windward (2008a)
2006	EW – Recontamination Monitoring 2006	21	metals, SVOCs, Aroclors, pesticides, grain size, conventionals	Windward (2007b)
2005	Post Dredge Monitoring – 2005 Phase 1	9	metals, SVOCs, Aroclors, pesticides, grain size, conventionals	Anchor and Windward (2005a)
2005	USCG (Pier 36-37 slip and Berth Alpha)	11	metals, SVOCs, Aroclors, grain size, conventionals	Hart Crowser (2005)
2001	EW/Harbor Island Nature and Extent – Phase 1	62	metals, SVOCs, Aroclors, pesticides, grain size, conventionals	Windward (2002b)
2001	EW/Harbor Island Nature and Extent – Phase 2	22	metals, SVOCs, Aroclors, pesticides, grain size, conventionals	Windward (2002b)
2000	T-18 – post-dredge monitoring	13	metals, organometals, PAHs, Aroclors, pesticides, conventionals	Windward (2001)
1995	Harbor Island SRI	3	metals, SEM metals, organometals, SVOCs, Aroclors, pesticides, grain size, conventionals	EVS (1996a, b)

Note: Some of these data were excluded if a sediment sampling location was resampled at a later date within 10 ft of the original location; if analytes were available for both sampling events, the older data were excluded.

[a] Composite samples were analyzed for PCB congeners and dioxins/furans. Intertidal areas were sampled to create three MIS samples. Subtidal composite sediment samples were created for 13 areas that cover the entire study site.

EW – East Waterway
MIS – multi-increment sampling
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
PDM – post-dredge monitoring
RCM – recontamination monitoring
SEM – simultaneously extracted metals
SRI – supplemental remedial investigation
SVOC – semivolatile organic compound
T-18 – Terminal 18
T-30 – Terminal 30
USCG – US Coast Guard

Based on the above considerations, Table A.2-9 presents a summary of the sediment data used in the ERA, and Table A.2-10 presents data excluded from the ERA. The locations of the surface sediment samples listed in Table A.2-9 are presented on Maps A.2-2 and A.2-3.

Table A.2-10. Summary of surface sediment data excluded from the EW ERA

Sampling Date	Sampling Event	Number of Samples	Reason for Exclusion	Source
2009	EW Surface Sed Comps	4	lower depth is > 10 cm	Windward (2010b)
2009	EW Surface Sed	3	collected from reference area outside the study area	Windward (2010b)
2009	EW Benthic Tissue 09	8	composite samples[a]	Windward (2009a)
2008	EW Benthic Tissue 08	13	composite samples[a]	Windward (2009a)
2008	EW Clam Survey	5	lower depth is > 10 cm	Windward (2010b)
2008	EW – Recontamination Monitoring 2006	10	lower depth is < 5 cm; selected 0-10-cm-samples at the same location	Windward (2008b)
2008	EW T-30 PDM 2008	5	data superseded by 2009 event	Windward (2010b)
2005	EW Pre-Sand Placement Monitoring	37	samples no longer represent the surface layer	Anchor and Windward (2005b)
2005	PostDredgeMonitoring-2005	6	samples no longer represent the surface layer	Anchor and Windward (2005b)
2005	USCG_P36 PostDredge Sed Char	2	lower depth is < 5 cm	Hart Crowser (2005)
2001	EW/HI Nature and Extent Phase 2	2	area was subsequently dredged	Windward (2002a)
1997	Pier 36/37 – surface	3	area was subsequently dredged	Tetra Tech (1997)
1996	KC CSO 96	6	lower depth is < 5 cm, or area was subsequently dredged	King County (1997)
1996	Pier36-underpier	3	lower depth is < 5 cm	Tetra Tech (1996)
1995	KC CSO 95	7	lower depth is < 5 cm, or area was subsequently dredged	King County (1997)
1995	HI RI 95	15	area was subsequently dredged	EVS (1996a, b)

[a] These composite samples were collected over large areas and were co-located with benthic invertebrate tissue samples. They were not included in the ERA because the SMS apply to individual locations and individual grab samples were available from all of these areas.

CSO – combined sewer overflow
ERA – ecological risk assessment
EW – East Waterway
HI – Harbor Island
KC – King County
PDM – post-dredge monitoring
RI – remedial investigation
USCG – US Coast Guard

A.2.4.1.2 Tissue chemistry

A variety of tissue samples have been collected from the EW (Table A.2-11). Most of these data were collected as part of the 2008 and 2009 SRI sampling, which included the collection of juvenile Chinook salmon, English sole, brown rockfish, shiner surfperch, crabs, mussels, shrimp, benthic invertebrates, and clams. Three historical studies included the collection of tissue data and were considered acceptable for use based on a data quality review (Anchor and Windward 2008) (Table A.2-11). Only tissue data collected in 1995 or later are included in the ERA tissue dataset in order to represent current site conditions. No historical tissue datasets were excluded from the ERA dataset. The tissue dataset does not include fish tissue data that were analyzed as fillets. Locations for samples listed in Table A.2-11 are presented on Maps A.2-4, A.2-5, and A.2-6.

Table A.2-11. Summary of tissue data used in the EW ERA

Species	Year of Sample Collection	Sampling Event	No. of Samples	No. of Individuals per Sample	Sample Type	Analytes	Source
English sole	2008	EW-Fish Collection 2008	11	5	whole body	PCBs (Aroclors), pesticides, SVOCs, metals, inorganic arsenic, butyltins, lipids, dioxins/furans (subset of samples), PCB congeners (subset of samples)	Windward (2010a, c)
	2005	EW-Fish Collection 2005	2	5	skinless fillet and remainder[a]	PCBs (Aroclors), mercury, lipids,	Windward (2006b)
Brown rockfish	2008	EW-Fish Collection 2008	13	1	whole body	PCBs (Aroclors), pesticides, SVOCs, metals, inorganic arsenic, butyltins, lipids, dioxins/furans (subset of samples), PCB congeners (subset of samples)	Windward (2010a, c)
	2005	EW-Fish Collection 2005	2	1		PCBs (Aroclors), mercury, lipids,	Windward (2006b)
Shiner surfperch	2008	EW-Fish Collection 2008	8	10	whole body	PCBs (Aroclors), pesticides, SVOCs, metals, inorganic arsenic, butyltins, lipids, dioxins/furans (subset of samples), PCB congeners (subset of samples)	Windward (2010a, c)
	2005	EW-Fish Collection 2005	3	6 to 8		PCBs (Aroclors), mercury, lipids,	Windward (2006b)
Juvenile Chinook salmon	2009	EW-Chinook sampling 2009	1	165	stomach contents	metals, PAHs	Windward (2010d)
		EW-Chinook sampling 2009	6	4 to 36	whole body	PCBs (Aroclors), pesticides, SVOCs, metals, butyltins, lipids, PCB congeners and, dioxins/furans	Windward (2010d)
	2002	EW-Salmon	6	7 to 8		PCBs (Aroclors), mercury, lipids	Windward (2002c)

Table A.2-11. Summary of tissue data used in the EW ERA (cont.)

Species	Year of Sample Collection	Sampling Event	No. of Samples	No. of Individuals per Sample	Sample Type	Analytes	Source
Dungeness crab[b]	2008	EW-Fish Collection 2008	1	7	edible meat	PCBs (Aroclors), pesticides, SVOCs, metals, inorganic arsenic, butyltins, lipids, dioxins/furans (subset of samples), PCB congeners (subset of samples)	Windward (2010a, c)
		EW-Fish Collection 2008			hepatopancreas	PCBs (Aroclors), pesticides, SVOCs, metals, inorganic arsenic, butyltins, lipids, dioxins/furans (subset of samples), PCB congeners (subset of samples)	Windward (2010a, c)
Red rock crab[b]	2008	EW-Fish Collection 2008	8	7	edible meat	PCBs (Aroclors), pesticides, SVOCs, metals, inorganic arsenic, butyltins, lipids, dioxins/furans (subset of samples), PCB congeners (subset of samples)	Windward (2010a, c)
	2008	EW-Fish Collection 2008	8	7	hepatopancreas	PCBs (Aroclors), pesticides, SVOCs, metals, inorganic arsenic, butyltins, lipids, dioxins/furans (subset of samples), PCB congeners (subset of samples)	Windward (2010a, c)
Mussels	2008	EW-Fish Collection 2008	11	89 to 101	soft tissue	PCBs (Aroclors), pesticides, SVOCs, metals, inorganic arsenic, butyltins, lipids, dioxins/furans (subset of samples), PCB congeners (subset of samples)	Windward (2010a, c)
	1997	KC WQA	3	50 to 100		PCBs (Aroclors), SVOCs, pesticides, metals, butyltins, lipids, solids	King County (1999)
	1996	KC WQA	3	50 to 100		PCBs (Aroclors), SVOCs, pesticides, metals, butyltins, lipids, solids	King County (1999)
Shrimp	2008	EW-Fish Collection 2008	1	26	whole body	PCBs (Aroclors), pesticides, SVOCs, metals, inorganic arsenic, butyltins, lipids, dioxins/furans (subset of samples), PCB congeners (subset of samples)	Windward (2010a, c)
Clams[c]	2008	EW-Clam Survey	22	1 to 15	soft tissue	PCBs, TBT, mercury	Windward (2010a, b)
Benthic invertebrates	2008	EW Benthic Survey	13	not determined	whole body	PCBs (Aroclors), PAHs, metals, butyltins, lipids	Windward (2009a)
Sand sole[e]	2005	EW-Fish Collection 2005	6	1	whole body	PCBs (Aroclors), metals, lipids, solids	Windward (2006b)

Table A.2-11. Summary of tissue data used in the EW ERA (cont.)

[a] The results for the fillet composite samples and the remainder composite samples were weighted based on the fraction of the whole-body mass represented by each sample in order to calculate whole-body results (Windward 2006b).

[b] Data from hepatopancreas composite samples were mathematically combined with data from composite samples of edible meat to form composite samples of edible meat plus hepatopancreas. Whole-body (i.e., edible meat plus hepatopancreas) crab concentrations were calculated using the relative weights and concentrations of the edible meat and hepatopancreas.

[c] Geoduck tissue-residue data were used to quantify the exposures of human consumers in the EW HHRA; but in this ERA, they were addressed only in an uncertainty analysis for the benthic invertebrate evaluation because of the lack of relevant toxicity data for geoducks. Because geoducks are not a component of the diets of any of the ecological receptors, their tissue data were not used to quantify exposures to consumers.

[d] Twelve separate samples were analyzed for butyltins as shown on Map A.2-6.

[e] Sand sole data were evaluated in the uncertainty analysis as a surrogate for brown rockfish data.

ERA – ecological risk assessment
EW – East Waterway
FS – feasibility study
HHRA – human health risk assessment
KC – King County
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
SRI – supplemental remedial investigation
SVOC – semivolatile organic compound
TBT – tributyltin
WQA – water quality assessment

A.2.4.1.3 Surface water chemistry

The surface water dataset for the ERA consists of data collected in 2008 and 2009 for the SRI and data collected in 1996–1997 by King County as part of its water quality assessment (WQA) (Table A.2-12). King County collected 188 samples from three locations along a transect in the EW near the Hanford CSO outfall. Sampling was conducted on a weekly basis from October 1996 to June 1997. The SRI dataset consists of 59 samples collected during five separate sampling events (two dry season events, two wet season events, and one storm event) at five locations (Map A.2-7). The semivolatile organic compound (SVOC) data collected by King County as part of its WQA were not used in this ERA because the dataset consisted largely of non-detected results with higher analytical detection limits compared with the SRI detection limits for many of the compounds; the WQA data will be presented in the SRI. The metals datasets from both King County WQA and SRI samples were combined for use in this ERA because the data were considered comparable based on a visual evaluation of graphs that showed an overlap in the ranges in concentrations in the two datasets (Windward 2010f). The graphs are provided in Attachment 1.

Table A.2-12. Summary of surface water data used in the EW ERA

Year of Sample Collection	Sampling Event	No. of Samples Analyzed	Analytes	Source
2008–2009	SRI/FS	59	metals (filtered and unfiltered), PCBs congeners, SVOCs, TBT, and conventionals	Windward (2009e)
1996–1997	King County WQA	188[a]	metals (filtered and unfiltered) and conventionals	King County (1999)

[a] Samples analyzed only for conventional parameters are not included in the number of samples analyzed.

ERA – ecological risk assessment
EW – East Waterway
FS – feasibility study
PCB – polychlorinated biphenyl
SRI – supplemental remedial investigation
SVOC – semivolatile organic compound
TBT – tributyltin
WQA – water quality assessment

A.2.4.1.4 Porewater chemistry

The ERA includes an evaluation of risk to benthic invertebrates from exposure to volatile organic compounds (VOCs) in porewater. Therefore, sediment porewater data were collected from the EW and analyzed for VOCs for the EW SRI. No historical data are available for VOCs in EW porewater. Thirteen porewater samples were collected in July 2010 from four intertidal areas in the EW (Map A.2-7).

A.2.4.1.5 Sediment toxicity tests

Bioassays have been conducted as part of multiple projects to characterize the toxicity of EW sediment in the biologically active zone (i.e., the top 10 cm of the sediment column) and to assess the eligibility of dredged material (typically in 1.2-m depth intervals) to be placed in open-water disposal sites. Bioassay test results are included in the ERA if they were conducted with sediment collected within the top 10 cm and in accordance with Puget Sound Estuary Program (PSEP) protocols (PSEP 1995) and have concurrently collected chemistry data. Results from bioassays conducted for dredged material assessments are not included in the ERA because the sediment was composited over a 1.2-m depth horizon from multiple locations, and thus does not represent the benthic invertebrate exposure regime. In addition, bioassay results were not included if the tests were conducted with sediment from locations that were subsequently dredged. There were 9 locations that had chemistry and bioassay results and were subsequently resampled for chemistry only (Map A.2-8). The more recent chemistry replaced the older chemistry data for the location and the chemistry datasets were compared in terms of their SMS exceedance status to determine whether or not to retain the bioassay data for the location. In six locations, there were differences in the sediment chemistry that were sufficient to suggest that the older bioassay data may no longer be representative of site conditions and therefore the bioassay results were not retained in the ERA dataset. For three locations, the chemistry results were consistent in terms of their SMS exceedance status and therefore, the bioassay results were retained. The re-occupied locations with bioassay results and chemistry results that were resampled for chemistry only are summarized in Table 2-13.

Table A.2-13. Sediment samples that were initially sampled for chemistry and bioassays and resampled for chemistry only

Initial Sample	Bioassay Result	Chemistry result		Current Sample	Chemistry Result		Retain Bioassay Result in ERA Dataset?
		>SQS	>CSL		>SQS	>CSL	
EW-130	Fail (larval - SQS)	none	none	EW09-SS-001	none	none	yes
EW-127	Fail (*Neanthes*-SQS, larval - SQS)	total PCBs	none	EW09-SS-014	none	none	no
EW-124	Fail (larval -CSL)	total PCBs	none	EW09-SS-017	total PCBs	none	yes
EW-125	Fail (larval -CSL)	total PCBs	none	EW09-SS-019	total PCBs	none	yes
EW-122	Pass	mercury, benzyl alcohol	2,4-dimethly phenol, BEHP	EW09-SS-024	none	none	no
EW-121	Pass	Benzyl alcohol, total PCBs	2,4-dimethly phenol	EW09-SS-025	none	none	no
EW-123	Pass	mercury, total PCBs	2,4-dimethly phenol	EW09-SS-026	total PCBs	none	no
EW-137	Fail (larval -CSL)	zinc, total PCBs, butylbenzyl phthalate	mercury, 1,4-dichlorobenzene, 2,4-dimethylphenol, BEHP	EW09-SS-101	mercury, butylbenzyl phthalate, BEHP, total PCBs	1,4-dichlorobenzene	no
T-18 PDM-02	Pass	total PCBs	none	EW09-SS-111	none	none	no

BEHP – bis(2-ethylhexyl) phthalate
CSL – cleanup screening level
EW – East Waterway
SQS – sediment quality standards
PCB – polychlorinated biphenyl

Fifty- one surface sediment bioassay samples from three studies met the acceptance criteria and were used in the EW ERA (Table A.2-14). Each of the studies listed in Table A.2-14 included the following three types of bioassays: the acute (10-day) amphipod survival test using the amphipod *Eohaustorius estuarius*, the acute (48-hour) bivalve larvae normal survival test using the blue mussel *Mytilus galloprovincialis*, and the chronic (20-day) juvenile polychaete survival and growth test using *Neanthes arenaceodentata*.

Table A.2-14. Sediment bioassays conducted with EW surface sediment and used in the ERA

Event	Sampling Dates	No. of Bioassay Samples	Source
EW SRI Sampling	2009	11[a]	Windward (2010e)
EW/Harbor Island Nature and Extent – Phases 1 and 2	2001	34	Windward (2002b)
T-18 – PDM	2000	6	Windward (2001)

[a] Two samples were tested using the Pacific oyster (*Crassostrea gigas*) rather than the blue mussel *(Mytilus galloprovincialis)* for the acute (48-hour bivalve larvae normal survival test.

EW – East Waterway

ERA – ecological risk assessment

PCB – polychlorinated biphenyl

PDM – post-dredge monitoring

T-18 – Terminal 18

A.2.4.2 Data reduction

Data reduction refers to computational methods used to aggregate data. Data that were selected from those datasets presented in Tables A.2-9, A.2-11, and A.2-12 for determining exposures were on a dry-weight basis for sediment chemistry, on a dw or wet-weight basis for tissue chemistry (depending upon the method used for a particular ROC), and on a mass-per-unit-volume basis for water and porewater chemistry. All concentrations qualified as estimated (i.e., J-qualified data) were assumed to indicate the positive identification of the chemical and were used without modification in subsequent calculations. Less than 1% of the data were rejected by data validators for quality issues and flagged with an R-qualifier. R-qualified data are considered by EPA to be of insufficient quality and unusable in risk assessments under CERCLA.

Additional procedures related to averaging, the selection of the best data points when multiple data are available, the selection of significant figures and rounding procedures, and the calculation of totals for chemical groupings (i.e., PCBs, PAHs, dichlorodiphenyltrichloroethane [DDTs] and dioxins/furans) are described in the following subsections. These data reduction methods are consistent with those applied in the LDW ERA.

A.2.4.2.1 Averaging duplicate or replicate samples

Chemical concentrations obtained from the analysis of laboratory duplicates or replicates (i.e., two or more analyses performed on the same sample) were averaged for a closer representation of the "true" concentration compared with the results of a single analysis. Averaging rules were dependent on whether the individual chemical concentrations were detected or undetected. If all concentrations for a given chemical were detected, the values were averaged arithmetically. If all concentrations for a given chemical were undetected, the minimum reporting limit (RL) was reported. If the results were a combination of detected and undetected concentrations, any two or more detected concentrations were averaged arithmetically, and the undetected concentrations were excluded. If the combined concentrations consisted of a single detected concentration and one or more undetected concentrations, the detected concentration was reported. The latter two rules were applied regardless of whether the RL was higher or lower than the detected concentration.

Identical averaging rules were applied in situations in which multiple sediment samples were collected from the same location at the same time, such as field duplicate samples. In these instances, a single "average" result for each chemical was generated for that sediment sampling location.

A.2.4.2.2 Selection of best results

In some instances, the laboratory generates more than one result for a chemical for a given sample. Multiple results can occur for several reasons, including: 1) the original result did not meet the laboratory's internal quality control (QC) guidelines, and a reanalysis was performed; 2) the original result did not meet other project data quality objectives, such as a sufficiently low RL, and a reanalysis was performed; or 3) two different analytical methods were used for that chemical. In each case, a single best result was selected for use. The procedures for selecting the best result differed depending on whether a single or multiple analytical methods were used for a given chemical. For the same analytical method, if the results were:

- Detected and not qualified, the result from the lowest dilution was selected, unless multiple results from the same dilution were available, in which case the result with the highest concentration was selected.
- A combination of estimated (i.e., J-qualified) and unqualified detected results, the unqualified result was selected. This situation most commonly occurred when the original result was outside of calibration range, thus requiring a dilution. No results outside the calibration range were used in the ERA.
- All estimated, then the "best result" was selected using best professional judgment in consideration of the rationale for qualification. For example, a result qualified based on laboratory replicate results outside of QC objectives for precision would be preferred to a qualified result that was outside the calibration range.

- A combination of detected and undetected results, the detected result was selected. If there was more than one detected result, the applicable rules for multiple results (as discussed above) were followed.
- All undetected results, the lowest RL was selected.

If the multiple results were from different analytical methods, the result from the preferred method specified in the QAPP or based on the consensus of the professional opinions of project chemists was selected. Attachment 1 provides a detailed discussion of the samples and analytes with multiple results.

A.2.4.2.3 Significant figures and rounding

Analytical laboratories reported results with various numbers of significant figures depending on the QAPP instructions, the instrument, the parameter, and the concentration relative to the RL. The reported (or assessed) precision of each observation was explicitly stored in the project database by recording the number of significant figures assigned by the laboratory. Tracking of significant figures becomes important when calculating averages and performing other data summaries.

When a calculation involves addition, such as totaling PCBs or PAHs, the calculation can be only as precise as the least precise number that went into the calculation. For example (assuming two significant figures):

> 210 + 19 = 229 would be reported as 230 because 19 is reported to only 2 significant digits, and the enhanced precision of the trailing 0 in the number 210 is not significant.

When a calculation involves multiplication or division, such as carbon normalization, the original figures for each value are carried through the calculation (i.e., individual values are not adjusted to a standard number of significant figures; instead the appropriate adjustment is made to the resultant value at the end of the calculation). The result is rounded at the end of the calculation to reflect the value used in the calculation with the fewest significant figures. For example:

> 59.9 × 1.2 = 71.88 would be reported as 72 because there are 2 significant figures in the number 1.2.

When rounding, if the number following the last significant figure is less than 5, the digit is left unchanged. If the number following the last significant figure is equal to or greater than 5, the digit is increased by 1.

A.2.4.2.4 Calculating totals

Concentrations for several chemical sums were calculated as follows:

- Total PCBs were calculated using only detected concentrations for seven Aroclor mixtures (1016, 1221, 1232, 1242, 1248, 1254, and 1260)[7] in accordance with SMS (Washington Administrative Code [WAC] 173-204). For individual samples in which none of the seven Aroclor mixtures were detected, total PCBs were given a value equal to the highest RL of the seven Aroclors.
- Toxic equivalents (TEQs) were used for totaling certain groups of chemicals, specifically dioxin/furan congeners and dioxin-like co-planar PCB congeners. The 2,3,7,8-tetrachlorodibenzo-*p*-dioxin (TCDD) toxic equivalency factors (TEFs) for coplanar PCBs and certain polychlorinated dibenzo-*p*-dioxin or polychlorinated dibenzofuran (dioxin/furan) congeners are presented in Table A.2-15. The TEFs relate the toxicity of the coplanar PCB congeners and certain dioxin/furan congeners to the toxicity of 2,3,7,8-TCDD. PCB and dioxin/furan TEQ totals were calculated for each sample by summing the products of the concentrations of each individual congener and its specific TEF. Congeners that were undetected for a given sample were assigned a value equal to one-half the sample-specific RL for use in the TEQ calculation.
- Total DDTs were calculated from detected concentrations of three to six isomers: 2,4′-dichlorodiphenyldichloroethane (DDD), 2,4′-dichlorodiphenyl-dichloroethylene (DDE), 2,4′-DDT, 4,4′-DDD, 4,4′-DDE, and 4,4′-DDT. For samples in which all individual isomers were undetected, the single highest RL for that sample was assigned to represent the sum of the three to six isomers.

[7] For several sediment samples, Aroclors 1262 and 1268 were also included in the total PCB calculation, although these Aroclors are rarely quantified.

Table A.2-15. Toxic equivalency and potency equivalency factors for dioxins/furans and PCB congeners

Compound	TEF for Mammals	TEF for Fish	TEF for Birds
Dioxins/Furans[a]			
2,3,7,8-tetrachlorodibenzo-*p*-dioxin	1	1	1
1,2,3,7,8-pentachlorodibenzo-*p*-dioxin	1	1	1
1,2,3,4,7,8-hexachlorodibenzo-*p*-dioxin	0.1	0.5	0.05
1,2,3,6,7,8-hexachlorodibenzo-*p*-dioxin	0.1	0.01	0.01
1,2,3,7,8,9-hexachlorodibenzo-*p*-dioxin	0.1	0.01	0.1
1,2,3,4,6,7,8-heptachlorodibenzo-*p*-dioxin	0.01	0.001	< 0.001
Octachlorodibenzo-*p*-dioxin	0.0003	< 0.0001	0.0001
2,3,7,8-tetrachlorodibenzofuran	0.1	0.05	1
1,2,3,7,8-pentachlorodibenzofuran	0.03	0.05	0.1
2,3,4,7,8-pentachlorodibenzofuran	0.3	0.5	1
1,2,3,4,7,8-hexachlorodibenzofuran	0.1	0.1	0.1
1,2,3,6,7,8-hexachlorodibenzofuran	0.1	0.1	0.1
1,2,3,7,8,9-hexachlorodibenzofuran	0.1	0.1	0.1
2,3,4,6,7,8-hexachlorodibenzofuran	0.1	0.1	0.1
1,2,3,4,6,7,8-heptachlorodibenzofuran	0.01	0.01	0.01
1,2,3,4,7,8,9-heptachlorodibenzofuran	0.01	0.01	0.01
Octachlorodibenzofuran	0.0003	<0.0001	0.0001
PCBs[a]			
PCB 77	0.0001	0.0005	0.1
PCB 81	0.0003	0.0001	0.05
PCB 105	0.00003	0.005	0.1
PCB 114	0.00003	0.00005	0.001
PCB 118	0.00003	< 0.000005	0.0001
PCB 123	0.00003	< 0.000005	0.0001
PCB 126	0.1	< 0.000005	0.00001
PCB 156	0.00003	< 0.000005	0.00001
PCB 157	0.00003	< 0.000005	0.0001
PCB 167	0.00003	< 0.000005	0.0001
PCB 169	0.03	< 0.000005	0.00001
PCB 189	0.00003	< 0.000005	0.00001

[a] TEFs for dioxins/furans and PCB congeners are from the World Health Organization (Van den Berg et al. 2006; 1998).

PCB – polychlorinated biphenyl

PEF – potency equivalency factor

TEF – toxic equivalency factor

A.2.4.3 Suitability of data for risk assessment

A.2.4.3.1 Representativeness of site-related contamination and receptor exposure

Sediment

Sediment studies within the EW have been designed for both the reconnaissance (e.g., EW/Harbor Island Nature and Extent – Phases 1 and 2) and focused (e.g., Slip 27) investigations of areas of concern. In addition, a significant amount of sediment chemistry data has been collected as part of this SRI. Most of the events outside of the EW SRI/FS process focused primarily on subtidal sediments. The representativeness of the existing dataset was evaluated during the design of the SRI surface sediment sampling conducted in 2009 (Windward 2010e). The combined dataset was designed to be representative of surface sediment throughout the EW.

Tissue

Within the EW, samples of benthic invertebrates, shrimp, crab, clams, mussels, shiner surfperch, sand sole, English sole, juvenile Chinook salmon, and brown rockfish were collected from multiple locations throughout the waterway. Most of the samples used in the ERA were collected during the 2008 tissue sample collection efforts (Windward 2010c). The EW is relatively small, and given the variety of collection locations and the objective to evaluate the exposure of fish as receptors and prey throughout the EW, the available tissue chemistry data adequately represent site-related exposures.

Water

A significant amount of water chemistry data have been collected as part of the SRI (Windward 2009b) and by King County as part of its WQA (King County 1999). The SRI water data were collected from four locations through the EW and at various times to represent a variety of environmental conditions (i.e., different seasons, depths, and flow rates) in the EW. The King County WQA data were collected on weekly basis from October 1996 to June 1997. These samples were collected from three locations and two water depths along a transect near the Hanford No. 2 CSO outfall (Map A.2-7). The multiple sampling regimes provided an adequate characterization of surface water chemistry in the EW throughout the year.

A.2.4.3.2 QA/QC results

All datasets used in the ERA have been validated by the authors of the individual studies or by outside third parties. Summaries of the data validation reviews that have been conducted are presented in the EISR (Anchor and Windward 2008). Data validation reports for samples collected by the EWG for the SRI are included in the data reports (Windward 2010b, c, e, 2009b).

A.2.5 Selection of Chemicals of Potential Concern

This section presents the results of a risk-based screening process that was conducted to identify the COIs and COPCs for each of the ROCs. Through this screening process, a clear distinction was made between those chemical/receptor pairs that should be evaluated in greater detail using more realistic assumptions (Sections A.3 through A.6) and pairs for which no additional analysis is warranted. The COI and COPC selection processes are summarized in Table A.2-16, and the results are discussed separately for benthic invertebrates (Section A.2.5.1), fish (Section A.2.5.2), and wildlife (Section A.2.5.3).

Table A.2-16. COI and COPC selection processes

ROC	Medium	COI Selection	COPC Selection
Benthic invertebrate community	surface sediment	selected if either: • an SQS criterion or DMMP guideline was available and the chemical was detected in any sediment sample. • detected in > 5% of surface sediment samples analyzed for the chemical (i.e., any more frequently detected chemical, regardless of availability of regulatory criterion or guideline).	COI was retained as COPC if the maximum detected concentration[a] exceeded either the SMS SQS criteria or DMMP SL guideline.
	tissue	PCBs, mercury, and TBT	COI was retained as COPC if the maximum detected concentration[a] exceeded the aquatic invertebrate tissue NOAEL-based TRV.
	porewater	selected if VOCs were detected in any porewater sample[b]	COI was retained as COPC if the maximum detected concentration exceeded chronic WQC or literature TRVs.[a]
	surface water	selected if detected in any surface water sample	COI was retained as COPC if the maximum detected concentration in water[a] exceeded chronic WQC or literature TRVs.
Crab[c]	tissue	selected if two of the following three criteria were met: • detected in > 5% of surface sediment samples analyzed for the chemical • identified as a bioaccumulative chemical by EPA (2000a) • detected in any crab tissue sample from the EW	COI was retained as COPC if the maximum detected concentration in crab tissue[a] exceeded the tissue NOAEL-based TRV for crab or other decapods.[d]
	surface water	selected if detected in any surface water sample	COI was retained as COPC if the maximum detected concentration in water[a] exceeded chronic WQC or literature TRVs.
Juvenile Chinook salmon, English sole, and brown rockfish	tissue	selected if two of the following three criteria were met: • detected in > 5% of surface sediment samples analyzed for the chemical • identified as a bioaccumulative chemical by EPA (2000a) • detected in any fish tissue sample from the EW	COI was retained as COPC if the maximum detected concentration in fish tissue[a] exceeded the fish tissue NOAEL-based TRV.
	diet	selected if two criteria were met: • detected in > 5% of surface sediment samples analyzed for the chemical • detected in any fish tissue or prey sample from the EW	COI was retained as COPC if it was a PAH or a dietary metal[e] and the maximum detected concentration in diet[a, f] exceeded the dietary NOAEL-based TRV for fish.
	surface water	selected if detected in any surface water sample	COI was retained as COPC if the maximum detected concentration in water[a] exceeded chronic WQC values or

Table A.2-16. COI and COPC selection processes (cont.)

ROC	Medium	COI Selection	COPC Selection
			literature TRVs.
Osprey, pigeon guillemot, river otter, and harbor seal	diet	selected if two of the following three criteria were met: • detected in > 5% of surface sediment samples analyzed for the chemical • identified as a bioaccumulative chemical by EPA (2000a) • detected in any prey tissue sample from the EW	COI was retained as COPC if the maximum dietary dose[a, g] for a COI exceeded the dietary NOAEL-based TRV for birds or mammals.

[a] Detection limits that exceeded the screening criteria for COPC selection are discussed in the uncertainty section.

[b] Porewater samples were analyzed only for VOCs because VOCs do not have a high affinity for sediment as a result of their generally low OC-normalized partition coefficients(Mabey et al. 1982). Therefore, the exposure of sediment-dwelling organisms (i.e., benthic invertebrates) to VOCs is most appropriately assessed through the analysis of sediment porewater rather than bulk sediment.

[c] The direct contact sediment pathway was not evaluated for crab because data are generally not available for the toxic effects of sediment contact on crab.

[d] TRVs based on a broader search of effects on aquatic invertebrates were derived if no crab- or decapod-specific TRV was available.

[e] Dietary metals include arsenic, antimony, cadmium, chromium, cobalt, copper, lead, molybdenum, nickel, silver, thallium, and zinc.

[f] The maximum concentration of a COI in the fish diet (except juvenile Chinook) was calculated as a weighted concentration consisting of 10% of the maximum sediment concentration (to account for exposure via incidental sediment ingestion) plus 90% of the maximum prey tissue concentration. For juvenile Chinook, no incidental sediment ingestion was assumed; therefore, their dietary exposure was based on the maximum invertebrate tissue concentration. Note that 10% sediment ingestion is a conservative assumption used for this screening step and that lower sediment ingestion rates were assumed in the exposure analysis.

[g] The maximum dietary dose of a COI for wildlife species was calculated using the maximum concentration in prey and a site use factor of 1.

COI – chemical of interest
COPC – chemical of potential concern
DMMP – Dredged Material Management Program
EPA – US Environmental Protection Agency
EW – East Waterway
NOAEL – no-observed-adverse-effect level
OC – organic carbon
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
ROC – receptor of concern
SL – screening level
SMS – Washington State Sediment Management Standards
SQS – sediment quality standard
TBT – tributyltin
TRV – toxicity reference value
VOC – volatile organic compound
WQC –water quality criteria

In general, the COPC selection process was based on a comparison of the maximum detected COI concentrations with established sediment and water quality criteria, guidelines, or benchmarks or with toxicity reference values (TRVs) derived from the scientific literature (Table A.2-15). The established screening values as well as the literature-based TRVs used for screening are described in more detail within this section for benthic invertebrates (Section A.2.5.1), fish (Section A.2.5.2), and wildlife (Section A.2.5.3). Some general information on the derivation of TRVs for no-observed-adverse-effect-level (NOAEL) and lowest observed-adverse-effect-level (LOAEL) are discussed below because they apply to all literature-based TRVS presented for benthic invertebrates, fish, and wildlife.

The NOAEL represents the maximum level at which adverse effects have not been observed, and the LOAEL represents the lowest level above which effects have been observed in a toxicity study. To identify the NOAEL TRV, the LOAEL was first selected from among the list of possible TRVs for an ROC (see Attachments 3 through 6). The LOAEL was selected if it was the lowest concentration at which an effect for any of the three endpoints evaluated and a clear dose-response relationship were observed. Then, the NOAEL was selected as the highest no-effect level below the selected LOAEL with the same endpoint. If no NOAEL with the same endpoint as that of the selected LOAEL was available, the NOAEL was selected as the highest NOAEL below the selected LOAEL based on another endpoint (i.e., survival, growth, or reproduction).

For COIs that had no NOAELs lower than the selected LOAEL, the NOAEL was determined using the following uncertainty factors according to EPA Region 10 guidance (1997b):

- Acute or subchronic LOAEL/10
- Chronic or critical life stage[8] LOAEL/5
- Concentration that is lethal to 50% of an exposed population (LC50) (or similar)/50

Requirements for toxicity studies for selecting literature-based TRVs are discussed in the following sections for each ROC's screening process.

[8] 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 more than 10 weeks for birds and more than 1 year for mammals (Sample et al. 1996). For fish, chronic exposure duration was assumed to be 28 days or more. A critical life stage is one that occurs during reproduction, gestation, or development (Sample et al. 1996).

A.2.5.1 Benthic invertebrates

This section presents the screening process that was used to identify COPCs for the benthic invertebrate community and for crab. For the benthic invertebrate community, screening was conducted using surface sediment, tissue, surface water, and porewater data. For crab, screening was conducted using tissue and surface water data.

A.2.5.1.1 Benthic invertebrate community COPCs based on surface sediment data

The benthic invertebrate community surface sediment screen was conducted in two steps. First, any chemical detected in surface sediment for which an SMS criterion (Ecology 1995) or Dredged Material Management Program (DMMP) guideline (USACE et al. 2000) was available was identified as a chemical of interest (COI) (Table A.2-16). In addition, 9 detected chemicals without SMS criteria or DMMP guidelines were also identified as COIs because they had detection frequencies > 5% (Table A.2-17).

Table A.2-17. Surface sediment COIs for the benthic invertebrate community

COIs		
Metals		
Antimony	Copper	Silver
Arsenic	Lead	Vanadium[a]
Cadmium	Mercury	Zinc
Chromium	Molybdenum[a]	
Cobalt[a]	Nickel	
Organometals		
Monobutyltin[a]	Dibutyltin[a]	TBT[a]
PAHs		
1-Methylnaphthalene[a]	Total benzo fluoranthenes	Indeno(1,2,3-cd)pyrene
2-Methylnaphthalene	Benzo(g,h,i)perylene	Naphthalene
Acenaphthene	Chrysene	Phenanthrene
Acenaphthylene	Dibenzo(a,h)anthracene	Pyrene
Anthracene	Dibenzofuran	total HPAH
Benzo(a)anthracene	Fluoranthene	total LPAH
Benzo(a)pyrene	Fluorene	
Phthalates		
Bis(2-ethylhexyl) phthalate	Diethyl phthalate	Di-n-butyl phthalate
Butyl benzyl phthalate	Dimethyl phthalate	Di-n-octyl phthalate
SVOCs		
1,2,4-Trichlorobenzene	2-Methylphenol	n-Nitrosodiphenylamine
1,2-Dichlorobenzene	4-Methylphenol	Pentachlorophenol
1,3-Dichlorobenzene	Benzoic acid	Phenol
1,4-Dichlorobenzene	Benzyl alcohol	
2,4-Dimethylphenol	Carbazole[a]	

Table A.2-17. Surface sediment COIs for the benthic invertebrate community (cont.)

COIs		
PCBs		
Total PCBs[b]		
Dioxins/Furans[a]		
Dioxins/furans		
Organochlorine Pesticides		
Aldrin	Total chlordane[c]	Total DDTs[c]

[a] Chemical does not have an SMS criterion or DMMP guideline and was identified as a COI because detection frequency was > 5% in surface sediment samples that were analyzed for the chemical.

[b] Calculated as the sum of PCB Aroclors.

[c] trans-Nonachlor was the detected component of total chlordane and 4,4'-DDE and 4,4'-DDE were the detected components of total DDTs. These individual compounds were not identified as COIs because there were no SMS or DMMP values for these compounds, and they were not detected in > 5% of the surface sediment samples that were analyzed for pesticides. Total chlordane and total DDTs were identified as COIs because they had DMMP values.

COI – chemical of interest
DDD – dichlorodiphenyldichloroethane
DDT – dichlorodiphenyltrichloroethane
DMMP – Dredged Material Management Program
HPAH – high-molecular-weight polycyclic aromatic hydrocarbon
LPAH – low-molecular-weight polycyclic aromatic hydrocarbon
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
SMS – Washington State Sediment Management Standards
SVOC – semivolatile organic compound
TBT – tributyltin

In the second step of the COPC screening process, the maximum detected concentration of each COI in EW surface sediment was compared with SMS sediment quality standards (SQS) criteria or with DMMP screening levels (SLs) if SQS criteria were not available (TableA.2-18). SQS criteria and DMMP guidelines 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 the derivation of the SQS criteria and DMMP guidelines were field measures of benthic infaunal abundance, laboratory toxicity tests with marine benthic invertebrates (i.e., amphipod survival and abnormal development of oyster larvae), and laboratory toxicity tests with marine aquatic bacteria (Microtox™ [decrease in luminescence from the bacteria *Vibrio fisheri*]). Representatives of these groups are found throughout the EW. Under the provisions of SMS and DMMP, surface sediment with chemical concentrations equal to or less than all the SQS values is designated as having no adverse effects on biological resources (WAC 173-204-310(1)(a)).

Table A.2-18. Benthic invertebrate community COPC screening results based on surface sediment data

COI	Maximum Concentrations in Units Comparable to Screening Criteria[a]	Unit	SQS or SL[b]	Unit	Selected as a Sediment COPC?[c]
Metals					
Antimony	44	mg/kg dw	150[d]	mg/kg dw	no
Arsenic	**<u>241</u>**	mg/kg dw	57	mg/kg dw	yes
Cadmium	**<u>6.76</u>**	mg/kg dw	5.1	mg/kg dw	yes
Chromium	82	mg/kg dw	260	mg/kg dw	no
Cobalt	16	mg/kg dw	na	na	no
Copper	272	mg/kg dw	390	mg/kg dw	no
Lead	208	mg/kg dw	450	mg/kg dw	no
Mercury	**<u>1.07</u>**	mg/kg dw	0.41	mg/kg dw	yes
Molybdenum	5	mg/kg dw	na	na	no
Nickel	56	mg/kg dw	140[d]	mg/kg dw	no
Silver	6	mg/kg dw	6.1	mg/kg dw	no
Vanadium	94.1	mg/kg dw	na	na	no
Zinc	**<u>1,230 J</u>**	mg/kg dw	410	mg/kg dw	yes
Organometals					
Monobutyltin	8.4	mg/kg dw	na	na	no
Dibutyltin	63	mg/kg dw	na	na	no
TBT	6,000	mg/kg dw	na	na	no
PAHs					
2-Methylnaphthalene	**<u>85</u>**	mg/kg OC	38	mg/kg OC	yes
Acenaphthene	**<u>230</u>**	mg/kg OC	16	mg/kg OC	yes
Acenaphthylene	53	mg/kg OC	66	mg/kg OC	no
Anthracene	**<u>230</u>**	mg/kg OC	220	mg/kg OC	yes
Benzo(a)anthracene	**<u>350</u>**	mg/kg OC	110	mg/kg OC	yes
Benzo(a)pyrene	**<u>240</u>**	mg/kg OC	99	mg/kg OC	yes
Benzo(g,h,i)perylene	**<u>55</u>**	mg/kg OC	31	mg/kg OC	yes
Total benzofluoranthenes	**<u>915</u>**	mg/kg OC	230	mg/kg OC	yes

Table A.2-18. Benthic invertebrate community COPC screening results based on surface sediment data (cont.)

COI	Maximum Concentrations in Units Comparable to Screening Criteria[a]	Unit	SQS or SL[b]	Unit	Selected as a Sediment COPC?[c]
Chrysene	**1,100**	mg/kg OC	110	mg/kg OC	yes
Dibenzo(a,h)anthracene	**21**	mg/kg OC	12	mg/kg OC	yes
Dibenzofuran	**160**	mg/kg OC	15	mg/kg OC	yes
Fluoranthene	**6,400**	mg/kg OC	160	mg/kg OC	yes
Fluorene	**220**	mg/kg OC	23	mg/kg OC	yes
Indeno (1,2,3,-c,d)pyrene	**58 J**	mg/kg OC	34	mg/kg OC	yes
Naphthalene	91	mg/kg OC	99	mg/kg OC	no
Phenanthrene	**780**	mg/kg OC	100	mg/kg OC	yes
Pyrene	**3,500**	mg/kg OC	1,000	mg/kg OC	yes
Total HPAH	**12,500 J**	mg/kg OC	960	mg/kg OC	yes
Total LPAH	**1,330**	mg/kg OC	370	mg/kg OC	yes
Phthalates					
Bis(2-ethylhexyl) phthalate	**1,900**	mg/kg OC	47	mg/kg OC	yes
Butyl benzyl phthalate	**14**	mg/kg OC	4.9	mg/kg OC	yes
Diethylphthalate	5.3	mg/kg OC	61	mg/kg OC	no
Dimethyl phthalate	4.2	mg/kg OC	53	mg/kg OC	no
Di-n-butyl phthalate	**2,600**	mg/kg OC	220	mg/kg OC	yes
Di –n-octyl phthalate	5.8	mg/kg OC	58	mg/kg OC	no
Other SVOCs					
1,2,4-Trichlorobenzene	0.67	mg/kg OC	0.81	mg/kg OC	no
1,2-Dichlorobenzene	0.78	mg/kg OC	2.3	mg/kg OC	no
1,3-Dichlorobenzene	20	µg/kg dw	170[d]	µg/kg dw	no
1,4-Dichlorobenzene	**1,100**	mg/kg OC	3.1	mg/kg OC	yes
2,4-Dimethylphenol	**90 J**	µg/kg dw	29	µg/kg dw	yes
2-Methylphenol	38	µg/kg dw	63	µg/kg dw	no
4-Methylphenol	200	µg/kg dw	670	µg/kg dw	no
Benzoic acid	340	µg/kg dw	650	µg/kg dw	no
Benzyl alcohol	38	µg/kg dw	57	µg/kg dw	no
Carbazole	2,200	µg/kg dw	na	na	

Table A.2-18. Benthic invertebrate community COPC screening results based on surface sediment data (cont.)

COI	Maximum Concentrations in Units Comparable to Screening Criteria[a]	Unit	SQS or SL[b]	Unit	Selected as a Sediment COPC?[c]
n-Nitrosodiphenylamine	**<u>180</u>**	µg/kg dw	28[e]	µg/kg dw	yes
Pentachlorophenol	<u>110</u>	µg/kg dw	360	µg/kg dw	no
Phenol	**<u>630</u>**	µg/kg dw	420	µg/kg dw	yes
PCBs					
Total PCBs	**<u>840</u>**	mg/kg OC	12	mg/kg OC	yes
Dioxins/Furans					
Dioxin/furan TEQ[f]	<u>30.6</u>	ng/kg dw	na	na	no
Organochlorine Pesticides					
Aldrin	2.1	µg/kg dw	10[d]	µg/kg dw	no
Total chlordane	4.4	µg/kg dw	10[d]	µg/kg dw	no
Total DDTs	**<u>32</u>**	µg/kg dw	6.9[d]	µg/kg dw	yes

[a] Units are presented in dry weight for COIs with screening values in dw and for COIs without screening criteria (i.e., cobalt, molybdenum, vanadium, organometals, carbazole, and dioxins/furans).

[b] SQS criteria unless otherwise noted.

[c] COIs were identified as COPCs if the maximum surface sediment concentrations exceeded the SQS (or SL for antimony, nickel, 1,3-dichlorobenzene, aldrin, total chlordane, and total DDTs).

[d] SL guideline.

[e] Comparison was based on a dry-weight LAET because the OC content of the sample with the maximum concentration was too low to be considered for OC normalization.

[f] Dioxin/furan TEQ was calculated using mammalian TEFs from Van den Berg et al. (2006) and one-half the detection limit for undetected congeners.

AET – apparent effects threshold
COI – chemical of interest
COPC – chemical of potential concern
DDT – dichlorodiphenyltrichloroethane
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 or not available
OC – organic carbon
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
SL – screening level
SMS – Washington State Sediment Management Standards
SQS – sediment quality standard
SVOC – semi-volatile organic compound
TEQ – toxic equivalent
TBT – tributyltin
TOC – total organic carbon

<u>Bold and underline</u> identify the maximum surface sediment concentrations that are greater than the SQS or SL.

The scientific literature was searched for toxicologically based TRVs or screening criteria for the nine chemicals or chemical groups detected at frequencies greater than 5% in sediment samples analyzed for these chemicals but which had no SQS or SL values (i.e., cobalt, molybdenum, vanadium, TBT, monobutyltin, dibutyltin, 1-methylnapththalene, carbazole, and dioxins/furans). No other TRVs were found for these chemicals. DMMP guidelines for TBT were not used because they are based on TBT concentrations in interstitial water rather than whole sediment, since the available evidence indicates that sediment concentrations are not as useful in predicting effects to the benthic invertebrates (USACE et al. 2008). Therefore, risk to benthic invertebrates from exposure to butyltins was addressed using the tissue residues rather than sediment concentrations. For dioxins/furans, the DMMP recently developed new interim guidelines for sediment (WDNR et al. 2010). These guidelines are not based on concentrations at which adverse effects have been observed in benthic organisms but instead are based on Puget Sound non-urban background concentrations based on an evaluation of human health risks from seafood consumption using tribal consumption rates. The objectives of these guidelines are to provide consistency with the narrative human health requirements in the SMS and to reduce bioaccumulative risk to human and ecological receptors from dioxin/furan exposure (WDNR et al. 2010). Therefore, these guidelines are not appropriate for use as ecological risk thresholds for benthic organisms.[9] Risks from the chemicals without sediment TRVs and undetected chemicals with RLs greater than the SQS are discussed in the uncertainty analysis in Section A.6.1.1.1.

Many SQS sediment values for organic compounds are expressed as concentrations normalized to 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 ≤ 0.5% or ≥ 4.0%. In these cases, dry-weight chemical concentrations were compared with the lowest AET (LAET) expressed on a dry-weight basis, which is functionally equivalent to the SQS. For example, the maximum concentration of n-nitrosodiphenylamine was detected in a sample with TOC < 0.5%, and the LAET was used in the screening rather than the OC-normalized SQS (Table A.2-17). If the maximum concentration of a chemical exceeded its SQS criterion or SL guideline, that chemical was identified as a COPC.

As presented in Table A.2-17, 30 chemicals or chemical groups had concentrations in surface sediment that exceeded their respective SQS or SL values. The concentration of 1-methylnaphthalene exceeded the SQS for 2-methylnaphthalene, to which it was compared because 1-methylnaphthalene has no SQS of its own. However, 1-methynaphthalene was not identified as a COPC because it exceeded the SQS only once at a concentration of 82 mg/kg OC and in the same sample in which

[9] Although dixions/furans were not screened in as a COPC for the benthic invertebrate community, they were screened in as COPCs for fish and wildlife in Sections A.2.5.2 and A.2.5.3, respectively and were therefore addressed in the ERA.

2-methynaphthalene was detected at a concentration of 85 mg/kg OC. The remaining 29 chemicals with concentrations that exceeded the SQS were selected as COPCs for the benthic invertebrate community based on the surface sediment evaluation (Table A.2-17).

A.2.5.1.2 Benthic invertebrate community COPCs based on tissue data

For the assessment of benthic invertebrate tissue, PCBs, mercury, and tributyltin (TBT) were selected as COIs. For TBT, the tissue-residue evaluation was the only method used to determine if TBT was a COPC for benthic organisms. TBT could not be evaluated based on sediment toxicity because no SMS criterion or DMMP guideline was available for TBT in sediment. PCBs and mercury were identified as COIs for the benthic invertebrate tissue-residue evaluation because of their potential to bioaccumulate and their prevalence in sediment samples collected from the EW. The comparison of sediment concentrations with SQS was the primary risk evaluation for PCBs and mercury. The comparison of benthic invertebrate tissue with their respective tissue-residue TRVs was used in the risk characterization a secondary evaluation for determining potential effects on benthic invertebrates from PCBs and mercury.

To screen the three COIs and select COPCs for benthic invertebrate tissue, the maximum exposure concentration of each of the three tissue COIs was compared with the tissue TRV for that chemical. If the maximum tissue concentration was greater than the TRV, the chemical was identified as a COPC for benthic invertebrates. TRVs were obtained by searching the scientific literature for adverse effect levels for tissue concentrations in benthic invertebrates. The literature search included BIOSIS, EPA's ECOTOX database, the US Army Corps of Engineers' (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). The following considerations were made in determining whether toxicological data were acceptable for benthic invertebrates:

- The COI had to be chemically analyzed in tissue as part of the study (preference was given to measured over nominal concentrations).
- All selected TRVs were based on laboratory toxicological studies. Studies using field-collected data (i.e., field-collected organisms) were not considered acceptable. Field studies were not used to derive TRVs because adverse effects observed in organisms during 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.

- The test organisms were macroinvertebrate species, with preference given to benthic invertebrates over pelagic invertebrates.
- Controlled laboratory studies of single chemical exposure with statistically significant responses were given preference. Studies with clear dose-response relationships were also given preference.

After the literature search was conducted, all acceptable studies for TRV derivation were compiled, and presented in Attachment 3.

For each tissue COI, both NOAEL and LOAEL TRVs were derived using methods described in the introduction to Section 2.5. The NOAEL and LOAEL TRVs for benthic invertebrate tissues are identified in Table A.2-19. The NOAEL for total PCBs was estimated from the LOAEL using the uncertainty factors discussed in the introduction to Section 2.5, because a NOAEL was not available. To identify COPCs, maximum concentrations of COIs in benthic invertebrate tissue samples were compared with tissue-residue NOAELs[10] (Table A.2-20). TBT and total PCBs had maximum tissue concentrations that exceeded their NOAEL TRVs and were therefore identified as benthic invertebrate tissue COPCs for further evaluation in the ERA.

Table A.2-19. Benthic invertebrate tissue-residue TRVs

COI	Test Species	NOAEL (mg/kg ww)	LOAEL (mg/kg ww)	Endpoint	Source
Mercury	slipper limpet	na	8.0	growth	Thain (1984)
	banded mystery snail	6.0	na	growth, survival	Tessier (1996)
TBT	gastropod	0.024[a]	0.12[b]	reproduction	Gibbs et al. (1988)
Total PCBs	grass shrimp	0.11[c, d]	1.1[c]	survival	Hansen et al. (1974b)

[a] Calculated from the chronic LOAEL by dividing by 5.

[b] The LOAEL represents the TBT concentration associated with sterility in female gastropods resulting from imposex.

[c] The TRV for total PCBs was derived from the study with the lowest LOAEL using any Aroclor. Test organisms in the study were exposed to Aroclor 1016.

[d] Calculated from acute LOAEL by dividing by 10.

COI – chemical of interest
LOAEL – lowest-observed-adverse-effect level
NOAEL – no-observed-adverse-effect level
PCB – polychlorinated biphenyl
TBT – tributyltin
TRV – toxicity reference value
ww – wet weight

[10] LOAELs were used to evaluate COPCs in the risk characterization of the EW ERA.

Table A.2-20. Benthic invertebrate community COPC screening results based on tissue-residue data

COI	Maximum Concentration in Benthic Invertebrate Tissue (mg/kg ww)	NOAEL TRV (mg/kg ww)	Selected as Tissue COPC?
Mercury	0.06	6.0	no
TBT	**<u>0.39</u>**	0.024	yes
Total PCBs	**<u>0.38</u>**	0.11	yes

COI – chemical of interest
COPC – chemical of potential concern
NOAEL – no-observed-adverse-effect level
PCB – polychlorinated biphenyl
TBT – tributyltin
TRV – toxicity reference value
ww – wet weight

<u>Bold and underline</u> identify the maximum benthic invertebrate tissue concentrations that are greater than the NOAEL TRV.

A.2.5.1.3 Benthic invertebrate community COPCs based on surface water data

For surface water exposure of the benthic invertebrate community, any chemical detected in surface water was identified as a COI (Table A.2-21).

Table A.2-21. Surface water COIs for the benthic invertebrate community

COIs		
Metals		
Antimony	Cobalt	Selenium
Arsenic	Copper	Silver
Beryllium	Lead	Thallium
Cadmium	Mercury	Vanadium
Chromium	Nickel	Zinc
Organometals		
Monobutyltin	Dibutyltin	TBT
PAHs		
1-Methylnaphthalene	Benzo(a)anthracene	Fluorene
2-Methylnaphthalene	Chrysene	Naphthalene
Acenaphthene	Dibenzofuran	Phenanthrene
Anthracene	Fluoranthene	Pyrene
SVOCs		
1,4-Dichlorobenzene	Diethyl phthalate	Benzoic acid
Bis(2-ethylhexyl) phthalate	Di-n-butyl phthalate	
Total PCBs[a]		

[a] Calculated as the sum of PCB congeners
COI – chemical of interest
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
SVOC – semivolatile organic compound
TBT – tributyltin

To identify COPCs for the surface water exposure of the benthic invertebrate community, the maximum detected concentration of a COI was compared with its Washington State marine chronic water quality criteria (WQC) for the protection of aquatic life (WAC 173-201A-240). If a Washington State WQC was not available for a particular chemical, the federal WQC for marine chronic effects was used (EPA 2009). If no Washington State or federal WQC were available, alternative screening values from other EPA regions and programs were used, including the Tier II freshwater aquatic benchmarks developed by Suter and Tsao (1996) (Table A.2-22). A priority was given to screening values based on marine organisms and chronic effects related to survival, growth, and reproduction. The maximum concentrations of cadmium, mercury, and TBT in surface water samples exceeded the corresponding screening values, and these chemicals were identified as surface water COPCs for the benthic invertebrate community.

Table A.2-22. Benthic invertebrate community COPC screening results based on surface water data

COI	Maximum Chemical Concentration (µg/L)	Marine Chronic WQC or Other Screening Value (µg/L)	Screening Value Source[a]	Selected as a Surface Water COPC?
Metals				
Antimony (total)	0.15	30	Tier II	no
Arsenic (dissolved)	1.43	36	WQC	no
Beryllium (total)	0.015	0.66	Tier II	no
Cadmium (dissolved)	**37.8**	9.3	WQC	yes
Chromium (dissolved)	1.15	50	WQC for hexavalent chromium	no
Cobalt (total)	2.13	23	Tier II	no
Copper (dissolved)	2.44	3.1	WQC	no
Lead (dissolved)	0.814	8.1	WQC	no
Mercury (total)	**0.0277**	0.025	WQC	yes
Nickel (dissolved)	0.855	8.2	WQC	no
Selenium (dissolved)	0.38	71	WQC	no
Silver (total)	0.019	0.36	Tier II	no
Thallium (total)	0.021	12	Tier II	no
Vanadium (total)	9.29	20	Tier II	no
Zinc (dissolved)	7.79	81	WQC	no
Organometals				
Monobutyltin	0.036	na	na	no
Dibutyltin	0.015	na	na	no
TBT	**0.01**	0.0074	WQC[b]	yes

Table A.2-22. Benthic invertebrate community COPC screening results based on surface water data (cont.)

COI	Maximum Chemical Concentration (µg/L)	Marine Chronic WQC or Other Screening Value (µg/L)	Screening Value Source[a]	Selected as a Surface Water COPC?
PAHs				
1-Methylnaphthalene	0.091	2.1	Tier II	no
2-Methylnaphthalene	1	330	EPA Region 5 ESL	no
Acenaphthene	0.2	64	lowest marine FCV from EPA (2003d)	no
Anthracene	0.057	0.73	Tier II	no
Benzo(a)anthracene	0.02	0.027	Tier II	no
Chrysene	0.024	0.1	BCMOE	no
Dibenzofuran	0.13	3.7	Tier II	no
Fluoranthene	0.19	14.4	lowest marine FCV from EPA (2003d)	no
Fluorene	0.16	3.9	Tier II	no
Naphthalene	12	12	Tier II	no
Phenanthrene	0.9	8.1	lowest marine FCV from EPA (2003d)	no
Pyrene	0.12	4.53	lowest marine FCV from EPA (2003d)	no
SVOCs				
1,4-Dichlorobenzene	3.1	15	Tier II	no
Bis(2-ethylhexyl) phthalate	7.8	16	Canadian WQG	no
Diethyl phthalate	2.2	110	EPA Region 5 ESL	no
PCBs				
Total PCBs	0.0058	0.03	WQC	no

Note: Aquatic life WQC for metals (except mercury) were based on dissolved concentrations and thus were compared with dissolved concentrations in EW surface water. Tier II benchmarks for metals were based on total concentrations and thus were compared with total concentrations in EW surface water. WQC or Tier II benchmarks for mercury or organic compounds were based on total concentrations.

[a] WQC for all COIs are Washington State criteria unless otherwise noted.

[b] Federal WQC.

BCMOE – British Columbia Ministry of Environment
COI – chemical of interest
COPC – chemical of potential concern
EPA – US Environmental Protection Agency
ESL – ecological screening level
EW – East Waterway
FCV – final chronic value
na – not applicable or not available
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
SVOC – semivolatile organic compound
TBT – tributyltin
TRV – toxicity reference value
WQC – water quality criteria
WQG – water quality guidelines

Bold and underline identify the maximum surface water concentrations that are greater than the water screening values.

A.2.5.1.4 Benthic invertebrate community COPCs based on the porewater data

For porewater exposure of the benthic invertebrate community, any of the VOCs detected in porewater was identified as a COI. Three chemicals were detected and considered to be COIs: naphthalene, benzene, and cis-1,2-dichloroethene (Table A.2-23). To identify COPCs for the porewater exposure of the benthic invertebrate community, the maximum detected concentration of a COI was compared with its Tier II freshwater aquatic benchmark developed by Suter and Tsao (1996) because no Washington State or federal WQC are available for these COIs (Table A.2-23). The maximum concentration of naphthalene in porewater samples exceeded the corresponding Tier II values and was identified as the only porewater COPC for the benthic invertebrate community.

Table A.2-23. Benthic invertebrate community COPC screening results based on porewater data

COI	Maximum Porewater Concentration (µg/L)	Tier II Value	Selected as a Porewater COPC?
Naphthalene	**<u>48</u>**	12	yes
Benzene	0.30	130	no
cis-1,2-Dichloroethene	0.30	590	no

Note: Tier II benchmarks for organic compounds were based on total concentrations and were compared with total concentrations in EW porewater.

COI – chemical of interest

COPC – chemical of potential concern

<u>Bold and underline</u> identify the maximum porewater concentrations that are greater than the porewater screening values.

A.2.5.1.5 Crab COPCs based on crab tissue-residue data

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

- Detection in at least 5% of EW surface sediment samples analyzed for the chemical
- Identification as a bioaccumulative chemical by EPA (2000a)
- Detection in any EW crab tissue sample

Table A.2-24 presents a summary of the metals, organometals, and organic compounds identified as COIs for crab. Detailed results of the COI screening step for crab are presented in Attachment 4.

Table A.2-24. Tissue COIs for crab

COIs		
Metals		
Arsenic	Lead	Selenium
Cadmium	Mercury	Silver
Chromium	Molybdenum	Vanadium
Cobalt	Nickel	Zinc
Copper		
Organometals		
Dibutyltin	TBT	
PAHs		
1-Methylnaphthalene	Benzo(b)fluoranthene	Fluoranthene
2-Methylnaphthalene	Benzo(g,h,i)perylene	Fluorene
Acenaphthene	Benzo(k)fluoranthene	Indeno(1,2,3-cd)pyrene
Acenaphthylene	Chrysene	Naphthalene
Anthracene	Dibenzo(a,h)anthracene	Phenanthrene
Benzo(a)anthracene	Dibenzofuran	Pyrene
Benzo(a)pyrene		
SVOCs		
1,4-Dichlorobenzene	Phenol	
Total PCBs[a]		
Dioxins/Furans		
Organochlorine Pesticides		
4,4′-DDD	Heptachlor epoxide	Oxychlordane
4,4′-DDE	cis-Nonachlor	
Dieldrin	trans-Nonachlor	

[a] Calculated as the sum of PCB Aroclors.

COI – chemical of interest
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
SVOC – semivolatile organic compound
TBT – tributyltin

In the second step of the COPC screening process, the maximum concentration of each COI in the tissue of EW crab was compared with a TRV for that chemical. If the maximum crab tissue concentration was greater than the TRV, the chemical was identified as a COPC for crab. TRVs were obtained by searching the scientific literature for adverse effect levels for tissue concentrations in decapods. The literature search included BIOSIS, EPA’s ECOTOX 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).

The TRV search focused on chemical tissue-residue data associated with effects on decapods to support the tissue-residue evaluation for crab (see Section A.3.4). Toxicological data were determined to be acceptable for crab based on the following considerations:

- The COI had to be chemically analyzed in tissue as part of the study (preference was given to measured over nominal concentrations).
- All selected TRVs were based on laboratory toxicological studies. Studies using field-collected data (i.e., field-collected crab) were not considered acceptable. Field studies were not used to derive TRVs because adverse effects observed in organisms during 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.
- 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 statistically significant responses were given preference. Studies with clear dose-response relationships were also given preference.
- Studies with chronic exposure duration (30-plus days) were preferred if available.

After the literature search was conducted, all acceptable studies for TRV derivation were compiled (Attachment 3, Table 2). For each COI, TRVs were selected for both the NOAEL and LOAEL. TRV selection rules and uncertainty factors discussed in the introduction to Section A.2.5 were used. Based on the available literature and considerations outlined above, TRVs were developed for 15 of the 45 crab COIs (Tables A.2-25 and A.2-26). For an additional four COIs, TRVs for similar or related compounds were substituted (total chlordane for cis-nonachlor, trans-nonachlor, and oxychlordane; and 4,4′-DDT for 4,4′-DDD and 4,4’-DDE) because toxicological data were not available for those COIs. The remaining 26 COIs with no TRVs are discussed in the uncertainty analysis (Section A.6.1.2.2).

Table A.2-25. Results of TRV search for crab COIs

COIs		
Chemicals with TRVs		
Arsenic	Zinc	trans-Nonachlor[c]
Cadmium	TBT	Oxychlordane[c]
Chromium	Naphthalene	Heptachlor epoxide
Copper	1,4-Dichlorobenzene[a]	4,4′-DDD[d]
Mercury	Total PCBs	4,4′-DDE[d]
Silver	Dioxins/furans[b]	
Vanadium	cis-Nonachlor[c]	
Chemicals without TRVs		
Cobalt	Acenaphthene	Dibenzo(a,h)anthracene
Lead	Acenaphthylene	Dibenzofuran
Molybdenum	Anthracene	Fluoranthene
Nickel	Benzo(a)anthracene	Fluorene
Dibutyltin	Benzo(a)pyrene	Indeno(1,2,3-cd)pyrene
Selenium	Benzo(b)fluoranthene	Phenanthrene
1-Methylnaphthalene	Benzo(g,h,i)perylene	Pyrene
2-Methylnaphthalene	Benzo(k)fluoranthene	Dieldrin
Phenol	Chrysene	

[a] 1,4-Dichlorobenzene was not detected in crab tissue but was considered the be a COI because it was detected in at least 5% of EW surface sediment samples analyzed for 1,4-dichlorobenzene and was identified as a bioaccumulative chemical.

[b] A TRV for 2,3,7,8-TCDD was used for dioxins/furans.

[c] A TRV was not available for cis-nonachlor, trans-nonachlor, or oxychlordane, so the TRV for total chlordane was used as a surrogate.

[d] A TRV was not available for 4,4′-DDD or 4,4’-DDE, so the TRV for 4,4′-DDT was used as a surrogate.

COI – chemical of interest
DDD – dichlorodiphenyldichloroethane
DDT – dichlorodiphenyltrichloroethane
PCB – polychlorinated biphenyl
TBT – tributyltin
TCDD – tetrachlorodibenzo-*p*-dioxin
TRV – toxicity reference value

Table A.2-26. Selected tissue-residue TRVs for crab COIs

COI	Test Species	Tissue Type	NOAEL (mg/kg ww)	LOAEL (mg/kg ww)	Endpoint	Source
Arsenic	grass shrimp	whole body	1.28	na	survival	Lindsay and Sanders (1990)
	brown shrimp	whole body	na	21	survival	Madsen (1992)
Cadmium	virile crayfish	whole body	0.57[a]	5.7[b]	survival	Mirenda (1986b)
Chromium	juvenile sand crab	whole body	1	3.2	growth	Mortimer and Miller (1994)
Copper	banana prawn	whole body	2.6[a]	26	growth	Ahsanullah and Ying (1995)
Mercury	Norway lobster	hepatopancreas	0.99	na	survival	Canli and Furness (1995)
	shore crab	hepatopancreas	na	1	survival	Bianchini and Giles (1996)
Silver	American red crayfish	hepatopancreas	86.3	na	survival	Mann et al. (2004)
Vanadium	Monaco shrimp	whole body	0.6	na	survival	Miramand et al. (1981)
Zinc	virile crayfish	whole body	12.7	35.2	survival	Mirenda (1986a)
TBT	juvenile blue crab	whole body	0.12	na	growth	Rice et al. (1989)
1,4-Dichlorobenzene	sand crab	whole body	0.074[a]	0.74[b]	survival	Mortimer and Connell (1994)
Naphthalene	spot shrimp	whole body	0.005[a]	0.05	survival	Sanborn and Malins (1977)
Total PCBs[c]	grass shrimp	whole body	0.11[a]	1.1	survival	Hansen et al. (1974b)
Dioxins/furans (2,3,7,8-TCDD)	signal crayfish	whole body	0.0003[a]	0.003[d]	survival	Ashley et al. (1996)
Total chlordane[e]	pink shrimp	whole body	0.71	1.7	survival	Parrish et al. (1976)
Heptachlor epoxide	pink shrimp	whole body	0.054	0.18	survival	Schimmel et al. (1976)
4,4′-DDT	pink shrimp	whole body	na	0.06	survival	Nimmo et al. (1970)
	water nymph crayfish	whole body	0.046	na	survival	Johnson et al. (1971)

[a] Calculated from LOAEL by dividing by 10.
[b] Converted from dw to ww using a moisture content of 80% (Jarvinen and Ankley 1999).
[c] The TRV for total PCBs was derived from the study with the lowest LOAEL using any Aroclor. Hansen et al. (1974b) exposed test organisms to Aroclor 1016.
[d] Tissue-based LOAEL estimated based on injected dose.
[e] Parrish et al. (1976) exposed test organisms to technical grade chlordane.

COI – chemical of interest
DDT – dichlorodiphenyltrichloroethane
dw – dry weight
LOAEL – lowest-observed-adverse-effect level
na – not acceptable or not available
NOAEL – no-observed-adverse-effect level
PCB – polychlorinated biphenyl
TBT – tributyltin
TCDD – tetrachlorodibenzo-*p*-dioxin
TRV – toxicity reference value
ww – wet weight

To identify crab COPCs, maximum concentrations of COIs detected in the EW crab whole-body or hepatopancreas tissue samples were compared with their respective tissue-residue NOAEL TRVs (e.g., maximum concentrations detected in hepatopancreas tissue were compared with hepatopancreas NOAELs; estimated maximum whole-body concentrations [as calculated from hepatopancreas and edible muscle tissue concentrations] were compared with whole-body NOAELs).[11] NOAELs for naphthalene, 1,4-dichlorobenzene, total PCBs, and 2,3,7,8-TCDD were estimated as the LOAEL (based on survival data) divided by 10 because NOAELs were not available. If a COI was not detected, one-half the RL was compared with the NOAEL TRV.

Five chemicals (arsenic, cadmium, copper, zinc, and total PCBs) had maximum detected crab tissue concentrations that exceeded the NOAEL TRV (Table A.2-27) and were identified as COPCs for crab. One chemical (1,4-dichlorobenzene) was not detected in crab tissue but had a reporting limit that exceeded the NOAEL TRV; this chemical was not selected as a COPC but is discussed in the uncertainty analysis (Section A.6.1.2.2).

Table A.2-27. Crab COPC screening results based on tissue-residue data

COI	Maximum Concentration in Crab Tissue (mg/kg ww)[a]	NOAEL TRV (mg/kg ww)	Selected as a Crab Tissue COPC?
Arsenic	**6.81 J**	1.28	yes
Cadmium	**3.1**	0.57	yes
Chromium	0.1	1	no
Copper	**31.3**	2.6	yes
Mercury	0.12[b]	0.99[b]	no
Silver	0.53[b]	86.3[b]	no
Vanadium	0.3	0.6	no
Zinc	**59.0**	12.7	yes
TBT	0.013	0.12	no
Naphthalene	0.0037	0.005	no
1,4-Dichlorobenzene	0.085 U	0.074	no[c]
Total PCBs	**0.86**	0.11	yes
Dioxin/furan TEQ[d]	0.00000125	0.0003	no
Total chlordane	0.018 J	0.71	no
Heptachlor epoxide	0.00031	0.054	no
Total DDTs	0.0046 JN	0.046	no

a All concentrations are whole-body concentrations except as noted.

b Hepatopancreas concentration.

c 1,4-Dichlorobenzene was not detected in crab tissue but had a reporting limit that exceeded the NOAEL TRV; this chemical was not selected as a COPC but is discussed in the uncertainty analysis (Section A.6.1.2.2).

11 LOAELs were used to evaluate COPCs in the risk characterization of the EW ERA.

[d] The maximum dioxin/furan TEQ in crab tissue was compared with the NOAEL TRV for 2,3,7,8-TCDD. The dioxin/furan TEQ in crab tissue was calculated using fish TEFs from Van den Berg et al. (2006) and one-half the detection limit for non-detected congeners.

COI – chemical of interest
COPC – chemical of potential concern
DDT – dichlorodiphenyltrichloroethane
J – estimated concentration
N – tentative identification
NOAEL – no-observed-adverse-effect level
PCB – polychlorinated biphenyl
TBT – tributyltin
TCDD – tetrachlorodibenzo-*p*-dioxin
TEF – toxic equivalency factor
TEQ – toxic equivalent
TRV – toxicity reference value
U – not detected at given concentration
ww – wet weight

Bold and underline identify the maximum crab tissue concentrations that are greater than the NOAEL TRV.

A.2.5.1.6 Crab COPCs based on surface water data

For surface water exposure of crab, both COIs and COPCs were identified using the same process as that used for the benthic invertebrate community. Surface water COIs were identified as any chemical detected in surface water. The resulting COIs are the same as those identified for benthic invertebrates and are presented in Section A.2.5.1.3 (Table A.2-21).

Surface water COPCs were identified as any COI with a maximum detected concentration in EW surface water that exceeded its Washington State marine chronic WQC for the protection of aquatic life (WAC 173-201A-240) or other appropriate TRVs. The surface water COPCs identified for crab were cadmium, mercury, and TBT (Table A.2-22).

A.2.5.1.7 Summary of COPC screening results for the benthic invertebrate community and crab

Based on the COPC screen, 30 chemicals were included in the risk characterization for the benthic invertebrate community based on surface sediment chemical concentrations (Table A.2-28). A smaller number of chemicals were included as COPCs in the risk characterization for the benthic invertebrate community based on their concentrations in other media: cadmium, mercury, and TBT in surface water; TBT and total PCBs in tissue; and naphthalene in porewater. For crab, five chemicals (arsenic, cadmium, copper, zinc, and total PCBs) were identified as COPCs based on concentrations in crab tissue, and three chemicals were identified as COPCs (cadmium, mercury, and TBT) based on concentrations in surface water.

Table A.2-28. COPCs for the benthic invertebrate community and crab

COPC	COPC Identified for the Benthic Invertebrate Community by Evaluation Type				COPC Identified for Crab by Evaluation Type	
	Surface Sediment	Tissue Residue	Surface Water	Porewater	Tissue Residue	Surface Water
Metals						
Arsenic	X				X	
Cadmium	X		X		X	X
Copper					X	
Mercury	X		X			X
Zinc	X				X	
Organometals						
TBT		X	X			X
PAHs						
1-Methylnaphthalene						
2-Methylnaphthalene	X					
Acenaphthene	X					
Benzo(a)anthracene	X					
Benzo(a)pyrene	X					
Benzo(g,h,i)perylene	X					
Chrysene	X					
Dibenzo (a,h)anthracene	X					
Dibenzofuran	X					
Fluoranthene	X					
Fluorene	X					
Indeno (1,2,3,-c,d)pyrene	X					
Naphthalene				X		
Phenanthrene	X					
Pyrene	X					
Total benzofluoranthenes	X					
Total HPAH	X					
Total LPAH	X					

Table A.2-28 COPCs for the benthic invertebrate community and crab (cont.)

COPC	COPC Identified for the Benthic Invertebrate Community by Evaluation Type				COPC Identified for Crab by Evaluation Type	
	Surface Sediment	Tissue Residue	Surface Water	Porewater	Tissue Residue	Surface Water
Other SVOCs						
Bis(2-ethylhexyl) phthalate	X					
Butyl benzyl phthalate	X					
Di-n-butyl phthalate	X					
1,4-Dichlorobenzene	X					
2,4-Dimethylphenol	X					
n-Nitrosodiphenylamine	X					
Phenol	X					
PCBs						
Total PCBs	X	X			X	
Organochlorine Pesticides						
Total DDTs	X					

COPC – chemical of potential concern'
DDT – dichlorodiphenyltrichloroethane
HPAH – high-molecular-weight polycyclic aromatic hydrocarbon
LPAH – low-molecular-weight polycyclic aromatic hydrocarbon
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
SVOC – semivolatile organic compound
TBT – tributyltin
VOC – volatile organic compound

A.2.5.2 Fish

This section describes the COPC selection process and results for fish ROCs. COPCs were identified separately for the three exposure media used to assess risk to fish (tissue, diet, and surface water), as described in the following subsections.

A.2.5.2.1 Fish COPCs based on tissue-residue data

The tissue-residue evaluation was used to identify COPCs in tissue for the following chemicals: selenium, mercury, butyltins, pesticides, SVOCs (excluding PAHs), total PCBs, and dioxins/furans. The evaluation of tissue concentrations based on a tissue-residue evaluation integrates all exposure pathways and reduces the uncertainty associated with the relative uptake and depuration rates of a chemical.

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

- Detection in at least 5% of EW surface sediment samples analyzed for the chemical
- Identification as a bioaccumulative chemical in EPA (2000a)
- Detection in any fish tissue sample collected from the EW

Chemicals identified as tissue-residue COIs for fish are summarized in Table A.2-29. Detailed results of the COI screening step for fish are presented in Attachment 2.

In the second step of the COPC screening process, the maximum tissue concentration of each COI was compared with a tissue-residue NOAEL for that chemical. If the maximum tissue concentration was greater than the NOAEL, the chemical was identified as a fish tissue COPC. For each COI, the scientific literature was searched to identify NOAEL and LOAEL TRVs. The literature search included BIOSIS, EPA's ECOTOX database, and USACE's ERED. 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).

Table A.2-29. Fish Tissue COIs

Fish Tissue COIs		
Metals		
Mercury	Selenium	
Organometals		
Dibutyltin	TBT	
SVOCs (excluding PAHs)		
1,4-Dichlorobenzene		
PCBs		
Total PCBs[a]		
Dioxins/Furans		
Dioxins/furans		
Organochlorine Pesticides		
4,4′-DDD	alpha-BHC	Heptachlor epoxide
4,4′-DDE	alpha-Chlordane	Mirex
4,4′-DDT	beta-Chlordane	cis-Nonachlor
Dieldrin	beta-Endosulfan	trans-Nonachlor

[a] Calculated as the sum of PCB Aroclors.

BHC – benzene hexachloride
COI – chemical of interest
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
SVOC – semivolatile organic compound
TBT – tributyltin

Databases were searched for tissue-residue studies, and acceptable NOAELs and LOAELs were compiled for fish. Toxicological data were judged to be acceptable based on the following considerations:

- The data were used only if the COI was chemically analyzed in tissue as part of the study, with preference given to measured over nominal concentrations.
- 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 adverse effects observed in organisms during field studies may be attributed to the presence of multiple chemicals and/or other uncontrolled environmental factors, rather than to a single test chemical.[12]

[12] The uncertainty associated with not including TRVs derived from field-collected data is addressed in the uncertainty analysis (Section A.6.2.1.2).

- Selected TRVs were based preferentially on dietary, sediment, or water exposure studies. Studies conducted using intraperitoneal (IP) or egg injection or oral gavage as an exposure route 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 egg-to-adult conversion factors from the literature.
- Controlled laboratory studies of single chemical exposure with statistically significant responses were given preference. Studies with clear dose-response relationships were also given preference.
- Chronic exposure duration studies were preferred if available.

After the literature search was conducted, all acceptable studies for TRV derivation were compiled (Attachment 5). For each COI, TRVs were selected for both the NOAEL and the LOAEL. TRV selection rules and uncertainty factors discussed in the introduction to Section A.2.5 were used. Acceptable TRVs or surrogate TRVs were identified for 18 of the 19 tissue-residue COIs (Table A.2-30). Surrogate TRVs were assigned to COIs that had no toxicity data if data for related compounds were available. In these cases, the NOAEL TRV for the related compound was considered acceptable for screening the identified COIs. The selected TRVs are presented in Table A.2-31. Early or reproductive life stages are generally considered to be the most sensitive life stages for toxicity to fish, so early life-stage toxicity tests are considered to be protective of adult fish (for example, see Macek and Sleight 1977). However, several toxicological studies have demonstrated an equal or greater sensitivity of adult fish with respect to both growth and survival (for example see Giesy et al. 2002). Therefore, there may be some uncertainty in the TRVs for fish tissue if the most sensitive life stage was not tested for a particular species and chemical.

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

Fish Tissue COIs		
COIs with TRVs		
Mercury	Total PCBs	beta-Endosulfan[d]
Selenium	4,4-DDD[b]	beta-Chlordane[c]
Dibutyltin	4,4′-DDE[b]	Heptachlor epoxide[e]
TBT	4,4′-DDT[b]	Mirex
1,4-Dichlorobenzene	Dieldrin	cis-Nonachlor[c]
Dioxins/furans[a]	alpha-Chlordane[c]	trans-Nonachlor[c]
COIs without TRVs		
alpha-BHC		

a Dioxins/furans were evaluated on a TEQ basis based on their cumulative toxicity to fish relative to that of 2,3,7,8-TCDD according to the methods of Van den Berg et al. (2006).

b TRVs were not available for 4,4′-DDD, 4,4′-DDE, or 4,4′-DDT, so the available TRV for total DDTs was used to screen these compounds.

c A TRV was not available for cis-nonachlor, trans-nonachlor, alpha-chlordane, or beta-chlordane, so the available TRV for total chlordane was used to screen these compounds.

d A TRV was not available for beta-endosulfan, so the available TRV for endosulfan was used to screen this compound.

e A TRV was not available for heptachlor epoxide, so the available TRV for heptachlor was used to screen this compound.

BHC – benzene hexachloride
COI – chemical of interest
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
PCB – polychlorinated biphenyl
TBT – tributyltin
TEQ – toxic equivalent
TCDD – tetrachlorodibenzo-*p*-dioxin
TRV – toxicity reference value

Table A.2-31. TRVs selected for fish COIs for the tissue-residue evaluation

COI	Test Species	NOAEL (mg/kg ww)	LOAEL (mg/kg ww)	Endpoint	Source
Mercury	fathead minnow	na	0.39[a]	reproduction	Hammerschmidt et al. (2002)
	golden shiner	0.23	na	survival behavior	Webber and Haines (2003)
Selenium	Chinook salmon	1.1[b]	2.1[b]	growth	Hamilton et al. (1990)
Dibutyltin	guppy	5.5	na	growth, survival	Wester (1990)
TBT	rainbow trout	0.029[c]	0.29	growth	Triebskorn et al. (1994)
1,4-Dichlorobenzene	rainbow trout	30[d]	1,505	survival	Kaiser et al.(1984)
Total PCBs	common barbel	0.10 – 0.53[e]	0.520 – 2.64[e]	reproduction	Hugla and Thome (1999)
	pinfish	1.4[c, f]	14[f]	survival	Hansen et al. (1971)
2,3,7,8-TCDD	rainbow trout	0.000046	0.000085	growth	Fisk et al (1997)
Total DDTs	cutthroat trout	0.9[g]	1.8[g]	survival	Allison et al. (1964)
Dieldrin	rainbow trout	0.12	0.20	survival	Shubat and Curtis (1986)
Endosulfan	spot	0.0031[c,h]	0.031[h]	survival	Schimmel et al. (1977)
Total chlordane	goldfish	0.71	na	survival	Moore et al. (1977)
	goldfish	na	1.36	survival	Feroz and Khan (1979)
Heptachlor epoxide	bluegill	0.08[c]	0.8	growth	Andrews et al. (1966)
Mirex	fathead minnow	129	156	reproduction	Buckler et al. (1981)

a TRV was based on exposure to methylmercury. Methylmercury was not analyzed in any of the whole-body fish tissue samples from the EW; the total mercury concentration in EW tissue samples was compared with the methylmercury TRV.

b Dry-weight concentration was converted to wet weight assuming 80% moisture content.

c NOAEL was estimated using an uncertainty factor of 10 (acute/subchronic LOAEL to NOAEL).

d NOAEL was estimated using an uncertainty factor of 50 (LC50 to NOAEL).

e A range of LOAELs was selected because of uncertainties associated with this study. The NOAEL range was estimated by applying an uncertainty factor of 5 to the chronic LOAEL range. This TRV was not applied to juvenile Chinook salmon because it was based on a reproductive endpoint, and juvenile Chinook salmon are not present in the EW during their reproductive life stage.

f This TRV was applied only to juvenile Chinook salmon because the lowest PCB TRV (Hugla and Thome 1999) was based on a reproductive endpoint, and juvenile Chinook salmon do not reproduce in the EW.

g The LOAEL is the 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 is the tissue concentration at 111 days in fish exposed to 0.03 and 0.01 mg/L DDT in water. No mortality was observed at these doses compared with the negative control.

h No TRV was identified for beta-endosulfan, so the available TRV for endosulfan was used to screen for beta-endosulfan; endosulfan was not identified as a COI.

COI – chemical of interest
DDT – dichlorodiphenyltrichloroethane
LC50 – concentration that is lethal to 50% of an exposed population
LOAEL – lowest-observed-adverse-effect level
NOAEL – no-observed-adverse-effect level
na– not applicable or not available
PCB – polychlorinated biphenyl
TBT – tributyltin
TCDD – tetrachlorodibenzo-*p*-dioxin
TRV – toxicity reference value
ww – wet weight

To select COPCs from the list of fish tissue COIs, maximum COI concentrations in fish tissue were compared with NOAEL TRVs from the literature. This screen was conducted in two parts. First, the maximum whole-body fish tissue concentration for each COI (or one-half the RL for each non-detected COI) in any fish ROC species was compared with its respective NOAEL TRV. COIs with maximum tissue concentrations greater than the NOAEL TRV were identified as fish tissue 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.4).

Four chemicals were identified as COPCs for fish tissue: mercury, TBT, total PCBs, and beta-endosulfan (Table A.2-32).

Table A.2-32. Results of COPC screen for fish tissue

COI	Maximum Chemical Concentration in Fish Tissue (mg/kg ww)	NOAEL TRV (mg/kg ww)	Selected as a Fish Tissue COPC?
Mercury	**0.418**	0.23	yes
Selenium	0.85	1.1	no
Dibutyltin	0.024	5.5	no
TBT as ion	**0.42**	0.029	yes
1,4-Dichlorobenzene	4.8	30	no
Total PCBs	**7.9**	0.10	yes
PCB TEQ[a]	0.00000489	0.000046	no
Dioxin/furan TEQ[a]	0.0000026	0.000046	no
Total TEQ[a]	0.00000673	0.000046	no
Total DDTs	0.054	0.9	no
4,4′-DDD	0.0056	0.9[b]	no
4,4′-DDE	0.049	0.9[b]	no
4,4′-DDT	0.0014	0.9[b]	no
Dieldrin	0.00066	0.12	no
Total chlordane	0.0137	0.71	no
alpha-Chlordane	0.0014	0.71[c]	no
beta-Chlordane	0.00079	0.71[c]	no
beta-Endosulfan	**0.013**	0.0031[d]	yes
Heptachlor epoxide	0.00024	0.08	no
Mirex	0.00076	129	no
cis-Nonachlor	0.0028	0.71[c]	no
trans-Nonachlor	0.009	0.71[c]	no

a Dioxin-like PCBs and dioxins/furans were evaluated on a TEQ basis (i.e., PCB TEQ and dioxin/furan TEQ, respectively) based on a their cumulative toxicity to fish relative to that of 2,3,7,8-TCDD according to the methods of Van den Berg et al. (1998). In addition, the toxicity of both dioxin-like PCBs and dioxins/furans were evaluated (i.e., total TEQ).

b TRV is for total DDTs, which was used as a surrogate.

c TRV is for total chlordane, which was used as a surrogate.

d TRV is for endosulfan, which was used as a surrogate.

Table A.2-32. Results of COPC screen for fish tissue (cont.)

COI – chemical of interest
DDT – dichlorodiphenyltrichloroethane
J – estimated concentration
NOAEL – no-observed-adverse-effect level
PCB – polychlorinated biphenyl
RL – reporting limit
ROC – receptor of concern
TBT – tributyltin
TEQ – toxic equivalent
TRV – toxicity reference value
U – not detected at given concentration
ww – wet weight

Bold and underline identify the maximum fish tissue concentrations that are greater than the NOAEL.

ROC-specific COPCs were then determined from the general fish COPCs presented above. The TRVs used to screen for COPCs in fish tissue were the same for all three fish ROCs except total PCBs. A juvenile Chinook-specific TRV was identified for total PCBs because the study providing the selected total 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 EW 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 life stage or exposure regime of the migratory juvenile Chinook salmon in the EW. The lowest LOAEL for a non-reproductive endpoint was 14 mg/kg ww for survival of pinfish (Hansen et al. 1971). There was no survival NOAEL below the selected LOAEL reported in this study, so a NOAEL of 1.4 mg/kg ww was extrapolated from the LOAEL using an uncertainty factor of 10. This extrapolated NOAEL (1.4 mg/kg ww) was selected as the screening NOAEL TRV for juvenile Chinook salmon. This TRV is likely protective of juvenile Chinook salmon growth. A study by Mauck et al. (1978) reported no effects on growth in brook trout fry exposed to Aroclor 1254 at an aqueous concentration of 0.69 µg/L. Reduced growth was observed for fish exposed to 1.5 µg/L after 48 days, however tissue residue concentrations were not measured until 118 days of exposure at this concentration. Residue concentrations associated with no effects (0.69 µg/L) ranged from 1.8 mg/kg ww in fish exposed for 7 days to 31 mg/kg ww in fish exposed for 118 days. The lowest effects concentrations (1.5 µg/L) ranged from 3.2 mg/kg ww for 7 days to 71 mg/kg ww for 118 days.

Based on the ROC-specific screen, in which maximum concentrations in the tissue of each ROC were compared with the NOAEL TRVs, the ROC-COPC pairs identified for further evaluation in this baseline ERA are listed below and detailed in Table A.2-33. There were no COPCs identified for juvenile Chinook salmon. TBT and total PCBs were identified as COPCs for English sole and brown rockfish, and mercury and beta-endosulfan were also identified as COPCs for brown rockfish.

Table A.2-33. ROC-specific COPC screening results based on fish tissue-residue data

ROC	COI	Maximum Fish Tissue Concentration (mg/kg ww)	NOAEL TRV (mg/kg ww)	Selected as a ROC-Specific Tissue COPC?
Juvenile Chinook salmon	mercury	0.043	0.23	no
	TBT	0.0037 U[a]	0.029	no
	total PCBs	0.0915	1.4	no
	beta-endosulfan	0.002 U[a]	0.0031	no
English sole	mercury	0.042	0.23	no
	TBT	**<u>0.038</u>**	0.029	yes
	total PCBs	**<u>7.9</u>**	0.104	yes
	beta-endosulfan	0.0018	0.0031	no
Brown rockfish	mercury	**<u>0.418</u>**	0.23	yes
	TBT	**<u>0.42</u>**	0.029	yes
	total PCBs	**<u>6.2</u>**	0.104	yes
	beta-endosulfan	**<u>0.013</u>**	0.0031	yes

a Value reported as one-half the maximum RL.

COI – chemical of interest
COPC – chemical of potential concern
NOAEL – no-observed-adverse-effect level
PCB – polychlorinated biphenyl
RL – reporting limit
ROC – receptor of concern
TBT – tributyltin
TRV – toxicity reference value
U – not detected at given concentration
ww – wet weight

<u>Bold and underline</u> identify the maximum fish tissue concentrations that are greater than the NOAEL.

A.2.5.2.2 Fish COPCs based on fish dietary exposure concentrations

The fish dietary evaluation was used to identify COPCs for PAHs and metals (except mercury, selenium, and butyltins). For PAHs and most metals, which are highly regulated or metabolized by fish (Varanasi 1989; Bury et al. 2003, as cited in Meyer et al. 2005), concentrations in the fish diet more accurately assess potential risk than concentrations in fish tissue. Therefore, COPCs from among these chemicals were selected by comparing concentrations in the tissue of fish prey with dietary TRVs for fish. Chemicals included in the fish tissue-residue evaluation described in Section A.2.5.2.1 were not included in the dietary evaluation. COPCs for the fish dietary evaluation were identified using a two-step process. The first step was to identify chemicals as COIs if they met the following two criteria:

- Detection in at least 5% of EW surface sediment samples analyzed for the chemical
- Detection in any fish prey tissue sample collected from the EW

Chemicals identified as fish dietary COIs are summarized in Table A.2-34. Detailed results of the COI screening step for the fish dietary evaluation are presented in Attachment 5.

Table A.2-34. Chemicals identified as dietary COIs for fish

Dietary COIs for Fish		
Metals		
Arsenic	Copper	Silver
Cadmium	Lead	Vanadium
Chromium	Molybdenum	Zinc
Cobalt	Nickel	
PAHs		
1-Methylnaphthalene	Benzo(b)fluoranthene	Fluoranthene
2-Methylnaphthalene	Benzo(g,h,i)perylene	Fluorene
Acenaphthene	Benzo(k)fluoranthene	Indeno(1,2,3-cd)pyrene
Acenaphthylene	Chrysene	Naphthalene
Anthracene	Dibenzo(a,h)anthracene	Phenanthrene
Benzo(a)anthracene	Dibenzofuran	Pyrene
Benzo(a)pyrene		

COI – chemical of interest
PAH – polycyclic aromatic hydrocarbon

In the second step of the COPC screening process, the maximum dietary concentration of each COI in prey tissue was compared with a dietary NOAEL TRV for that chemical. If the maximum dietary concentration was greater than the NOAEL TRV, the chemical was identified as a dietary COPC. For each COI, the scientific literature was searched to identify NOAEL and LOAEL TRVs. The TRVs were based on concentrations of COIs in the diet of fish. An alternative method of evaluating the dietary exposure of an ROC is to calculate the rate of chemical uptake as µg/g fish bw/day; risks are then evaluated by comparing this dose with a dose-based TRV. Because fish prey consumption is variable, the use of a dietary dose approach is becoming more prevalent as a way to normalize dietary exposure to body weight 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 the 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 to calculate fish exposure and effects in this ERA and is consistent with the approach used in the LDW ERA.

The literature search for dietary TRVs included BIOSIS and EPA's ECOTOX 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).

Databases were searched for dietary studies. Toxicological data were determined to be acceptable for fish based on the following considerations:

- 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 adverse effects observed in organisms during 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 exposure studies. Studies conducted using oral gavage as an exposure route were not considered representative of the ROC exposure conditions, and were therefore used only if no other studies were available.

- Controlled laboratory studies of single chemical exposure with statistically significant responses were given preference. Studies with clear dose-response relationships were also given preference. Chronic exposure duration studies were preferred if available.

After the literature search was conducted, all acceptable studies for TRV derivation were compiled, as presented in Attachment 5. For each COI, TRVs were selected for both the NOAEL and the LOAEL. TRV selection rules and uncertainty factors discussed in the introduction to Section A.2.5 were used. Acceptable TRVs or surrogate TRVs were identified for 27of the 31 dietary COIs (Table A.2-35). The total PAH TRV was used as a surrogate TRV for 18 individual PAHs that are included as components of the mixture associated with the total PAH TRV but for which there are no specific toxicity values. In these cases, the NOAEL TRV for total PAHs was considered acceptable for screening the individual PAH COIs.

Table A.2-35. Results of TRV search for fish dietary COIs

COIs for Fish		
Dietary COIs with TRVs		
Arsenic	1-Methylnaphthalene[a]	Benzo(k)fluoranthene[a]
Cadmium	2-Methylnaphthalene[a]	Chrysene[a]
Chromium	Acenaphthene[a]	Dibenzo(a,h)anthracene[a]
Copper	Acenaphthylene[a]	Fluoranthene[a]
Lead	Anthracene[a]	Fluorene[a]
Silver	Benzo(a)anthracene[a]	Indeno(1,2,3-cd)pyrene[a]
Vanadium	Benzo(a)pyrene[a]	Naphthalene[a]
Zinc	Benzo(b)fluoranthene[a]	Phenanthrene[a]
Total PAHs[a]	Benzo(g,h,i)perylene[a]	Pyrene[a]

Table A.2-35. Results of TRV search for fish dietary COIs (cont.)

COIs for Fish		
Dietary COIs without TRVs		
Cobalt	Dibenzofuran	Molybdenum
Nickel		

[a] TRVs were based on PAH mixture derived from Meador et al. (2006). Data are not available for individual PAHs, with the exception of benzo(a)pyrene. Individual PAHs were included in the mixtures of PAHs evaluated by Meador et al. (2006).

COI – chemical of interest

PAH – polycyclic aromatic hydrocarbon

TRV – toxicity reference value

Selected TRVs for the fish dietary COIs are presented in Table A.2-36. Chemicals with no TRVs are discussed in the uncertainty analysis (Section A.6.2.1.2).

Table A.2-36. TRVs selected for fish COIs for a dietary evaluation

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	rockfish	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	2.0[a]	10.2	growth	Hilton and Bettger (1988)
Zinc	rainbow trout	1,900	na	growth, survival	Mount et al. (1994)
	rainbow trout	1,000[b]	2,000	growth	Takeda and Shimma (1977)
Benzo(a)pyrene	rainbow trout	1.5	2	growth	Kim et al.(2008)
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 not selected due to higher NOAEL from Mount et al. (1994).

[c] Mixture comprised the following 21 PAHs included in the 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 LOAEL was identified in the literature search; selected NOAEL was the highest unbounded NOAEL in the literature reviewed)

NOAEL – no-observed-adverse-effect level

PAH – polycyclic aromatic hydrocarbon

TRV – toxicity reference value

For the selection of COPCs for fish dietary tissue, maximum detected concentrations of COIs in fish diets (expressed as mg/kg dw) were compared with the selected dietary NOAEL reported in the literature (also expressed as mg/kg dw). Consistent with the LDW ERA, 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 prey concentration for each COI (Equation 2-1), except in the case of juvenile Chinook salmon, which do not consume appreciable amounts of sediment during foraging (Cordell 2001). Prey data for juvenile Chinook salmon were represented by the maximum concentration in benthic invertebrate tissue or juvenile Chinook salmon stomach contents. Prey data for English sole were represented by maximum concentrations in benthic invertebrate tissue. Prey data for brown rockfish were represented by the maximum concentration in benthic invertebrates, Dungeness crab, red rock crab, shiner surfperch, or shrimp tissue.[13]

For English sole and brown rockfish, 10% 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 (arsenic, cadmium, chromium, copper, and vanadium) evaluated using the dietary evaluation were selected as COPCs for all three fish ROCs (TableA.2-37). In addition, benzo(a)pyrene was retained as a COPCs for English sole and brown rockfish but not for juvenile Chinook salmon (Table A.2-37).

Table A.2-37. Fish COPC screening results based on dietary exposure concentrations

COI	Maximum EW-Wide Sediment Concentration (mg/kg dw)	Maximum EW-Wide Prey Concentration (mg/kg dw)	Maximum Dietary Exposure Concentration (mg/kg dw)	NOAEL TRV (mg/kg dw)	Selected as a Dietary COPC?
Juvenile Chinook Salmon					
Arsenic	na	32.6	**32.6**	20	yes
Cadmium	na	2	**2**	0.1	yes
Chromium	na	45.1	**45.1**	9.42	yes
Copper	na	155	**155**	50	yes

[13] Juvenile Chinook salmon tissue data were not used because brown rockfish are not expected to eat juvenile salmon. In addition, maximum concentrations of fish dietary COIs were lower in the tissue of juvenile Chinook salmon than in the tissue of benthic invertebrates, Dungeness crabs, red rock crabs, shiner surfperch, or shrimp.

Table A.2-37. Fish COPC screening results based on dietary exposure concentrations (cont.)

COI	Maximum EW-Wide Sediment Concentration (mg/kg dw)	Maximum EW-Wide Prey Concentration (mg/kg dw)	Maximum Dietary Exposure Concentration (mg/kg dw)	NOAEL TRV (mg/kg dw)	Selected as a Dietary COPC?
Lead	na	86.8	86.8	7,040	no
Molybdenum	na	10	10	1,500	no
Silver	na	0.4	0.4	3,000	no
Vanadium	na	31	**<u>31</u>**	2.04	yes
Zinc	na	339	339	1,900	no
Benzo(a)pyrene	na	1.2	1.2	1.5	no
Total PAHs[a]	na	16	16	324	no
English Sole					
Arsenic	241	32.6	**<u>53.4</u>**	20	yes
Cadmium	5.7	2	**<u>2.4</u>**	0.1	yes
Chromium	76.2	45.1	**<u>48.2</u>**	9.42	yes
Copper	272	155	**<u>167</u>**	50	yes
Lead	208	86.8	98.9	7,040	no
Molybdenum	5	10	10	1,500	no
Silver	6	0.4	1	3,000	no
Vanadium	94.1	31	**<u>37</u>**	2.04	yes
Zinc	1,230	339	428	1,900	no
Benzo(a)pyrene	7.8	1.2	**<u>1.9</u>**	1.5	yes
Total PAHs[a]	155	16	30	324	no
Brown Rockfish					
Arsenic	241	35.1	**<u>55.7</u>**	20	yes
Cadmium	5.7	19	**<u>18</u>**	0.1	yes
Chromium	76.2	45.1	**<u>48.2</u>**	9.42	yes
Copper	272	203	**<u>210</u>**	50	yes
Lead	208	86.8	99	7,040	no
Molybdenum	5	10	10	1,500	no
Silver	6	1.7	2	3,000	no
Vanadium	94.1	31	**<u>37</u>**	2.04	yes
Zinc	1,230	346	434	1,900	no
Benzo(a)pyrene	7.8	1.2	**<u>1.9</u>**	1.5	yes
Total PAHs[a]	155	16	30	324	no

[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
COPC – chemical of potential concern
dw – dry weight
EW – East Waterway
na – not applicable or not available
NOAEL – no-observed-adverse-effect level
PAH – polycyclic aromatic hydrocarbon
TRV – toxicity reference value

<u>Bold and underline</u> identify the maximum dietary exposure concentrations that are greater than the NOAEL.

A.2.5.2.3 Fish COPCs based on surface water data

For surface water exposure of fish, both COIs and COPCs for fish were identified using the same process as that used to identify COIs and COPCs for surface water exposures of benthic invertebrates. Surface water COIs were identified as any chemical detected in surface water. The resulting COIs are the same as those identified for benthic invertebrates and are presented in Section A.2.5.1.3 (Table A.2-21).

Surface water COPCs were identified as any COI with a maximum detected concentration in EW surface water that exceeded its Washington State marine chronic WQC for the protection of aquatic life (WAC 173-201A-240) or other appropriate TRVs. The surface water COPCs identified for all fish ROCs were cadmium, mercury, and TBT (Table A.2-22).

A.2.5.2.4 Fish COPC summary

COPCs for fish were identified separately based on three different approaches: diet, tissue-residue, and surface water. COPCs for surface water exposure were identified from all chemical classes. COPCs for dietary exposure were identified from PAHs and metals (except butyltins, mercury, and selenium). COPCs for the tissue-residue evaluation were identified from the remaining group of chemicals, including selenium, mercury, butyltins, and all organic chemicals except PAHs.

Based on the two-step screening process for each exposure evaluation, the COPCs identified for fish ROCs are presented in Table A.2-38.

Table A.2-38. COPCs selected for fish

COPC	COPCs Identified for Fish by Evaluation Type		
	Tissue Residue	Diet	Surface Water
Arsenic		X[a]	
Cadmium		X[a]	X[a]
Chromium		X[a]	
Copper		X[a]	
Mercury	X[b]		X[a]
Vanadium		X[a]	
TBT	X[c]		X[a]
Benzo(a)pyrene		X[c]	
Total PCBs	X[c]		
beta-Endosulfan	X[b]		

a Identified as a COPC for all three fish ROCs: juvenile Chinook salmon, English sole, and brown rockfish.

b Identified as a COPC for only brown rockfish.

c Identified as a COPC for brown rockfish and English sole but not juvenile Chinook salmon.

COPC – chemical of potential concern
ROC – receptor of 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 if they met at least two of the following three criteria:

- Detection in at least 5% of EW surface sediment samples analyzed for the chemical
- Identification as a bioaccumulative chemical in aquatic organisms by EPA (2000a)
- Detection in any wildlife prey tissue sample collected from the EW

Table A.2-39 presents a summary of 55 chemicals identified as COIs as a result of this screen. Detailed results of the COI screening step for wildlife are presented in Attachment 2.

Table A.2-39. Chemicals identified as COIs for wildlife ROCs

COIs		
Metals		
Arsenic	Lead	Silver
Cadmium	Mercury	Vanadium
Chromium	Molybdenum	Zinc
Cobalt	Nickel	
Copper	Selenium	
Organometals		
Monobutyltin	Dibutyltin	TBT
PAHs		
1-Methylnaphthalene	Benzo(b)fluoranthene	Fluoranthene
2-Methylnaphthalene	Benzo(g,h,i)perylene	Fluorene
Acenaphthene	Benzo(k)fluoranthene	Indeno(1,2,3-cd)pyrene
Acenaphthylene	Chrysene	Naphthalene
Anthracene	Dibenzo(a,h)anthracene	Phenanthrene
Benzo(a)anthracene	Dibenzofuran	Pyrene
Benzo(a)pyrene		
Other SVOCs		
1,4-Dichlorobenzene	Pentachlorophenol	Phenol
PCBs		
Total PCBs [a]		
PCB congeners		
Dioxins/Furans		
Dioxins/furans		

Table A.2-39. Chemicals identified as COIs for wildlife ROCs (cont.)

COIs		
Organochlorine Pesticides		
4,4′-DDD	alpha-Chlordane	Mirex
4,4′-DDE	beta-Chlordane	cis-Nonachlor
4,4′-DDT	beta-Endosulfan	trans-Nonachlor
Dieldrin	Heptachlor	Oxychlordane
alpha-BHC	Heptachlor epoxide	

[a] Calculated as the sum of Aroclors.

BHC – benzene hexachloride
COI – chemical of interest
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
ROC – receptor of concern
SVOC – semivolatile organic compound
TBT – tributyltin

In the second step of the COPC screening process, the maximum dietary dose of each COI based on the ingestion of prey was compared with a dose-based NOAEL for that chemical. Other pathways such as direct contact with sediment or water are considered insignificant (see Section A.2.6) and therefore were not included. The incidental sediment ingestion and water ingestion pathways are such a small component of the overall dose that using the maximum prey concentration to calculate the maximum exposure dose results in an appropriate screening level. If the maximum exposure dose was greater than the NOAEL, the chemical was identified as a COPC for wildlife. For each COI, the scientific literature was searched to identify TRVs. The literature search included BIOSIS databases, EPA's ECOTOX database, the National Library of Medicine's TOXNET database, the USFWS's Contaminant Review series, the Oak Ridge National Laboratory's Risk Assessment Information System, and EPA's Integrated Risk Information System (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).

For wildlife ROCs, chemical exposure was evaluated as a daily dietary dose and expressed in mg/kg bw/day. In some cases, the toxicity literature presented data only as a concentration in food, so these values were converted to a daily dose using the animal's body weight and ingestion rate. The following guidelines were considered in the selection of TRVs for wildlife:

- 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 were 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 Japanese 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. Other reproductive effects (e.g., hatchability) based on studies with chickens or Japanese quail were considered for TRV selection.
- Toxicity studies conducted with chemical forms not likely found in the EW, such as the fungicide methylmercury dicyandiamide, were not used to develop TRVs. Toxicity of these chemical forms is not comparable with the toxicity of forms of chemicals present in the EW.
- Controlled laboratory studies of single chemical exposure with statistically significant responses were given preference. Studies with clear dose-response relationships were also given preference.
- Chronic exposure duration studies were preferred if available.

After the literature search was conducted, all acceptable studies for TRV derivation were compiled, as presented in Attachments 7 and 8. For each COI, available TRVs were selected for both the NOAEL and the LOAEL. TRV selection rules and uncertainty factors discussed in the introduction to Section A.2.5 were used.

Of the 55 chemicals or chemical groups identified as COIs, TRVs were identified for 49 COIs for birds and 34 COIs for mammals (Tables A.2-40 and A.2-41, respectively). Selected TRVs are presented in Table A.2-42 for birds and in Table A.2-43 for mammals.

Table A.2-40. Results of TRV search for COIs for birds

COIs for Birds		
COIs with TRVs		
Arsenic	Acenaphthylene[a]	Pentachlorophenol
Cadmium	Anthracene[a]	Total PCBs[b]
Chromium	Benzo(a)anthracene[a]	Dioxins/furans[c]
Cobalt	Benzo(a)pyrene[a]	4,4′-DDD[d]
Copper	Benzo(b)fluoranthene[a]	4,4′-DDE[d]
Lead	Benzo(g,h,i)perylene[a]	4,4′-DDT[d]
Mercury	Benzo(k)fluoranthene[a]	Dieldrin
Molybdenum	Chrysene[a]	alpha-BHC[e]
Nickel	Dibenzo(a,h)anthracene[a]	alpha-Chlordane[f]
Selenium	Dibenzofuran[a]	beta-Chlordane[f]
Vanadium	Fluoranthene[a]	beta-Endosulfan[g]
Zinc	Fluorene[a]	Heptachlor
TBT	Indeno(1,2,3-cd)pyrene[a]	Heptachlor epoxide[h]
Total PAHs[a]	Naphthalene[a]	cis-Nonachlor[f]
1-Methylnaphthalene[a]	Phenanthrene[a]	trans-Nonachlor[f]
2-Methylnaphthalene[a]	Pyrene[a]	Oxychlordane[f]
Acenaphthene[a]		
COIs without TRVs		
Silver	Dibutyltin	Phenol
Monobutyltin	1,4-Dichlorobenzene	Mirex

[a] TRVs were based on PAH mixtures derived from Patton and Dieter (1980). Data are not available for individual PAHs, with the exception of benzo(a)anthracene, benzo(a)pyrene, naphthalene, and pyrene. The available TRV for total PAHs was used for those PAHs without TRVs.

[b] The lowest LOAEL for any PCB Aroclor or PCB mixture was used as a TRV for comparison to the maximum dose of total PCBs in bird diets.

[c] A TRV for 2,3,7,8-TCDD was used for comparison to the maximum dose of the dioxins/furan TEQ in bird diets.

[d] The lowest LOAEL from any study based on exposure to DDD, DDE, DDT, or a mixture of these compounds was used as the TRV and was compared with the maximum dose of total DDTs for pigeon guillemot and osprey.

[e] A TRV for alpha-BHC was not available, so a TRV for gamma-BHC was used.

[f] The only avian toxicity studies for chlordane or related compounds were conducted with technical chlordane, which is a mixture including these compounds, or with gamma-chlordane. The LOAEL for technical chlordane (i.e., total chlordane) was used as the TRV and was compared to the maximum dose of total chlordane compounds in bird diets.

[g] A TRV for beta-endosulfan was not available, so the TRV for endosulfan was used.

[h] A TRV for heptachlor epoxide was not available, so the TRV for heptachlor was used.

BHC – benzene hexachloride
COI – chemical of interest
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
LOAEL – lowest-observed-adverse-effect level
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
TBT – tributyltin
TCDD – tetrachlorodibenzo-*p*-dioxin
TEQ – toxic equivalent
TRV – toxicity reference value

Table A.2-41. Results of TRV search for COIs for mammals

COIs for Mammals		
Chemicals with TRVs		
Arsenic	Dibutyltin	Dieldrin
Cadmium	TBT	alpha-BHC[d]
Chromium	1-Methylnaphthalene	alpha-Chlordane[e]
Cobalt	2-Methylnaphthalene	beta-Chlordane[e]
Copper	Benzo(a)pyrene	beta-Endosulfan[f]
Lead	Naphthalene	Heptachlor
Mercury	Phenol	Heptachlor epoxide[g]
Molybdenum	Total PCBs[a]	cis-Nonachlor[e]
Nickel	Dioxins/furans[b]	trans-Nonachlor[e]
Selenium	4,4′DDD[c]	Oxychlordane[e]
Vanadium	4,4′-DDE[c]	
Zinc	4,4′-DDT[c]	
Chemicals without TRVs		
Silver	Benzo(g,h,i)perylene	Indeno(1,2,3-cd)pyrene
Monobutyltin	Benzo(k)fluoranthene	Phenanthrene
Acenaphthene	Chrysene	Pyrene
Acenaphthylene	Dibenzo(a,h)anthracene	1,4-Dichlorobenzene
Anthracene	Dibenzofuran	Pentachlorophenol
Benzo(a)anthracene	Fluoranthene	Mirex
Benzo(b)fluoranthene	Fluorene	

a The lowest LOAEL for any PCB Aroclor or PCB mixture was used as a TRV for comparison to the maximum dose of total PCBs in bird diets.

b A TRV for 2,3,7,8-TCDD was used for dioxins/furans.

c The lowest LOAEL from any study based on exposure to DDD, DDE, DDT, or a mixture of these compounds was used as the TRV and was compared with the maximum dose of total DDTs for harbor seal and river otter.

d A TRV for alpha-BHC was not available, so a TRV for gamma-BHC was used.

e The only mammalian toxicity studies for chlordane or related compounds were conducted with technical chlordane, which is a mixture including these compounds. The LOAEL for technical chlordane (i.e., total chlordane) was used as the TRV and was compared to the maximum dose of total chlordane compounds in mammal diets.

f A TRV for beta-endosulfan was not available, so the TRV for endosulfan was used.

g A TRV for heptachlor epoxide was not available, so the TRV for heptachlor was used.

BHC – benzene hexachloride
COI – chemical of interest
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
LOAEL – lowest-observed-adverse-effect level
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
TBT – tributyltin
TCDD – tetrachlorodibenzo-*p*-dioxin
TRV – toxicity reference value

Table A.2-42. 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	mallard	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, survival	Pattee (1984)
Mercury	American kestrel	0.0146[b]	0.073	reproduction	Albers et al. (2007)
Molybdenum	chicken	6.0[b]	30	reproduction	Lepore and Miller (1965)
Nickel	mallard	77	107	survival, growth	Cain and Pafford (1981)
Selenium	mallard	0.50	0.82	reproduction	Heinz et al. (1987); Heinz et al. (1989)
Vanadium	chicken	1.2	2.3	growth	Ousterhout and Berg (1981)
Zinc	chicken	82	124	growth	Roberson and Schaible (1960)
Organometals					
TBT	Japanese quail	1.3	3.2	reproduction	Coenen et al. (1992)
PAHs					
Benzo(a)anthracene	bobwhite quail	0.58	na	survival	Brausch et al. (2010)
Benzo(a)pyrene	pigeon	0.28[b]	1.4	reproduction	Hough el al. (1993)
Total PAHs[c]	mallard	8.0	40	growth	Patton and Dieter (1980)
SVOCs					
Pentachlorophenol	chicken	22	63	growth	Prescott el al. (1982)
PCBs					
PCBs[d]	screech owl	0.49	na	reproduction	McLane and Hughes (1980)

Table A.2-42. TRVs selected for bird COIs (cont.)

COI	Test Species	NOAEL (mg/kg bw/day)	LOAEL (mg/kg bw/day)	Endpoint	Source
	ringed turtle dove	na	1.4	reproduction	Peakall et al. (1972); Peakall and Peakall (1973)
Dioxins/Furans					
2,3,7,8-TCDD	ring-necked pheasant	0.000014	0.00014	reproduction	Nosek et al. (1992)
Organochlorine Pesticides					
Total chlordane	bobwhite quail	0.6	na	growth, survival	Ludke (1976)
	bobwhite quail	na	20	survival	Hill et al. (1975); Heath et al. (1972)
Total DDT[e]	barn owl	0.064[b]	0.32	reproduction	Mendenhall et al. (1983)
Dieldrin	quail	0.080	0.12	survival	DeWitt (1956)
Endosulfan	gray partridge	10	na	reproduction	Abiola (1992)
alpha-BHC[f]	mallard	1.6	3.6	reproduction	Chakravarty and Lahiri (1986)
Heptachlor	bobwhite quail	0.5[a]	5	survival	Hill et al (1975); Heath et al. (1972)

[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] Mallards were exposed to a paraffin wax mixture containing a mixture of PAHs that did not included benzo(a)pyrene.
[d] The NOAEL was based on exposure of screech owls to PCB Aroclor 1248 and the LOAEL was based on exposure of ringed turtle doves to PCB Aroclor 1254.
[e] Barn owls were exposed to DDE.
[f] Mallards were exposed to gamma-BHC; toxicity data for alpha-BHC were not available.

bw – body weight
BHC – benzene hexachloride
COI – chemical of interest
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
LOAEL – lowest-observed-adverse-effect level
na – not applicable or 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
TCDD – tetrachlorodibenzo-*p*-dioxin
TRV – toxicity reference value

Table A.2-43. TRVs selected for mammal COIs

COI	Test Species	NOAEL (mg/kg bw/day)	LOAEL (mg/kg bw/day)	Endpoint	Source
Metals					
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	0.10[a]	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[b]	0.0084	growth	Verschuuren et al. (1976)
Molybdenum	mouse	0.258[a]	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	0.080	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					
Dibutyltin	rat	na	7.6	reproduction, growth	Ema et al. (2003)
	rat	3.8	ns	growth	Harazono and Ema (2003)
TBT	rat	0.4	2.0	reproduction	Omura et al. (2001)
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[b]	10	reproduction	MacKenzie and Angevine (1981)
Naphthalene	rat	50	150	growth	Navarro et al. (1991)

Table A.2-43. TRVs selected for mammal COIs (cont.)

COI	Test Species	NOAEL (mg/kg bw/day)	LOAEL (mg/kg bw/day)	Endpoint	Source
Other SVOCs					
Phenol	rat	60[c]	120[c]	growth	Argus Research Laboratories (1997), as cited in IRIS (EPA 2006b)
	rat	60[c]	120[c]	reproduction	Charles River Laboratories (1988) and NTP (1983), as cited in IRIS (EPA 2006b)
PCBs					
PCBs	mink	0.045[d]	0.089	reproduction	Brunstrom et al. (2001)
Dioxins/Furans					
2,3,7,8-TCDD	guinea pig	6.5×10^{-7}	4.9×10^{-6}	growth	DeCaprio et al. (1986)
Organochlorine Pesticides					
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[b]	0.19	reproduction	Treon and Cleveland (1955)
Endosulfan	mouse	0.84	2.5	growth and survival	Hack et al. (1995)
alpha-BHC[e]	rat	64	na	growth	Srinivasan et al. (1991)
Heptachlor	mink	1.0	1.8	survival, growth and reproduction	Crum et al. (1993)

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

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

c Both studies had the same LOAEL and NOAEL.

d NOAEL was 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.

e Rats were exposed to gamma-BHC; toxicity data were not available for alpha-BHC.

BHC – benzene hexachloride
bw – body weight
COI – chemical of interest
DDT – dichlorodiphenyltrichloroethane
IRIS – Integrated Risk Information System
LOAEL – lowest-observed-adverse-effect level
na – not applicable or 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
TCDD – tetrachlorodibenzo-*p*-dioxin
TRV – toxicity reference value

To identify COPCs for birds and mammals, the estimated maximum dietary doses (expressed in mg/kg bw/day) for each of the four wildlife ROCs were compared with the NOAEL TRVs. The maximum dietary doses were calculated using the following equation:

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

Where:

Dose	=	amount of COI ingested per day via food (mg COI/kg bw/day)
FIR	=	food ingestion rate (kg ww food/day)
C_{food}	=	maximum concentration in prey tissue (mg COI/kg ww)
BW	=	wildlife species body weight (kg ww)

The maximum daily ingested doses for birds and mammals were calculated using the maximum detected tissue concentrations in any of the prey species in each of the ROC species' diets.[14] Incidental ingestion of sediment and water ingestion were not considered in the COPC selection for wildlife ROCs because of the small amount of sediment and water assumed to be ingested and the conservative approach of using the maximum tissue concentration.[15] It was assumed that each ROC obtained all of its diet from the EW (i.e., 100% site use). If a COI was not detected, one-half the maximum RL was used in the calculation. The body weights and food ingestion rates used to calculate the maximum dietary doses for each wildlife ROC are presented in Section A.5. The derivation of these values is described in detail in Section A.5.1.2.

For pigeon guillemot, the maximum dietary exposures to mercury, total PCBs, PCB TEQ and total TEQ exceeded their respective NOAEL TRVs (Table A.2-44). For osprey, total PCBs was the only COI with a maximum exposure that exceeded its NOAEL TRV (Table A.2-45). Based on this screening evaluation, total PCBs is a COPC for both pigeon guillemot and osprey; and mercury, PCB TEQ, and total TEQ are COPCs for pigeon guillemot (Table A.2-44); these COPCs are further evaluated in the risk characterization.

[14] The prey species in each of the ROC's diets are described in detail in the wildlife exposure assessment (Section A.5, Table A.5-1)

[15] Incidental ingestion of sediment and water ingestion pathways were an average of 2.9 and 0.05%, respectively, of the total ingested dose of each COC for each wildlife receptor in the risk calculations conducted in Section A.6.3.

Table A.2-44. Results of COPC screen for pigeon guillemot

COI	Maximum Chemical Concentration in Prey (mg/kg ww)	Calculated Maximum Dietary Dose (mg/kg bw/day)[a]	NOAEL TRV (mg/kg bw/day)	Selected as a COPC?
Metals				
Arsenic	6.81 J	1.4	10	no
Cadmium	3.1	0.62	1.5	no
Chromium	1.0	0.20	1.0	no
Cobalt	0.36 J	0.072	2.31	no
Copper	31.3	6.3	21	no
Lead	1.2 J	0.24	5.82	no
Mercury	0.418	**0.084**	0.0146	yes
Molybdenum	0.6 J	0.12	6.0	no
Nickel	2.3	0.46	77	no
Selenium	1.36	0.27	0.50	no
Vanadium	0.82 J	0.16	1.2	no
Zinc	59.0	12	82	no
Organometals				
TBT	0.42	0.084	1.3	no
PAHs				
Benzo(a)anthracene	0.095	0.019	0.58	no
Benzo(a)pyrene	0.13 J	0.026	0.28	no
Total PAHs	1.04 J	0.21	8.0	no
SVOCs				
Pentachlorophenol	0.0082 J	0.0016	22	no
PCBs and Dioxins/Furans				
Total PCBs	7.9 J	**1.6**	0.49	yes
PCB TEQ	9.4×10^{-5}	**1.9×10^{-5}**	1.4×10^{-5}	yes
Dioxin and furan TEQ[b]	9.2×10^{-6} J	1.8×10^{-6}	1.4×10^{-5}	no
Total TEQ	9.8×10^{-5}	**2.0×10^{-5}**	1.4×10^{-5}	yes
Organochlorine Pesticides				
Total DDTs	0.054 J	0.011	0.064	no
Dieldrin	0.00076 J	0.00015	0.08	no
alpha-BHC	0.00058 J	0.00012	1.6[c]	no
Total chlordane	0.018 J	0.0036	0.6	no
beta-Endosulfan	0.013	0.0026	10[d]	no
Heptachlor	0.0001 J	0.000020	0.5	no
Heptachlor epoxide	0.00031 J	0.000062	0.5[e]	no

[a] Calculated using Equation 2-2.

[b] The doses from the maximum PCB TEQ, dioxin and furan TEQ, and total TEQ in crab tissue were compared with the NOAEL TRV for 2,3,7,8-TCDD. The TEQs were calculated using bird TEFs from Van den Berg et al. (2006) and one-half the RL for non-detected congeners.

Table A.2-44. Results of COPC screen for pigeon guillemot (cont.)

[c] TRV is for gamma-BHC, which was used as a surrogate for alpha-BHC.
[d] TRV is for endosulfan, which was used as a surrogate for beta-endosulfan.
[e] TRV is for heptachlor, which was used as a surrogate for heptachlor epoxide.

BHC – benzene hexachloride
bw – body weight
COI – chemical of interest
COPC – chemical of potential concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
J – estimated concentration
NOAEL – no-observed-adverse-effect level
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
RL – reporting limit
SVOC – semivolatile organic compound
TBT – tributyltin
TCDD – tetrachlorodibenzo-*p*-dioxin
TEF – toxic equivalency factor
TEQ – toxic equivalent
TRV – toxicity reference value
ww – wet weight

Bold and underline identify the maximum dietary exposure concentrations that are greater than the NOAEL.

Table A.2-45. Results of COPC screen for osprey

COI	Maximum Chemical Concentration in Prey (mg/kg ww)	Calculated Maximum Dietary Dose (mg/kg bw/day)[a]	NOAEL TRV (mg/kg bw/day)	Selected as a COPC?
Metals				
Arsenic	1.24 J	0.26	10	no
Cadmium	0.04 U	0.008	1.6	no
Chromium	0.4	0.084	1.0	no
Cobalt	0.12	0.025	2.31	no
Copper	3.16	0.67	21	no
Lead	0.4 U	0.085	5.82	no
Mercury	0.05	0.011	0.0146	no
Molybdenum	0.4	0.085	6.0	no
Nickel	0.2 U	0.042	77	no
Selenium	0.6 J	0.13	0.50	no
Vanadium	0.28	0.059	1.2	no
Zinc	46.2	9.8	82	no
Organometals				
TBT	0.067	0.014	1.3	no
PAHs				
Benzo(a)anthracene	0.0048	0.0010	0.58	no
Benzo(a)pyrene	0.00068 J	0.00014	0.28	no
Total PAHs	0.0454 J	0.010	8.0	no
SVOCs				
Pentachlorophenol	0.38 U	0.080	22	no

Table A.2-45. Results of COPC screen for osprey (cont.)

COI	Maximum Chemical Concentration in Prey (mg/kg ww)	Calculated Maximum Dietary Dose (mg/kg bw/day)[a]	NOAEL TRV (mg/kg bw/day)	Selected as a COPC?
PCBs and Dioxins/Furans				
Total PCBs	5.4	**<u>1.1</u>**	0.49	yes
PCB TEQ	3.9×10^{-5}	8.2×10^{-6}	1.4×10^{-5}	no
Dioxin/furan TEQ[b]	4.94×10^{-6} J	1.0×10^{-6}	1.4×10^{-5}	no
Total TEQ	4.4×10^{-5}	9.2×10^{-6}	1.4×10^{-5}	no
Organochlorine Pesticides				
Total DDTs	0.011 J	0.0023	0.064	no
Dieldrin	0.00076 J	0.00016	0.08	no
alpha-BHC	0.001 U	0.00021	1.6[c]	no
Total chlordane	0.003 J	0.00064	0.6	no
beta-Endosulfan	0.002 U	0.00042	10[d]	no
Heptachlor	0.001 U	0.00021	0.5	no
Heptachlor epoxide	0.001 U	0.00021	0.5[e]	no

[a] Calculated using Equation 2-2.

[b] The doses from the maximum PCB TEQ, dioxin/furan TEQ, and total TEQ in prey tissue were compared with the NOAEL TRV for 2,3,7,8-TCDD. The TEQs were calculated using bird TEFs from Van den Berg et al. (2006) and one-half the RL for non-detected congeners.

[c] TRV is for gamma-BHC, which was used as a surrogate for alpha-BHC.

[d] TRV is for endosulfan, which was used as a surrogate for beta-endosulfan.

[e] TRV is for heptachlor, which was used as a surrogate for heptachlor epoxide.

BHC – benzene hexachloride
bw – body weight
COI – chemical of interest
COPC – chemical of potential concern
DDT – dichlorodiphenyltrichloroethane
J – estimated concentration
NOAEL – no-observed-adverse-effect level
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
RL – reporting limit
SVOC – semivolatile organic compound
TBT – tributyltin
TCDD – tetrachlorodibenzo-*p*-dioxin
TEF – toxic equivalency factor
TEQ – toxic equivalent
TRV – toxicity reference value
U – not detected at given concentration
ww – wet weight

<u>Bold and underline</u> identify the maximum dietary exposure concentrations that are greater than the NOAEL TRV.

For river otter, estimated maximum exposure doses of mercury, selenium, total PCBs, PCB TEQ, and total TEQ were greater than their respective NOAEL TRVs (Table A.2-46). For harbor seal, estimated maximum exposure doses of mercury, total PCBs, PCB TEQ, and total TEQ exceeded their respective NOAEL TRVs (Table A.2-47). Risks associated with these COPCs are evaluated in the risk characterization. A summary of the COPCs that were evaluated for each wildlife ROC in the risk characterization is presented in Table A.2-48.

Table A.2-46. Results of COPC screen for river otter

COI	Maximum Chemical Concentration in Prey (mg/kg ww)	Calculated Maximum Dietary Dose (mg/kg bw/day)[a]	NOAEL TRV (mg/kg bw/day)	Selected as a COPC?
Metals				
Arsenic	6.81 J	0.89	2.6	no
Cadmium	3.1	0.40	3.5	no
Chromium	1.0	0.13	1,466	no
Cobalt	0.36 J	0.047	0.10	no
Copper	31.3	4.1	18	no
Lead	1.2 J	0.16	11	no
Mercury	0.418	**0.054**	0.0017	yes
Molybdenum	0.6 J	0.078	0.258	no
Nickel	1.2	0.16	8.4	no
Selenium	1.36	**0.18**	0.055	yes
Vanadium	0.82 J	0.11	1.05	no
Zinc	59.0	7.7	160	no
Organometals				
Dibutyltin	0.0253	0.0033	3.8	no
TBT	0.42	0.055	0.4	no
PAHs				
1- Methylnaphthalene	0.015	0.0020	150	no
2- Methylnaphthalene	0.043	0.0056	54	no
Benzo(a)pyrene	0.13 J	0.017	2.0	no
Naphthalene	1.04 J	0.017	50	no
SVOCs				
Phenol	0.67	0.087	60	no
PCBs and Dioxins/Furans				
Total PCBs	7.9 J	**1.0**	0.045	yes
PCB TEQ	6.0×10^{-5}	**7.8×10^{-6}**	6.5×10^{-7}	yes
Dioxin/furan TEQ[b]	3.0×10^{-6} J	3.9×10^{-7}	6.5×10^{-7}	no
Total TEQ	6.2×10^{-5}	**8.1×10^{-6}**	6.5×10^{-7}	yes
Organochlorine Pesticides				
Total DDTs	0.054 J	0.0070	1.2	no
Dieldrin	0.00076 J	0.00010	0.038	no
alpha-BHC	0.00058 J	0.000076	64[c]	no
Total chlordane	0.018 J	0.0023	0.18	no
beta-Endosulfan	0.013	0.0017	0.84[d]	no
Heptachlor	0.0001 J	0.000013	1.0	no
Heptachlor epoxide	0.00031 J	0.000040	1.0[e]	no

[a] Calculated using Equation 2-2.

Table A.2-46. Results of COPC screen for river otter (cont.)

[b] The doses from the maximum PCB TEQ, dioxin/furan TEQ, and total TEQ in prey tissue were compared to the NOAEL TRV for 2,3,7,8-TCDD. The TEQs were calculated using mammalian TEFs from Van den Berg et al. (2006) and one-half the RL for non-detected congeners.

[c] TRV is for gamma-BHC, which was used as a surrogate for alpha-BHC.

[d] TRV is for endosulfan, which was used as a surrogate for beta-endosulfan.

[e] TRV is for heptachlor, which was used as a surrogate for heptachlor epoxide.

bw – body weight
COI – chemical of interest
COPC – chemical of potential concern
DDT – dichlorodiphenyltrichloroethane
J – estimated concentration
NOAEL – no-observed-adverse-effect level
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
RL – reporting limit
SVOC – semivolatile organic compound
TBT – tributyltin
TCDD – tetrachlorodibenzo-*p*-dioxin
TEF – toxic equivalency factor
TEQ – toxic equivalent
TRV – toxicity reference value
ww – wet weight

Bold and underline identify the maximum dietary exposure concentrations that are greater than the NOAEL TRV.

Table A.2-47. Results of COPC screen for harbor seal

COI	Maximum Chemical Concentration in Prey (mg/kg ww)	Calculated Maximum Dietary Dose (mg/kg bw/day)[a]	NOAEL TRV (mg/kg bw/day)	Selected as a COPC?
Metals				
Arsenic	4.18 J	0.13	2.6	no
Cadmium	0.04	0.0012	3.5	no
Chromium	0.6	0.019	1,466	no
Cobalt	0.12	0.0037	0.10	no
Copper	3.16	0.10	18	no
Lead	0.4 U	0.012	11	no
Mercury	0.418	**<u>0.013</u>**	0.0017	yes
Molybdenum	0.4	0.012	0.258	no
Nickel	1.0	0.031	8.4	no
Selenium	0.85	0.026	0.055	no
Vanadium	0.49	0.015	1.05	no
Zinc	46.2	1.4	160	no
Organometals				
Dibutyltin	0.024	0.00074	3.8	no
TBT	0.42	0.013	0.4	no
PAHs				
1-Methylnaphthalene	0.0031	0.00010	150	no
2- Methylnaphthalene	0.0037	0.00011	54	no
Benzo(a)pyrene	0.0079	0.00025	2.0	no
Naphthalene	0.0047 J	0.00015	50	no
SVOCs				
Phenol	0.65 U	0.020	60	no
PCBs and Dioxins/Furans				
Total PCBs	7.9 J	**<u>0.25</u>**	0.045	yes
PCB TEQ	6.0×10^{-5}	**<u>1.8×10^{-6}</u>**	6.5×10^{-7}	yes
Dioxin/furan TEQ[b]	3.0×10^{-6} J	9.3×10^{-8}	6.5×10^{-7}	no

Table A.2-47. Results of COPC screen for harbor seal (cont.)

COI	Maximum Chemical Concentration in Prey (mg/kg ww)	Calculated Maximum Dietary Dose (mg/kg bw/day)[a]	NOAEL TRV (mg/kg bw/day)	Selected as a COPC?
Total TEQ	6.2×10^{-5}	**<u>1.9×10^{-6}</u>**	6.5×10^{-7}	yes
Organochlorine Pesticides				
Total DDTs	0.054 J	0.0017	1.2	no
Dieldrin	0.00076 J	0.000024	0.038	no
alpha-BHC	0.00058 J	0.000018	64[c]	no
Total chlordane	0.0137	0.00042	0.18	no
beta-Endosulfan	0.013	0.00040	0.84[d]	no
Heptachlor	0.001 J	0.000031	1.0	no
Heptachlor epoxide	0.00014 J	0.0000043	1.0[e]	no

a Calculated using Equation 2-2.

b The doses from the maximum PCB TEQ, dioxin/furan TEQ, and total TEQ in crab tissue were compared to the NOAEL TRV for 2,3,7,8-TCDD. The TEQs were calculated using mammalian TEFs from Van den Berg et al. (2006) and one-half the RL for non-detected congeners.

c TRV is for gamma-BHC, which was used as a surrogate for alpha-BHC.

d TRV is for endosulfan, which was used as a surrogate for beta-endosulfan.

e TRV is for heptachlor, which was used as a surrogate for heptachlor epoxide.

bw – body weight
COI – chemical of interest
COPC – chemical of potential concern
DDT – dichlorodiphenyltrichloroethane
J – estimated concentration
N – tentative identification
NOAEL – no-observed-adverse-effect level
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
RL – reporting limit
SVOC – semivolatile organic compound
TBT – tributyltin
TCDD – tetrachlorodibenzo-*p*-dioxin
TEF – toxic equivalency factor
TEQ – toxic equivalent
TRV – toxicity reference value
U – not detected at given concentration
ww – wet weight

<u>Bold and underline</u> identify the maximum dietary exposure concentrations that are greater than the NOAEL TRV.

Table A.2-48. COPCs for birds and mammals

COPC	COPCs Identified for Birds and Mammals			
	Pigeon Guillemot	Osprey	River Otter	Harbor Seal
Mercury	X		X	X
Selenium			X	
Total PCBs	X	X	X	X
PCB TEQ	X		X	X
Total TEQ	X		X	X

COPC – chemical of potential concern
PCB – polychlorinated biphenyl
TEQ – toxic equivalent

A.2.6 CONCEPTUAL SITE MODEL

This section presents the CSM for the EW ERA, which is a graphical representation of exposure media, transport mechanisms, exposure pathways, exposure routes, and ROCs. It provides the basis for developing exposure scenarios that are evaluated in the exposure assessment component of this ERA.

For the purposes of this ERA, sediment is the assumed to be the primary source of chemicals for all exposures at the site, regardless of the actual exposure medium (e.g., tissue, sediment, surface water). The exposure assessment for each ROC therefore focuses on scenarios that include direct or indirect pathways for sediment-associated chemicals. Examples of direct pathways include ingestion of sediment or direct contact with sediment. Indirect pathways include the ingestion of aquatic biota that have been exposed to contaminated media. Because of the potential flux of chemicals from sediment to surface water, ecological receptors may also be indirectly exposed to sediment-associated chemicals through ingestion of surface water or contact with surface water.

To understand the potential exposure pathways of a chemical from sediment to biota, including upper-trophic-level ROCs, knowledge of general food web relationships is important. Figure A.2-2 shows a generalized food web diagram for the EW, including uptake from sediment. A more specific food web diagram (Figure A.2-3) shows the interrelationships for the selected ROCs; this figure presents only the ROCs and does not include all potential prey types of each ROC or all potential ecological receptors in EW. The relationship among trophic levels illustrates the pathways for chemical transfer through the ingestion of prey. Figures A.2-2 and A.2-3 are used to further illustrate exposure pathways through the food web; the components of the food web are represented as "biota" in the CSM (as shown in Figures A.2-4 and A.2-5).

For chemicals to pose a risk to a particular ROC, the exposure pathway must be complete. Identifying complete exposure pathways prior to conducting a quantitative evaluation allows for a focused evaluation (EPA 1997a, b). An exposure pathway is considered complete if a chemical can travel from a source to an ecological receptor 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 major exposures of ecological receptors sensitive to a chemical (EPA 1997a, b) are identified as having more importance than pathways that are likely to provide a minor fraction of the total exposure of a ROC to a chemical.

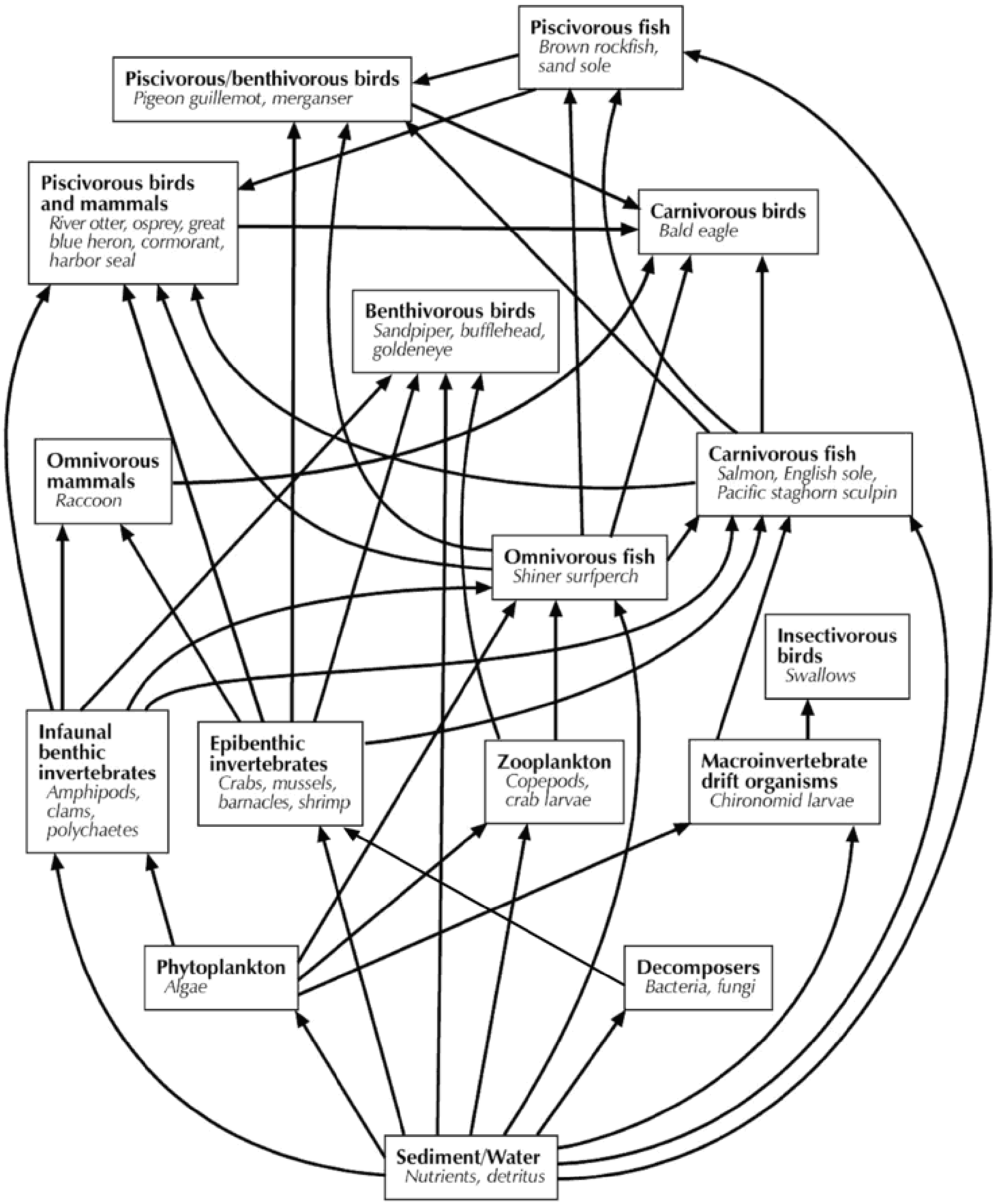
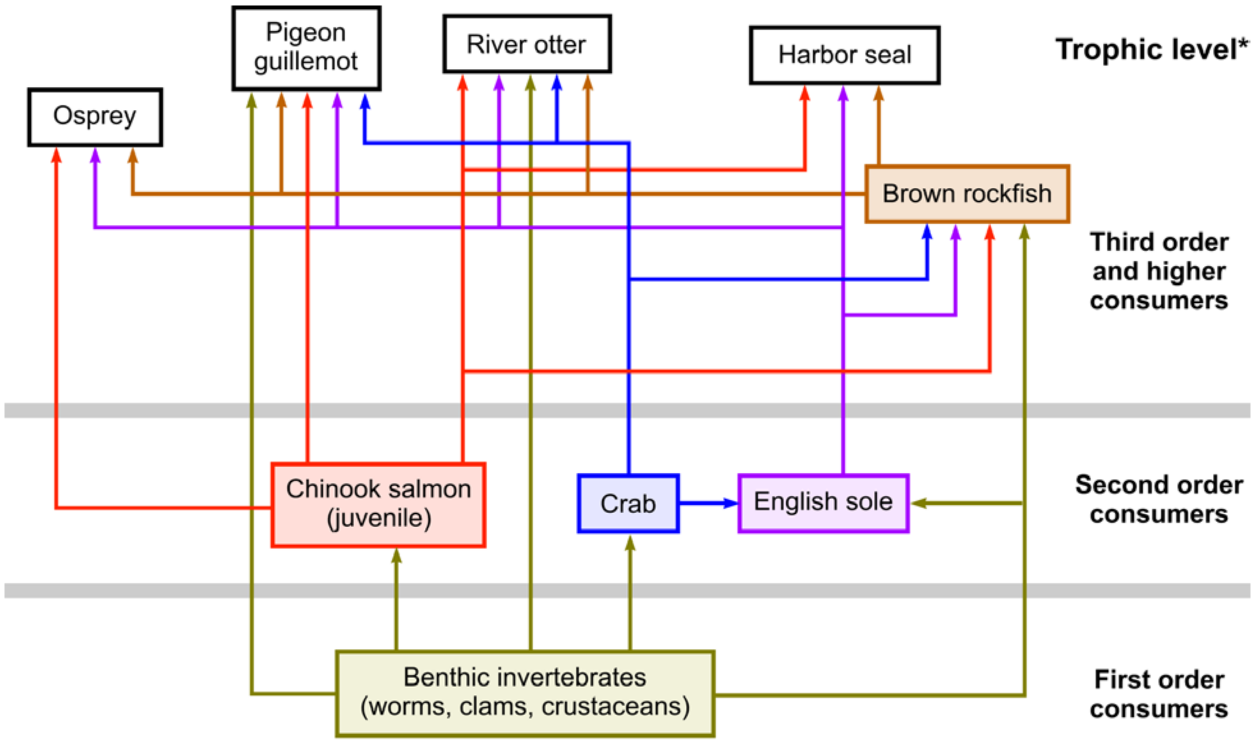


Figure A.2-2. Generalized food web diagram for the East Waterway



* Not all species feed at only one trophic level. In addition, feeding is site-specific and trophic levels tend to increase with the organism's age. Therefore, the trophic level identified for each species is approximate.

Figure A.2-3. Food web diagram illustrating connections between receptors of concern

Exposure pathways for ROCs to sediment-associated chemicals in the EW 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. Figures A.2-4 and A.2-5 present the CSMs for fish and the benthic invertebrate community and for wildlife, respectively.

- **Complete and significant –** There is a direct link between the ROC and chemical via this pathway, and the specific pathway is considered to be potentially important. Pathways classified as complete and significant are addressed in greater detail in the exposure and effects assessment sections of this ERA.
- **Complete and significance unknown –** There is a direct link between the ROC and the chemical via this pathway; however, there are insufficient data available to quantify the significance of the pathway in the overall assessment of exposure. Pathways classified as complete and significance unknown are discussed qualitatively in the uncertainty analysis sections of this ERA.

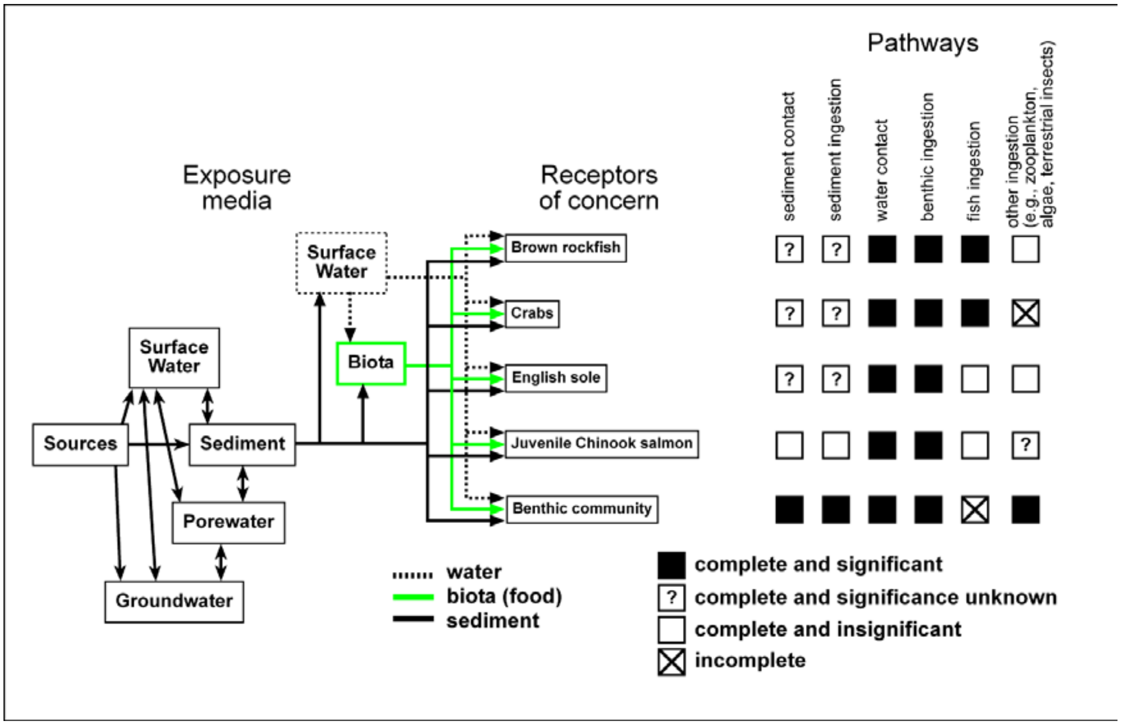


Figure A.2-4. Conceptual site model for fish, the benthic invertebrate community, and crab

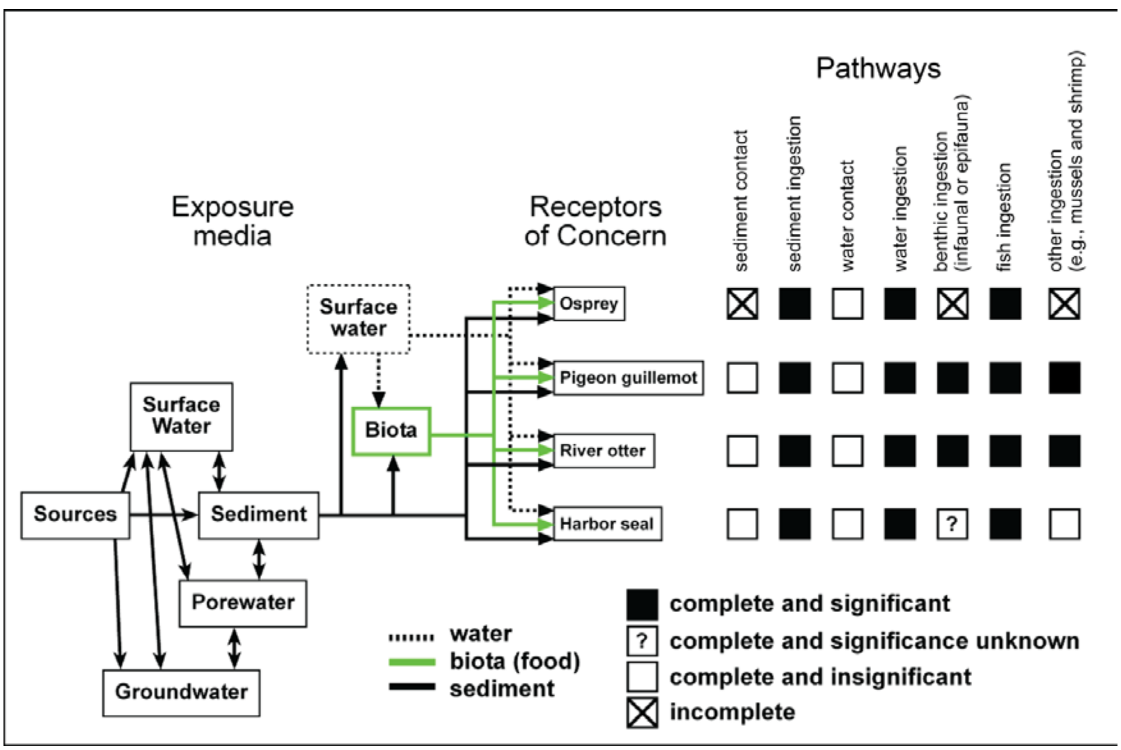


Figure A.2-5. Conceptual site model for wildlife

- **Complete and insignificant –** There is a direct link between the ROC 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 are not evaluated in this ERA.
- **Incomplete –** There is no direct pathway between the ROC and the chemical. Pathways classified as incomplete will are not evaluated in this ERA.

For the benthic invertebrate community, complete and significant pathways include sediment contact, sediment ingestion, prey ingestion, and surface water contact (Figure A.2-4). Risks to the benthic invertebrate community from sediment and surface water contact are addressed directly in this ERA. Risks from sediment and prey ingestion are evaluated indirectly through the evaluation of bioaccumulative compounds in benthic tissue residue, which integrates all exposure pathways. For crab, prey ingestion and surface water contact are complete and significant, but the significance of the sediment ingestion and sediment contact exposure pathways are unknown. All of these pathways are evaluated using the tissue-residue evaluation for crab.

For fish, the most important exposure pathway for sediment-associated chemicals in the EW is that of prey ingestion (Figure A.2-4), which is addressed through the evaluation chemicals in prey items using a dietary approach. Water contact is also a complete and significant pathway for fish and is addressed in this ERA. Sediment contact and sediment ingestion are complete pathways for fish; they are insignificant for juvenile Chinook salmon, and their significance is minor for brown rockfish and English sole and is addressed in the exposure assessment for these two ROCs. For most fish COPCs, exposure pathways are assessed through a tissue-residue evaluation, which integrates all forms of exposure, including from water, sediment, and diet.

For wildlife, the ingestion of prey, surface water, and sediment are all complete and significant pathways that are addressed in the ERA, although the surface water and sediment ingestion pathways are typically a very small portion of the overall exposure when compared with the prey ingestion pathway (Figure A.2-5). The sediment and surface water contact pathways are considered complete but insignificant for all wildlife ROCs; and sediment contact pathway is considered incomplete for osprey. The water and sediment contact pathways are considered insignificant compared with other pathways for wildlife. The feathers and fur on birds and mammals limit direct exposure of their skin, although some areas are more exposed (e.g., the legs and feet and under the wings for birds).

A.2.7 Assessment Endpoints and Measures of Effect and Exposure

An assessment endpoint is 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., important to tribal, commercial, and sport fisheries, valued for its beauty, or intrinsic value by the general public). An assessment endpoint must define both the valued entity and the attribute of the entity to be protected. Assessment endpoints provide direction for the risk assessment and are the basis for the analyses; they are typically selected for a population of organisms based on organism-level attributes. Survival, growth, or reproduction are the most commonly used assessment endpoints, although biomarker, behavioral, and histological endpoints may also be used when they are linked to direct effects on the assessment endpoints. In this ERA, the populations of receptor organisms are assumed to be those individuals that occur within the boundaries of the EW. For species with home ranges smaller than the EW, all individuals are assumed to reside solely within the EW. For species with home ranges larger than the EW, the individuals present in the EW are assumed to represent a distinct population with regard to chemical exposure, although they are not assumed to forage solely within the EW. For juvenile Chinook salmon, which is a threatened and endangered (T&E) species, risks to individual organisms are important (EPA 1998), although specific EPA guidance is not available for the evaluation of risk to individual organisms. Endpoints selected for the EW ERA were amenable to evaluation using acceptable historical data or data collected from the EW specifically for risk assessment purposes.

Survival, growth, and reproduction are the measurement endpoints that were evaluated for all ROCs in this ERA. Reproductive effects considered for juvenile Chinook salmon were limited to effects associated with exposure during the smolt or juvenile life stages because spawning and earlier life stages do not occur in the EW, and the majority of exposure as adults occurs in the Pacific Ocean and Puget Sound. No data have been identified linking exposure of salmon as juveniles to chemicals that could later cause reproductive effects in adults. In addition, adult salmon are exposed to a variety of chemicals in Puget Sound and the Pacific Ocean for much longer periods of time compared to the very limited time spent in the EW. Therefore, the fraction of the total maternal chemical burden in adults accumulated from exposure to EW sources that would be passed on to embryonic life stages is very small compared to the fraction accumulated from other sources (O'Neill et al. 1998).

Biomarker, behavioral, and histological endpoints were considered for inclusion as measurement endpoints only if they could be linked to adverse effects on the ecologically relevant assessment endpoints, such as growth, mortality, or reproduction. Typically, ERAs focus on ecological effects that integrate an overall response by an organism (e.g., survival, growth, or reproduction), rather than indicators of a biochemical response (i.e., biomarkers) that may or may not result in an ecologically relevant effect. For biomarkers to be useful in determining sediment-associated risk, there must be clear dose-response data relating exposure to ecologically significant effects. Biomarkers such as deoxyribonucleic acid (DNA) adducts and cytochrome P450 1A (CYP1A) induction do not have clearly associated or quantifiable effects data and are thus categorized as a measure of exposure rather than as a measure of effect.

Research is ongoing in the area of biomarkers to better understand their significance for potential use in ERAs.

Risk associated with each assessment endpoint were evaluated through measures of exposure and measures of effect, which are defined in EPA (1998) ERA guidelines as follows:

- **Measures of exposure –** Measures of stressor existence and movement in the environment and their contact or co-occurrence with the assessment endpoint
- **Measures of effect –** Measurable changes in an attribute of an assessment endpoint or its surrogate in response to a stressor to which it is exposed

Together, each unique combination of the assessment endpoint, measure of exposure, and measure of effect constitutes a line of evidence to evaluate risk for each ROC. Lines of evidence for the various EW ROCs are presented in Table A.2-49.

Table A.2-49. Lines of evidence and methods of risk evaluation for the selected ecological receptors of concern

ROC	Assessment Endpoint	Line of Evidence: Measure of Exposure	Line of Evidence: Measure of Effect	Method of Evaluation
Benthic Invertebrates				
Benthic invertebrate community (infauna/ epifauna)	maintenance of the benthic invertebrate community in EW sediment	chemical concentrations in surface sediment	SMS and toxicity-based regional guidelines (where no standards are available)	Compare measured chemical concentrations in sediment to Washington State SMS or DMMP guidelines.
			site-specific sediment toxicity tests (survival and growth) relative to reference area sediment toxicity tests	Compare 10-day amphipod survival in site sediment to amphipod survival in reference area sediment.
				Compare 48-hr echinoderm embryo or bivalve larvae normal survival in site sediment elutriates with normal embryo/larval survival in reference area sediment.
				Compare 20-day polychaete growth in site sediment with polychaete growth in reference area sediment.
		VOC concentrations in porewater	WQC or other water TRVs based on survival and growth	Compare chemical concentrations in porewater to WQC or other relevant TRVs.
		PCB, mercury, and TBT concentrations in benthic invertebrate tissue (field-collected)	tissue-residue TRVs based on survival, growth, and reproduction	Compare measured tissue burdens to tissue-residue TRV.
		chemical concentrations in surface water	WQC or other water TRVs based on survival, growth and reproduction	Compare chemical concentrations in surface water to WQC or other relevant TRVs.
Cancrid crab	maintenance of crab populations in the EW	concentrations of chemicals in cancrid crab whole-body tissue	tissue-residue TRVs based survival, growth, and reproduction	Compare chemical concentrations measured in tissue to tissue-residue-based TRVs for crab or other decapods.

Table A.2-49. Lines of evidence and methods of risk evaluation for the selected ecological receptors of concern (cont.)

ROC	Assessment Endpoint	Line of Evidence		Method of Evaluation
		Measure of Exposure	Measure of Effect	
Fish				
Juvenile Chinook salmon	survival and growth of individual juvenile anadromous salmon in the EW	chemical concentrations in juvenile Chinook salmon whole-body tissue	tissue-residue TRVs based on survival and growth	Compare chemical concentrations in juvenile Chinook tissue to fish tissue-residue TRVs.
		chemical concentrations in prey (benthic invertebrate) tissue	dietary TRVs based on survival and growth	Compare chemical concentrations in juvenile Chinook salmon prey and juvenile Chinook salmon stomach contents to diet-based TRVs for fish.
		chemical concentrations in juvenile Chinook salmon stomach contents		
		chemical concentrations in surface water	WQC or other water TRVs based on survival and growth	Compare chemical concentrations in surface water to WQC or other relevant TRVs.
English sole	maintenance of benthivorous and planktivorous fish populations in the EW	chemical concentrations in English sole whole-body tissue	tissue-residue TRVs based on survival, growth, and reproduction	Compare chemical concentrations in English sole tissue to fish tissue-residue TRVs.
		chemical concentrations in prey (benthic invertebrate) tissue and surface sediment	dietary TRVs based on survival, growth, and reproduction	Compare chemical concentrations in English sole prey and incidentally ingested surface sediment collected throughout the EW to diet-based TRVs for fish.
		chemical concentrations in surface water	WQC or other water TRVs based on survival, growth, and reproduction	Compare chemical concentrations in surface water to WQC or other relevant TRVs.
Brown rockfish	maintenance of upper-trophic-level fish populations in the EW	chemical concentrations in brown rockfish whole-body tissue	tissue-residue TRVs based on survival, growth, and reproduction	Compare chemical concentrations in brown rockfish tissue to tissue-residue TRVs for fish.
		chemical concentrations in prey tissue (benthic invertebrate, shrimp, juvenile Chinook salmon, shiner surfperch) and surface sediment	dietary TRVs based on survival, growth, and reproduction	Compare chemical concentrations in brown rockfish prey and incidentally ingested surface sediment collected throughout the EW to diet-based TRVs for fish.
		chemical concentrations in surface water	WQC or other water TRVs based on survival, growth, and reproduction	Compare chemical concentrations in surface water to WQC or other relevant TRVs.

Table A.2-49. Lines of evidence and methods of risk evaluation for the selected ecological receptors of concern (cont.)

ROC	Assessment Endpoint	Line of Evidence		Method of Evaluation
		Measure of Exposure	Measure of Effect	
Wildlife				
Osprey	maintenance of piscivorous bird populations in the EW	chemical concentrations in prey fish tissue and surface water	dietary TRVs based on survival, growth, and reproduction of birds	Compare dietary dose calculated from chemical concentrations in fish, surface water, and incidentally ingested sediment to diet-based TRVs for birds.
Pigeon guillemot	maintenance of piscivorous/ benthivorous bird populations in the EW	chemical concentrations in prey (fish tissue, shrimp, crab, and mussels), surface sediment, and surface water	dietary TRVs based on survival, growth, and reproduction of birds	Compare dietary dose calculated from chemical concentrations in fish, invertebrates, incidentally ingested surface sediment, and surface water to diet-based TRVs for birds.
River otter	maintenance of piscivorous semi-aquatic mammal populations in the EW	chemical concentrations in prey (fish tissue, clams, crab, and mussels), surface sediment, and surface water	dietary TRVs based on survival, growth, and reproduction of mammals	Compare dietary dose calculated from chemical concentrations in fish, invertebrates, incidentally ingested surface sediment, and surface water to diet-based TRVs for mammals.
Harbor seal	maintenance of piscivorous marine mammal populations in the EW	chemical concentrations in prey fish tissue, surface sediment, and surface water	dietary TRVs based on survival, growth, and reproduction of mammals	Compare dietary dose calculated from chemical concentrations in fish, incidentally ingested surface sediment, and surface water to diet-based TRVs for mammals.

DMMP – Dredge Material Management Program
EW – East Waterway
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
ROC – receptor of concern
SMS – Washington State Sediment Management Standards
TBT – tributyltin
TRV – toxicity reference value
VOC – volatile organic compound
WQC –water quality criteria

The overall approach to this ERA as presented in Table A.2-49 was designed to address the following risk questions:

- Are concentrations of COPCs in surface sediment at levels that might pose unacceptable risks to the benthic invertebrate community in the EW?
- Are concentrations of COPCs in invertebrate tissue at levels that might pose unacceptable risks to the benthic invertebrate community in the EW?
- Are concentrations of COPCs in EW surface water at levels that might pose unacceptable risks to the benthic invertebrate community in the EW?
- Are concentrations of COPCs in EW porewater at levels that might cause an adverse effect on the benthic invertebrate community in the EW?
- Are concentrations of COPCs in crab tissues at levels that might cause an adverse effect on the crab population in the EW?
- Are concentrations of COPCs in the diet of fish that forage in the EW at concentrations that might cause an adverse effect on benthivorous, planktivorous, or upper-trophic-level fish populations or individual juvenile anadromous salmon in the EW?
- Are concentrations of COPCs in EW surface water at concentrations that might cause an adverse effect on benthivorous, planktivorous, or upper-trophic-level fish populations or individual juvenile anadromous salmon in the EW?
- Are concentrations of COPCs in the tissue of fish that forage in the EW at concentrations that might cause an adverse effect on benthivorous, planktivorous, or upper-trophic-level fish populations or individual juvenile anadromous salmon in the EW?
- Are concentrations of COPCs in the diet of birds or mammals that forage in the EW at levels that might cause an adverse effect on bird or mammal populations in the EW?

A.3 Exposure and Effects Assessment: Benthic Invertebrates

This section presents the exposure and effects assessment for the benthic invertebrate community and crab ROCs. The benthic invertebrate community assessments were based on evaluations using four types of data: surface sediment, tissue residue, surface water, and porewater. The exposure and effects assessments for the benthic invertebrate community are presented in Sections A.3.1 and A.3.2, respectively. The exposure and effects assessments for crab are presented in Sections A.3.3 and A.3.4, respectively. These assessments include the specific exposure and effects data used in the risk characterization. Summaries of both assessments are provided in Section A.3.5.

A.3.1 Benthic Invertebrate Community Exposure Assessment

The exposure assessment describes the exposure point concentrations (EPCs) used in each of the four evaluations (surface sediment, tissue residue, surface water, and porewater) used to characterize risk to the benthic invertebrate community. EPCs were developed for each COPC identified as a result of the COPC screen for each type of evaluation as summarized in Table A.2-26 in Section A.2.5.1.6.

A.3.1.1 Surface sediment exposure assessment

In this section, surface (i.e., top 10 cm) sediment data for COPCs are presented to characterize the exposure regime for the benthic invertebrate community. Summary statistics (i.e., concentrations and detection frequencies) for the 29 COPCs in the surface sediment dataset are 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. Therefore, the EPCs for the benthic community are equal to surface sediment chemical concentrations at each grab sampling location. Surface sediment sampling locations are shown on Map A.2-2. Chemicals with RLs greater than chemical criteria or without criteria are discussed in the uncertainty analysis.

Table A.3-1. Detection frequencies and chemical concentrations of surface sediment COPCs identified for the benthic invertebrate community

COPC	Unit	Detection Frequency (%)[a]	Detected Concentration		
			Minimum	Maximum	Mean
Metals					
Arsenic	mg/kg dw	162/231 (70)	2.3	241	10
Cadmium	mg/kg dw	155/231 (67)	0.126	6.76	0.9
Mercury	mg/kg dw	233/239 (97)	0.02 J	1.07 J	0.3
Zinc	mg/kg dw	231/231 (100)	25.3 J	1,230 J	100
PAHs					
2-Methylnaphthalene	µg/kg dw	87/240 (36)	9.7 J	2,800	77
Acenaphthene	µg/kg dw	126/240 (53)	10 J	3,000	170
Benzo(a)anthracene	µg/kg dw	226/240 (94)	9.8 J	9,000	350

Table A.3-1. Detection frequencies and chemical concentrations of surface sediment COPCs identified for the benthic invertebrate community (cont.)

COPC	Unit	Detection Frequency (%)[a]	Detected Concentration		
			Minimum	Maximum	Mean
Benzo(a)pyrene	µg/kg dw	225/240 (94)	15 J	7,800	340
Benzo(g,h,i)perylene	µg/kg dw	212/240 (88)	10 J	1,800	120
Total benzofluoranthenes	µg/kg dw	228/240 (95)	14 J	10,800	790
Chrysene	µg/kg dw	230/240 (96)	12 J	13,000	540
Dibenzo(a,h)anthracene	µg/kg dw	156/240 (65)	3.0 J	690	52
Dibenzofuran	µg/kg dw	107/240 (45)	7.1 J	1,700	110
Fluoranthene	µg/kg dw	233/240 (97)	12 J	75,000	1,100
Fluorene	µg/kg dw	144/240 (60)	8.6 J	3,800	140
Indeno(1,2,3-cd)pyrene	µg/kg dw	210/240 (88)	11 J	1,800	130
Phenanthrene	µg/kg dw	230/240 (96)	12 J	24,000	540
Pyrene	µg/kg dw	235/240 (98)	18 J	41,000	920
Total HPAH	µg/kg dw	237/240 (99)	3.0 J	148,000 J	4,200
Total LPAH	µg/kg dw	230/240 (96)	12 J	41,000	1,000
Phthalates					
Bis(2-ethylhexyl) phthalate	µg/kg dw	207/231 (90)	18 J	37,000	440
Butyl benzyl phthalate	µg/kg dw	101/231 (44)	4.8 J	290	41
Di-n-butyl phthalate	µg/kg dw	32/231 (14)	11 J	48,000	1,500
Other SVOCs					
1,4-Dichlorobenzene	µg/kg dw	146/231 (63)	1.9	15,000	190
2,4-Dimethylphenol	µg/kg dw	14/231 (6)	6.1	90 J	16
n-Nitrosodiphenylamine	µg/kg dw	2/231 (< 1)	160 J	180	170
Phenol	µg/kg dw	94/231 (41)	13 J	630	110
PCBs					
Total PCBs	µg/kg dw	227/240 (95)	6.0	8,400	520
Organochlorine Pesticides					
Total DDTs	µg/kg dw	8/143 (5.6)	2.3	32	8.8

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

COPC – chemical of potential concern
DDT – dichlorodiphenyltrichloroethane
dw – dry weight
EW – East Waterway
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

A.3.1.2 Tissue-residue exposure assessment

TBT and total PCBs were identified as tissue-residue COPCs for benthic invertebrates based on the COPC screen for the benthic invertebrate community presented in Section A.2.5.1.2. Benthic invertebrates are relatively immobile, so exposure was evaluated on an area-specific basis. The areas represented by each composite benthic invertebrate sample are shown on Map A.2-5 for total PCBs and Map A.2-6 for TBT. One composite sample was collected for each specific area and COPC. The EPCs for TBT and total PCBs are represented by concentrations in the benthic invertebrate samples collected and composited from those areas (i.e., one concentration for each COPC for each area) (Table A.3-2).

Table A.3-2. EPCs for the tissue-residue evaluation for the benthic invertebrate community

Area	EPC (mg/kg ww)[a]
TBT	
Area 2W	0.020
Area 3N	0.14
Area 3S	0.089
Area 4N	0.10
Area 4S	0.090
Area 5	0.39
Area 6	0.091
Area 8N	0.10
Area 8S	0.092
Area 9	0.088
Area 10N	0.057
Area 10S	0.098

Area	EPC (mg/kg ww)[a]
Total PCBs	
Area 1	0.11
Area 2E	0.38
Area 2W	0.093
Area 3	0.24
Area 4	0.15 J
Area 5	0.29 J
Area 6	0.21
Area 7	0.26 J
Area 8	0.164
Area 9	0.23
Area 10	0.25
Area 11	0.18
Area 12	0.115

[a] The EPC is equivalent to the COPC concentration in the composite sample from each area.

COPC – chemical of potential concern

EPC – exposure point concentration

PCB – polychlorinated biphenyl

TBT – tributyltin

ww – wet weight

A.3.1.3 Surface water exposure assessment

Three chemicals were identified as surface water COPCs for the benthic invertebrate community based on the COPC screen presented in Section A.2.5.1.3: cadmium, mercury, and TBT. The EW benthic invertebrate community is relatively immobile so exposure was evaluated using EPCs calculated on a location-specific basis (i.e., samples were grouped by location). In addition, as a more conservative approach, exposure was evaluated based on EPCs for each individual water sample to represent a range of conditions that were present at the time of sampling at each location.

For the location-specific evaluation, EPCs were calculated using water samples collected from 1 m above the sediment surface to represent conditions at the bottom of the water column to which benthic invertebrates would be exposed. ProUCL was used to calculate 95% upper confidence limits on the mean (UCLs) for the location-specific EPCs if there were at least six detected concentrations. The maximum concentration was used if there were fewer than six detected concentrations, as recommended by ProUCL. The ProUCL software uses both detected and undetected values and creates interpolated values for non-detects based on the perceived distribution of the detected concentrations. Once any necessary interpolation has been performed, the software analyzes the resulting data distribution to determine the most appropriate 95% UCL and makes a recommendation. Location-specific EPCs for surface water collected from 1 m above the sediment surface are presented in Table A.3-3.

Table A.3-3. Location-specific surface water EPCs for the benthic invertebrate community for samples collected 1 m from the bottom of the water column

COPC	Location ID	Detection Frequency	Mean Value (µg/L)[a]	Maximum Detection (µg/L)	Maximum RL (µg/L)	EPC (µg/L)	Statistic Used
Cadmium (dissolved)	EW-SW-1	8/8	4.8	37.8	na	37.8	maximum detect[b]
	EW-SW-2 and HNF/C	17/18	0.07	0.078	0.088	0.073	95% KM (t) UCL
	EW-SW-3	5/5	0.072	0.079 J	na	0.079	maximum detect
	EW-SW-4	0/1	0.044	na	0.088	0.088	maximum RL
	EW-SW-5	5/5	0.072	0.091	na	0.091	maximum detect
	EW-SW-6	6/6	0.067	0.078	na	0.074	95% Student's-t UCL
	HNF/E	12/12	0.0733	0.0827	na	0.0757	95% Student's-t UCL
	HNF/W	12/12	0.0732	0.0783	na	0.0751	95% Student's-t UCL
Mercury (dissolved)	EW-SW-1	3/8	0.00042	0.00110	0.00054	0.0011	maximum detect
	EW-SW-2 and HNF/C	2/9	0.0002	0.000340	0.00041	0.00034	maximum detect
	EW-SW-3	1/5	0.00026	0.00044	0.00054	0.00044	maximum detect
	EW-SW-4	0/1	0.00021	na	0.00041	0.00041	maximum RL
	EW-SW-5	0/5	0.0002	na	0.00041	0.00041	maximum RL
	EW-SW-6	0/6	0.0002	na	0.00041	0.00041	maximum RL

Table A.3-3. Location-specific surface water EPCs for the benthic invertebrate community for samples collected 1 m from the bottom of the water column (cont.)

COPC	Location ID	Detection Frequency	Mean Value (µg/L)[a]	Maximum Detection (µg/L)	Maximum RL (µg/L)	EPC (µg/L)	Statistic Used
TBT	EW-SW-1	0/8	0.0045	na	0.01	0.0050	maximum RL
	EW-SW-2 and HNF/C	1/6	0.0052	0.010 J	0.01	0.010	maximum detect
	EW-SW-3	0/5	0.0044	na	0.01	0.01	maximum RL
	EW-SW-4	0/1	0.005	na	0.01	0.01	maximum RL
	EW-SW-5	0/5	0.0042	na	0.01	0.01	maximum RL
	EW-SW-6	0/6	0.0042	na	0.01	0.01	maximum RL

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

[b] The maximum detected concentration was used for the cadmium EPC at EW-SW-1 because the 95% UCL was higher than the maximum value. The maximum value is considered an anomalous value; the total cadmium concentration in the same sample was 1.45 µg/L, and the dissolved cadmium concentration in the field duplicate collected from the same location and at the same time was 0.076 µg/L.

COPC – chemical of potential concern
EPC – exposure point concentration
ID – identification
J – estimated concentration
KM – Kaplan-Meier
RL – reporting limit
TBT – tributyltin
UCL – upper confidence limit on the mean

EPCs for individual water samples were equivalent to the detected COPC concentrations in the individual samples. The summary statistics for COPC concentrations (i.e., EPCs) in individual water samples collected from 1 m above the sediment surface are presented in Table A.3-4.

Table A.3-4. Summary statistics for surface water EPCs for the benthic invertebrate community for individual water samples collected 1 m from the bottom of the water column

COPC	Number of Samples	Number of Detects	Detection Frequency (%)	Concentration (µg/L)		
				Range of Detects	Range of RLs for Non-Detects	Mean Detect
Cadmium (dissolved)	67	65	97	0.055 – 37.8	0.088	0.65
Mercury (dissolved)	39	6	18	0.000170 – 0.00110	0.0001 – 0.00054	0.00054
TBT	31	1	3.2	0.010 J	0.008 – 0.01	na

COPC – chemical of potential concern
EPC – exposure point concentration
J – estimated concentration
na – not applicable (only one detection)
RL – reporting limit
TBT – tributyltin

A.3.1.4 Porewater exposure assessment

Naphthalene was the only porewater COPC identified for the benthic invertebrate community as a result of the COPC screen presented in Section A.2.5.1.4. Exposure was evaluated on a location-specific basis because benthic invertebrates are relatively immobile. For each location, the EPC for naphthalene is represented by the detected concentration in the porewater sample collected from that location (Map A.2-7; Table A.3-5). Naphthalene was detected in 2 of the 12 porewater samples at concentrations of 3.4 and 48 µg/L.

Table A.3-5. Summary statistics for location-specific porewater EPCs for the benthic invertebrate community

COPC	Number of Samples	Number of Detects	Detection Frequency (%)	Concentration (µg/L)	
				Range of Detects	RL for Non-Detects
Naphthalene	12	2	17	3.4 – 48	0.50

COPC – chemical of potential concern
EPC – exposure point concentration
RL – reporting limit

A.3.2 Benthic Invertebrate Community Effects Assessment

This section presents the effects assessment for the benthic invertebrate community based on the four types of evaluations (surface sediment, tissue residue, surface water, and porewater). The potential effects of surface sediment-associated COPCs were evaluated through:

- Comparisons of surface sediment chemical concentrations with SQS and cleanup screening levels (CSL) chemical criteria from the SMS or, if no SMS criteria available, then DMMP SLs and maximum levels (MLs) (Section A.3.2.1.1)
- Site-specific sediment toxicity tests (Section A.3.2.1.2.)

Effects associated with tissue residues were evaluated by comparing COPC concentrations in benthic invertebrate tissue with tissue-residue-based TRVs from the literature (Section A.3.2.2). The evaluation of effects associated with exposure to surface water (Section A.3.2.3) and porewater (Section A.3.2.4) was based on a comparison of COPC concentrations in surface water and porewater with Washington State or federal WQC. When neither state nor federal criteria were available, a TRV was selected from the scientific literature. Information on the effects of COPCs presented in this section is used in combination with the exposure data to characterize risk. The risk characterization and associated uncertainties are discussed in Section A.6.1.

A.3.2.1 Surface sediment effects assessment

This section presents the SMS criteria and DMMP guidelines used to evaluate risk based on sediment chemistry and also presents the results from the site-specific toxicity tests.

A.3.2.1.1 Surface sediment chemistry

Potential effects on the benthic invertebrate community were estimated by comparing the COPC concentrations in EW surface sediment with the SMS SQS and CSL criteria (WAC 173-204) or, when not available, the DMMP guidelines (USACE et al. 2008). This only occurred for total DDT, which has no SMS criteria. Both the SMS criteria and DMMP guidelines are based on AETs developed for PSEP (Barrick et al. 1988). AETs were derived using data from Puget Sound field-collected sediment samples that contained diverse chemical mixtures; these samples were analyzed simultaneously for chemistry, toxicity, and benthic community characteristics. The data used to derive the 1988 AETs were collected from various locations in Puget Sound between March 1982 and September 1986. AETs were developed for four endpoints[16] (i.e., amphipod mortality, abnormal development of oyster larvae, benthic invertebrate community major taxa abundance, and Microtox® bioluminescence) for 47 chemicals. An AET is the highest "no effect" chemical-specific sediment concentration above which a significant adverse biological effect for the endpoint always occurred among the several hundred samples used in its derivation. Thus, four sets of AETs were developed for each chemical, one for each toxicity test endpoint. The methods used to calculate the AETs are described by Barrick et al. (1988) and Gries and Waldow (1996).

In general, SQS were set at the LAET for each chemical; the CSLs were based on second lowest AET (2LAET) for each chemical. The SQS/LAET corresponds to a sediment concentration below which no adverse effects to biological resources are anticipated; the CSL/2LAET corresponds to a sediment concentration above which minor adverse effects are expected. Chemical concentrations that fall between the SQS and the CSL are generally interpreted as having a potential for minor adverse effects.

The DMMP ML value for total DDTs was based on an AET calculated for benthic community abundance. The SL value for total DDTs is 10% of the ML value. Table A.3-6 presents the SMS criteria and DMMP guidelines for COPCs used in the effects assessment. For certain non-ionic organic compounds, SMS criteria are expressed on an OC-normalized basis for comparison with OC-normalized chemical concentrations in sediment samples when the TOC content is > 0.5 and < 4.0%. In this ERA, samples with TOC content outside of this range were evaluated on a dry-weight basis and compared with dry-weight AETs, as presented in Table A.3-6.

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

Table A.3-6. SMS criteria and dry-weight-equivalent AETs

COPC	SMS Criteria			AETs		
	SQS	CSL	Unit	LAET	2LAET	Unit
Metals						
Arsenic	57	93	mg/kg dw	57	93	mg/kg dw
Cadmium	5.1	6.7	mg/kg dw	5.1	6.7	mg/kg dw
Mercury	0.41	0.59	mg/kg dw	0.41	0.59	mg/kg dw
Zinc	410	960	mg/kg dw	410	960	mg/kg dw
PAHs						
2-Methylnaphthalene	38	64	mg/kg OC	670	1,400	µg/kg dw
Acenaphthene	16	57	mg/kg OC	500	730	µg/kg dw
Benzo(a)anthracene	110	270	mg/kg OC	1,300	1,600	µg/kg dw
Total benzofluoranthenes	230	450	mg/kg OC	3,200	3,600	µg/kg dw
Benzo(a)pyrene	99	210	mg/kg OC	1,600	3,000	µg/kg dw
Benzo(g,h,i)perylene	31	78	mg/kg OC	670	720	µg/kg dw
Chrysene	110	460	mg/kg OC	1,400	1,800	µg/kg dw
Dibenzo (a,h)anthracene	12	33	mg/kg OC	230	540	µg/kg dw
Dibenzofuran	15	58	mg/kg OC	540	540	µg/kg dw
Fluoranthene	160	1,200	mg/kg OC	1,700	2,500	µg/kg dw
Fluorene	23	79	mg/kg OC	540	1,000	µg/kg dw
Indeno (1,2,3,-c,d)pyrene	34	88	mg/kg OC	600	690	µg/kg dw
Phenanthrene	100	480	mg/kg OC	1,500	5,400	µg/kg dw
Pyrene	1,000	1,400	mg/kg OC	2,600	3,300	µg/kg dw
HPAH	960	5,300	mg/kg OC	12,000	17,000	µg/kg dw
LPAH	370	780	mg/kg OC	5,200	13,000	µg/kg dw
Phthalates						
Bis(2-ethylhexyl) phthalate	47	78	mg/kg OC	1,300	1,900	µg/kg dw
Butyl benzyl phthalate	4.9	64	mg/kg OC	63	900	µg/kg dw
Di-n-butyl phthalate	220	260	mg/kg OC	1,400	na	µg/kg dw
Other SVOCs						
1,4-Dichlorobenzene	3.1	9	mg/kg OC	110	120	µg/kg dw
2,4-Dimethylphenol	29	29	µg/kg dw	29	72	µg/kg dw
Dibenzofuran	15	58	mg/kg OC	540	700	µg/kg dw
n-Nitrosodiphenylamine	11	11	mg/kg OC	28	40	µg/kg dw
Phenol	420	1,200	µg/kg dw	420	1,200	µg/kg dw
PCBs						
Total PCBs	12	65	mg/kg OC	130	1,000	µg/kg dw
Chlorinated Pesticides						
Total DDTs	na	na	µg/kg dw	6.9[a]	69[b]	µg/kg dw

Source: Washington State Sediment Management Standards (WAC 173-204); Barrick et al. (1988); and DMMP guidelines (USACE et al. 2008).

[a] The value presented in the DMMP SL, which was set at 10% of the ML.

Table A.3-6. SMS criteria and dry-weight-equivalent AETs (cont.)

[b] The value presented is the DMMP ML, which was based on benthic community abundance.

AET – apparent effects threshold
COPC – chemical of potential concern
CSL – cleanup screening level
DMMP – Dredged Material Management Program
dw – dry weight
HPAH – high-molecular-weight polycyclic aromatic hydrocarbon
LPAH – low-molecular-weight polycyclic aromatic hydrocarbon
LAET – lowest apparent effect threshold
2LAET – second lowest apparent effect threshold
ML – maximum level
na – not available
OC – organic carbon
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
SL – screening level
SMS – Washington State Sediment Management Standards
SQS – sediment quality standard
SVOC – semivolatile organic compound

A.3.2.1.2 Site-specific sediment toxicity assessment

This section describes the results of site-specific toxicity tests conducted on EW sediment samples to assess the potential effects of sediment-associated chemicals on benthic invertebrates. As identified in Section A.2, toxicity data from three studies (both historical and current) were used in this evaluation (see Table A.2-13 for details on the accepted studies).

Three types of toxicity tests were conducted according to Puget Sound protocols and interpreted according to SMS (Ecology 1995, 2008). Tests were conducted with surface sediment (0 to 10 cm) collected at 51 locations in the EW (Map A.2-2). 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

The results from the three sediment toxicity tests were evaluated using the SMS rules for marine toxicity tests (Ecology 1996, 1995). The biological effects criteria for designating either SQS or CSL effects levels are summarized in Table A.3-7. Test responses less than or equal to the SQS effects level indicate that COPCs in sediment are not expected to adversely affect benthic organisms, test responses greater than SQS and less than or equal to the CSL indicate minor adverse effects, and test responses greater than the CSL indicate that adverse effects are expected to occur. An exceedance of the SQS biological effects criteria in any two toxicity tests at one location is considered to be a CSL exceedance at that location (WAC 173-204-420(3)).

Table A.3-7. SMS biological effects criteria for marine sediment toxicity tests

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

a Mean normal survivorship is a combined measure of mortality and abnormality (i.e., the number of normal larvae relative to the initial number of organisms).

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

CSL – cleanup screening level

SMS – Washington State Sediment Management Standards

SQS – sediment quality standards

For the amphipod mortality endpoint, 6 of the 51samples failed the SMS biological effects criteria at the CSL level, and an additional one sample failed the SQS level (Table A.3-8). For the polychaete growth endpoint, 1 of the 51 samples failed the biological effects criteria at the CSL level, and 6 other samples failed the SQS effects level. For the bivalve survival and development endpoint, 16 of the 51 samples failed the CSL biological effects criteria and 7 failed the SQS effects level (Table A.3-8). The locations of the SMS failures in the EW are shown on Map A.3-1.

Table A.3-8. Site-specific toxicity test results for EW surface sediment samples and SMS designations

Location ID	Sample ID	Amphipod			Polychaete		Larval		Overall SMS Designation
		Mortality (%)	Site Mortality Relative to Reference Mortality (% difference)	SMS Designation	Site Growth Relative to Reference Growth (% difference)	SMS Designation	Site NCMA Relative to Reference NCMA (% difference)	SMS Designation	
2001 East Waterway Nature and Extent Study									
2160	EW-100	2	-3	no exceedance	76.8	no exceedance	-40.8	no exceedance	no exceedance
2166	EW-101	3	-2	no exceedance	82.1	no exceedance	-36.1	no exceedance	no exceedance
2167	EW-102	5	0	no exceedance	71.2	no exceedance	-21.2	no exceedance	no exceedance
2168	EW-103	9	7	no exceedance	96	no exceedance	69.9	CSL	CSL
2169	EW-104	11	6	no exceedance	123	no exceedance	3.9	no exceedance	no exceedance
2170	EW-105	2	-3	no exceedance	72.3	no exceedance	-33.8	no exceedance	no exceedance
2128	EW-106	2	-3	no exceedance	71.7	no exceedance	-35.1	no exceedance	no exceedance
2129	EW-107	0	-5	no exceedance	88.6	no exceedance	-6.1	no exceedance	no exceedance
2130	EW-108	0	-5	no exceedance	91.3	no exceedance	-33.9	no exceedance	no exceedance
2131	EW-109	10	5	no exceedance	69.4	SQS	47.6	CSL	CSL
2132	EW-110	0	-2	no exceedance	113.6	no exceedance	67.8	CSL	CSL
2133	EW-111	3	1	no exceedance	75.4	no exceedance	-7.7	no exceedance	no exceedance
2134	EW-112	2	-3	no exceedance	121	no exceedance	-48.4	no exceedance	no exceedance
2135	EW-113	13	8	no exceedance	110	no exceedance	23	SQS	SQS
2136	EW-114	8	3	no exceedance	87.1	no exceedance	-26.4	no exceedance	no exceedance
2137	EW-115	2	-3	no exceedance	113	no exceedance	-39.7	no exceedance	no exceedance
2139	EW-116	100	98	CSL	-4.5	CSL	102.4	CSL	CSL
2140	EW-117	1	-1	no exceedance	123	no exceedance	56.8	CSL	CSL
2141	EW-118	5	0	no exceedance	87.5	no exceedance	-12.5	no exceedance	no exceedance
2142	EW-119	11	4	no exceedance	80.4	no exceedance	10.9	no exceedance	no exceedance
2143	EW-120	8	3	no exceedance	101	no exceedance	5.4	no exceedance	no exceedance
2147	EW-124	10	5	no exceedance	108	no exceedance	37.3	CSL	CSL

Table A.3-8. Site-specific toxicity test results for EW surface sediment samples and SMS designations (cont.)

Location ID	Sample ID	Amphipod			Polychaete		Larval		Overall SMS Designation
		Mortality (%)	Site Mortality Relative to Reference Mortality (% difference)	SMS Designation	Site Growth Relative to Reference Growth (% difference)	SMS Designation	Site NCMA Relative to Reference NCMA (% difference)	SMS Designation	
2148	EW-125	10	3	no exceedance	70.5	no exceedance	32.5	CSL	CSL
2150	EW-126	0	-2	no exceedance	111	no exceedance	21.1	SQS	SQS
2152	EW-128	14	9	no exceedance	76.8	no exceedance	2.7	no exceedance	no exceedance
2154	EW-130	12	5	no exceedance	84.8	no exceedance	27.9	SQS	SQS
2156	EW-132	6	4	no exceedance	77.5	no exceedance	89.4	CSL	CSL
2157	EW-133	7	5	no exceedance	122	no exceedance	82.8	CSL	CSL
2158	EW-134	14	9	no exceedance	87.3	no exceedance	31.3	CSL	CSL
2159	EW-135	22	17	no exceedance	77.7	no exceedance	43.9	CSL	CSL
2161	EW-136	13	6	no exceedance	94.6	no exceedance	38.2	CSL	CSL
2163	EW-138	17	10	no exceedance	61.2	SQS	28.5	SQS	CSL
2164	EW-141	22	15	no exceedance	98.9	no exceedance	23.9	SQS	SQS
2165	EW-142	4	-1	no exceedance	84.4	no exceedance	7.8	no exceedance	no exceedance
2000 T-18 PDM									
1	PDM-01	8	4	no exceedance	69	SQS	22.4	SQS	CSL
3	PDM-03	17	13	no exceedance	106	no exceedance	17.3	SQS	SQS
6	PDM-06	40	36	CSL	91.4	no exceedance	42.5	CSL	CSL
8	PDM-08	45	41	CSL	69.8	SQS	37.3	CSL	CSL
10	PDM-10	24	20	no exceedance	81.9	no exceedance	43.6	CSL	CSL
14	PDM-15	77	73	CSL	65.1	SQS	58.2	CSL	CSL
2009 East Waterway SRI/FS									
SS-005	EW09-SS-005-010	10	4	no exceedance	87.6	no exceedance	103.5	no exceedance	no exceedance
SS-030	EW09-SS-030-010	26	24	SQS	72.6	no exceedance	102.0	no exceedance	SQS
SS-032	EW09-SS-032-010	14	12	no exceedance	81.5	no exceedance	98.8	no exceedance	no exceedance
SS-033	EW09-SS-033-010	35	33	CSL	78.5	no exceedance	102.5	no exceedance	CSL
SS-034	EW09-SS-034-010	18	16	no exceedance	88.1	no exceedance	94.9	no exceedance	no exceedance
SS-035	EW09-SS-035-010	33	31	CSL	75.6	no exceedance	94.9	no exceedance	CSL

Table A.3-8. Site-specific toxicity test results for EW surface sediment samples and SMS designations (cont.)

Location ID	Sample ID	Amphipod			Polychaete		Larval		Overall SMS Designation
		Mortality (%)	Site Mortality Relative to Reference Mortality (% difference)	SMS Designation	Site Growth Relative to Reference Growth (% difference)	SMS Designation	Site NCMA Relative to Reference NCMA (% difference)	SMS Designation	
SS-217	EW09-SS-217-010	2	0	no exceedance	102.6	no exceedance	100.9	no exceedance	no exceedance
SS-218	EW09-SS-218-010	3	1	no exceedance	108.7	no exceedance	99.1	no exceedance	no exceedance
SS-220	EW09-SS-220-010	0	-2	no exceedance	92.2	no exceedance	93.0	no exceedance	no exceedance
SS-015	EW09-SS-015-010	6	0	no exceedance	59.5	SQS	94.9	no exceedance	SQS
SS-215	EW09-SS-215-010	13	7	no exceedance	76.7	no exceedance	102.4	no exceedance	no exceedance

[a] NCMA is the number of normal larvae in the site/reference test at the end of the exposure period relative to the number of normal larvae in the seawater control at the end of the exposure period.

CSL – cleanup screening level

CSO – combined sewer overflow

FS – feasibility study

ID – identification

NCMA – normalized combined mortality and abnormality

PDM – post-dredge monitoring

RI – remedial investigation

SMS – Washington State Sediment Management Standards

SQS – sediment quality standard

SRI – supplemental remedial investigation

T-18 – Terminal 18

Based on the combined evaluation of all three toxicity test results at each location, 24 of the 51 sampling locations are not expected to result in adverse effects on any of the test organisms (all tests less than SQS), 7 of the 51 sampling locations have the potential for minor adverse effects (greater than SQS in one test and less than CSL in all tests), and 20 of the 51 sampling locations are expected to result in adverse effects (greater than the CSL in at least one test or greater than SQS in at least two tests) (Table A.3-8).

A.3.2.2 Tissue-residue effects assessment

TRVs were selected for the two COPCs (TBT and PCBs) identified for the tissue-residue evaluation for benthic invertebrates. The following subsections summarize the toxicity studies reviewed and the NOAEL and LOAEL TRVs selected for these COPCs.

A.3.2.2.1 TBT

Eleven studies that evaluated the toxicity of TBT to benthic invertebrates and the associated concentrations in tissue were identified as acceptable for TRV derivation (Table A.3-9). In these studies, a variety of invertebrates, including amphipods, gastropods, polychaetes, clams, mussels, and oysters, were exposed to TBT in water or sediment. The sterilization of female gastropods as a result of imposex was observed by Gibbs et al. (1988) at a tissue TBT concentration of 0.12 mg/kg ww after chronic exposure to TBT in water. Reduced growth was observed at TBT concentrations in tissue that ranged from 0.54 to 2.9 mg/kg ww, and reduced survival was observed at concentrations that ranged from 3.4 to 21 mg/kg ww. The imposex observed in gastropod snails has not been observed in other species at those low tissue concentrations and may occur by a unique mechanism specific to gastropods (Meador and Rice 2001). Gastropod snails were observed during benthic invertebrate tissue sampling conducted in the EW (Windward 2009a). The gastropod LOAEL was selected as the LOAEL TRV to represent the benthic invertebrate species most sensitive to TBT. However, it should be noted that this TRV may not be applicable to the remainder of the benthic invertebrate community where adverse effects have not been observed at levels associated with imposex in gastropods.. Because no NOAEL was reported in Gibbs et al. (1988), the NOAEL TRV was calculated as 0.024 mg/kg ww using an uncertainty factor of 5 applied to the LOAEL TRV, based on EPA Region 10 guidelines for chronic data.

Table A.3-9. Benthic invertebrate toxicity studies reviewed for the selection of tissue-residue TRVs for TBT

Chemical	Test Species	NOAEL (mg/kg ww)	LOAEL (mg/kg ww)	Exposure Route and Duration	Effect	Source
Tributyltin chloride	dog whelk (*Nucella lapillus*)	na	**<u>0.12</u>**	water for 2 years	sterilization of females	Gibbs et al. (1988)
Tributyltin chloride	polychaete (*Armandia brevis*)	0.22	0.54	sediment for 42 days	reduced growth	Meador and Rice (2001)
Tributyltin chloride	blue mussel	0.79	1.1	water for 96 hours	reduced growth	Widdows and Page (1993)
Tributyltin chloride	polychaete (*Neanthes arenaceodentata)*	0.60	1.3	water for 10 weeks	reduced growth and reproductive success	Moore et al.(1991)
Bis(tributyltin) oxide	Pacific oyster	na	1.3	water and sediment for 56 days	reduced growth	Waldock and Thain (1983)
Tributyltin chloride	blue mussel	na	1.8	water for 36 hours	reduced growth	Page et al. (1998)
TBT (form not specified)	soft-shell clam	na	2.9	water for 28 days	reduced growth	Kure and Depledge (1994)
Tributyltin chloride	polychaete (*Neanthes arenaceodentata)*	1.3	3.4	water for 10 weeks	reduced survival	Moore et al.(1991)
Tributyltin chloride	amphipod (*Hyalella azteca*)	na	6.4	water for 4 weeks	reduced survival (LC50)	Borgmann et al. (1996)
Tributyltin chloride	amphipod (*Eohaustorius washingtonianus*)	na	9.0	water for 10 days	reduced survival (LC50)	Meador (1997)
Tributyltin chloride	polychaete (*Armandia brevis*)	na	9.4	water for 10 days	reduced survival (LC50)	Meador (1997)
Tributyltin chloride	amphipod (*Eohaustorius estuarius*)	na	12	water for 10 days	reduced survival (LC50)	Meador (1993)
Tributyltin chloride	zebra mussel	12.6	na	water for 35 days	no effect on growth or survival	van Slooten and Tarradellas (1994)
Tributyltin chloride	amphipod (*Rhepoxynius abronius)*	na	15	water for 10 days	reduced survival (LC50)	Meador (1997)
Tributyltin chloride	amphipod (*Rhepoxynius abronius)*	na	16	water for 4 days	reduced survival (LC50)	Meador (1993)
Tributyltin chloride	amphipod (*Rhepoxynius abronius)*	na	19	sediment for 10 days	reduced survival (LC50)	Meador et al. (1997)
Tributyltin chloride	polychaete (*Armandia brevis*)	na	21	sediment for 10 days	reduced survival (LC50)	Meador et al. (1997)
Tributyltin chloride	amphipod (*Hyalella azteca*)	na	23.2	water for 4 weeks	reduced survival (LC50)	Borgmann et al. (1996)

LC50 – concentration that is lethal to 50% of an exposed population
LOAEL – lowest-observed-adverse-effect level
na – not available
NOAEL – no-observed-adverse-effect level
TBT – tributyltin
TRV – toxicity reference value
ww – wet weight

<u>Bold and underline</u> identify the LOAEL selected as the TRV.

A.3.2.2.2 PCBs

Seven studies were identified as being acceptable for TRV derivation for the PCB tissue-residue evaluation for benthic invertebrates (Table A.3-10). These studies evaluated growth, survival, and reproductive endpoints in a variety of invertebrates (i.e., amphipods, polychaetes, shrimp, oysters, and crayfish) exposed to PCB Aroclors in water or sediment. Reduced survival was observed at total PCB concentrations in tissue that ranged from 1.1 to 42 mg/kg ww, and reduced growth was observed at total PCB concentrations that ranged from 32 to 119 mg/kg ww (Table A.3-10). Only two studies evaluated reproductive effects; LOAELs in these studies, which were conducted with amphipods, ranged from 76 to 552 mg/kg ww (Table A.3-10).

Table A.3-10. Benthic invertebrate toxicity studies reviewed for the selection of tissue-residue TRVs for PCBs

Chemical	Test Species	NOAEL (mg/kg ww)	LOAEL (mg/kg ww)	Exposure Route and Duration	Effect	Source
Aroclor 1016	grass shrimp	na	**<u>1.1</u>**	water for 96 hours	reduced survival	Hansen et al. (1974b)
Aroclor 1254	pink shrimp	1.3	3.9	water for 48 hours	reduced survival	Duke et al. (1970)
Aroclor 1254	pink shrimp	na	16	water for 20 days	reduced survival	Duke et al. (1970)
Aroclor 1254	grass shrimp	18	27	water for 16 days	reduced survival	Nimmo et al (1974)
Aroclor 1242	amphipod (*Hyalella azteca*)	28.9	na	water for 11 weeks	no effect on survival, growth or reproduction	Borgmann et al. (1990)
Aroclor 1016	American oyster	4.0	32	water for 96 hours	reduced growth	Hansen et al. (1974b)
Aroclor 1254	Eastern oyster	8.1	33	water for 96 hours	reduced growth	Duke et al. (1970)
Aroclor 1254	polychaete (*Armandia brevis*)	na	36	sediment for 28 days	reduced growth	Rice et al. (2000)
Aroclor 1016	brown shrimp	3.8	42	water for 96 hours	reduced survival	Hansen et al. (1974b)
Aroclor 1242	amphipod (*Gammarus pseudolimnaeus*)	na	76	water for 60 days	reduced number of young per adult	Nebeker and Puglisi (1974)
Aroclor 1254	Eastern oyster	101	119	water for 24 weeks	reduced growth	Lowe et al. (1972)
Aroclor 1248	amphipod (*Gammarus pseudolimnaeus*)	127	552	water for 60 days	reduced number of young per adult	Nebeker and Puglisi (1974)

LOAEL – lowest-observed-adverse-effect level
na – not available
NOAEL – no-observed-adverse-effect level
PCB – polychlorinated biphenyl
TRV – toxicity reference value
ww – wet weight

<u>Bold and underline</u> identify the LOAEL selected as the LOAEL TRV.

The lowest LOAEL of 1.1 mg/kg ww in tissue was from a study in which the survival of grass shrimp was reduced after exposed to Aroclor 1016 in water for 96 hours. This LOAEL was selected as the LOAEL TRV. There was no NOAEL below this lowest LOAEL, so the acute LOAEL TRV was divided by 10 to derive a NOAEL TRV of 0.11 mg/kg ww.

A.3.2.3 Surface water effects assessment

This section describes the effects data used to evaluate risk to benthic invertebrates from exposure to the three surface water COPCs (cadmium, mercury, and TBT). Surface water TRVs were based on Washington State or federal chronic WQC for marine waters. Marine criteria were used because salinity in the EW is consistently greater than 5 ppt in the water column above the sediment surface. In addition, the benthic community of the EW was composed of estuarine organisms (Section A.2.2.2). Methods for the derivation of the WQC in general are described in Section A.3.2.3.1. The WQC for cadmium, mercury, and TBT are discussed in more detail in the remaining subsections.

A.3.2.3.1 Derivation of WQC

This section discusses the methods used to derive the Washington State and federal WQC values. Because the Washington State WQC are based on federal WQC, this section describes the federal method for WQC derivation. A detailed description of the WQC methodology is presented in Stephan et al. (1985).

A chronic criterion is derived after an acute criterion has been derived. The general method for calculating an acute criterion involves compiling 48- to 96-hr LC50 or EC50 (concentration that causes a non-lethal effect in 50% of an exposed population) values for fish and aquatic invertebrates. A dataset with a minimum of eight specific families (including both fish and aquatic invertebrates) is required for criteria development so that a diverse spectrum of aquatic species can be represented. If this species diversity requirement is met, the species mean acute value (SMAV) is calculated as the geometric mean of LC50 and EC50 values for each species. Then, for each genus, the geometric mean of the genus mean acute value (GMAV) is calculated. The 5th percentile GMAV, which may be an extrapolated value below the lowest GMAV, is the final acute value (FAV). If the FAV is greater than the SMAV for a recreationally or economically important species, such as rainbow trout, the FAV is set equal to the SMAV for that species. Finally, in order to "not severely adversely affect too many of the organisms," the FAV is then divided by two to derive the criterion maximum concentration, or acute criterion (Stephan et al. 1985).

To develop a chronic criterion, chronic toxicity test data (longer-term survival, growth, or reproduction) based on no-observed-effect concentrations (NOECs) and/or lowest-observed-effect concentrations (LOECs) must be available for at least three taxa.[17] Most often the chronic criterion is set by applying a final acute-to-chronic ratio (ACR) to the FAV. The final ACR is derived from individual ACRs for species for which corresponding acute and chronic values are available. The final chronic value (FCV) is calculated as the FAV divided by the ACR. The criterion continuous concentration, or chronic criterion, is then typically based on the FCV, although it may be lowered for one of three reasons:

- If the chemical is a bioaccumulative substance and the calculated water concentration that is protective of people or wildlife that consume fish and shellfish is lower than the FCV
- If plants or algae are shown to be more sensitive than fish or aquatic invertebrates
- If a particular fish or aquatic invertebrate species or genus is shown to be more sensitive than indicated by the resulting FCV

A.3.2.3.2 Cadmium

The Washington State marine chronic WQC value for cadmium is 9.3 µg/L, which is based on the dissolved fraction. This value is based on the federal WQC published for cadmium in 1985 (EPA 1985a). Toxicity data for 33 invertebrate species and 5 fish species were used to develop the marine acute criterion for cadmium (EPA 1985a). The SMAVs for the 38 marine genera ranged from 41.29 µg/L for a mysid to 135,000 µg/L for an oligochaete. The FAV (85.09 µg/L) was divided by two to obtain the acute criterion of 42.55 µg/L for total cadmium, or 42 µg/L for dissolved cadmium using the total-to-dissolved conversion factor of 0.994 for cadmium and rounding down to the nearest whole number. Chronic data were available from three toxicity tests conducted with the marine invertebrate *Americamysis bahia* (classified as *Mysidopsis bahia* when the federal WQC for cadmium were published in 1985). The ACR was calculated as 9.105, resulting in a FCV and chronic criterion of 9.3 µg/L (85.09 µg/L divided by 9.105). This chronic criterion of 9.3 µg/L for dissolved cadmium was selected as the TRV for the benthic invertebrate community surface water evaluation.

A.3.2.3.3 Mercury

Washington State WQC chronic value for mercury is 0.025 µg/L, which is based on the total fraction. The marine acute criterion for mercury was developed using data on the acute toxicity of mercuric chloride to 29 genera of marine organisms, including annelids, mollusks, crustaceans, echinoderms, and fishes (EPA 1985b). Acute values ranged from 3.5 µg/L for a mysid species to 1,678 µg/L for winter flounder. The FAV of 4.125 µg/L was divided by two to obtain an acute criterion of 2.1 µg/L for total

[17] NOECs and LOECs are equivalent to NOAELs and LOAELs but are used here because they are the more commonly used terms for water toxicity data. The terms NOEC and LOEC are therefore used in this ERA for TRVs based on water toxicity.

mercury, or 1.8 µg/L for dissolved mercury using the total-to-dissolved conversion factor of 0.85 for mercury. The final ACR of 3.731 was selected based on ACRs for *Daphnia* and a mysid species. There is some uncertainty in this ACR because data for fathead minnow indicate that fish may have substantially higher ACRs. The FCV calculated using the ACR of 3.731 was 1.106 µg/L for total mercury, or 0.94 µg/L for dissolved mercury using the total-to-dissolved conversion factor of 0.85 for mercury. However, this value was not selected as the chronic criterion in the 1985 document; instead the chronic criterion of 0.025 µg/L was selected based on the final residue value in marine water calculated using a bioconcentration factor and the action level for the protection of human health (1.0 mg/kg in seafood). This value of 0.025 µg/L has been selected by Washington State as the chronic WQC; it represents the concentration of total mercury because it accounts for bioconcentration of any form of mercury from the water column. However, this value was not used in the ERA because it is not based on the toxicity of mercury to fish and aquatic invertebrates but rather a threshold value for the consumption of fish by people. Instead, the federal marine chronic WQC value for the protection of aquatic life (0.94 µg/L for dissolved mercury) was used as the TRV for the benthic invertebrate community surface water evaluation. This value is based on the federal WQC published for mercury in 1985 (EPA 1985b).

A.3.2.3.4 TBT

There is no Washington State marine chronic WQC for TBT, so the federal marine chronic WQC was used in the risk evaluation. The federal marine chronic WQC for TBT is 0.0074 µg/L. This value is based on the federal WQC published for TBT in 2003 (EPA 2003a). The marine acute criterion for TBT was developed using data on the acute toxicity of TBT to 26 species of invertebrates and 7 species of fish. Acute values ranged from 0.24 µg/L for juvenile copepods to 282.2 µg/L for Pacific oysters. The FAV of 0.835 µg/L was divided by two to obtain a marine acute WQC of 0.42 µg/L. The final ACR of 12.69 was calculated based on ACRs for *Daphnia magna*, fathead minnow, a mysid species, and a copepod species. The FCV calculated using the ACR of 12.69 was 0.066 µg/L. However, this value of 0.066 µg/L was not selected as the federal chronic criterion because additional data indicated that there were effects on invertebrates at substantially lower concentrations (i.e., imposex in snails, particularly in the Atlantic dogwinkle; growth effects in commercially important bivalve molluscs, and survival of ecologically important copepods). Therefore, the lowest concentration associated with adverse effects on snails, molluscs, or copepods (0.0074 µg/L), was selected as the marine chronic WQC to protect these organisms. This value was the NOEC in a study in which egg capsule production was decreased in Atlantic dogwinkle. This chronic criterion of 0.0074 µg/L was selected as the TRV for the benthic invertebrate community surface water evaluation.

A.3.2.4 Porewater effects assessment

Naphthalene was the only COPC for porewater based on the COPC screen presented in Section A.2.5.1.4. Federal acute and chronic marine WQC could not be calculated for naphthalene because of the lack of data for a variety of aquatic species, although numerous studies on the effects of naphthalene on aquatic organisms were found by searching EPA's ecotoxicology database (ECOTOX 2010). ECOTOX contains effects data from individual studies published in the scientific literature. A TRV was found by searching the ECOTOX database for the effects of naphthalene on growth, reproduction, and survival endpoints in fish and aquatic invertebrates (see Attachment 4 for results from the ECOTOX search).

Effects from exposure to naphthalene in water were observed in 63 studies conducted with 28 aquatic invertebrate species and 14 fish species. The observed effect concentrations ranged from 8 to 220,000 µg/L. Only three studies were conducted with infaunal invertebrates (marine bivalves and polychaete worms); reported effect concentrations ranged from 3,500 to 74,000 µg/L after acute exposures of these species. The lowest effect concentration found in the ECOTOX search was from a study in which larvae of Dungeness crab and spot shrimp were exposed to naphthalene at concentrations that ranged from 8 to 12 µg/L in a flow-through system (Sanborn and Malins 1977). At 18 to 24 hours, narcosis was observed; and within 24 to 36 hours, all organisms died at all exposure concentrations tested. The study with the next lowest effect concentration exposed rainbow trout embryo-larvae to naphthalene under flow-through conditions from the time of fertilization through 4 days post-hatch (Black et al. 1983). An LC50 of 110 µg/L was calculated from this study. The LOEC of 8 µg/L from the Sanborn and Malins (1977) study was selected as the LOEC TRV. Because this was an acute effect concentration that resulted in mortality to all organisms tested, a NOEC TRV of 0.16 µg/L was estimated from the LOEC using an uncertainty factor of 50 according to EPA Region 10 guidance (1997b), as described in Section A.2.5.1.2.

A.3.3 Crab Exposure Assessment

The exposure assessment describes the EPCs used in the tissue-residue and surface water evaluations used to characterize risk to crab. EPCs were developed for each COPC identified as a result of the COPC screen, as summarized in Table A.2-27 in Section A.2.5.1.6.

A.3.3.1 Tissue-residue exposure assessment

Five chemicals were identified as COPCs for the crab tissue-residue evaluation based on the COPC screen presented in Section A.2.5.1.3: arsenic, cadmium, copper, zinc, and total PCBs. EPCs for crab were calculated using the nine composite crab samples collected throughout the EW during the SRI crab sampling event. Eight of the nine samples consisted of red rock crab, and one sample consisted of Dungeness crab. Crab composite samples were analyzed as edible meat and hepatopancreas tissues. Whole-body crab concentrations in each of the composite samples were calculated using the relative weights of and COPC concentrations in edible meat and hepatopancreas.[18] The EPCs were calculated as the 95% UCL of both the Dungeness crab and red rock crab data for each COPC using ProUCL software (Table A.3-11). ProUCL allows detected and non-detected values to be identified and creates interpolated values for non-detects based on the perceived distribution of the detected concentrations. Once any necessary interpolation has been performed, the software conducts an analysis of the data to determine the most appropriate UCL and makes a recommendation.

Table A.3-11. EPCs for crab tissue-residue evaluation

COPC	Detection Frequency	Concentration (mg/kg ww)			Statistic Used
		Mean Value	Maximum Detection	EPC	
Arsenic	9/9	4.64	6.81 J	5.19	95% Approximate Gamma UCL
Cadmium	9/9	2	3.1	3.61	95% Chebyshev (mean, Sd) UCL
Copper	9/9	27.5	31.3	29.1	95% Student's-t UCL
Zinc	9/9	48.5	59.0	53.4	95% Student's-t UCL
Total PCBs	9/9	0.30	0.86	0.45	95% Modified-t UCL

COPC – chemical of potential concern
EPC – exposure point concentration
J – estimated concentration
PCB – polychlorinated biphenyl
Sd – standard deviation
UCL – upper confidence limit on the mean
ww – wet weight

A.3.3.2 Surface water exposure assessment

Three chemicals were identified as surface water COPCs for crab based on the COPC screen presented in Section A.2.5.1.6: cadmium, mercury, and TBT. Dungeness and red rock crab are relatively mobile[19] so exposure was evaluated using EPCs calculated on a site-wide basis. In addition, as a more conservative approach, exposure was evaluated based on EPCs for individual water samples to represent conditions at each location at the time of sampling. Cadmium and mercury EPCs were based on the dissolved fraction because the TRVs were based on dissolved fraction; TBT EPCs were based on total concentrations.

For the site-wide evaluation, EPCs were calculated using all water samples collected from 1 m above the sediment surface, to represent conditions at the bottom of the water column to which crabs would be exposed. ProUCL was used to calculate 95% UCLs for the site-wide EPCs if there were at least six detected concentrations. The maximum

[18] 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 mean relative weights of these tissues in all crab samples for which data were available from the SRI 2008 sampling event.

[19] Dungeness crab in Oregon have been known to travel up to 91 km (Hildenbrand et al. 2011), and red rock crab were reported to have moved 3.1 km in 6 to 10 days (Carroll and Winn 1989).

concentration was used if there were fewer than six detected concentrations, as recommended by ProUCL. The ProUCL software uses both detected and undetected values and creates interpolated values for non-detects based on the perceived distribution of the detected concentrations. Once any necessary interpolation has been performed, the software analyzes the resulting data distribution to determine the most appropriate 95% UCL and makes a recommendation. Site-wide EPCs for surface water collected from 1 m above the sediment surface are presented in Table A.3-12. EPCs for individual water samples were equivalent to the detected COPC concentrations in the individual samples. The range of COPC concentrations (i.e., EPCs) detected in individual water samples collected from 1 m above the sediment surface is also presented in Table A.3-12.

Table A.3-12. Site-wide surface water EPCs for crab for samples collected 1 m from the bottom of the water column

COPC	Detection Frequency	Mean Value (µg/L)[a]	Range of Detects (µg/L)	Maximum RL (µg/L)	EPC (µg/L)	Statistic Used
Cadmium (dissolved)	65/67	0.63	0.055 – 37.8	0.088	3.1	95% KM (Chebyshev) UCL
Mercury (dissolved)	6/39	0.0003	0.000170 – 0.00110	0.00054	0.00040	95% KM (t) UCL
TBT	1/31	na	0.010 J	0.01	0.01[b]	maximum detect

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

b No UCL was calculated for TBT because there was only one detected concentration; thus, the EPC is equal to the detected concentration.

COPC – chemical of potential concern
EPC – exposure point concentration
na – not applicable; only one detected value
RL – reporting limit
TBT – tributyltin
UCL – upper confidence limit on the mean

A.3.4 Crab Effects Assessment

This section summarizes the toxicity literature for the COPCs identified for crab and presents the TRVs selected for crab. The literature search and guidelines for TRV selection for crab are described in detail in Section A.2.5.1.5. Toxicological data presented in this section and exposure data presented in Section A.4.1 are evaluated together in Section A.6.2 to characterize risks to crab.

A.3.4.1 Tissue-residue effects assessment

TRVs were selected for the five COPCs identified for the tissue-residue evaluation for crabs (arsenic, cadmium, copper, zinc, and total PCBs). The following subsections summarize the toxicity studies reviewed and the NOAEL and LOAEL TRVs selected for these COPCs.

A.3.4.1.1 Arsenic

Two studies that evaluated the toxicity associated with arsenic concentrations in decapod tissue were considered acceptable for TRV derivation for the tissue-residue evaluation for crabs (Table A.3-13). One of these studies reported an effect on the survival of brown shrimp at an arsenic concentration of 21 mg/kg ww in tissue, with no NOAEL reported (Madsen 1992). The other study reported a NOAEL for the survival of grass shrimp at an arsenic concentration of 1.28 mg/kg ww in tissue (Lindsay and Sanders 1990). These values of 1.28 and 21 mg/kg ww were selected as the NOAEL and LOAEL TRVs, respectively, for arsenic in crab tissue.

Table A.3-13. Crab or decapod toxicity studies reviewed for the selection of arsenic crab tissue-residue TRVs

Chemical	Test Species	NOAEL (mg/kg ww)	LOAEL (mg/kg ww)	Exposure Route and Duration	Effect	Source
Sodium arsenate	grass shrimp	**<u>1.28</u>**	na	water and diet for 28 days	no effect on survival	Lindsay and Sanders (1990)
Sodium arsenate	brown shrimp	na	**<u>21</u>**	water for 180 hours	reduced survival	Madsen (1992)

LOAEL – lowest-observed-adverse-effect level
na – not available
NOAEL – no-observed-adverse-effect level
TRV – toxicity reference value
ww – wet weight
<u>Bold and underline</u> identify the NOAEL and LOAEL selected as TRVs.

A.3.4.1.2 Cadmium

Nine studies that evaluated the toxicity associated with cadmium concentrations in whole-body tissue of decapods were considered acceptable for TRV derivation for the tissue-residue evaluation for crabs (Table A.3-14). Two additional studies evaluated the toxicity associated with cadmium concentrations in muscle or hepatopancreas tissue (Canli and Furness 1995; Dickson et al. 1982). The NOAEL and LOAEL were selected from the studies that reported whole-body concentrations. In the uncertainty section, cadmium concentrations in edible meat are compared with the only LOAEL (9.5 mg/kg ww) associated with cadmium in muscle tissue from a study with shore crab by Jennings and Rainbow (1979).

Table A.3-14. Crab or decapods toxicity studies reviewed for the selection of cadmium crab tissue-residue TRVs

Chemical by Tissue Type	Test Species	NOAEL (mg/kg ww)	LOAEL (mg/kg ww)	Exposure Route and Duration	Effect	Source
Whole Body						
Cadmium chloride	grass shrimp	0.6	na	sediment for 14 days	no effect on survival	Rule and Alden (1982)
Cadmium (form not specified)	grass shrimp	2.0	2.6	water for 21 days	reduced survival	Vernberg et al. (1977)
Cadmium chloride	virile crayfish	na	**<u>5.7</u>**[a]	water for 2 weeks	reduced survival	Mirenda (1986b)
Cadmium chloride	fiddler crab	na	6.4[a]	water for 24 days	reduced survival	Weis (1978)
Cadmium (form not specified)	soldier crab	na	7.4	sediment for 21 days	reduced survival	Weimin et al.(1994)
Cadmium chloride	grass shrimp	na	9.0	water for 6 weeks	reduced survival	Pesch and Stewart (1980)
Cadmium chloride	grass shrimp	14.9	22	water for 5 months	reduced survival	Thorp et al. (1979)
Cadmium chloride	shore crab	8.4	23	water for 40 days	reduced survival	Jennings and Rainbow (1979)
Cadmium chloride	northern clearwater crayfish	534	na	water for 8 days	no effect on survival	Gillespie et al.(1977)
Muscle						
Cadmium chloride	Norway lobster	0.13	na	diet for 50 days	no effect on survival	Canli and Furness (1995)
Cadmium chloride	Norway lobster	0.58	na	water for 30 days	no effect on survival	Canli and Furness (1995)
Cadmium chloride	White River crayfish	1.4	na	water for 21 days	no effect on survival	Dickson et al.(1982)
Cadmium chloride	shore crab	4.9	9.5	water for 40 days	reduced survival	Jennings and Rainbow (1979)
Hepatopancreas						
Cadmium chloride	Norway lobster	5.7	na	diet for 50 days	no effect on survival	Canli and Furness (1995)
Cadmium chloride	Norway lobster	46	na	water for 30 days	no effect on survival	Canli and Furness (1995)

[a] These studies did not indicate whether cadmium concentrations in tissue were based on wet weight or dry weight, so dry weight was assumed. A moisture content of 80% was used to convert concentrations from dry weight to wet weight.

LOAEL – lowest-observed-adverse-effect level

na – not available

NOAEL – no-observed-adverse-effect level

TRV – toxicity reference value

ww – wet weight

<u>Bold and underline</u> identify the NOAEL and LOAEL selected as TRVs.

The nine studies considered in the derivation of the cadmium tissue-residue TRV evaluated whole-body tissue concentrations associated with the survival of crabs, shrimp, and crayfish. In the study with the lowest LOAEL for whole-body tissue concentrations (Vernberg et al. 1977), grass shrimp were exposed to cadmium at concentrations of 0 and 50 µg/L in water for 21 days at four salinity levels (5, 10, 20, and 30 parts per thousand). Mortality was observed and tissue concentrations were measured at days 3, 7, 14, and 21. Mortality was < 10% at all salinity levels for both the cadmium-exposed shrimp and control shrimp with one exception; mortality was > 20% beginning at Day 7 at the 5 ppt salinity level for the cadmium-exposed shrimp. Statistical tests were not conducted, so it is not known whether mortality in exposed shrimp was significantly different than that in the control. In addition, there was a clear relationship between salinity and tissue cadmium concentrations, with an increase in cadmium tissue concentrations as salinity decreased. Because a salinity of 5 ppt (the level at which potential but uncertain effects were observed) is lower than that generally measured in the EW and effects are expected to be lower at higher salinities, this study was not used to derive a cadmium tissue-residue TRV.

The next lowest LOAEL was from a study that exposed crayfish to cadmium in water for 14 days, with 25% mortality observed at a cadmium whole-body tissue concentration of 28.3 mg/kg, although the study did not indicate if the concentration was wet or dry weight (Mirenda 1986b). Based on the assumption that the concentration was dry weight and the moisture content of the tissue was 80%, the LOAEL for a survival endpoint was calculated as 5.7 mg/kg ww. The next higher LOAEL from Weis (1978) had the same uncertainty as did Mirenda (1986b) regarding wet vs. dry weight. The study by Weis (1978) exposed fiddler crab to cadmium in water for 24 days, and 20% mortality was observed at a whole-body tissue cadmium concentration of 31.9 mg/kg, equivalent to 6.4 mg/kg ww, assuming the reported concentration was in dry weight and the moisture content was 80%. Because these two studies had the same uncertainties, the lower LOAEL from Mirenda (1986b) (5.7 mg/kg ww) was selected as the LOAEL TRV. The LOAEL was unbounded so a NOAEL was calculated as 0.57 mg/kg ww using an acute uncertainty factor of 10.

A.3.4.1.3 Copper

Four studies that evaluated the toxicity associated with copper concentrations in decapod tissue were identified as acceptable for TRV derivation for the tissue-residue evaluation for crab (Table A.3-15). One study was based on the growth endpoint, and the other three studies were based on the survival endpoint. Ahsanullah and Ying (1995) observed a reduction in the growth of banana prawn exposed to copper in water for 2 weeks, with an associated copper concentration in tissue of 26 mg/kg ww; this was the lowest LOAEL of the four studies. In the remaining studies, no effects were observed in grass shrimp or rusty crayfish at concentrations of 40 and 50 mg/kg ww, respectively; and survival was reduced in Australian ghost shrimp at a tissue concentration of 145.9 mg/kg ww (Table A.3-15). The lowest LOAEL of 26 mg/kg ww was selected as the LOAEL TRV. There was no NOAEL at a concentration lower than the selected LOAEL TRV, so an uncertainty factor of 10 was used with the LOAEL to derive a NOAEL TRV of 2.6 mg/kg ww.

Table A.3-15. Crab or decapod toxicity studies reviewed for the selection of copper crab tissue-residue TRVs

Chemical	Test Species	NOAEL (mg/kg ww)	LOAEL (mg/kg ww)	Exposure Route and Duration	Effect	Source
Copper (form not specified)	banana prawn	na	**<u>26</u>**	water for 2 weeks	reduced growth	Ahsanullah and Ying (1995)
Copper sulfate	grass shrimp	40	na	sediment for 2 weeks	no effect on survival	Rule and Alden (1996)
Copper (form not specified)	rusty crayfish	50	na	water for 48 hours	no effect on survival	Evans (1980)
Copper (form not specified)	Australian ghost shrimp	na	145.9	water for 2 weeks	reduced survival	Ahsanulla et al. (1981)

LOAEL – lowest-observed-adverse-effect level
na – not available
NOAEL – no-observed-adverse-effect level
TRV – toxicity reference value
ww – wet weight
<u>Bold and underline</u> identify the LOAEL selected as the LOAEL TRV.

A.3.4.1.4 Zinc

Only one toxicity study that evaluated adverse effects associated with zinc in decapods tissue was identified. In this study (Mirenda 1986a), crayfish (*Orconectes virilis*) were exposed to zinc in water for 2 weeks. The zinc tissue-residue concentration of 35.2 mg/kg ww was selected as the LOAEL TRV because the mortality rate was significantly higher in organisms with this tissue concentration than in control organisms. The zinc tissue concentration of 12.7 mg/kg ww was selected as the NOAEL TRV because the mortality rate observed in organisms with this tissue concentration was not significantly higher than that in control organisms.

A.3.4.1.5 PCBs

Five studies that evaluated the effects of PCBs on decapod species were identified (Table A.3-16). Three shrimp species (brown, grass, and pink) were exposed to Aroclor 1254, and two shrimp species (grass and brown) were exposed to Aroclor 1016 in water for 2 to 20 days. Effects on survival were assessed in all studies. Whole-body tissue concentrations ranged from 1.3 to 18 mg/kg ww for exposures associated with no effects on survival and from 1.1 to 42 mg/kg ww for exposures associated with effects on survival. The effects of PCBs (Aroclor 1016 and 1254) on the survival of three other decapod species (crayfish, blue crab, and horseshoe crab) were also evaluated. No effects were observed in these species at whole-body tissue concentrations that ranged from 1.2 to 31.9 mg/kg ww. The lowest LOAEL tissue concentration of 1.1 mg/kg ww was selected as the PCB LOAEL TRV for the tissue-residue evaluation for crab. Because there was no NOAEL below this LOAEL, the LOAEL TRV was divided by 10 (Canli and Furness 1995) to derive a NOAEL TRV of 0.11 mg/kg ww.

Table A.3-16. Crab or decapods toxicity studies reviewed for the selection of PCB crab tissue-residue TRVs

Chemical	Test Species	NOAEL (mg/kg ww)	LOAEL (mg/kg ww)	Exposure Route and Duration	Effect	Source
Aroclor 1016	grass shrimp	na	**<u>1.1</u>**	water for 96 hours	reduced survival	Hansen et al. (1974b)
Aroclor 1254	crayfish	1.2	na	water for 21 days	no effect on survival	Sanders and Chandler (1972)
Aroclor 1254	pink shrimp	1.3	3.9	water for 48 hours	reduced survival	Duke et al. (1970)
Aroclor 1254	pink shrimp	na	16	water for 20 days	reduced survival	Duke et al. (1970)
Aroclor 1254	grass shrimp	18	27	water for 16 days	reduced survival	Nimmo et al (1974)
Aroclor 1016	horseshoe crab	na	31.9	water for 96 days	reduced survival	Neff and Giam (1977)
Aroclor 1016	brown shrimp	3.8	42	water for 96 hours	reduced survival	Hansen et al. (1974b)
Aroclor 1254	blue crab	23	na	water for 20 days	no effect on survival	Duke et al. (1970)

LOAEL – lowest-observed-adverse-effect level
na – not available
NOAEL – no-observed-adverse-effect level
PCB – polychlorinated biphenyl
TRV – toxicity reference value
ww – wet weight
<u>Bold and underline</u> identify the LOAEL selected as the LOAELTRV.

A.3.4.2 Surface water effects assessment

TRVs were selected for the three COPCs identified for the surface water evaluation for crab (cadmium, mercury, and TBT). The following subsections summarize the toxicity studies reviewed and the chronic TRVs selected for these COPCs.

A.3.4.2.1 Cadmium

For cadmium, the Washington State marine chronic WQC value is 9.3 µg/L, which is based on the dissolved fraction of cadmium (EPA 2001). The derivation of this value is described in detail in Section A.3.2.3.2. This criterion is based on data for 33 invertebrate species and 5 fish species and is designed to protect 95% of the species in the aquatic community. The marine chronic WQC of 9.3 µg/L was selected as the TRV for crab for the surface water evaluation.

A.3.4.2.1 Mercury

The federal marine chronic WQC for mercury for the protection of aquatic life is 0.94 µg/L, which is based on the dissolved fraction of mercury in water (EPA 1985b). The derivation of this value is described in detail in Section A.3.2.3.2. The FCV was based on acute data for 25 invertebrate species and 7 fish species and acute-to-chronic ratios for two invertebrate species and two fish species. This marine chronic criterion of 0.94 µg/L was selected as the TRV for crab for the surface water evaluation.

A.3.4.2.1 TBT

The federal marine chronic WQC for TBT is 0.0074 µg/L (EPA 2003a), as described in Section A.3.2.3.2. The FCV is 0.066 µg/L based on acute data for 26 invertebrate species and 7 fish species and chronic data for three invertebrate species and one fish species. However, the FCV was not selected by EPA as the marine chronic WQC because additional data indicated that there were effects on dogwinkle, commercially important bivalves, and ecologically important copepods at lower concentrations. Thus, a value of 0.0074 µg/L was selected by EPA as the marine chronic WQC based on a no observed effect level for dogwinkle. However, the marine chronic federal WQC for TBT is overly conservative when applied to crab, which did not show sensitivity below 0.76 µg/L.[20] Therefore, the FCV of 0.066 µg/L, which includes data from crab and decapods in its calculation, will be protective of crab. Because the marine chronic WQC was set to a lower value to be protective of particularly sensitive species and because crab do not share these particular sensitivities, the FCV of 0.066 µg/L from the federal WQC for TBT was selected instead of the marine chronic criterion as the TRV for crab for the surface water evaluation.

A.3.5 SUMMARY OF EXPOSURE AND EFFECTS ASSESSMENTS

This section presents a summary of the exposure and effects assessments for the benthic invertebrate community (Section A.3.5.1) and for crab (Section A.3.5.2).

A.3.5.1 Benthic invertebrate community

The exposure of the benthic invertebrate community to COPCs was evaluated using surface sediment, benthic invertebrate tissue, surface water, and porewater data, as follows:

- **Surface sediment –** Exposure to 44 surface sediment COPCs was assessed based on the frequency and magnitude of detected concentrations at each surface sediment sampling location in the EW (Table A.3-1).
- **Benthic invertebrate tissue –** Exposure to two benthic invertebrate tissue COPCs (TBT and PCBs) was evaluated using benthic invertebrate tissue data collected as composite samples from each tissue sampling area (Table A.3-2).
- **Surface water –** Exposure to three surface water COPCs (cadmium, mercury, and TBT) was evaluated using the frequency and magnitude of detected concentrations in individual surface water samples collected from 1 m above the sediment surface (Tables A.3-3 and A.3-4).

[20] Chronic sensitivity data were based on crab species mean acute values divided by saltwater ACR for TBT.

- **Porewater –** Exposure to a single porewater COPC (naphthalene) was evaluated based on the frequency and magnitude of detected concentrations in each porewater sample (Table A.3-4).

The type of biological endpoints used to establish the SMS criteria and DMMP guidelines for the COPCs were discussed in the effects assessment. In addition, the approach for measuring site-specific toxicity was discussed, and results of site-specific toxicity testing conducted in the EW were presented. Of the 48 locations tested for toxicity, 24 locations did not exhibit the potential for adverse effects on any of the test organisms, 6 locations exhibited the potential for minor adverse effects, and 18 locations exhibited the potential for severe effects (Table A.3-7; Map A.3-1).

For the benthic invertebrate tissue effects assessment, NOAEL and LOAEL TRVs were selected, as summarized in Table A.3-17. For the surface water effects assessment, acute and chronic marine WQC were presented for cadmium, mercury, and TBT, as summarized in Table A.3-18. For the porewater effects assessment, NOEC and LOEC TRVs were derived for naphthalene, as shown in Table A.3-19.

Table A.3-17. Critical tissue-residue TRVs selected for benthic invertebrates

COPC	TRV (mg/kg ww)	
	NOAEL	LOAEL
TBT	0.024	0.12
Total PCBs	0.11	1.1

COPC – chemical of potential concern
LOAEL – lowest-observed-adverse-effect level
NOAEL – no-observed-adverse-effect level
PCB – polychlorinated biphenyl
TBT – tributyltin
TRV – toxicity reference value
ww – wet weight

Table A.3-18. Surface water TRVs selected for benthic invertebrates

COPC	Chronic TRV (µg/L)
Cadmium	9.3
Mercury	0.94
TBT	0.0074

COPC – chemical of potential concern
TBT – tributyltin
TRV – toxicity reference value

Table A.3-19. Porewater TRVs selected for benthic invertebrates

COPC	TRV (µg/L)	
	NOEC	LOEC
Naphthalene	0.16	8

COPC – chemical of potential concern
LOEC – lowest-observed-effect concentration
NOEC – no-observed-effect concentration
TRV – toxicity reference value

A.3.5.2 Crab

The crab exposure assessment estimated the exposure of crab via two approaches: tissue-residue and surface water. Based on the COPC screen presented in Section A.2.5.1, five chemicals were identified as COPCs for crabs for the tissue-residue evaluation (arsenic, cadmium, copper, zinc, and total PCBs), and three chemicals were identified as COPCs for the surface water evaluation (cadmium, mercury, and TBT).

The exposure of crab to these COPCs was based on concentrations of these chemicals in crab tissue and in surface water. Tissue-residue EPCs were calculated as 95% UCLs using the data from all nine composite crab samples collected on a site-wide basis. Crab tissue-residue EPCs are presented in Table A.3-11. Surface water EPCs were calculated as 95% UCLs using all surface water data collected from the bottom of the water column for a site-wide evaluation (Table A.3-12). EPCs equivalent to concentrations in individual water samples collected from the bottom of the water column were also used as exposure concentrations for crab.

The effects assessment presented critical tissue-residue TRVs for crabs and other decapods, as well as chronic surface water TRVs. A summary of selected NOAEL and LOAEL TRVs for tissue-residue and chronic TRVs for surface water are presented in Tables A.3-20 and A.3-21, respectively.

Table A.3-20. Critical tissue-residue TRVs selected for crab

COPC	TRV (mg/kg ww)	
	NOAEL	LOAEL
Arsenic	1.28	21
Cadmium	0.57	5.7
Copper	2.6	26
Zinc	12.7	35.2
Total PCBs	0.11	1.1

COPC – chemical of potential concern
LOAEL – lowest-observed-adverse-effect level
NOAEL – no-observed-adverse-effect level
TRV – toxicity reference value
ww – wet weight

Table A.3-21. Surface water TRVs selected for crab

COPC	Chronic TRV (µg/L)
Cadmium	9.3
Mercury	0.94
TBT	0.066

COPC – chemical of potential concern
TBT – tributyltin
TRV – toxicity reference value

A.4 Exposure and Effects Assessment: Fish

This section presents the exposure and effects assessment for fish in the EW based on three approaches: critical tissue residue, dietary, and surface water. The exposure assessment is presented as Section A.4.1, and the effects assessment is presented as Section A.4.2. The data used in the application of the three approaches, as well as in the risk characterization and uncertainty analysis for fish (Section A.6.2), are also presented in this section.

A.4.1 Exposure Assessment

This section describes the methods used to quantify exposures of the fish ROCs (i.e., English sole, brown rockfish, and juvenile Chinook salmon) and presents the EPCs that were used in both the tissue-residue and the dietary evaluations for each of the fish ROCs. In addition, this section also presents the surface water EPCs used for the surface water evaluation. COPCs for fish selected as a result of the COPC screen are presented in Table A.4-1.

Table A.4-1. COPCs selected for fish

COPC	COPCs Identified for Fish by Evaluation Type		
	Tissue Residue	Diet	Surface Water
Arsenic		X[a]	
Cadmium		X[a]	X[a]
Chromium		X[a]	
Copper		X[a]	
Mercury	X[b]		X[a]
Vanadium		X[a]	
TBT	X[c]		X[a]
Benzo(a)pyrene		X[c]	
Total PCBs	X[c]		
beta-Endosulfan	X[b]		

a Identified as a COPC for all three fish ROCs: juvenile Chinook salmon, English sole, and brown rockfish.
b Identified as a COPC only for brown rockfish.
c Identified as a COPC for brown rockfish and English sole but not juvenile Chinook salmon.
COPC – chemical of potential concern
PCB – polychlorinated biphenyl
TBT – tributyltin

A.4.1.1 Tissue-residue exposure assessment

The following chemicals were identified as COPCs for two of the three fish ROCs for the EW following the COPC screen for the tissue-residue evaluation (Section A.2.5.2.2):

- **English sole –** TBT and total PCBs
- **Brown rockfish –** mercury, TBT, total PCBs, and beta-endosulfan

No chemicals were identified as COPCs for the tissue-residue evaluation for juvenile Chinook salmon (i.e., the third fish ROC) as a result of the COPC screen.

In order to characterize risks based on the tissue-residue evaluation, whole-body tissue EPCs (in mg/kg ww) were compared with TRVs to estimate the potential for adverse effects associated with chemicals in tissue; the whole-body tissue EPCs used for the exposure component are presented in Table A.4-2. The whole-body tissue concentration integrates the exposure of a fish from all pathways (e.g., direct sediment and water contact and diet) within its foraging range. However, the foraging ranges of the fish ROCs are not precisely known. For the purpose of the ERA, the English sole foraging range was assumed to be the entire EW, and therefore, the English sole were collected from throughout the site and analyzed as site-wide composites of whole fish to reflect site-wide integration of exposures. Because the brown rockfish foraging range is smaller than the size of the EW, brown rockfish tissue data were collected from discrete locations throughout the site and analyzed as individual whole fish to reflect integrated exposure at those locations. Therefore, brown rockfish data were evaluated in two ways: on a site-wide basis (combining all individual brown rockfish data) and on a location-specific basis (using detected concentrations in each individual brown rockfish). For the site-wide evaluations of each species, EPCs for tissue were calculated as the 95% UCLs of the site-wide data for English sole and brown rockfish using ProUCL (Version 4.0). The ProUCL software uses both detected and undetected values and creates interpolated values for non-detects based on the perceived distribution of the detected concentrations. Once any necessary interpolation is performed, the software analyses the resulting data distribution to determine the most appropriate 95% UCL and makes a recommendation. For the evaluation of brown rockfish on a smaller spatial scale (i.e., discrete sample locations), EPCs were equivalent to concentrations of COPCs in the tissue of each individual brown rockfish sample (Table A.4-3).

Table A.4-2. Site-wide EPCs for English sole and brown rockfish for the tissue-residue evaluation

COPC	Detection Frequency	Mean (mg/kg ww)[a]	Maximum Detection (mg/kg ww)	EPC (mg/kg ww)	Statistic Used
Mercury[b]					
Brown rockfish	15/15	0.2	0.418	0.21	95% Approximate Gamma UCL
TBT					
English sole	11/11	0.026	0.038	0.030	95% Student's-t UCL
Brown rockfish	13/13	0.160	0.420	0.22	95% Approximate Gamma UCL

Table A.4-2. Site-wide EPCs for English sole and brown rockfish for the tissue-residue evaluation (cont.)

COPC	Detection Frequency	Mean (mg/kg ww)[a]	Maximum Detection (mg/kg ww)	EPC (mg/kg ww)	Statistic Used
Total PCBs					
English sole	13/13	3.20	7.90 J	4.1	95% Approximate Gamma UCL
Brown rockfish	15/15	2.00	6.20	4.0	95% H-UCL
Beta-Endosulfan					
Brown rockfish	6/9	0.0028	0.013	0.0056	95% KM (BCA) UCL

[a] The mean is calculated as the average of the detected concentrations and one-half the RL for non-detected results.

[b] Mercury was not identified as a COPC for English sole.

BCA – bias-corrected accelerated
COPC – chemical of potential concern
EPC – exposure point concentration
J – estimated concentration
KM – Kaplan-Meier
PCB – polychlorinated biphenyl
TBT – tributyltin
UCL – upper confidence limit on the mean
ww – wet weight

Table A.4-3. EPCs for individual brown rockfish for the tissue-residue evaluation

Brown Rockfish Sample ID	Concentration (mg/kg ww)			
	Mercury	TBT	Total PCBs	beta-Endosulfan
EW-08-SB002-BR-01	0.067	0.1	0.71	na
EW-08-SB002-BR-02	0.073	0.09	0.74 J	na
EW-08-SB003-BR-03	0.066	0.14	2.8 J	na
EW-08-SB004-BR-04	0.181	0.061 J	0.6	0.0017
EW-08-SB005-BR-05	0.23	0.42	0.61 J	0.0012
EW-08-SB006-BR-06	0.235	0.18	0.5 J	0.0013
EW-08-SB007-BR-07	0.12	0.29	2.2	0.0033
EW-08-SB008-BR-08	0.418	0.3	4.3	0.013
EW-08-SB009-BR-09	0.105	0.1	2.9	0.003
EW-08-SB012-BR-10	0.082	0.12	2	0.00045 U
EW-08-SB011-BR-11	0.12	0.12	2.8	0.00045 U
EW-08-SB012-BR-12	0.04	0.1	0.57 J	na
EW-08-SB013-BR-13	0.26	0.038	0.4 J	0.00047 U
EW-TR003-RF-01-WB-01-05	0.235	na	6.2	na
EW-TR003-RF-01-WB-02-05	0.07 J	na	2.9	na

ID – identification
J – estimated concentration
na – not analyzed
PCB – polychlorinated biphenyl
U – not detected at given concentration (value equals one-half the reporting limit)
TBT – tributyltin
ww – wet weight

A.4.1.2 Dietary exposure assessment

Chemicals were identified as COPCs for the three fish ROCs for the dietary evaluation (Section A.2.5.2.2) as follows:

- **English sole –** arsenic, cadmium, chromium, copper, vanadium, and benzo(a)pyrene
- **Brown rockfish –** arsenic, cadmium, chromium, copper, vanadium, and benzo(a)pyrene
- **Juvenile Chinook salmon –** arsenic, cadmium, chromium, copper, and vanadium

For the dietary evaluation, the primary exposure route is assumed to be the ingestion of prey. Incidental sediment ingestion is also considered an exposure route for English sole and brown rockfish.

A.4.1.2.1 Methods

In the risk characterization for the dietary evaluation, the estimated COPC concentrations in EW fish diets (in mg/kg dw) were compared with TRVs based on dietary concentrations associated with the presence or absence of adverse effects. Thus, exposures used for the dietary evaluation were estimated as the concentrations of COPCs in the dietary items, including sediment ingested incidentally.

COPC concentrations in the diet of each fish ROC were calculated as the weighted average of EPC concentrations in sediment and prey tissue using Equation 4-1.

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

Where:

C_{diet}	=	COPC concentration in the diet (mg/kg dw)
X_i	=	proportion of a particular prey item or sediment in the diet (unitless)
EPC_i	=	exposure point concentration of COPC in the prey item or sediment (mg/kg dw)
n	=	number of dietary items

The exposure assumptions (i.e., proportions of prey in the diet) are discussed in Section A.4.1.2.1, and the EPCs and calculated dietary concentrations are discussed in Section A.4.1.2.2.

A.4.1.2.2 Exposure assumptions

The assumed relative proportions of each prey item or sediment in each fish ROC's diet are summarized in Table A.4-4. The rationale for these proportions and additional ROC-specific exposure assumptions for each ROC are described in the subsections that follow.

Table A.4-4. Proportions of prey items and sediment in dietary exposure estimates for fish ROCs

ROC	Prey Item	Proportion in Diet (unitless)	Source
Juvenile Chinook salmon	benthic invertebrates	1.0	Windward (2004a), Cordell et al. (1997)
English sole	benthic invertebrates	0.99	Fresh et al. (1979), Wingert et al. (1979)
	sediment	0.01	Johnson (2006), Lange (2006)
Brown rockfish	benthic invertebrates	0.095	Wingert et al. (1979)
	coonstripe shrimp	0.47	
	crabs	0.045	
	shiner surfperch	0.38	
	sediment	0.01	Lange (2006)

COPC – chemical of potential concern
EW – East Waterway
ROC – receptor of concern

Juvenile Chinook Salmon

Stomach contents analyses of juvenile Chinook salmon from the LDW indicate that juvenile Chinook salmon typically ingest benthic invertebrates such as amphipods, worms, and clam siphons,[21] as well as drift organisms and zooplankton (Cordell et al. 1997, 1999, 2001). For the purpose of this ERA, juvenile Chinook salmon in the EW were assumed to ingest only benthic invertebrates. These same dietary assumptions were used in the LDW ERA. Because benthic invertebrates live in close contact with sediment, they have a greater potential for sediment exposure than do other juvenile Chinook salmon prey items; therefore, this assumption may overestimate exposure. Juvenile Chinook salmon from the LDW were found to have no appreciable amounts of sediment in their stomachs (Cordell 2001); therefore, it was assumed that juvenile Chinook salmon from the EW do not ingest sediment. The entire EW benthic invertebrate dataset, which consists of composite samples collected primarily from subtidal areas in the EW during the 2008 SRI sampling event, was used to calculate an EPC for each COPC, even though juvenile Chinook salmon generally do not use deep-

[21] EW data on larger individual clams were not included in the exposure calculations because the benthic invertebrate composite samples included clams < 2.0 cm, which were assumed to represent the sizes of clams consumed by the fish ROCs.

water subtidal habitats (Tabor et al. 2004). This uncertainty in the dietary evaluation for juvenile Chinook salmon is discussed in Section 6.2.1.2. For juvenile Chinook salmon, dietary exposure was also estimated independently based on the chemical analysis of the single composited juvenile Chinook salmon stomach contents sample collected from the EW during the 2009 SRI sampling event. This composite sample included stomach contents from 146[22] juvenile Chinook salmon.

English Sole

Stomach contents analyses of English sole collected from Puget Sound showed that English sole ingest benthic invertebrates such as polychaetes, amphipods, and mollusks almost exclusively (Fresh et al. 1979; Wingert et al. 1979). Based on these analyses, all English sole prey were assumed to be small benthic invertebrates. Thus, the same benthic invertebrate site-wide EPCs developed for juvenile Chinook salmon were used for English sole (i.e., the UCL of the entire EW benthic invertebrate dataset). In addition, incidental sediment ingestion of 1% was assumed based on observations of the stomach contents of English sole and other bottom-feeding fish in Puget Sound (Johnson 2006; Lange 2006). These same dietary assumptions were used in the LDW ERA.

Brown Rockfish

Stomach contents analyses of brown rockfish collected from Puget Sound showed that they primarily ingest shrimp and small fish, and smaller amounts of crabs and benthic invertebrates such as amphipods and isopods (Wingert et al. 1979). Percentages of these items in the brown rockfish diet were approximately 47% shrimp, 38% small fish, 4.5% crabs, and 9.5% benthic invertebrates (8.5% amphipods and 1.0% isopods) based on an analysis of the Index of Relative Importance (Wingert et al. 1979).[23] Data for COPC concentrations in these dietary items from the EW were available to develop EPCs for the dietary evaluation for brown rockfish, as follows:

- **Shrimp –** Data from the single composite shrimp sample that consisted of 26 individual shrimp collected from throughout the EW in 2008 (Windward 2010c) were used to represent shrimp in the brown rockfish diet
- **Small fish –** Data for 11 shiner surfperch composite samples collected from the EW in 2005 Windward (2006b) and 2008 (Windward 2010c) were used to represent small fish in the brown rockfish diet. Shiner surfperch were selected as the representative prey fish because they are numerically dominant in the fish community in the EW, and surfperch are important prey for brown rockfish (Matthews 1990b). Shiner surfperch also feed primarily on benthic organisms so

[22] A total of 165 fish were sampled for stomach contents. Nineteen fish had no measurable stomach contents and 146 fish contributed mass to the composite sample. The individual fish stomach content masses ranged from 0.01 -0.502g with an average of 0.05g of stomach contents per fish.

[23] IRI is a metric used to determine dietary importance of food items, including numerical abundance, biomass, and frequency of occurrence in diet.

they are susceptible to the bioaccumulation of sediment-associated chemicals through the food chain (Wingert et al. 1979; Fresh et al. 1979; Miller et al. 1977b).

- **Crab –** Data for eight red rock crab composite samples and one Dungeness crab composite sample collected from the EW in 2008 (Windward 2010c) were used to represent crab in the brown rockfish diet.[24] A single EPC was calculated for crab using these nine composite samples. Available EW crab data are from crabs larger than those consumed by brown rockfish. These larger crabs are expected to have COPC concentrations similar to or higher than those of smaller crabs. Therefore, the use of these data would more likely overestimate rather than underestimate exposure for brown rockfish.
- **Benthic invertebrates –** Data for 13 benthic invertebrate composite samples (representing a mix of species) collected from throughout the EW in 2008 (Windward 2009a) were used to represent amphipods and isopods in the brown rockfish diet.

An incidental sediment ingestion of 1% of the total diet was assumed based on the primarily epifaunal diet of the brown rockfish (Lange 2006).

Tagging studies have shown that brown rockfish have small home ranges on the order of 30 to 1,500 m^2 (Matthews 1990b). Therefore, the exposure of brown rockfish was evaluated on a site-wide basis to evaluate risks to the EW population, as well as on a sample-specific basis to assess smaller-scale exposures and risks to individual fish. For the individual dietary evaluation, three of the four brown rockfish prey types listed above (i.e., shrimp, small fish, and crab) have home ranges larger than those of brown rockfish, and thus the dietary EPCs represent site-wide exposures. However, for benthic invertebrates, data were collected from 13 separate areas within the EW, so data from the area closest to each individual brown rockfish sample were used in the dietary evaluation (see Map A.4-1). In addition, co-located sediment samples were collected with the individual brown rockfish samples, so those data were used for the incidental sediment ingestion component of the diet for the individual brown rockfish dietary evaluation (Map A.4-1).

A.4.1.2.3 Dietary EPCs

The dietary evaluation was conducted on a site-wide basis for all three fish ROCs, and thus site-wide EPCs were used for tissue and sediment in the dietary calculations. In addition, the dietary evaluation was conducted on a sample-specific basis for individual rockfish, using location-specific EPCs in the dietary calculations when available (i.e., for benthic invertebrates and sediment).

For the site-wide analyses, one site-wide EPC was calculated for each COPC in each ROC prey item, and one site-wide EPC was calculated for each COPC in sediment. The

[24] Composite samples were collected for both edible meat and hepatopancreas. Data from these two tissue types were mathematically combined to estimate whole-body concentrations for each composite sample using the relative weights and concentrations of the edible meat and hepatopancreas.

EPC for each prey item was estimated as the 95% UCL based on all samples collected from the EW for that prey item (Table A.4-5). The ProUCL software uses detected and undetected values and creates interpolated values for non-detects based on the perceived distribution of the detected concentrations. Once any necessary interpolation is performed, the software analyses the resulting data distribution to determine the most appropriate 95% UCL and makes a recommendation. The maximum concentration was used as the EPC if fewer than six samples were available, based on ProUCL guidance.

Table A.4-5. Site-wide EPCs for prey tissue, juvenile Chinook salmon stomach contents, and ingested sediment for fish ROCs

COPC	Detection Frequency	Mean (mg/kg dw)[a]	Maximum Detection (mg/kg dw)	Maximum RL (mg/kg dw)	EPC (mg/kg dw)	Statistic Used
Arsenic						
Benthic invertebrates	13/13	17	32.6	na	20	95% Student's-t UCL
Shiner surfperch	8/8	3.78	4.38 J	na	4.35	95% Student's-t UCL
Crab	9/9	26.9	35.1 J	na	29.4	95% Student's-t UCL
Shrimp	1/1	na	18.2	na	18.2	maximum detect
Juvenile Chinook salmon stomach contents	1/1	na	3.55	na	3.55	maximum detect
Sediment	162/231	9	241	20 U	12	95% KM (BCA) UCL
Cadmium						
Benthic invertebrates	6/13	1	2	3 U	1.2	95% KM (t) UCL
Shiner surfperch	0/8	0.1	nd	0.3 U	0.15[c]	half maximum RL
Crab	9/9	10	19	na	17	95% Student's-t UCL
Shrimp	1/1	na	0.70	na	0.70	maximum detect
Juvenile Chinook salmon stomach contents	1/1	na	0.488	na	0.488	maximum detect
Sediment	155/231	0.7	6.76	1 U	0.71	95% KM (BCA) UCL
Chromium						
Benthic invertebrates	13/13	20	45.1	na	29	95% Chebyshev (mean, Sd) UCL
Shiner surfperch	6/8	0.7	1	0.7 UJ	1.0[c]	maximum detect
Crab	8/9	0.5	0.7	0.3 UJ	0.38	95% KM (Chebyshev) UCL
Shrimp	1/1	na	2	na	2	maximum detect
Juvenile Chinook salmon stomach contents	1/1	na	1.59	na	1.59	maximum detect
Sediment	231/231	30	82 J	na	29	95% modified-t UCL
Copper						
Benthic invertebrates	13/13	97.5	155	na	110	95% Student's-t UCL
Shiner surfperch	8/8	5.98	11.3	na	8.00	95% Approximate Gamma UCL

Table A.4-5. Site-wide EPCs for prey tissue, juvenile Chinook salmon stomach contents, and ingested sediment for fish ROCs (cont.)

COPC	Detection Frequency	Mean (mg/kg dw)[a]	Maximum Detection (mg/kg dw)	Maximum RL (mg/kg dw)	EPC (mg/kg dw)	Statistic Used
Crab	9/9	162	203	na	178	95% Student's-t UCL
Shrimp	1/1	na	109	na	109	maximum detect
Juvenile Chinook salmon stomach contents	1/1	na	17.3	na	17.3	maximum detect
Sediment	231/231	59	272 J	na	62	95% Approximate Gamma UCL
Vanadium						
Benthic invertebrates	13/13	20	31	na	19	95% Approximate Gamma UCL
Shiner surfperch	4/8	0.4	1.0	0.4 U	1.0[d]	maximum detect
Crab	9/9	0.8	2	na	1.1	95% modified-t UCL
Shrimp	1/1	na	1	na	1	maximum detect
Juvenile Chinook salmon stomach contents	1/1	na	1.47	na	1.47	maximum detect
Sediment	111/111	57	94.1	na	59	95% Student's-t UCL
Benzo(a)pyrene						
Benthic invertebrates	13/13	0.50	1.20	na	0.68	95% Student's-t UCL
Shiner surfperch	1/8	0.0023	0.0024 J	0.013 U	0.0065[b]	half maximum RL
Crab	7/9	0.095	0.670 J	0.340 U	0.83	99% KM (Chebyshev) UCL
Shrimp	0/1	na	na	0.33	0.17[b]	half maximum RL
Juvenile Chinook salmon stomach contents	nd	nd	nd	nd	nd	nd
Sediment	225/240	0.320	7.80	0.061 U	0.50	95% KM (Chebyshev) UCL

[a] The mean is calculated as the average of the detected concentrations and one-half the RL for non-detected results.

[b] COPC was not detected; EPC is equal to one-half the maximum detection limit.

[c] ProUCL did not recommend a UCL for these data because there were only two distinct detected values (i.e., four detected concentrations were 0.7 mg/kg dw, and two detected concentrations were 1.0 mg/kg dw). ProUCL indicated that these data were not adequate to compute meaningful and reliable test statistics. Therefore, the maximum detected concentration was used as the EPC.

[d] The EPC is the maximum concentration because there were fewer than six detected concentrations.

BCA – bias-corrected accelerated
COPC – chemical of potential concern
dw – dry weight
EPC – exposure point concentration
KM – Kaplan-Meier
na – not applicable
nd – no data
RL – reporting limit
ROC – receptor of concern
Sd – standard deviation
UCL – upper confidence limit on the mean

The EPCs for surface sediment were calculated using both subtidal and intertidal data. Most of the surface sediment samples were collected from subtidal areas as individual grab samples, whereas only three composite samples were collected in intertidal areas using the MIS method. Because of the different sampling methods for the intertidal and subtidal samples, the site-wide EPC was calculated as a weighted average of the intertidal and subtidal EPCs based on the percentage of EW area represented by the intertidal samples (2.7%) and the subtidal samples (97.3%). EPCs for subtidal areas were calculated as the UCL of all subtidal data using ProUCL, as described in the previous paragraph for prey tissue. EPCs for intertidal areas were equivalent to the maximum concentration because there were fewer than the recommended number of samples (i.e., at least six) for calculating a UCL. The site-wide surface sediment EPCs are presented in Table A.4-6; the subtidal and intertidal EPCs used in calculating the site-wide EPCs are also presented for informational purposes.

Table A.4-6. Site-wide EPCs in EW surface sediment for fish ROCs

COPC	Exposure Area[a]	Detection Frequency	Mean (mg/kg dw)[a]	Maximum Detection (mg/kg dw)	Maximum RL (mg/kg dw)	EPC (mg/kg dw)	Statistic Used
Arsenic	intertidal	3/3	10	13	na	13	maximum detect
	subtidal	152/221	9.3	241	20 U	12	95% KM (BCA) UCL
	site-wide	155/224	9.3	241	20 U	12	weighted site-wide value[b]
Cadmium	intertidal	3/3	0.6	0.6	na	0.6	maximum detect
	subtidal	152/221	0.65	6.76	1 U	0.73	95% KM (BCA) UCL
	site-wide	155/224	0.65	6.76	1 U	0.73	weighted site-wide value[b]
Chromium	intertidal	3/3	31.2	44.8	na	45	maximum detect
	subtidal	221/221	28	82 J	na	29	95% Modified-t UCL
	site-wide	224/224	28	82 J	na	29	weighted site-wide value[b]
Copper	intertidal	3/3	39.5	41.4	na	41	maximum detect
	subtidal	221/221	59	272 J	na	70	95% Chebyshev (mean, Sd) UCL
	site-wide	224/224	58	272 J	na	69	weighted site-wide value[b]
Vanadium	intertidal	3/3	41	46 J	na	46	maximum detect
	subtidal	101/101	57	94.1	na	59	95% Student's-t UCL
	site-wide	104/104	57	94.1	na	59	weighted site-wide value[b]
Benzo(a)pyrene	intertidal	3/3	0.76	1.4	na	1.4	maximum detect
	subtidal	215/230	0.29	2.8	0.061 U	0.42	95% KM (Chebyshev) UCL
	site-wide	218/233	0.30	2.8	0.061 U	0.45	weighted site-wide value[b]

[a] The mean is calculated as the average of the detected concentrations and one-half the RL for non-detected results. Mean concentrations for the site-wide area are the weighted average of the intertidal mean (2.7% of the exposure area) and the subtidal mean (97.3% of the exposure area).

[b] The site-wide EPC is the weighted average of the intertidal EPC (2.7% of the exposure area) and the subtidal EPC (97.3% of the exposure area).

BCA – bias-corrected accelerated bootstrap method
COPC – chemical of potential concern
dw – dry weight
EPC – exposure point concentration
EW – East Waterway
J – estimated concentration
KM – Kaplan-Meier
na – not applicable (i.e., all results were detects)
ROC – receptor of concern
Sd – standard deviation
U – not detected at given concentration
UCL – upper confidence limit on the mean

For the individual rockfish dietary analysis, location-specific EPCs were available for benthic invertebrates and sediment. For benthic invertebrates, the EPC for each individual rockfish was equal to the concentration in the composite sample collected from the area closest to the individual rockfish sample (Map A.4-1). For sediment, the EPC for each individual rockfish was equal to the concentration in the co-located sediment sample collected with each individual rockfish sample (Map A.4-1). EPCs for surfperch, crab, and shrimp were available only as site-wide concentrations and were calculated as discussed above and presented in Table A.4-6. EPCs for dietary items for individual brown rockfish are presented in Table A.4-7.

Table A.4-7. EPCs for prey tissue and sediment ingested by individual brown rockfish

COPC	EPC by Tissue Type or Sediment (mg/kg dw)				
	Benthic Invertebrates[a]	Surfperch[b]	Crab[b]	Shrimp[b, c]	Sediment[a]
Arsenic	11 – 21.4	4.35	29.4	18.2	3.1 – 26.2
Cadmium	0.9 – 2.0	0.15	17	0.70	0.2 – 6.8
Chromium	9.0 – 45.1	1.0	0.38	2	20.1 – 82.0
Copper	67.2 – 140	8.00	178	109	27.3 – 272
Vanadium	10 – 31	1.0	1.1	1	42.1 – 91.5
Benzo(a)pyrene	0.14 – 1.2	0.0065	0.83	0.17	0.050 – 2.8

[a] Range of concentrations in benthic invertebrate and sediment samples used in individual brown rockfish dietary calculations.

[b] Site-wide EPC for surfperch, crab, and shrimp; derivation of these EPCs is presented in Table A.4-6.

[c] Concentration in single site-wide composite shrimp sample.

COPC – chemical of potential concern
dw – dry weight
EPC – exposure point concentration
ID – identification

Using Equation 4-1 and the methods described in Section A.4.1.2.1, dietary COPC concentrations were calculated for each fish ROC on a site-wide basis as well as for brown rockfish on an individual basis (Tables A.4-8 and A.4-9).

Table A.4-8. Calculated dietary EPCs for fish ROCs for use in dietary evaluation

ROC	Dietary Concentration (mg/kg dw)					
	Arsenic	Cadmium	Chromium	Copper	Vanadium	Benzo(a)pyrene
Juvenile Chinook salmon	20	1.2	29	110	19	na
English sole	20	1.2	29	110	19	0.68
Brown rockfish	14	1.3	4.4	73	3.3	0.19

COPC – chemical of potential concern
dw – dry weight
EPC – exposure point concentration
ROC – receptor of concern

Table A.4-9. Calculated dietary EPCs for individual brown rockfish for use in dietary evaluation

Sample ID	Dietary Concentration (mg/kg dw)					
	Arsenic	Cadmium	Chromium	Copper	Vanadium	Benzo(a)pyrene
EW-08-SB002-BR-01	7.9	2.5	3.2	25	2.9	0.089
EW-08-SB002-BR-02	7.9	2.5	3.2	25	2.9	0.089
EW-08-SB003-BR-03	7.9	2.5	3.2	25	2.8	0.090
EW-08-SB004-BR-04	8.5	3.6	3.1	31	3.9	0.33
EW-08-SB005-BR-05	8.5	3.0	2.4	29	4.1	0.067
EW-08-SB006-BR-06	8.0	2.6	2.5	26	3.5	0.064
EW-08-SB007-BR-07	8.3	3.5	2.6	33	3.7	0.087
EW-08-SB008-BR-08	8.1	2.6	2.1	33	4.2	0.091
EW-08-SB009-BR-09	9.4	1.9	2.7	27	3.5	0.084
EW-08-SB012-BR-10	7.3	2.2	2.4	26	2.9	0.12
EW-08-SB011-BR-11	8.6	3.8	2.6	29	4.5	0.14
EW-08-SB012-BR-12	8.4	2.6	2.1	27	4.3	0.17
EW-08-SB013-BR-13	7.7	2.1	6.3	32	5.2	0.20

dw – dry weight
EPC – exposure point concentration
ID – identification

A.4.1.3 Surface water exposure assessment

Three chemicals (cadmium, mercury, and TBT) were identified as surface water COPCs for the fish ROCs (Section A.2.5.2.3). EPCs were calculated as the 95% UCLs using all the site-wide surface water data for each COPC to represent exposure throughout the site, thus accounting for a variety of seasons and water flow conditions. In addition, EPCs based on detected COPC concentrations in individual water samples were used to represent conditions at that location at the time of sampling as a more conservative analysis. Cadmium and mercury EPCs were based on the dissolved fraction because the TRVs were based on the dissolved fraction; TBT EPCs were based on total concentrations.

For the site-wide evaluation, EPCs were calculated using all water samples for each COPC. ProUCL was used to calculate 95% UCLs for the site-wide EPCs if there were at least six detected concentrations. The maximum concentration was used if there were fewer than six detected concentrations, as recommended by ProUCL. The ProUCL software uses both detected and undetected values and creates interpolated values for non-detects based on the perceived distribution of the detected concentrations. Once any necessary interpolation has been performed, the software analyzes the resulting data distribution to determine the most appropriate 95% UCL and makes a recommendation. Site-wide EPCs for surface water based on samples collected throughout the EW are presented in Table A.4-10. EPCs for individual water samples were equivalent to the detected COPC concentrations in the individual samples. The range of COPC detected concentrations (i.e., EPCs) in individual water samples collected from throughout the EW are also presented in Table A.4-10.

Table A.4-10. Site-wide surface water EPCs for fish

COPC	Detection Frequency	Mean (µg/L)[a]	Range of Detects (µg/L)	Maximum RL (µg/L)	EPC (µg/L)	Statistic Used
Cadmium (dissolved)	126/130	0.40	0.009 J – 37.8	0.088	0.94	95% KM (BCA) UCL
Mercury (dissolved)	23/68	0.0003	0.00013 – 0.00146	0.00054	0.00039	95% KM (t) UCL
TBT	1/59	na	0.010 J	0.01	0.01[b]	Maximum detect

[a] The mean is calculated as the average of the detected concentrations and one-half the RL for non-detected results.

[b] No UCL was calculated for TBT because there was only one detected concentration; thus, the EPC is equal to the detected concentration.

BCA – bias-corrected accelerated
COPC – chemical of potential concern
EPC – exposure point concentration
KM – Kaplan-Meier
na – not applicable (only one detected value)
RL – reporting limit
TBT – tributyltin
UCL – upper confidence limit on the mean

A.4.2 Effects Assessment

This section summarizes the toxicity literature for the COPCs identified for fish and presents the TRVs selected for fish. The literature search and guidelines for TRV selection for fish ROCs are described in detail in Section A.2.5.2. Toxicological data presented in this section and exposure data presented in Section A.4.1 are evaluated together in Section A.6.2 to characterize risks to fish.

A.4.2.1 TRVs for tissue-residue effects

TRVs were selected for COPCs identified for the tissue-residue evaluation for fish: mercury, TBT, total PCBs, and beta-endosulfan. The following subsections summarize the toxicity studies reviewed and the NOAEL and LOAEL TRVs selected for these COPCs.

A.4.2.1.1 Mercury

Mercury was identified as a COPC only for brown rockfish (Section A.2.5.2.1). Nineteen toxicity studies were identified as being acceptable for TRV selection for mercury: these studies measured tissue concentrations and toxicological effects associated with exposure to methylmercury (the most toxic form of mercury) or inorganic mercury (Table A.4-11). The pathways of exposure included sediment, diet, and water. No studies that used brown rockfish were identified. Tissue concentrations of mercury were associated with adverse effects in seven species (mummichog, golden shiner, fathead minnow, mosquitofish, brook trout, creek chub, goldfish, and rainbow trout).[25] Adverse effects included increased mortality, reduced adult and offspring growth, impaired fertilization success, reduced hatchability, and reduced offspring survival. Whole-body tissue-residue LOAELs ranged from 0.39 mg/kg ww for reduced spawning success of fathead minnow (Hammerschmidt et al. 2002) to 11.2 mg/kg ww for rainbow trout survival (Niimi and Kissoon 1994). The NOAELs ranged from 0.2 mg/kg ww for mortality of guppy (Kudo and Mortimer 1979) to 29 mg/kg ww for mortality of rainbow trout (Rodgers and Beamish 1982).

Table A.4-11. Whole-body fish toxicity studies reviewed for the selection of tissue-residue TRVs for mercury

Chemical[a]	Test Species	NOAEL (mg/kg ww)	LOAEL (mg/kg ww)	Exposure Route and Duration	Effect	Source
Mercuric chloride	guppy	0.2	na	sediment and water for 20 days	no effect on survival	Kudo and Mortimer (1979)
Methylmercuric chloride	fathead minnow	na	**0.39**	diet in multiple generations	reduced spawning success	Hammerschmidt et al. (2002)
Methylmercuric chloride	mummichog	0.2	0.47	water for 42 days	reduced survival	Matta et al. (2001)
Methylmercuric chloride	golden shiner	**0.23**	0.536[b]	diet for 90 days	altered predator avoidance (potential for reduced survival)	Webber and Haines (2003)
Methylmercuric chloride	fathead minnow	na	0.714	diet for at least 21 days	reduced spawning success	Sandheinrich and Miller (2006)
Mercuric chloride	fathead minnow	0.8	1.31	diet for 60 days	reduced growth	Snarski and Olson (1982)
Methylmercuric chloride	brook trout	2.7	3.4	water for 756 days	reduced number of viable eggs	McKim et al. (1976)
Mercuric chloride	creek chub	na	3.72	water for 48 hours	reduced survival	Kim et al. (1977)
Mercuric chloride	fathead minnow	2.75	4.18	water for 60 days	reduced survival	Snarski and Olson (1982)
Mercuric chloride	goldfish	na	4.4	water for 4 days	reduced survival	Heisinger et al. (1979)
Mercuric chloride	fathead minnow	2.84	4.47	water for 287 days	reduced spawning	Snarski and Olson (1982)
Methylmercury	rainbow trout	5.0	na	water for 84 days	no effect on growth or survival	Lock (1975)
Methylmercuric chloride	rainbow trout	8.63	na	water for 24 days	no effect on growth	Phillips and Buhler (1978)

[25] Total mercury analyses, rather than speciation analyses, are generally conducted for fish tissue because the predominant form of mercury is methylmercury (Bloom 1992; Grieb et al. 1990). Fish tissue samples in the ERA dataset were analyzed for total mercury.

Table A.4-11. Whole-body fish toxicity studies reviewed for the selection of tissue-residue TRVs for mercury (cont.)

Chemical[a]	Test Species	NOAEL (mg/kg ww)	LOAEL (mg/kg ww)	Exposure Route and Duration	Effect	Source
Methylmercuric chloride	brook trout	9.4	na	water for 756 days	no effect on growth or survival	McKim et al. (1976)
Methylmercuric chloride	rainbow trout	na	10	diet for 84 days	reduced growth	Rodgers and Beamish (1982)
Methylmercuric chloride	fathead minnow	10.9	na	water for 336 days	no effect on growth or survival	Olson et al. (1975)
Methylmercuric chloride	mummichog	1.1	11	diet for 42 days	reduced fertilization success	Matta et al. (2001)
Methylmercuric chloride	rainbow trout	na	11.2	water for 12 to 33 days	no effect on survival	Niimi and Kissoon (1994)
Methylmercuric chloride	rainbow trout	12	na	water for 75 days	no effect on growth or survival	Niimi and Lowe-Jinde (1984)
Methylmercuric chloride	rainbow trout	29	na	diet for 84 days	no effect on mortality	Rodgers and Beamish (1982)

[a] Represents the form of mercury to which fish were exposed.

[b] The abstract of this paper indicated that the LOAEL whole-body mercury concentration was 0.518 mg/kg ww rather than 0.536 mg/kg ww, as presented in the body of the paper.

LOAEL – lowest-observed-adverse-effect level

na – not available

NOAEL – no-observed-adverse-effect level

TRV – toxicity reference value

ww – wet weight

Bold and underline identify the NOAEL and LOAEL selected as TRVs.

The LOAEL of 0.39 mg/kg ww based on a reproductive endpoint from Hammerschmidt et al. (2002) was selected as the LOAEL TRV for mercury in fish. This selected LOAEL was the lowest whole-body tissue residue reported in the literature considered acceptable for TRV derivation. There was no NOAEL below the LOAEL for a reproductive endpoint; therefore, the highest NOAEL below the LOAEL was taken from a study with an endpoint related to survival. This NOAEL of 0.23 mg/kg ww was reported in golden shiners exposed to dietary methylmercury for 90 days (Webber and Haines 2003). This study measured predator avoidance behavior, which is assumed to reduce survival. At the LOAEL, fish broke the water surface and broke from the shoal, thus clearly increasing predator access and resulting in a high likelihood of decreased survival in the wild. The golden shiner predator avoidance NOAEL of 0.23 mg/kg ww was selected as the NOAEL TRV.

A.4.2.1.2 TBT

TBT was identified as a COPC for English sole and brown rockfish. Two studies that evaluated the toxicity associated with TBT measured in whole-body fish were available (Table A.4-12). These two studies reported whole-body tissue concentrations associated with adverse effects following the exposure of rainbow trout to TBT in water (Triebskorn et al. 1994) and the exposure of Japanese medaka to TBT in the diet

(Nirmala et al. 1999). No studies that used the English sole or brown rockfish were available.

Table A.4-12. Fish toxicity studies reviewed for the selection of whole-body tissue-residue TRVs for TBT

Chemical	Test Species	Whole-Body NOAEL (mg/kg ww)[a]	Whole-body LOAEL (mg/kg ww)[a]	Exposure Route and Duration	Effect	Source
Studies Reporting Whole-Body NOAELs and LOAELs						
Tributyltin oxide	rainbow trout	na	**<u>0.29</u>**	water exposure for 21 days	reduced body weight	Triebskorn et al. (1994)
Tributyltin oxide	Japanese medaka	na	2.39	maternal exposure to 1 mg/kg dw in food for 3 weeks	reduced hatching, swim-up, and embryonic success	Nirmala et al. (1999)
Studies Reporting Only Egg NOAELs and LOAELs						
Tributyltin oxide	Japanese whiting (egg)[a]	na	0.73 – 1.37[b]	dietary exposure for 30 days	reduced floating egg rate, hatchability, and number of viable larvae	Shimasaki et al. (2006)
Tributyltin oxide	Japanese medaka (egg)[a]	na	1.05[c]	maternal exposure to 5 mg/kg dw in food for 3 weeks	reduced swim-up success and hatchability	Nakayama et al. (2005)

[a] Whole-body NOAELs and LOAELs were estimated using egg-to-adult conversion factors for studies that reported concentrations in eggs rather than whole-body tissue.

[b] Whole-body maternal tissue concentrations were estimated from egg concentrations of 0.085 and 0.16 mg/kg ww reported in Shimasaki et al. (2006) using an egg-to-adult conversion factor of 8.6 based on the data for Japanese medaka reported in Nirmala et al. (1999).

[c] Whole-body maternal tissue concentrations were estimated from egg concentrations of 0.123 mg/kg ww reported in Nakayama et al. (2005) using an egg-to-adult conversion factor of 8.6 based on data for Japanese medaka reported in Nirmala et al. (1999).

dw – dry weight
LOAEL – lowest-observed-adverse-effect level
na – not available
NOAEL – no-observed-adverse-effect level
TBT – tributyltin
TRV – toxicity reference value
ww – wet weight
<u>Bold and underline</u> identify the LOAEL selected as the TRV.

Whole-body tissue LOAELs were 0.29 mg/kg ww for reduced body weight in rainbow trout following 21 days of aqueous exposure to TBT (Triebskorn et al. 1994) and 2.39 µg/kg ww for reduced hatchability and early-life stage mortality of Japanese medaka offspring spawned from parents exposed to dietary TBT for 3 weeks during reproduction (Nirmala et al. 1999). At the lowest LOAEL, the head and body TBT concentrations were 0.35 and 0.27 mg/kg ww, respectively. The whole- body LOAEL of 0.29 mg/kg ww was calculated assuming a head-to-body mass ratio of 1:4. A lower LOAEL (0.159 mg TBT/kg ww) for reduced body weight in Japanese flounder larvae following 65 days of dietary exposure to TBT was identified (Shimasaki et al. 2003) but did not meet TRV study selection criteria because the reported tissue concentrations were whole-body samples minus intestines, livers, kidneys, and gall bladders. Additional uncertainties associated with this study included:

- Parental fish were experimentally manipulated to produce only female offspring, which were subsequently used for the TBT toxicity experiments.
- High mortality was observed in both the negative control and TBT-exposed groups (survival was 42% in the control group and 57% in the LOAEL group). In standardized fish toxicity tests, control survival less than 90% generally invalidates the test (e.g., ASTM 1996).

Another study with a lower LOAEL (0.047 mg TBT/kg ww) for zebrafish masculization (Santos et al. 2006) was not included because the reproductive significance of the increased proportion of males in the TBT-exposed population (82%of test fish were male) relative to the negative control population (63%of control fish were male) was uncertain. Both zebrafish and the Japanese flounder used in Shimasaki et al. (2003) are fish species that are known to undergo sex reversal in response to environmental and chemical stressors.

Two other studies in Table A.4-12 were based on the toxicity associated with TBT measured in eggs rather than whole-body fish. The egg tissue concentrations were converted to whole-body tissue concentrations using an egg-to-adult conversion factor of 8.6 from Nirmala et al. (1999).

Based on the available literature, the lowest acceptable LOAEL of 0.29 mg/kg ww reported in Triebskorn et al. (1994) was selected as the LOAEL TRV. No acceptable NOAELs were identified so a NOAEL of 0.029 mg/kg ww was selected based on the LOAEL divided by a safety factor of 10 for a subchronic exposure duration.

A.4.2.1.3 PCBs

Eighteen studies on the potential adverse effects of PCB mixtures on fish were reviewed (Table A.4-13). In four of these studies, effects were based on concentrations in fish eggs or embryos rather than in whole-body fish (Fisher et al. 1994; Freeman and Idler 1975; Hendricks et al. 1981; McCarthy et al. 2003). For the purpose of this ERA, the egg tissue concentrations were converted to whole-body tissue concentrations using egg-to-adult conversion factors presented in Niimi (1983), as noted in Table A.4-13.

Table A.4-13. Fish toxicity studies reviewed for the selection of whole-body tissue-residue TRVs for PCBs

Chemical	Test Species	Tissue Analyzed	Whole-body NOAEL (mg/kg ww)[a]	Whole-body LOAEL (mg/kg ww)[a]	Exposure Route and Duration	Effect	Source
Studies Reporting Whole-Body NOAELs and LOAELs							
Aroclor 1260	common barbel	whole body	na	**0.520**[b]	maternal exposure for 50 days	reduced fecundity	Hugla and Thome (1999)
Aroclor 1254	juvenile Chinook salmon	whole body	0.980	na	17 mg/kg ww in food for 4 weeks	no effect on growth or survival	Powell et al. (2003)
Aroclor 1260	common barbel	whole body	**0.520**[b]	**2.64**[b]	maternal exposure for 75 days	lack of spawning in first reproductive season; egg and larval mortality	Hugla and Thome (1999)
Aroclor 1254	rainbow trout (14 weeks)	whole body	8.0	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	1.9	9.3	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)
Aroclor 1254	pinfish	whole body	na	14	water for 14 to 35 days	reduced survival	Hansen et al. (1971)
Aroclor 1268	mummichog (adult)	whole body	15	na	15 µg/g in food for 6 weeks	no effect on fertilization, hatching, or larval survival	Matta et al. (2001)
Clophen A50	common minnow	whole body	na	25	diet for 40 days	reduction in time to hatch, fry mortality	Bengtsson (1980)
Aroclor 1260	channel catfish	whole body	32	na	diet for 193 days	no effect on growth or survival	Mayer et al. (1977)
Aroclor 1254	spot	whole body	27	46	1 and 5 µg/L in water for 20 days	reduced survival	Hansen et al. (1971)
Aroclor 1260	fathead minnow	whole body	na	50	water for 30 days	reduced offspring body weight	DeFoe et al. (1978)
Aroclor 1254	brook trout embryos	whole body	31	71[c]	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)
Aroclor 1016	sheepshead minnow	whole body	77	na	10 µg/L in water for 2 weeks	no effect on fertilization success, survival of embryos, or fry survival	Hansen et al. (1975)
Aroclor 1016	pinfish	whole body	na	106	21 µg/L in water for 33 days	50% mortality	Hansen et al. (1974b)

Table A.4-13. Fish toxicity studies reviewed for the selection of whole-body tissue-residue TRVs for PCBs (cont.)

Chemical	Test Species	Tissue Analyzed	Whole-body NOAEL (mg/kg ww)[a]	Whole-body LOAEL (mg/kg ww)[a]	Exposure Route and Duration	Effect	Source
Aroclor 1254: 1260 mixture	juvenile rainbow trout	whole body	120	na	2.9 µg/L in water for 90 days	no effect on survival	Mayer et al. (1985)
Aroclor 1254: 1260 mixture	juvenile rainbow trout	whole body	70	120	1.5 and 2.9 µg/L in water for 90 days	reduced growth	Mayer et al. (1985)
Aroclor 1254	brook trout embryos	whole body	71	125	1.5 and 3.1 µg/L in 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	77	200	10 and 32 µg/L in water for 2 weeks	reduced fry survival	Hansen et al. (1975)
Clophen A50	goldfish	whole body	na	250	4,000 µg/L in water for 5 to 21 days	reduced survival	Hattula and Karlog (1972)
Aroclor 1254	fathead minnow	whole body	na	196 (male)	1.8 µg/L in water for 8 months	reduced spawning	Nebeker et al. (1974)
Aroclor 1242, 1254, or 1260	fathead minnow (6 months)	whole body	na	1.86 – 749	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)
Studies Reporting Only Egg and Embryo NOAELs and LOAELs							
1:1:1:1 Aroclor 1016, 1221, 1254, and 1260 mixture	Atlantic salmon	embryo	na	0.857	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)
Aroclor 1254	rainbow trout	embryo	na	1.64[d]	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	3.2[e]	maternal transfer	reduced larval growth	McCarthy et al. (2003)
Aroclor 1254	brook trout	embryo	na	77.9[f]	200 µg/L in water for 21 days	reduced hatchability (75%)	Freeman and Idler (1975)

[a] Whole-body NOAELs and LOAELs were estimated using egg-to-adult conversion factors for studies that reported concentrations in eggs rather than whole-body tissue.

[b] 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). 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.

[c] At the LOAEL, growth was significantly less than control at 48 days after hatching 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 (1.8 mg/kg ww) and low effects (3.2 mg/kg ww) were lower than the concentration at 118 days post-hatch.

Table A.4-13. Fish toxicity studies reviewed for the selection of whole-body tissue-residue TRVs for PCBs (cont.)

[d] Whole-body maternal tissue concentrations estimated from egg concentration of 7.69 mg/kg ww reported in Hendricks et al. (1981) using an egg-to-adult conversion factor of 4.69, based on rainbow trout data reported in Niimi (1983).

[e] Whole-body maternal tissue concentrations estimated from egg concentration of 8.67 mg/kg ww reported in McCarthy et al. (2003) using an egg-to-adult conversion factor of 2.71, based on the average of five fish species reported in Niimi (1983).

[f] Whole-body maternal tissue concentrations estimated from egg concentration of 365.4 mg/kg ww reported in Freeman and Idler (1975) using an egg-to-adult conversion factor of 4.69, based on rainbow trout data reported in Niimi (1983).

LOAEL – lowest-observed-adverse-effect level

na – not available

NOAEL – no-observed-adverse-effect level

PCB – polychlorinated biphenyl

TRV – toxicity reference value

ww – wet weight

Bold and underline identify the NOAEL and LOAEL selected as TRVs.

None of the studies in Table A.4-13 used English sole or brown rockfish, the two fish ROCs for which PCBs is a COPC. Concentrations of PCBs in tissue were reported in 17 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, common minnow, and spot). Adverse effects included reduced body weight; 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. In Table A.4-13, the whole-body NOAELs and LOAELs estimated from eggs or embryos are presented separately from the measured whole-body NOAELs and LOAELs because of the high uncertainty associated with converting the egg or embryo concentrations to whole-body tissue concentrations. There is uncertainty in the egg-to-adult conversion factor because the study used species and exposures that may have been different from the toxicity data to which they were applied, and there was little to no replication. [26] Because of the uncertainty in the estimated whole-body NOAELs and LOAELs, these studies were not selected as TRVs but are discussed below and in the uncertainty analysis (Section A.6.2).

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 0.520 mg/kg ww for reduced barbel fecundity (Hugla and Thome 1999) to 749 mg/kg ww for mortality of fathead minnows (van Wezel et al. 1995).

In the study that reported the lowest LOAEL, Hugla and Thome (1999) exposed 3- to 5-year-old common barbel from the University of Liege hatchery to 2.5 mg/kg PCBs in food for 50 days or to 12.5 mg/kg PCBs in food for 75 days (nominal concentrations) and analyzed the effects on reproduction. Fish were reared at elevated temperatures (Leroy 2007). Treatments were not replicated; 16 fish in each treatment were exposed in a single tank (Leroy 2007). Spawning success was monitored during 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[27] were reported following 50 or 75 days of exposure. During the first reproductive season, no spawning was reported at the high exposure level, and no adverse effects were reported for the lower exposure level. One year following exposure, significant reductions in fecundity were reported at both exposure levels corresponding to whole-body concentrations of 0.520 and 2.64 mg/kg ww for the low and high exposure levels, respectively. Egg mortality in the high-level dietary exposure group was close to 100% and was significantly higher than the control (which had a mean egg mortality of 52.4%), and

[26] Niimi (1983) calculated egg-to-adult ratios in five species of field-collected fish and eggs: rainbow trout, white sucker, white bass, smallmouth bass, and yellow perch.

[27] 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).

egg and larval mortality significantly increased as PCB concentrations in eggs increased. 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-dosed fish finally did spawn. After the second spawning, average fecundity of fish that received both the higher and lower doses was similar, 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, two LOAELs were selected for PCBs: 0.520 to 2.64 mg/kg ww. Additional effects data are discussed below for comparison.

In the study that reported the next higher LOAEL, Hansen et al. (1974a) exposed 20 female and 10 male adult sheepshead minnows for 4 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 9.3 mg/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, which corresponded to a tissue concentration of 1.9 mg/kg ww. Uncertainties associated with this study are discussed in the uncertainty analysis (Section A.6.2.2.2).

Among the studies reviewed for this ERA, whole-body effects concentrations extrapolated from eggs and embryos ranged from 0.857 to 77.9 mg/kg ww. The lowest value was for reduced fry body weight after embryos were exposed to 625 µg/L of PCBs in water for 48 hours (Fisher et al. 1994). The highest value was for reduced hatchability for brook trout embryos exposed to 200 µg/L of PCBs in water for 21 days (Freeman and Idler 1975). NOAELs were not identified in Fisher et al. (1994) and (Freeman and Idler 1975).

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 the comparison of whole-body concentrations in more mature fish to egg and embryo concentrations are discussed in the uncertainty analysis (Section A.6.2.2.2).

Whole-body NOAELs ranged from 0.980 mg/kg ww, at which no effect on growth or survival was reported for juvenile Chinook salmon (Powell et al. 2003), to 120 mg/kg ww, at which no effect on survival was reported for 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 0.104 to 0.528 mg/kg ww was estimated by applying an uncertainty factor of 5 to the range of chronic reproductive effects concentrations.

A.4.2.1.4 beta-Endosulfan

One study that evaluated the toxicity of endosulfan to fish was available (Table A.4-14). In this study by Schimmel et al. (1977), spot, pinfish, and mullet were exposed to endosulfan in water for 96 hours. The lowest LOAEL in the study (0.031 mg/kg ww) was reported for spot and was selected as the LOAEL TRV. No lower NOAEL was identified, so a NOAEL of 0.0031 mg/kg ww was derived as the NOAEL TRV using a safety factor of 10 because this was an acute study. There is some uncertainty associated with this TRV because fish were exposed to technical endosulfan, which contains 70% alpha-endosulfan and 30% beta-endosulfan, as discussed in the uncertainty analysis for fish (Section A.6.2).

Table A.4-14. Fish toxicity studies reviewed for the selection of whole-body tissue-residue TRVs for beta-endosulfan

Chemical	Test Species	NOAEL (mg/kg ww)	LOAEL (mg/kg ww)	Exposure Route and Duration	Effect	Source
Technical endosulfan[a]	spot	na	**<u>0.031</u>**	water for 96 hours	reduced survival (65%)	Schimmel et al. (1977)
Technical endosulfan[a]	pinfish	0.195	0.272	water for 96 hours	reduced survival (65%)	Schimmel et al. (1977)
Technical endosulfan[a]	mullet	na	0.360	water for 96 hours	reduced survival (60%)	Schimmel et al. (1977)

[a] Technical endosulfan is 70% alpha-endosulfan and 30% beta-endosulfan.

LOAEL – lowest-observed-adverse-effect level

na – not available

NOAEL – no-observed-adverse-effect level

TRV – toxicity reference value

ww – wet weight

<u>Bold and underline</u> identify the LOAEL selected as the TRV.

A.4.2.2 TRVs for dietary effects

TRVs were selected for six COPCs identified for the dietary evaluation for fish, as follows: arsenic, cadmium, chromium, copper, vanadium, and benzo(a)pyrene. The following subsections summarize the toxicity studies reviewed and the NOAEL and LOAEL TRVs selected for these COPCs.

A.4.2.2.1 Arsenic

Six toxicity studies that evaluated the effects of dietary arsenic on fish were identified (Table A.4-15). All six studies reported reductions in growth among juvenile rainbow trout and striped bass following dietary exposure to arsenic; no dietary toxicity data were available for the survival or reproductive endpoints. 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 to 30 mg/kg of dietary arsenic had significantly less weight gain than did control fish.

Table A.4-15. Fish toxicity studies reviewed for the selection of dietary TRVs for arsenic

Chemical	Test Species	NOAEL (mg/kg dw)[a]	LOAEL (mg/kg dw)[a]	Exposure Duration	Effect	Source
Sodium arsenite	juvenile rainbow trout	**<u>20</u>**[b]	**<u>30</u>**	6 weeks	reduced body weight	Oladimeji et al. (1984)
Disodium arsenate heptahydrate	juvenile rainbow trout	8	44	16 weeks	reduced body weight	Cockell et al. (1991)
Disodium arsenate heptahydrate	juvenile rainbow trout	na	49	24 weeks	reduced body weight	Cockell et al. (1991)
Disodium arsenate heptahydrate	juvenile rainbow trout	na	55	8 days	reduced body weight	Cockell et al. (1992)
Disodium arsenate	juvenile rainbow trout	na	58	12 days	reduced body weight	Cockell and Bettger (1993)
Disodium arsenate heptahydrate	juvenile rainbow trout	32	60	12 days	reduced body weight	Cockell et al. (1992)
Disodium arsenate heptahydrate	juvenile rainbow trout	33[c]	65	24 weeks	reduced body weight	Cockell et al. (1991)
Disodium arsenate	juvenile rainbow trout	na	137	8 days	reduced body weight	Cockell and Hilton (1988)
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)

[a] Concentrations are for elemental arsenic.

[b] Concentrations in figure and text in study did not agree: 20 mg/kg dw was mentioned both as an effect level and a no-effect level in the text; however, it was shown in the figure to be not significant. The NOAEL was assumed to be 20 mg/kg dw.

[c] 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

TRV – toxicity reference value

<u>Bold and underline</u> identify the NOAEL and LOAEL selected as TRVs.

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) 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 (Table A.4-16).[28] LOAELs ranged from 0.5 mg/kg dw for the growth of juvenile rockfish fed dietary cadmium for 60 days (Kang et al. 2005; Kim et al. 2004) to 2,265 mg/kg dw for the survival of juvenile rainbow trout following dietary exposure for 30 days (Szebedinsky et al. 2001). 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 based on the TRV selection guidelines used in this ERA. 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 have been 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 786 mg/kg dw for survival 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.

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[28] Note that Kim et al. (2004) and Kang et al. (2005) are the same study reported in two separate publications.

Table A.4-16. Fish toxicity studies reviewed for the selection of dietary TRVs for cadmium

Chemical	Test Species	NOAEL (mg/kg dw)[a]	LOAEL (mg/kg dw)[a]	Exposure Duration	Effect	Source
Cadmium nitrate	juvenile rockfish	na	**<u>0.5[b]</u>**	60 days	reduced growth rate and condition factor	Kim et al. (2004); Kang et al. (2005)
Cadmium chloride	rainbow trout fry	55	na	60 days	no effect on body weight, length, or survival	Mount et al. (1994)
Cadmium nitrate	juvenile rockfish	125[b]	na	60 days	no effect on survival	Kim et al. (2004) Kang et al. (2005)
Cadmium chloride	guppy	171[c]	na	10 to 30 days	no effect on growth	Hatakeyama and Yasuno (1982)
Cadmium chloride	adult guppy	210	na	2 months	no effect on fry survival or premature embryos	Hatakeyama and Yasuno (1987)
Cadmium	Atlantic salmon	250[b]	na	4 weeks	no effect on growth rate (body weight)	Lundebye et al. (1999)
Cadmium chloride	guppy (2 months old)	274	na	30 days	no effect on body weight	Hatakeyama and Yasuno (1987)
Cadmium chloride	juvenile rainbow trout	294[b]	na	15 to 30 days	no effect on growth rate or survival	Baldisserotto et al. (2005)
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)
Cadmium chloride	guppy (1 month old)	na	1,250	7 months	reduced female growth and survival	Hatakeyama and Yasuno (1987)
Cadmium nitrate	juvenile rainbow trout	786[b]	1,395[b]	30 days	57% survival	Szebedinsky et al. (2001)

[a] Concentrations are for elemental cadmium.

[b] Dietary dose was not reported as wet weight or dry weight and was assumed to be a dry-weight concentration.

[c] Body length was reduced at day 10 in fish fed 171 mg/kg dw in diet as compared with that of the control but not at 20 days.

dw – dry weight

na – not available

NOAEL – no-observed-adverse-effect level

<u>Bold and underline</u> identify the LOAEL selected as the TRV.

There was high variability in the toxicological data reviewed for the 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 resulted in increased uncertainty associated with the selected TRV.

Of the nine studies available for cadmium, five studies were conducted with salmonid species (Baldisserotto et al. 2005; Franklin et al. 2005; Lundebye et al. 1999; Mount et al. 1994; Szebedinsky et al. 2001). These studies provided species-specific information for juvenile Chinook salmon, which serves as an ROC representing all out-migrating juvenile salmonids. The only LOAEL reported in these studies was 1,395 mg/kg dw for the mortality of rainbow trout (Szebedinsky et al. 2001). 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) and ranged from 55 mg/kg dw for the growth of rainbow trout (Mount et al. 1994) to 786 mg/kg dw, also for the survival of rainbow trout (Szebedinsky et al. 2001). In comparison, the selected NOAEL and LOAEL TRVs are 0.1 mg/kg dw and 0.5 mg/kg dw, respectively, which are substantially lower than the values from the salmonid studies. These data suggest that the cadmium TRVs selected for juvenile Chinook salmon may result in an overestimate of risk for this ROC, as discussed further in the uncertainty analysis (Section A.6.2.1.2).

A.4.2.2.3 Chromium

Only one study that evaluated the effects of dietary chromium on fish (Table A.4-17) was available. There were no adverse effects on growth in this study, which exposed grey mullet to chromium in the diet, so only a NOAEL was determined. Because this is an unbounded NOAEL (i.e., no available LOAEL from the literature), there is no information to indicate the dietary concentration at which effects might occur. The NOAEL of 9.42 mg/kg dw as selected as the NOAEL TRV. The high uncertainty associated with the evaluation of risks to fish from dietary chromium because of the lack of an adverse effect level is discussed in the uncertainty analysis.

Table A.4-17. Fish toxicity studies reviewed for the selection of dietary TRVs for chromium

Chemical	Test Species	NOAEL (mg/kg dw)	LOAEL (mg/kg dw)	Exposure Duration	Effect	Source
Chromium (III)	grey mullet	**<u>9.42</u>**[a]	na	8 weeks	reduced growth	Walsh et al. (1994)

[a] Concentration is for elemental chromium.
dw – dry weight
LOAEL – lowest-observed-adverse-effect level
na – not available
NOAEL – no-observed-adverse-effect level
<u>Bold and underline</u> identify the NOAEL selected as the TRV.

A.4.2.2.4 Copper

Sixteen toxicity studies that exposed fish to dietary copper were evaluated for TRV selection (Table A.4-18). 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 the growth of channel catfish (Murai et al. 1981) to 2,397 mg/kg dw for the growth of grey mullet (Baker et al. 1998). NOAELs ranged from 8 mg/kg dw for the growth of channel catfish (Murai et al. 1981) to 1,368 mg/kg dw for growth, reproduction and mortality in zebrafish (Alsop et al. 2007).

Table A.4-18. Fish toxicity studies reviewed for the selection of dietary TRVs for copper

Chemical	Test Species	NOAEL (mg/kg dw)[a]	LOAEL (mg/kg dw)[a]	Exposure Duration	Effect	Source
Copper sulfate	channel catfish fingerling	8	16	16 weeks	reduced growth	Murai et al. (1981)
Copper sulfate pentahydrate	channel catfish fingerling	40	na	13 weeks	no effect on growth	Gatlin and Wilson (1986)
Copper sulfate	juvenile rockfish	**<u>50</u>**	**<u>100</u>**	60 days	reduced growth rate	Kang et al. (2005)
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)
Copper sulfate	Channel fingerlings	246	na	30 days	No effect on survival or growth	Erickson et al. (2010)
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)
Copper sulfate pentahydrate	juvenile rainbow trout	287	730	8 weeks	reduced growth	Lanno et al. (1985b)
Copper sulfate pentahydrate	juvenile rainbow trout	730	na	8 weeks	no effect on survival	Lanno et al. (1985b)
Copper sulfate pentahydrate	juvenile rainbow trout	na	796	16 weeks	reduced growth	Lanno et al. (1985a)
Copper chloride	rainbow trout fry	440	830[b]	60 days	reduced survival	Mount et al. (1994)
Copper sulfate	Atlantic salmon fry	638	868	3 months	reduced growth	Berntssen et al. (1999a)
Copper chloride	rainbow trout fry	1,000	na	60 days	no effect on body weight or length	Mount et al. (1994)
Copper sulfate pentahydrate	juvenile rainbow trout	1,042	na	28 days	no effect on survival or growth	Kamunde et al. (2001)
Copper sulfate pentahydrate	zebrafish	1,368	na	260 days	no effect on survival, growth, reproduction	Alsop et al. (2007)
Copper sulfate pentahydrate	juvenile grey mullet	na	2,397	67 days	reduced growth	Baker et al. (1998)

[a] Concentrations are for elemental copper.

[b] As reported in Mount et al. (1994), reduced survival at the LOAEL (830 mg/kg dw) was likely to the result of elevated copper concentrations in water rather than in the diet.

dw – dry weight
na – not available
LOAEL – lowest-observed-adverse-effect level
NOAEL – no-observed-adverse-effect level

<u>Bold and underline</u> identify the NOAEL and LOAEL selected as TRVs.

The lowest LOAEL reported in the reviewed literature was from 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 as compared with that of the control group, but a significant reduction in body weight was not observed in fish fed 8 mg/kg dw relative to the control group. 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; Erickson et al. 2010). 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. (2010) 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. The results of Erickson et al. (2010) 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. Other toxic effects on fish associated with exposure to copper include altered locomotion as a result of hyperactivity and interference with the ability of fish to respond positively to L-alanine, an important constituent of prey odors (Eisler 1997).

A.4.2.2.5 Vanadium

One study that evaluated the toxicological effects of dietary vanadium on fish was identified (Table A.4-19). In this study, 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; the ratio of food ingested to body weight gained was significantly greater for fish in this group than for fish in the control group. Therefore, the reduction in growth observed at this LOAEL was not attributed to reduced food intake. No NOAEL was reported by Hilton and Bettger (1988); therefore, a NOAEL was estimated from the chronic LOAEL using an uncertainty factor of 5. The resulting NOAEL of 2.04 mg/kg dw was selected as the NOAEL TRV. Because of the paucity of toxicological data on the dietary effects of vanadium to fish, there is uncertainty associated with the selected TRVs.

Table A.4-19. Fish toxicity studies reviewed for the selection of dietary TRVs for vanadium

Chemical	Test Species	NOAEL (mg/kg dw)[a]	LOAEL (mg/kg dw)[a]	Exposure Duration	Effect	Source
Sodium orthovanadate	juvenile rainbow trout	na	**<u>10.2</u>**	12 weeks	reduced body weight	Hilton and Bettger (1988)

[a] Concentrations are for elemental vanadium.
dw – dry weight
LOAEL – lowest-observed-adverse-effect level
na – not available
NOAEL – no-observed-adverse-effect level
<u>Bold and underline</u> identify the LOAEL selected as the TRV.

A.4.2.2.6 Benzo(a)pyrene

Benzo(a)pyrene was identified as a COPC for English sole and brown rockfish. Five studies that evaluated the toxicological effects of dietary benzo(a)pyrene on fish were found and determined to be acceptable for TRV derivation (Table A.4-20). Adverse effects on growth were reported in rockfish, rainbow trout, and English sole following exposure to dietary benzo(a)pyrene. No effects on survival were observed, and none of the studies evaluated reproductive effects. LOAELs ranged from 2.0 to 1,000 mg/kg dw (reduced growth effects), and NOAELs ranged from 1.5 to 1,000 mg/kg dw (no effects on growth or survival).

Table A.4-20. Fish toxicity studies reviewed for the selection of dietary TRVs for benzo(a)pyrene

Chemical	Test Species	NOAEL (mg/kg dw)	LOAEL (mg/kg dw)	Exposure Duration	Effect	Source
Benzo(a)pyrene	rockfish	**<u>1.5</u>**	**<u>2.0</u>**	30 days	reduced growth	Kim et al. (2008)
Benzo(a)pyrene	English sole	47	116	10 to 12 days	reduced growth	Rice et al.(2000)
Benzo(a)pyrene	areolated grouper	81	na	4 weeks	no effect on survival or growth	Wu et al. (2003)
Benzo(a)pyrene	rainbow trout	100	1,000	28 days	reduced growth	Hart and Heddle (1991)
Benzo(a)pyrene	rainbow trout	na	1,000	18 months	reduced growth	Hendricks et al. (1985)
Benzo(a)pyrene	rainbow trout	1,000	na	18 months	no effect on survival	Hendricks et al. (1985)

dw – dry weight
LOAEL – lowest-observed-adverse-effect level
na – not available
NOAEL – no-observed-adverse-effect level
TRV – toxicity reference value
<u>Bold and underline</u> identify the NOAEL and LOAEL selected as TRVs.

The lowest LOAEL reported in the reviewed literature was based on Kim et al. (2008). In this study, a significant decrease in growth rate compared with that of the control group was reported for juvenile rockfish (*Sebastes schlegeli*) exposed to 2.0 mg/kg dw of benzo(a)pyrene in a prepared diet for 30 days. A significant reduction in growth rate relative to controls was not observed in fish fed at the next lowest dose of 1.5 mg/kg dw. The NOAEL and LOAEL from this study (1.5 and 2.0 mg/kg dw, respectively) were selected as the TRVs for benzo(a)pyrene.

A.4.2.3 TRVs for surface water effects

This section describes the effects data used to evaluate the risk to fish from exposure to the three surface water COPCs (cadmium, mercury, and TBT).

A.4.2.3.1 Cadmium

For cadmium, the Washington State marine chronic WQC value is 9.3 µg/L, which is based on the dissolved fraction of cadmium (EPA 2001). The derivation of this value is described in detail in Section A.3.2.3.2. This criterion is based on invertebrate and fish data; the SMAVs used to develop the marine acute criterion were based on 33 invertebrate species and 5 fish species and was designed to protect 95% of the species in the aquatic community. The marine chronic WQC of 9.3 µg/L was selected as the TRV for fish for the surface water evaluation.

A.4.2.3.1 Mercury

The federal marine chronic WQC for mercury for the protection of aquatic life is 0.94 µg/L, which is based on the dissolved fraction in water (EPA 1985b). The derivation of this value is described in detail in Section A.3.2.3.2. The FCV was based on acute data for 25 invertebrate species and 7 fish species and acute-to-chronic ratios for two invertebrate species and two fish species. This marine chronic criterion of 0.94 µg/L was selected as the TRV for fish for the surface water evaluation.

A.4.2.3.1 TBT

The federal marine chronic WQC for TBT is 0.0074 µg/L (EPA 2003a), as described in Section A.3.2.3.2. The FCV was 0.066 µg/L based on acute data for 26 invertebrate species and 7 fish species and chronic data for three invertebrate species and one fish species. However, this value was not selected as the marine chronic WQC because additional data indicated that there were effects on dogwinkle, commercially important bivalves, and ecologically important copepods at lower concentrations. Thus, a value of 0.0074 µg/L was selected as the federal marine chronic criterion based on a NOEC for dogwinkle. However, this value is not directly applicable to fish because the fish test species demonstrated sensitivity at concentrations above 0.12µg/L.[29] Instead, the FCV, which includes data from fish species in its calculation, is more relevant for evaluating

[29] Chronic values were based on fish species mean acute values divided by saltwater ACR for TBT and included a chronic study with fathead minnows.

risks to fish. Because the federal WQC was set to a lower value to be protective of particularly sensitive invertebrate species and because fish species do not share these particular sensitivities, the FCV of 0.066 µg/L from the federal WQC for TBT was selected as the TRV for fish for the surface water evaluation instead of the marine chronic criterion.

A.4.3 Summary of Fish Exposure and Effects Assessment

A.4.3.1 Exposure assessment

The fish exposure assessment estimated the exposure of fish via three exposure approaches: tissue residue, dietary, and surface water. Exposure concentrations for these three types of evaluations were calculated as concentrations in whole-body tissue, diet, and surface water for each COPC identified for the three fish ROCs: juvenile Chinook salmon, English sole, and brown rockfish.

The tissue-residue evaluation was used for chemicals that bioaccumulate and persist in fish tissue. COPCs identified in the screening process for the tissue-residue evaluation were TBT and total PCBs for both English sole and brown rockfish. Mercury and beta-endosulfan were also identified as COPCs for brown rockfish. Whole-body tissue exposure concentrations for COPCs are presented in Tables A.4-2 and A.4-3.

The dietary evaluation was used for chemicals that are highly regulated (e.g., most metals) or metabolized (i.e., PAHs) by fish. COPCs identified in the screening process were arsenic, cadmium, chromium, copper, mercury, and vanadium for all three fish ROCs. Benzo(a)pyrene was also identified as a COPC for English sole and brown rockfish. Estimated dietary concentrations of COPCs are presented in Tables A.4-8 and A.4-9.

The surface water evaluation was used for chemicals that were identified as COPCs in surface water during the screening process (i.e., detected in surface water and exceeded the screening threshold; see Section 2.5.1.3). The COPCs identified for all three fish ROCs were cadmium, mercury, and TBT. The exposure concentrations of COPCs in water are presented in Tables A.4-10 and A.4-11.

A.4.3.2 Effects assessment

The effects assessment presented TRVs selected for the tissue-residue, dietary and water evaluations. The TRVs for the tissue-residue and dietary evaluations were based on a search of the toxicity literature and selection of the most appropriate NOAELs and LOAELs based on a set of guidelines for TRV selection. The TRVs used for the surface water evaluation were the Washington State marine chronic WQC for cadmium, the federal marine chronic WQC for mercury, and the FCV for TBT. A summary of the TRVs selected for the three fish ROCs is presented in Tables A.4-21 and A.4-22.

Table A.4-21. TRVs selected for the tissue-residue and dietary evaluations

COPC	TRVs (mg/kg)[a]		ROCs Evaluated		
	NOAEL	LOAEL	Juvenile Chinook Salmon	English Sole	Brown Rockfish
Tissue Residue					
Mercury	0.23	0.47			X
TBT	0.018	0.159		X	X
Total PCBs	0.10 – 0.53	0.520 – 2.64		X	X
beta-endosulfan	0.0031	0.031			X
Dietary					
Arsenic	20	30	X	X	X
Cadmium	0.1	0.5	X	X	X
Chromium	9.42	na	X	X	X
Copper	50	100	X	X	X
Vanadium	2.04	10.2	X	X	X
Benzo(a)pyrene	1.5	2.0			X

[a] Tissue-residue concentrations are wet weight; dietary concentrations are dry weight.

COPC – chemical of potential concern
LOAEL – lowest-observed-adverse-effect level
na – not available
NOAEL – no-observed-adverse-effect level

PCB – polychlorinated biphenyl
ROC – receptor of concern
TBT – tributyltin
TRV – toxicity reference value

Table A.4-22. TRVs for the surface water evaluation

COPC	TRV (µg/L)	ROCs Evaluated		
		Juvenile Chinook Salmon	English Sole	Brown Rockfish
Cadmium	9.3	X	X	X
Mercury	0.94	X	X	X
TBT	0.066	X	X	X

COPC – chemical of potential concern
ROC – receptor of concern
TBT – tributyltin
TRV – toxicity reference value

A.5 Exposure and Effects Assessment: Wildlife

This section presents exposure and effects assessment for aquatic-dependent wildlife potentially exposed to COPCs in EW. The exposure assessment (Section A.5.1) presents the methods and data used to estimate exposure, including the calculated dietary doses for each of the wildlife ROCs. The effects assessment (Section A.5.2) presents the TRVs used for risk characterization for wildlife.

A.5.1 Exposure Assessment

This exposure assessment presents the methods and results for calculating exposure doses of COPCs to the aquatic-dependent wildlife ROCs in the EW. The general methods used for evaluating risk to wildlife are the same as those used for the LDW ERA (Windward 2007c). COPCs selected for wildlife as a result of the COPC screen conducted in Section A.2.5.3 are presented in Table A.5-1.

Table A.5-1. COPCs for birds and mammals

COPC	COPCs Identified for Birds and Mammals			
	Pigeon Guillemot	Osprey	River Otter	Harbor Seal
Mercury	X		X	X
Selenium			X	
Total PCBs	X	X	X	X
PCB TEQ	X		X	X
Total TEQ	X		X	X

COPC – chemical of potential concern
PCB – polychlorinated biphenyl
TEQ – toxic equivalent

A.5.1.1 Methods

In this assessment, estimated doses of each chemical for each ROC are calculated for ingestion of prey, surface water, and sediment. Other exposure pathways considered in the conceptual site model in the problem formulation were determined to be insignificant relative to these primary exposure pathways.[30] Chemical doses were estimated using the following equation:

[30] Direct (or dermal) contact with sediment and direct contact with surface water were considered complete exposure pathways but were assumed to be insignificant because feathers on birds and fur on mammals limit direct contact of skin with contaminated media. Risks from sediment contact are considered to be insignificant relative to those from ingestion (EPA 2000b).

$$\text{Dose} = \frac{\left[\left(IR_{prey} \times EPC_{prey}\right) + \left(IR_{water} \times EPC_{water}\right) + \left(IR_{sed} \times EPC_{sed}\right)\right] \times SUF}{BW}$$ **Equation 5-1**

Where:

Dose	=	chemical dose via prey, water, and sediment (mg/kg bw/day)
IR_{prey}	=	prey ingestion rate (kg prey ww/day)
EPC_{prey}	=	exposure point concentration in prey items (mg/kg prey ww)
IR_{water}	=	water ingestion rate (L/day)
EPC_{water}	=	chemical concentration in water (mg/L)
IR_{sed}	=	sediment ingestion rate (kg dw/day)
EPC_{sed}	=	concentration in sediment (mg/kg dw)
SUF	=	site use factor (unitless); fraction of time that a receptor spends foraging in the EW relative to the entire home range
BW	=	ROC species body weight (kg ww)

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. All COPCs were assumed to have the same bioavailability in the field as in the laboratory toxicity study that provided the basis for the TRV in all media (i.e., bioavailability factors were not used in the dose calculations). To calculate the EPC_{prey} for total prey ingestion, the fraction of each prey item consumed by a receptor was multiplied by the concentration in that prey item as follows:

$$EPC_{prey} = (EPC_1 \times F_1) + (EPC_2 \times F_2)$$ **Equation 5-2**

Where:

EPC_{prey}	=	exposure point concentration in food (mg/kg ww)
$EPC_{1,2}$	=	exposure point concentration in each prey item (mg/kg ww)
$F_{1,2}$	=	fraction of the diet consisting of each prey item

The 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: exposure to total PCBs (typically an Aroclor mixture), which was evaluated based on the toxicity of PCBs, and exposure to dioxin-like PCB congeners. Dioxin-like PCB congeners have structural and toxicological similarities to dioxins and furans. The potency of each individual dioxin-like PCB congener relative to that of 2,3,7,8-TCDD was quantified by calculating TEQs, as described in Section A.2.4.2.4. A total TEQ was also calculated for the sum of dioxin-like PCB congeners and dioxin/furan congeners.

The TEFs used to calculate the TEQs in this ERA were developed by the World Health Organization (WHO) in 1998 (Van den Berg et al. 1998) from a database containing all available mammalian, bird, and fish studies involving the relative toxicity of dioxin-like compounds; mammal values were updated in 2005 (Van den Berg et al. 2006). The TEFs are presented in Table A.2-14 in Section A.2.4.2.4.

The PCB TEQ accounts for the toxicity of a subset of PCB congeners that have dioxin-like modes of toxic action. Other PCB congeners may produce toxic effects with different toxic mechanisms. Therefore, the PCB TEQ approach is not used as a surrogate for the total PCB approach, which captures all PCB toxicity mechanisms and modes of action. Total TEQ accounts for the cumulative exposure of wildlife to both dioxins and furans and dioxin-like PCB congeners in the EW.

A.5.1.2 Exposure assumptions

This section presents the exposure parameters used in Equation 5-1 and the dietary fractions used in Equation 5-2 to calculate the exposure dose for each ROC. Table A.5-2 presents the ingestion rates for prey, surface water, and surface sediment; site use factors; and body weights for each ROC. Average of male and female body weights were used to calculate exposure doses. Table A.5-3 presents the percentages of prey items in the diet of each ROC. The following subsections provide details regarding the assumptions and sources of information for the values presented in Tables A.5-2 and A.5-3.

Table A.5-2. Exposure factor values for wildlife ROCs

ROC	Body Weight (kg ww)[a]	Prey Ingestion Rate (kg/day ww)	Incidental Water Ingestion Rate (L/day)	Incidental Sediment Ingestion Rate (kg/day dw)	Site Use Factor (unitless)
Pigeon guillemot	0.485	0.097	0.036	0.0004	0.5
Osprey	1.7	0.36	0.084	0.00094	0.41
River otter	8.6	1.1	0.69	0.0046	0.23
Harbor seal	80.6	2.5	5.1	0.012	0.1

[a] Average of male and female body weights.

dw – dry weight

ROC – receptor of concern

ww – wet weight

Table A.5-3. Percentages of prey items in ROC diets used in wildlife exposure calculations

ROC	Percentage of Prey Item in Diet (by wet weight)[a]							
	Shiner Surfperch	English Sole	Juvenile Chinook Salmon	Brown Rockfish	Crab	Shrimp	Clam	Mussel
Pigeon guillemot	14.3 (25)	14.3 (25)	14.3 (0)	14.3 (25)	14.3 (25)	14.3 (0)	0	14.3 (0)
Osprey	50 (100)	0	50 (0)	0	0	0	0	0
River otter	22 (29.3)	22 (29.3)	22 (0)	22 (29.3)	10 (10)	0	1 (2)	1 (0)
Harbor seal	25(33.3)	25 (3.33)	25 (0)	25 (33.3)	0	0	0	0

a Values in parentheses were used for exposure calculations for PCB TEQ and total TEQ because juvenile Chinook salmon, shrimp, and mussels were not analyzed for PCB congeners. Percentages were determined as described in text.

PCB – polychlorinated biphenyl

ROC – receptor of concern

TEQ – toxic equivalent

A.5.1.2.1 Pigeon guillemot

Body Weight

Female pigeon guillemots in California that were weighed in April were slightly heavier than males (average of 0.487 and 0.483 kg, respectively), but females in Alaska that were weighed during egg laying (May to June) were substantially heavier than males (0.530 and 0.462 kg, respectively) (Ewins 1993). The average of the male and female pigeon guillemots weighed in April prior to breeding (0.485 kg) were used as the body weight in dose calculations for pigeon guillemot.

Prey Ingestion Rate

The prey ingestion rate for pigeon guillemot was estimated as 20% of the body weight (Ewins 1993). An ingestion rate of 0.097 kg ww/day was calculated based on the body weight of 0.485 kg (Table A.5-2).

Water Ingestion Rate

The water ingestion rate was estimated as a function of the pigeon guillemot's body weight, using an allometric equation recommended in *Wildlife Exposure Factors Handbook* (EPA 1993), as follows:

$$IR_{water} = 0.059 \times BW^{0.67}$$ **Equation 5-3**

Where:

IR_{water} = water ingestion rate (L/day)

BW = body weight (kg)

Based on the average male and female body weight of 485 g used in Equation 5-3, the water ingestion rate for pigeon guillemot was calculated as 0.036 L/day.

Incidental Sediment Ingestion Rate

Information on rates of incidental sediment ingestion by pigeon guillemot was not available. Pigeon guillemot may ingest a small amount of sediment while foraging for prey. However, the sediment ingestion rate is likely to be low because pigeon guillemot prey primarily on benthic fish and epibenthic invertebrates rather than on infaunal invertebrates. Thus, sediment ingestion was estimated to be 2% of the pigeon guillemot's prey ingestion rate. The sediment ingestion rate was calculated as 0.0004 kg/day on a dry weight basis, using the average moisture content of 79% measured in pigeon guillemot prey species collected from the EW.

Site Use

Pigeon guillemot are present in the Puget Sound region year-round (Seattle Audubon Society 2008), and their nests have been observed under the T-18 piers (Hotchkiss 2007). Limited data are available to estimate the foraging range of pigeon guillemot. Litzow and Piatt (2003) observed that radio-tagged pigeon guillemot foraged only in the area in which they nested, although the size of that area was not defined. Data summarized by Ewins (1993) indicate that most breeders feed within 7 km of their nests. Based on this limited information, it is assumed that pigeon guillemot nesting along the EW obtain approximately half of their diet from within the EW. Thus, the estimated site use factor for pigeon guillemot is 0.5.

Composition of Diet

Pigeon guillemot are known to feed on bottom-dwelling organisms in water up to 45 m deep (Ewins 1993). It is estimated that the optimal diving and foraging efficiency of pigeon guillemots is in water 10 to 20 m deep (Ewins 1993). Most foraging is conducted in benthic habitats, although pigeon guillemot also feed in the water column (Ewins 1993). Because no data have been found to indicate that pigeon guillemot do not also feed in shallower water, both subtidal and intertidal areas of the EW were considered to be foraging habitat.

Pigeon guillemot are "generalists" and have been known to feed on over 50 species of benthic fish and invertebrates (Kuletz 1998). Fish and invertebrates in the diets of pigeon guillemot from Alaska, Oregon, and British Columbia include sandeels, Pacific sandfish, capelin, cod, sculpin, gunnel, blennies, gadids, Pacific sand lance, prickleback, flatfish, Pacific herring, crab, shrimp, with occasional polychaetes, gastropods, and bivalve mollusks (Ewins 1993; Golet et al. 2000; Kuletz 1998; Litzow et al. 2000). There may be considerable variation in diet as a function of habitat and time of year, as well as variation among individuals and pairs (Ewins 1993; Kuletz 1998; Litzow et al. 2000). Because of the wide variety of species that may be consumed, it is assumed that pigeon guillemot that forage in the EW consume equal proportions of shiner surfperch, English sole, juvenile Chinook salmon, brown rockfish, crab, mussels, and shrimp.

A.5.1.2.2 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). The average of these male and female body weights (1.7 kg) was used in the dose calculations for osprey.

Prey Ingestion Rate

The prey ingestion rate for osprey was estimated to be 21% of the body weight (Poole 1983; as cited in EPA 1993). An ingestion rate of 0.36 kg ww/day was calculated based on the average male and female body weight of 1.7 kg.

Water Ingestion Rate

The water ingestion rate was estimated as a function of the osprey's body weight using the allometric equation recommended in *Wildlife Exposure Factors Handbook* (EPA 1993). This is the same equation as that used for pigeon guillemot (Equation 5-3). Based on the body weight of 1.7 kg and Equation 5-3, the water ingestion rate for osprey was calculated to be 0.084 L/day.

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 intertidal water. Because osprey catch fish from only about the top 1 m of the water surface (Poole and Spitzer 1983) they are not expected to contact sediment in deeper subtidal areas. Thus, sediment ingestion was estimated to be 1% of the osprey's prey ingestion rate, and it was assumed that only intertidal sediment is ingested. The sediment ingestion rate was calculated as 0.00094 kg/day on a dry-weight basis, using the average moisture content of 74% measured in osprey prey species collected from the EW.

Site Use

Two osprey nest boxes are located along the EW: one at T-104 and one at T-18 (Blomberg 2007). In 2006, the WDFW reported that 10 osprey nest sites were located along the Duwamish River, which included the two nests along the EW (Thompson 2006). The distance osprey travel from their nests to forage depends on the availability of fish near the nest (Van Daele and Van Daele 1982). Preliminary 2006 USGS data on the foraging locations of osprey from the two EW nests are available (Davis 2007). Osprey from the nest at T-18 caught prey fish in Lake Washington, the lower Duwamish River, and Elliott Bay (76.9, 15.4, and 7.7% of the total numbers of prey items were captured from each area, respectively); osprey from the nest at T-104 caught prey fish in the lower Duwamish River and Puget Sound (66.7 and 33.3%, respectively). Marine species identified included salmonids and surfperch. To calculate the site use factor, it was assumed that all lower Duwamish River fish for both nests were captured from the EW. The average percentage of EW fish assumed to be captured for both nests was 41%, which is the average of the percentages for the lower Duwamish River from the T-18 and the Terminal 101 nests; thus, the site use factor for osprey was 0.41.

Composition of Diet

Osprey feed almost exclusively on live fish; at least 99% of their prey items are 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 (Poole et al. 2002). A west-central Idaho osprey study reported that 89% of the 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 in 2006, osprey were observed returning with prey to two nests along the EW, one nest at T-18 and one at T-104 (Davis 2007). The average diets of osprey returning to the T-18 nest, based on numbers of fish, consisted of 15.4% salmonids, 76.9% freshwater fish, and 7.7% unknown fish; and the average diet of osprey frequenting the T-104 nest consisted of 50% surfperch and 50% salmonids. Based on these data, it was assumed that the osprey diet consists of the maximum percentage for fish species known to be present in the EW and documented at both nests, resulting in 50% surfperch and 50% salmonids.

A.5.1.2.3 River otter

Body Weight

Adult body weights of 9.2 and 7.9 kg were assumed for male and female river otters, respectively, based on a study by Melquist and Hornocker (1983), as cited in EPA (1993). The average of the male and female body weights (8.6 kg) was used in the dose calculations for river otter.

Prey Ingestion Rate

The prey ingestion rate for river otter was estimated as a function of the metabolic rate and the caloric content of the prey using the following equation (EPA 1993):

$$IR_{prey} = \frac{FMR}{ME} \times \frac{0.001\ \text{kg food}}{\text{g food}}$$ **Equation 5-4**

Where:

IR_{prey} = prey ingestion rate (kg/day ww)
FMR = free-living metabolic rate (kilocalories [kcal]/day)
ME = average metabolizable energy of the total diet (kcal/g ww)

The free-living metabolic rate (FMR) for river otter was calculated to be 1,180 kcal/day, using an equation developed by Nagy (1987), as cited in EPA (1993), for placental mammals:

$$FMR\ (\text{kcal/day}) = 0.800 \times BW^{0.813}$$ **Equation 5-5**

where body weight is expressed in grams. The value used for the average metabolizable energy (ME) of the total diet was the value presented by EPA (1993) for mink on a diet of fish at 1.1 kcal/g ww. The prey ingestion rate for river otters was calculated to be 1.1 kg ww/day based on Equations 5-4 and 5-5.

Water Ingestion Rate

The water ingestion rate 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):

$$IR_{water} = 0.099 \times BW^{0.90}$$ **Equation 5-6**

Where:

IR_{water} = water ingestion rate (L water/day)

BW = body weight (kg)

Based on an average male and female body weight of 8.6 kg and Equation 5-6, the water ingestion rate for river otters was calculated as 0.69 L/day.

Incidental Sediment Ingestion Rate

Data were not available to estimate the amount of sediment ingested incidentally by river otters. A small amount of sediment might 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 prey ingestion rate. It was assumed that river otters incidentally ingest sediment from both intertidal and subtidal areas of the EW. The sediment ingestion rate was calculated as 0.0046 kg/day on a dry-weight basis using the average moisture content of 79% measured in river otter prey species collected from the EW.

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 of the lower Duwamish River 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 (Spinola et al. 1999; as cited in EPA 2000c). This study showed that river otter ranges were from 1.5 to 22.4 km, with an average range of 10 km (6 mi) for individuals monitored in western New York State.

No studies that document the use of the EW by river otters were found. However, it was assumed that any river otters that may potentially inhabit Kellogg Island have a home range of 10 km, as documented by Spinola et al. (1999; as cited in EPA 2000c). Assuming river otters forage in the areas 5 km north and south of Kellogg Island equally and because the EW is approximately 2.3 km long (23% of 10 km), the site use factor for river otters was 0.23.

Composition of Diet

River otters are opportunistic carnivores that take advantage of prey that is most abundant and easiest to catch. Fish are their primary prey (Kurta 1995; Larsen 1984; Stenson et al. 1984; Wise et al. 1981). River otters catch fish by diving and ambushing or chasing, and obtain invertebrates by digging in the substrate (Coulter et al. 1984). In general, slower-moving fish, such as suckers, carp, chubs, and bullheads, are eaten most frequently (Kurta 1995; Wise et al. 1981). Studies conducted 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 that range from 7.6 to 41 cm in length (Gilbert and Nancekivell 1982; Greer 1955; both as cited in EPA 1993), although Towel (1974) found that many of the salmon preyed upon by river otters in western Oregon were estimated to be as large as 80 cm in length. These salmon were caught 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 the types of prey 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% crab, 2% invertebrates other than crab, 1% birds, and 1% mammals and plant material. Thus, for this assessment, it was assumed that river otters ingest 88% fish, 10% crabs, and 1% each of mussels and clams. Based on the 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 EW might be ingested. Because no site-specific information was available on the fish preference of river otters, it was assumed that shiner surfperch, English sole, juvenile Chinook salmon, and brown rockfish are ingested in equal proportions for the 88% of the river otter diet that is fish.

A.5.1.2.4 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). The average of the male and female body weights (80.6 kg) was used to calculate the exposure dose for harbor seal.

Prey Ingestion Rate

The prey ingestion rate for harbor seal 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):

$$IR_{prey} = 0.089 \times BW^{0.76}$$ **Equation 5-7**

Where:

IR_{prey} =prey ingestion rate (kg/day ww)
BW = body weight (kg)

Based on an average male and female body weight of 80.6 kg and Equation 5-7, the prey ingestion rate was calculated as 2.5 kg/day.

Water Ingestion Rate

The water ingestion rate was estimated as a function of the harbor seal's body weight, using the allometric equation recommended in EPA (1993), which is presented above for the river otter in Equation 5-6. Based on a body weight of 80.6 kg and Equation 5-6, the water ingestion rate for harbor seal was calculated as 5.1 L/day.

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 by harbor seals while foraging on bottom fish. Therefore, the sediment ingestion rate was assumed to be 2% of the prey ingestion rate. It was also assumed that harbor seal ingest sediment from both intertidal and subtidal areas of the EW. The sediment ingestion rate was calculated as 0.012 kg/day on a dry-weight basis using the average moisture content of 76% measured in harbor seal prey species collected from the EW.

Site Use Factor

Harbor seals are commonly seen in Elliott Bay and occasionally enter the EW (Kenney 1982). Harbor seals have been known 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 EW is located at Blakely Rocks off the southeast end of Bainbridge Island, approximately 12 km (7 mi) from the EW. Site-specific information on harbor seal use of the EW is limited. The

WDFW observed harbor seals infrequently in the EW during an intensive survey conducted from December 1998 to June 1999, which monitored the EW, West Waterway, and LDW up to the 16th Avenue South Bridge for the presence of sea lions and seals for a total of 307 hours on 52 separate days (WDFW 1999). The EW was monitored for a total of 28.25 hours on 29 separate days; one harbor seal was observed during this time. 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 EW 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 salmonids are abundant and available (NMFS 1997).

Data from the WDFW survey (1999) were used to establish a site use factor for risk calculations. Based on the one harbor seal observed during the survey, the following assumptions were used to develop a site use factor for seals in the EW: 1) the seal obtained all of its prey from the EW on the one day that it was observed in the EW during 29 days of monitoring; and 2) site use during the 29 days of monitoring from December through June accurately represents use during other times of the year. Based on these assumptions, the site use factor is equal to 1/29 or 0.03. For the purpose of the ERA, a site use factor of 0.1 was used for harbor seal in order to account for potentially higher use of the site by harbor seals during periods of salmon migration.

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). The lengths of fish ingested were generally between 4 and 28 cm (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 on the amount of each type of fish ingested was not available, it was assumed that juvenile Chinook salmon, English sole, shiner surfperch, and brown rockfish are ingested in equal proportions.

A.5.1.3 EPCs and dietary doses

This section presents the exposure point concentrations (EPCs) for prey tissue, surface sediment, and surface water that were used in Equations 5-1 and 5-2 to calculate dietary doses. The resulting dietary doses are also presented.

A.5.1.3.1 Prey tissue EPCs

Tissue data were available for eight species that are potential prey of wildlife ROCs in the EW: shiner surfperch, English sole, brown rockfish, juvenile Chinook salmon, crab, shrimp, clams, and mussels. These data are described in Section A.2.4.1.2. Whole-body tissue data were used in the ERA for fish, shrimp, and crab tissue;[31] and soft tissue data were used for clam and mussel tissue.

For all wildlife ROCs, it was assumed that the foraging area includes the entire EW based on home range information presented in Section A.5.1.2. Thus, for each prey type ingested by ROC species, the exposure concentration was calculated using tissue data from throughout the EW as a single dataset. The exposure concentration for each COPC in each tissue type was estimated as the 95% UCL using ProUCL (Version 4.0). The ProUCL software used for this analysis allows detected and undetected values to be indicated and creates interpolated values for non-detects based on the perceived distribution of the detected concentrations. Once any necessary interpolation has been performed, the software conducts an analysis of the data to determine the most appropriate UCL and makes a recommendation. The maximum concentration was used as the EPC if fewer than six samples were available, based on ProUCL guidance. These EPCs are shown in Table A.5-4.

[31] Crab edible meat and hepatopancreas samples were analyzed separately; crab shells were not included. Whole-body concentrations were estimated from these separate parts using relative weights and concentrations of the edible meat and hepatopancreas (Attachment 1).

Table A.5-4. EPCs in tissues of prey species ingested by pigeon guillemot, osprey, river otter, and harbor seal

COPC	Detection Frequency	Mean (mg/kg ww)[a]	Maximum Detection (mg/kg ww)	Maximum RL (mg/kg ww)	Tissue EPC (mg/kg ww)	Statistic Used
Mercury						
Shiner surfperch	11/11	0.04	0.05	na	0.042	95% Student's-t UCL
English sole	13/13	0.03	0.042	na	0.036	95% Student's-t UCL
Juvenile Chinook salmon	12/12	0.022	0.043 J	na	0.027	95% Student's-t UCL
Brown rockfish	15/15	0.2	0.418	na	0.21	95% Approximate Gamma UCL
Crab	9/9	0.05	0.12	na	0.069	95% Approximate Gamma UCL
Clam	10/10	0.02	0.03	na	0.021	95% Student's-t UCL
Mussel	7/17	0.008	0.015	0.01 U	0.012	95% KM (Percentile Bootstrap) UCL
Shrimp	1/1	na	0.03	na	0.03	maximum detect
Selenium						
Shiner surfperch	8/8	0.4	0.6 J	na	0.51	95% Modified-t UCL
English sole	11/11	0.6	0.68	na	0.62	95% Student's-t UCL
Juvenile Chinook salmon	6/6	0.36	0.37	na	0.36	95% KM (BCA) UCL
Brown rockfish	13/13	0.66	0.85	na	0.72	95% Student's-t UCL
Crab	9/9	1	1.36	na	1.2	95% Modified-t UCL
Clam	10/10	0.37	0.52	na	0.41	95% Student's-t UCL
Mussel	11/11	0.49	0.60	na	0.54	95% Student's-t UCL
Shrimp	1/1	na	0.5	na	0.5	maximum detect
Total PCBs						
Shiner surfperch	11/11	1.50	5.40	na	2.3	95% H-UCL
English sole	13/13	3.20	7.90 J	na	4.1	95% Approximate Gamma UCL
Juvenile Chinook salmon	12/12	0.059	0.0915	na	0.072	95% Student's-t UCL
Brown rockfish	15/15	2.00	6.20	na	4.0	95% H-UCL
Crab	9/9	0.300	0.860	na	0.45	95% Modified-t UCL
Clam	11/11	0.056	0.082	na	0.069	95% Student's-t UCL
Mussel	14/17	0.026	0.044 J	0.013 U	0.031	95% KM (Percentile Bootstrap) UCL

Table A.5-4. EPCs in tissues of prey species ingested by pigeon guillemot, osprey, river otter, and harbor seal (cont.)

COPC	Detection Frequency	Mean (mg/kg ww)[a]	Maximum Detection (mg/kg ww)	Maximum RL (mg/kg ww)	Tissue EPC (mg/kg ww)	Statistic Used
Shrimp	1/1	na	0.46	na	0.46	maximum detect
PCB TEQ (birds)[b]						
Shiner surfperch	3/3	3.67×10^{-5}	3.86×10^{-5}	na	3.86×10^{-5}	maximum detect
English sole	3/3	3.89×10^{-5}	4.26×10^{-5} J	na	4.26×10^{-5}	maximum detect
Juvenile Chinook salmon[c]	nd	nd	nd	nd	nd	nd
Brown rockfish	6/6	4.91×10^{-5}	9.36×10^{-5} J	na	7.25×10^{-5}	95% Student's-t UCL
Crab	3/3	1.17×10^{-5}	1.25×10^{-5}	na	1.25×10^{-5}	maximum detect
Clam	3/3	3.67×10^{-6}	6.39×10^{-6}	na	6.39×10^{-6}	maximum detect
Mussel[d]	nd	nd	nd	nd	na	nd
Shrimp[c]	nd	nd	nd	nd	nd	nd
PCB TEQ (mammals)[e]						
Shiner surfperch	3/3	1.31×10^{-5}	1.43×10^{-5}	na	1.43×10^{-5}	maximum detect
English sole	3/3	3.5×10^{-5}	3.74×10^{-5} J	na	3.74×10^{-5}	maximum detect
Juvenile Chinook salmon[c]	nd	nd	nd	nd	na	nd
Brown rockfish	6/6	2.48×10^{-5}	5.95×10^{-5} J	na	4.01×10^{-5}	95% Student's-t UCL
Crab	3/3	4.83×10^{-6}	5.61×10^{-6}	na	5.61×10^{-6}	maximum detect
Clam	3/3	4.06×10^{-7}	7.34×10^{-7}	na	7.34×10^{-7}	maximum detect
Mussel[d]	nd	nd	nd	nd	nd	nd
Shrimp[c]	nd	nd	nd	nd	nd	nd
Total TEQ (birds)[b]						
Shiner surfperch	3/3	4.16×10^{-5}	4.35×10^{-5} J	na	4.35×10^{-5}	maximum detect
English Sole	3/3	4.32×10^{-5}	4.66×10^{-5} J	na	4.66×10^{-5}	maximum detect
Juvenile Chinook salmon[c]	nd	nd	nd	nd	nd	nd
Brown rockfish	6/6	5.55×10^{-5}	9.83×10^{-5} J	na	7.90×10^{-5}	95% Student's-t UCL
Crab	3/3	1.59×10^{-5}	1.66×10^{-5} J	na	1.66×10^{-5}	maximum detect
Clam	3/3	4.12×10^{-6}	7.04×10^{-6} J	na	7.04×10^{-6}	maximum detect

Table A.5-4. EPCs in tissues of prey species ingested by pigeon guillemot, osprey, river otter, and harbor seal (cont.)

COPC	Detection Frequency	Mean (mg/kg ww)[a]	Maximum Detection (mg/kg ww)	Maximum RL (mg/kg ww)	Tissue EPC (mg/kg ww)	Statistic Used
Mussel[d]	nd	nd	nd	nd	nd	nd
Shrimp[c]	nd	nd	nd	nd	nd	nd
Total TEQ (mammals)[e]						
Shiner surfperch	3/3	1.43×10^{-5}	1.56×10^{-5} J	na	1.56×10^{-5}	maximum detect
English sole	3/3	3.68×10^{-5}	3.90×10^{-5} J	na	3.90×10^{-5}	maximum detect
Juvenile Chinook salmon[c]	nd	nd	nd	nd	nd	nd
Brown rockfish	6/6	2.69×10^{-5}	6.18×10^{-5}J	na	4.25×10^{-5}	95% Student's-t UCL
Crab	3/3	6.03×10^{-6}	6.80×10^{-6} J	na	6.80×10^{-6}	maximum detect
Clam	3/3	6.88×10^{-7}	1.11×10^{-6} J	na	1.11×10^{-6}	maximum detect
Mussel[d]	nd	nd	nd	nd	nd	nd
Shrimp[c]	nd	nd	nd	nd	nd	nd

a The mean is calculated as the average of the detected concentrations and one-half the RL for non-detected results.

b PCB TEQs and total TEQs were calculated using TEFs for birds presented in Van den Berg et al. (1998).These TEFs are listed in Table A.2-14, and uncertainties associated with application of these TEFs are discussed in Section A.6.3.1.

c PCB congeners and dioxin/furan congeners were not analyzed in juvenile Chinook salmon or shrimp tissue because of limited sample mass.

d PCB congeners and dioxin/furan congeners were not analyzed in mussels because mussel tissue is a minor component of seafood ingestion for humans and otters and total PCB concentrations were relatively low in mussel tissues.

e PCB TEQs and total TEQs were calculated using TEFs for mammals presented in Van den Berg et al. (2006). These TEFs are listed in Table A.2-14, and uncertainties associated with application of these TEFs are discussed in Section A.6.3.1.

COPC – chemical of potential concern

EPC – exposure point concentration

na – not applicable (e.g., if there was only one sample, or if all results were detects)

nd – no data

PCB – polychlorinated biphenyl

TEF – toxic equivalency factor

TEQ – toxic equivalent

UCL – upper confidence limit on the mean

ww – wet weight

A.5.1.3.2 Sediment EPCs

Surface sediment data were used to estimate the COPC exposure resulting from incidental sediment ingestion. Pigeon guillemot, river otter, and harbor seal may forage throughout the water column in the EW and may incidentally ingest sediment in both shallow and deep areas. Osprey feed from the top 1 m of the water column, which would result in the potential incidental ingestion of sediment only in shallower intertidal areas. Most data in the EW surface sediment database were collected from subtidal areas as grab samples, whereas only three composite samples were collected in intertidal areas using the MIS method. Because of the different sampling methods for the intertidal and subtidal samples, the site-wide EPC was calculated as a weighted average of the intertidal and subtidal EPCs based on the percentage of EW area represented by the intertidal samples (2.7%) and the subtidal samples (97.3%). EPCs for subtidal areas were calculated as the UCL of all subtidal data using ProUCL, as described in Section 5.3.1.3.1 for prey tissue. EPCs for intertidal areas were equivalent to the maximum concentration because there were fewer than the recommended number of samples (i.e., at least six) for calculating a UCL. The site-wide EPCs were used for exposure of pigeon guillemot, river otter, and harbor seal, and intertidal UCLs were used for osprey. The calculated concentrations in surface sediment are presented in Table A.5-5.

Table A.5-5. EPCs in EW surface sediment used to estimate the exposure of wildlife ROCs

COPC	Exposure Area	Detection Frequency	Mean (mg/kg dw)	Maximum Detection (mg/kg dw)	Maximum RL (mg/kg dw)	EPC (mg/kg dw)	Statistic Used
Mercury	intertidal	3/3	0.08	0.10	na	0.10	maximum detect
	subtidal	223/229	0.3	1.07 J	0.070 U	0.30	95% KM (BCA) UCL
	site-wide	226/232	0.3[a]	1.07 J	0.070 U	0.29	weighted site-wide value[b]
Selenium	intertidal	0/3	0.3	na	0.6 U	0.3	one-half maximum RL
	subtidal	0/108	0.4	na	1 U	0.50	one-half maximum RL
	site-wide	0/111	0.4[a]	na	1 U	0.49	weighted site-wide value[b]
Total PCBs	intertidal	3/3	0.97	1.59	na	1.6	maximum detect
	subtidal	217/230	0.49	8.40	0.035 U	0.72	95% KM (Chebyshev) UCL
	site-wide	220/233	0.50[a]	8.40	0.035 U	0.74	weighted site-wide value[b]
PCB TEQ (bird)[b]	intertidal	3/3	5.86×10^{-5}	1.04×10^{-4} J	na	1.04×10^{-4}	maximum detect
	subtidal	13/13	3.31×10^{-5}	6.07×10^{-5} J	na	4.11×10^{-5}	95% Student's-t UCL
	site-wide	16/16	3.38×10^{-5} [a]	1.04×10^{-4} J	na	4.28×10^{-5}	weighted site-wide value[b]
PCB TEQ (mammal)[c]	intertidal	3/3	4.49×10^{-6}	6.31×10^{-6} J	na	6.31×10^{-6}	maximum detect
	subtidal	13/13	4.37×10^{-6}	9.5×10^{-6} J	na	5.62×10^{-6}	95% Student's-t UCL
	site-wide	16/16	4.37×10^{-6} [a]	9.5×10^{-6} J	na	5.64×10^{-6}	weighted site-wide value[b]
Total TEQ (bird)[b]	intertidal	3/3	7.30×10^{-5}	1.13×10^{-4} J	na	1.13×10^{-4}	maximum detect
	subtidal	13/13	5.39×10^{-5}	1.01×10^{-4} J	na	6.62×10^{-5}	95% Student's-t UCL
	site-wide	16/16	5.44×10^{-5} [a]	1.13×10^{-4} J	na	6.75×10^{-5}	weighted site-wide value[b]
Total TEQ (mammal)[c]	intertidal	3/3	1.65×10^{-5}	1.76×10^{-5} J	na	1.76×10^{-5}	maximum detect
	subtidal	13/13	2.01×10^{-5}	4.01×10^{-5} J	na	2.47×10^{-5}	95% Student's-t UCL
	site-wide	16/16	2.00×10^{-5} [a]	4.01×10^{-5} J	na	2.45×10^{-5}	weighted site-wide value[b]

Note: Site-wide EPCs were used for the exposure of pigeon guillemot, river otter, and harbor seal; intertidal EPCs were used for exposure of osprey as described in the text. Data for subtidal areas are presented for informational purposes.

[a] The mean is calculated as the average of the detected concentrations and one-half the RL for non-detected results. Mean concentrations for the site-wide area are the weighted average of the intertidal mean (2.7% of the exposure area) and the subtidal mean (97.3% of the exposure area).

[b] The site-wide EPC is the weighted average of the intertidal EPC (2.7% of the exposure area) and the subtidal EPC (97.3% of the exposure area).

Table A.5-5. EPCs in EW surface sediment used to estimate the exposure of wildlife ROCs (cont.)

[c] PCB TEQs and total TEQs were calculated using TEFs for birds as presented in Van den Berg et al. (1998). These TEFs are listed in Attachment 1, and uncertainties associated with application of these TEFs are discussed in Section A.6.3.1.2.

[d] PCB TEQs and total TEQs were calculated using TEFs for mammals presented in Van den Berg et al. (2006). These TEFs are listed in Table A.2-14, and uncertainties associated with application of these TEFs are discussed in Section A.6.3.3.2.

COPC – chemical of potential concern

dw – dry weight

EPC – exposure point concentration

EW – East Waterway

na – not applicable (COPC was not detected, or all results were detects)

PCB – polychlorinated biphenyl

RL – reporting limit

ROC – receptor of concern

TEF – toxic equivalency factor

TEQ – toxic equivalent

UCL – upper confidence limit

A.5.1.3.3 Water EPCs

Surface water data were used to estimate the COPC exposure from incidental water ingestion. Surface water EPCs were calculated as the 95% UCL of all surface water samples using ProUCL, as described in the subsection for prey tissue EPCs (Table A.5-6). Surface water samples were not analyzed for dioxin/furan congeners, so the PCB TEQ was used to represent the total TEQ. Uncertainties associated with the use of PCB TEQ instead of the total TEQ, which was not available, are expected to be low and are discussed in the uncertainty analyses for each ROC in Section 6.3.

Table A.5-6. EPCs in surface water used to estimate exposure of wildlife ROCs

COPC	EPC in Water (mg/L)
Mercury	5.8×10^{-6}
Selenium	1.4×10^{-4}
Total PCBs	1.63×10^{-6}
PCB TEQ	1.14×10^{-9}
Total TEQ[a]	1.14×10^{-9}

a Dioxin/furan congeners were not analyzed in surface water; therefore, the total TEQ in surface water is represented by the PCB TEQ.

EPC – exposure point concentration
COPC – chemical of potential concern
PCB – polychlorinated biphenyl
ROC – receptor of concern
TEQ – toxic equivalent

A.5.1.3.4 Estimated dietary doses

Exposures as dietary doses based on the ingestion of prey, water, and sediment were estimated for each wildlife ROC using Equation 5-1 and the information presented in preceding sections; exposure doses are summarized in Table A.5-7. Tables that contain all data used in these calculations for each ROC-COPC pair are presented in Attachment 6.

Table A.5-7. Exposure doses of COPCs for pigeon guillemot, osprey, river otter, and harbor seal

COPC	Dietary Exposure Dose (mg/kg bw/day)			
	Pigeon Guillemot	Osprey	River Otter	Harbor Seal
Mercury	0.0062	ne	0.0023	0.00025
Selenium	ne	ne	0.018	ne
Total PCBs	0.16	0.10	0.069	0.0081
PCB TEQs	4.2×10^{-6}	3.4×10^{-6}	7.4×10^{-7}	9.5×10^{-8}
Total TEQ	4.7×10^{-6}	3.8×10^{-6}	7.8×10^{-7}	1.0×10^{-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 on birds and mammals for each of the COPCs and describes the selection of TRVs for the wildlife ROCs. The literature search and guidelines for TRV selection for wildlife ROCs is described in detail in Section A.2. 5.3. Toxicological data presented in this section were used in combination with exposure doses (presented in Section A.5.1) to characterize risk, as presented in Section A.6.3.

In most of the studies reviewed, the toxicity data were related to a concentration in the dietary food item rather than as an ingested dose. Thus, the dietary food concentrations in the toxicity studies were converted to ingested doses using the toxicity test species-specific food ingestion rates and body weights, as follows:

$$\text{Dose} = \frac{(C_{food} \times IR_{food})}{BW} \qquad \textbf{Equation 5-8}$$

Where:

Dose	=	chemical dose via food (mg/kg bw/day)
IR_{food}	=	food ingestion rate (kg food dw/day)
C_{food}	=	chemical concentration in food (mg/kg food dw)
BW	=	test species body weight (kg ww)

If the food ingestion rates 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 food ingestion rate units using the moisture content in food (see footnotes to tables in Section A.5.2.1). The values for food ingestion rate, body weight, and food moisture content were obtained from the specific toxicity study, if available. If site-specific data were not available, generic data from the literature were used, as noted in tables in Section A.5.2.1.

A.5.2.1 TRVs for birds

This section presents the TRVs developed for the COPCs identified for bird ROCs. TRVs for total PCBs were developed from toxicity studies that exposed birds to PCB Aroclors. TRVs for PCB TEQ and total TEQ were developed from toxicity studies that exposed birds to 2,3,7,8-TCDD. The following subsections summarize the toxicity studies reviewed for mercury, total PCBs, and dioxins/furans (for the PCB TEQ and total TEQ) and the NOAEL and LOAEL TRVs selected for these COPCs based on the selection criteria discussed in Section A.2.5.1.2.

A.5.2.1.9 Mercury

Chronic effects of dietary mercury on birds include adverse effects on growth, development, reproduction, metabolism, and behavior; organomercury compounds, especially methylmercury, are more toxic than inorganic forms of mercury (Eisler 1987). Eight studies that evaluated the toxicity of dietary mercury in either the organic or the elemental form to birds were identified (Table A.5-8). When reviewing the toxicity literature for mercury, only forms of mercury relevant to the EW 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, methylmercury dicyandiamide) were not considered relevant because these forms of mercury are not expected to be present in the EW. 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 in the selection of TRVs.

Table A.5-8. Bird toxicity studies reviewed for the selection of mercury TRVs

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
Methyl-mercury chloride	American kestrel	na	**0.073**	1 month prior to egg laying through egg laying period	reduced number of fledglings	na	0.7 mg/kg dw	0.13 (Pattee 1984)	0.0136 kg dw/day	Albers (2007)
Methyl-mercury chloride	great egret (1 day old)	na	0.091	14 weeks	reduced growth	na	0.5 mg/kg ww	1.02 (Arizona Game & 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[c]	0.1 (NRC 1994)	0.0053 kg dw/day, galliformes (Nagy 2001)	Hill and Soares (1987)
Methyl-mercury chloride	zebra finch	0.72	1.4	76 days	reduced survival	2.5 mg/kg dw[d]	5 mg/kg dw[d]	0.012 (Dunning 1993)	0.0034 kg dw/day, passerines (Nagy 2001)	Scheuhammer (1988)
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[e]	na	0.13 (Pattee 1984)	0.0136 kg dw/day, Eurasian kestrel (Nagy 2001)	Peakall and Lincer (1972)
Mercuric chloride	Japanese quail (chicks)	na	62	5 days	reduced offspring hatchling survival	na	1,045 mg/kg ww[c]	0.1 (NRC 1994)	0.0053 kg dw/day, galliformes (Nagy 2001)	Hill and Soares (1987)

[a] No-effect and effect concentrations are presented in the units given in the studies reviewed. Table footnotes indicate how units were converted to wet weight or dry weight to correspond to the food ingestion rate unit 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.

[c] Effect concentration converted into dry weight assuming 10% moisture in prepared diet.

[d] No-effect and effect concentrations were converted into wet weight assuming 10% moisture in a prepared diet.

Table A.5-8. Bird toxicity studies reviewed for the selection of mercury TRVs (cont.)

[e] 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 an 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

TRV – toxicity reference value

ww – wet weight

Bold and underline identify 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 0.0146 mg/kg bw/day.

In the studies reviewed, adverse effects on reproduction, early-life-stage growth, or adult survival were reported from the dietary exposure of various bird species to mercury (Table A.5-8). LOAELs ranged from 0.073 mg/kg bw/day for reproductive effects in American kestrel (Albers et al. 2007) to 62 mg/kg bw/day for offspring mortality of Japanese quail (Hill and Soares 1987). The lowest LOAEL of 0.073 mg/kg bw/day mercury from Albers et al. (2007) was selected as the LOAEL TRV. There was no NOAEL from the study by Albers et al., and there was no NOAEL lower than the selected LOAEL from any other study in Table A.5-8. Therefore, an uncertainty factor of five was used to derive a NOAEL of 0.0146 mg/kg bw/day from the chronic LOAEL.

A.5.2.1.1 Total PCBs

Effects on birds from exposure to dietary PCBs include the disruption of normal patterns of growth, reproduction, metabolism, and behavior (Eisler 1986). The most sensitive effects are related to reproduction and include egg production, fertility, and hatching success (Eisler 1986). Of the laboratory bird 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 laboratory poultry studies, only data from laboratory wild bird 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 the 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 these data are not likely to predict risks accurately. Therefore, chicken toxicity data for reproductive endpoints were not used in this ERA.

Thirteen studies that evaluated the dietary toxicity of PCBs to birds were identified (Table A.5-9). Various species were studied, including American kestrel, screech owl, ring dove, Japanese quail, and mallard duck. 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.5-9. Bird toxicity studies reviewed for the selection of PCB TRVs

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
Aroclor 1248	American kestrel	na	0.35	5.5 months	reduced eggshell thickness	na	3[c]	0.13 (Pattee 1984)	0.0136 kg dw/day, Eurasian kestrel (Nagy 2001)	Lowe and Stendell (1991)
Aroclor 1248	screech owl	**0.49**	na	two generations	no effect on eggshell thickness, egg production, or hatching and fledging success	3[c]	na	0.181 (Dunning 1993)	0.0266 kg dw/day, carnivorous birds (Nagy 2001)	McLane and Hughes (1980)
Aroclor 1242	Japanese quail	na	0.60	45 days	reduced eggshell thickness	na	10[c]	0.09 (Dunning 1993)	0.0048 kg dw/day, galliformes (Nagy 2001)	Hill et al. (1976)
Aroclor 1254	ring dove	na	**1.4**	two generations	reduced hatching success in second generation	na	10[d]	0.155 (Sample et al. 1996)	0.0202 kg dw/day, all birds (Nagy 2001)	Peakall et al. (1972); Peakall and Peakall (1973)
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[e]	na[e]	na	na	Fernie et al. (2000; 2001)
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[e]	na[e]	na	na	Fernie et al. (2003a; 2003b; 2003c)
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 footnotes indicate how units were converted to wet weight or dry weight to correspond to the food ingestion rate unit for calculating NOAELs and LOAELs.

Table A.5-9. Bird toxicity studies reviewed for the selection of PCB TRVs (cont.)

[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.
[c] Effect and/or no-effect concentration was converted into dry weight assuming 10% moisture in prepared diet.
[d] Effect concentration was converted into dry weight assuming 9% moisture contents of seeds (EPA 1993)
[e] Body weight-normalized dose was estimated in the study.

bw – body weight
dw – dry weight
F1 – first generation
F2 – second generation
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 and underline identify 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 ring 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 in screech owls was reported (McLane and Hughes 1980), to 3.9 mg/kg bw/day, at which egg production and eggshell thinning in mallards were unaffected (Risebrough and Anderson 1975). The NOAEL of 0.49 mg/kg bw/day was selected as the NOAEL TRV.

A.5.2.1.2 2,3,7,8-TCDD

PCB TEQs and total TEQs are expressed as 2,3,7,8-TCDD equivalents; therefore, toxicity studies that involved exposing birds to 2,3,7,8-TCDD were reviewed. The effects of dioxins and furans reported in laboratory studies with various bird species included developmental toxicity, hepatotoxicity, endocrine disruption, immunotoxicity, and death (Kennedy et al. 1996).

No studies in the published literature that involved the 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.5-10.

Table A.5-10. Bird toxicity studies reviewed for the selection of the 2,3,7,8-TCDD TRVs

Chemical	Test Species	NOAEL (mg/kg bw/day)	LOAEL (mg/kg bw/day)	Exposure Duration	Effect	Source
2,3,7,8-TCDD	ring-necked pheasant	**<u>0.000014</u>**	**<u>0.00014</u>**	once per week for 10 weeks through IP injection	reduced body weight, egg production, and survival of adults and embryos	Nosek et al. (1992)
2,3,7,8-TCDD	cockerel	0.0001	0.001	20 to 21 days through oral intubation	decreased survival	Schwetz et al. (1973)

bw – body weight
IP – intraperitoneal
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
<u>Bold and underline</u> identify the NOAEL and LOAEL selected as the TRVs.

The lowest dose that resulted 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 by the authors. There is significant uncertainty with regard to 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 that in 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. The dose of 0.000014 mg/kg bw/day was selected as the NOAEL TRV. Uncertainties associated with the absence of relevant toxicological data for the 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.

A.5.2.2 TRVs for mammals

This section presents the TRVs developed for the COPCs identified for mammalian ROCs. TRVs for total PCBs were developed from toxicity studies with PCB Aroclors. TRVs for PCB TEQ and total TEQ were developed from toxicity studies with 2,3,7,8-TCDD. The following subsections summarize the toxicity studies reviewed for mercury, selenium, total PCBs and dioxins/furans (for the PCB TEQ and total TEQ) and the selected NOAEL and LOAEL TRVs for the COPCs based on the selection criteria discussed in Section A.2.5.1.2.

A.5.2.2.1 Mercury

Mercury has been reported to adversely affect reproduction, growth, development, behavior, blood and serum chemistry, motor coordination, vision, hearing, histology, and metabolism in mammals; organomercury compounds, especially methylmercury, are more toxic than inorganic forms of mercury (Eisler 1987). Three studies that evaluated the toxicity of dietary methylmercury to mammals were identified for the growth, reproduction, and survival endpoints (Table A.5-11). In these studies, adverse effects following the 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 on 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 of 0.0084 mg/kg bw was selected as the LOAEL TRV. Although toxicology data based on mink studies may be more representative with regard to the mammals using the EW, the LOAEL based on the rat study was selected because it was the most conservative effects threshold reported in the three studies reviewed and was based on a

Table A.5-11. Mammal toxicity studies reviewed for the selection of mercury TRVs

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)[a]	Food Ingestion Rate (kg ww/day)[b]	Source
Methylmercuric chloride	rat	na	**0.0084**	three generations	reduced growth	na	0.0799	0.16	0.016 (EPA 1993)	Verschuuren et al. (1976)
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)
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)
Methylmercury	mink	na	0.64	2 months	reduced growth, 100% mortality	na	5	1.2	0.15	Aulerich et al. (1974)

[a] Body weight is from the source study unless otherwise noted.

[b] Food ingestion rate from the source study unless otherwise noted.

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 and underline identify 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.

multi-generational study. No controlled laboratory studies in which mink were exposed to dietary mercury over a chronic period or during a critical life stage were found. 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.5.2.2.2 Selenium

Selenium is an essential nutrient, and deficiency in the diet may adversely affect 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 through dietary exposure (i.e., via food rather than drinking water or gavage) were identified (Table A.5-12). In these studies, adverse effects on growth or survival were reported following the 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 in which mammals were exposed to dietary selenium for a chronic exposure period or during a critical life stage were identified.

LOAELs ranged from 0.080 mg/kg bw/day, which resulted in reduced growth of rats (Halverson et al. 1966), to 5.8 mg/kg bw/day, which resulted 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 NOAEL was selected as the NOAEL TRV.

Table A.5-12. Mammal toxicity studies reviewed for the selection of selenium TRVs

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)[a]	Food Ingestion Rate (kg ww/day)[b]	Source
Sodium selenite	rat	**<u>0.055</u>**	**<u>0.080</u>**	6 weeks	reduced body weight	3.2	4.8	0.139 (NOAEL)[d] 0.129 (LOAEL)[d]	0.00238 (NOAEL)[c, d] 0.00215 (LOAEL)[c, d]	Halverson et al. (1966)
Sodium selenite	rat	0.13	0.14	6 weeks	reduced survival	8.0	9.6	0.129 (NOAEL) 0.1255 (LOAEL)	0.00215 (NOAEL)[d] 0.00186 (LOAEL)[d]	Halverson et al. (1966)
L-seleno-methionine	rat	na	0.16	110 days	reduced body weight	na	2	0.34	0.027 (EPA 1993)	Behne et al. (1992)
Selenite	rat	0.16	na	110 days	no effect on body weight	2	na	0.34	0.027 (EPA 1993)	Behne et al. (1992)
Sodium selenite, nano-Se, or organic selenium	rat	0.17	0.28	13 weeks	reduced body weight	na[e]	na[e]	na[e]	na[e]	Jia et al. (2005)
Seleno-methionine	hamster	0.36	0.76	21 days	reduced body weight	5.1	10.1	0.092 (NOAEL)[d] 0.091(LOAEL)[d]	0.00655 (NOAEL)[c, d] 0.0068 (LOAEL)[c, d]	Julius et al. (1983)
Sodium selenite	hamster	na	3.4	21 days	reduced body weight	na	40.25	0.074	0.0062[d]	Julius et al. (1983)
Sodium selenite	hamster	na	5.8	21 days	reduced female survival	na	80.24	0.062	0.0045[d]	Julius et al. (1983)

[a] Body weight is from the source study unless otherwise noted.
[b] Food ingestion rate is from the source study unless otherwise noted.
[c] Data presented in the study were not statistically evaluated.
[d] Body weight and ingestion rates used were specific to the no-effect and effect concentration test groups.
[e] Dietary dose was calculated in the source 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
<u>Bold and underline</u> identify the NOAEL and LOAEL selected as the TRVs.

A.5.2.2.3 Total PCBs

PCBs have been reported to produce a broad range of toxic effects in laboratory mammals under controlled exposure conditions, including mortality, 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 that, 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 tracts of most mammals readily absorb PCBs, but the absorption rate may be affected by the dose level and lipophilicity of the compound (Eisler 1986; Van den Berg et al. 1998). In addition, there is evidence of the placental transfer of PCBs in mammals (Eisler 1986b), and PCBs can also accumulate in the lipid portion of milk, resulting in the exposure of suckling young.

Adverse reproductive effects (e.g., reduced 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 through toxic action directly 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 produce a broad range of direct and indirect effects associated with reproductive functions. Direct effects on 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 a dose of 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 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.5-13). 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.

Table A.5-13. Mammal toxicity studies reviewed for the selection of PCB TRVs

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)[a]	Food Ingestion Rate (kg ww/day)[b]	Source
Clophen A50	mink	na	**0.089**	18 months	reduced offspring kit growth	na	0.1 mg/day[c]	1.12	na	Brunström et al. (2001)
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)
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 (composition 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[d]	na[d]	na[d]	na[d]	Kihlstrom et al. (1992)
Aroclor 1254	mink	1.2	1.8	28 days	reduced female growth	na[d]	na[d]	na[d]	na[d]	Aulerich et al. (1986)
Clophen A50	mink	na	2.0	3 months	all whelps stillborn	na[d]	na[d]	na[d]	na[d]	Kihlstrom et al. (1992)

Table A.5-13. Mammal toxicity studies reviewed for the selection of PCB TRVs (cont.)

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)[a]	Food Ingestion Rate (kg ww/day)[b]	Source
Aroclor 1254	mink	1.5	2.4	28 days	reduced male and female growth	na[d]	na[d]	na[d]	na[d]	Aulerich et al. (1986)
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)

[a] Body weight is from the source study unless otherwise noted.
[b] Food ingestion rate is from the source study unless otherwise noted.
[c] Dietary concentration was determined by dividing the daily dose by the body weight. Female mink were exposed to 0.24 mg Clophen A50 three times a week (0.1 mg/day).
[d] Dietary dose was calculated in the source 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 and underline identify 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.

Reported reproductive effect levels ranged from doses of 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 as compared with 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, as was used for other ROC-COPC pairs, because the large toxicity dataset for mink and PCBs indicates that the difference between a NOAEL and LOAEL is much less than fivefold. As presented in Table A.5-13, in the two studies that had both NOAELs and LOAELs (Aulerich and Ringer 1977; Aulerich et al. 1986), the LOAELs were higher than the NOAELs from the same studies by factors ranging from 1.5 to 2. In addition, dose-response plots of toxicity to mink exposed to PCBs show very steep transitions between PCB exposures that caused no adverse effects and those that resulted in severe adverse effects (EPA 2003), indicating that an uncertainty factor of 2 is more appropriate than an uncertainty factor of 5. The resulting NOAEL was 0.045 mg/kg bw/day.

A.5.2.2.4 2,3,7,8-TCDD

PCB TEQ and total TEQ 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.5-14). In these studies, adverse effects on the 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 dose because it was associated with 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.5-14. Mammal toxicity studies reviewed for the selection of 2,3,7,8-TCDD TRVs

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)[a]	Food Ingestion Rate (kg ww/day)[b]	Source
2,3,7,8-TCDD	Hartley guinea pig	**<u>6.5×10^{-7}</u>**	**<u>4.9×10^{-6}</u>**	90 days	reduced body weight	1.0×10^{-5}	7.6×10^{-5}	na[c]	na[c]	DeCaprio et al. (1986)
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)[d] 1.054 (LOAEL)[d]	0.18 (Bleavins and Aulerich 1981)	Hochstein et al. (2001)
2,3,7,8-TCDD	Sprague-Dawley rat	1.0×10^{-6}	1.0×10^{-5}	three generations	reduced litter size and F2 post-natal survival	na[c]	na[c]	na[c]	na[c]	Murray et al.(1979)
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[c]	na[c]	DeCaprio et al. (1986)
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)[d] 0.8183 (LOAEL)[d]	0.049 (NOAEL)[d] 0.050 (LOAEL)[d]	Hochstein et al. (2001)
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[c]	na[c]	na[c]	na[c]	Kociba et al.(1978)
2,3,7,8-TCDD	Sprague-Dawley rat	na	3.2×10^{-4}	13 weeks	reduced body weight	na[c]	na[c]	na[c]	na[c]	Van Birgelen et al. (1994)
2,3,7,8-TCDD	Sprague-Dawley rat	1.0×10^{-4}	na	three generations	reduced body weight	na[c]	na[c]	na[c]	na[c]	Murray et al.(1979)

a Body weight is from the source study unless otherwise noted.

b Food ingestion rate is from the source study unless otherwise noted.

c Dietary dose was calculated in the source study.

d Body weight and food ingestion rates used were specific to the no-effect and effect concentration test groups.

bw – body weight
F2 – second generation
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

<u>Bold and underline</u> identify the NOAEL and LOAEL selected as the TRVs.

A.5.3 SUMMARY OF WILDLIFE EXPOSURE AND EFFECTS ASSESSMENT

A.5.3.1 Exposure assessment

The exposure assessment provides an estimate of each wildlife ROC's exposure to COPCs through the ingestion of prey, surface water and surface sediment. 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 use were made using site-specific information, if available, along with general species life history information. Exposure doses were estimated using 95% UCL concentrations in prey tissue, surface sediment, and surface water. Exposure doses for wildlife were presented in Table A.5-7.

A.5.3.2 Effects assessment

The TRVs selected in the effects assessment 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 in order to select the most appropriate TRVs. TRVs for both no-effects and low-effects data were selected, as summarized in Table A.5-15.

Table A.5-15. Selected wildlife TRVs

COPC	NOAEL and LOAEL TRVs (mg/kg bw/day)							
	Pigeon Guillemot		Osprey		River Otter		Harbor Seal	
	NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL
Mercury	0.0146	0.073	ne	ne	0.0017	0.0084	0.0017	0.0084
Selenium	ne	ne	ne	ne	0.055	0.080	ne	ne
Total PCBs	0.49	1.4	0.49	1.4	0.045	0.089	0.045	0.089
PCB TEQ and total TEQ	1.4×10^{-5}	1.4×10^{-4}	1.4×10^{-5}	1.4×10^{-4}	6.5×10^{-7}	4.9×10^{-6}	6.5×10^{-7}	4.9×10^{-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
ROC – receptor of concern
TEQ – toxic equivalent
TRV – toxicity reference value

A.6 Risk Characterization and Uncertainty Analysis

This section presents the risk characterization for each ROC-COPC pair identified in the problem formulation (Section A.2) and discussed in the exposure and effects assessments (Sections A.3 through A.5). 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 conclusion sections include the identification of COCs for each of the various ROCs. This section is divided into subsections for each of the three major receptor groups: benthic invertebrates (Section A.6.1), fish (Section A.6.2), and wildlife (Section A.6.3).

A.6.1 BENTHIC INVERTEBRATES

This section characterizes risks to benthic invertebrates that are closely associated with sediment, such as amphipods, bivalves, and polychaetes, as well wider-ranging, higher-trophic-level macroinvertebrates, such as crab. Therefore, risk to benthic invertebrates was characterized separately for two ROCs: 1) the benthic invertebrate community (Section A.6.1.1) and 2) crab (Section A.6.1.2). The risk characterization for the benthic invertebrate community was based on four evaluations:

- Comparison of surface sediment chemistry data with available sediment chemical criteria and guidelines and through the use of site-specific sediment toxicity tests
- Comparison of benthic invertebrate tissue chemistry data with tissue-residue effects data from the literature
- Comparison of surface water chemistry data with Washington State or federal water quality standards and criteria
- Comparison of porewater chemistry data (VOCs only) with water effects data from the literature

Risk characterization for crab was based on a comparison of crab tissue chemistry data with tissue-residue effects data from the literature and the results of the surface water evaluation.

A.6.1.1 Benthic invertebrate community

The risk characterization results for each evaluation (sediment, tissue, surface water, and porewater) are provided in the following subsections, each of which presents a risk estimate and an uncertainty analysis. Each subsection also discusses risk conclusions based on an application of the uncertainty to the risk estimate.

A.6.1.1.1 Sediment

The potential for adverse effects on the benthic invertebrate community resulting from exposure to 29 surface sediment COPCs was evaluated through a comparison of COPC concentrations in EW surface sediment with SMS criteria (or DMMP guidelines for total DDTs) and site-specific toxicity testing. The SQS corresponds to a sediment concentration below which no adverse effects on biological resources are anticipated; the CSL of the SMS corresponds to a sediment concentration above which minor adverse effects are expected. Chemical concentrations that fall between the SQS and the CSL are generally interpreted as having a potential for minor adverse effects. Exceedance factors were calculated as follows:

$$EF_{sediment} = \frac{C_{sediment}}{\text{SMS or DMMP}} \qquad \textbf{Equation 6-1}$$

Where:

$EF_{sediment}$	=	exceedance factor for sediment chemistry
$C_{sediment}$	=	COPC concentration in sediment
SMS	=	SQS or CSL of the SMS
DMMP	=	SL or ML of the DMMP

For toxicity tests, SMS provide guidance for overall SQS and CSL biological effects exceedances at each sediment sampling location based on a weight-of-evidence approach using the three toxicity test results at each location. Specifically, the exceedance of the CSL biological effects criterion in a single toxicity test at a sediment sampling location is used as evidence of potential adverse effects on the benthic community; exceedances of the SQS biological effects criterion in two or more toxicity tests are considered equivalent to a CSL exceedance at the location. If one of the three types of toxicity test exceeds the SQS effects criterion and the other two are below the SQS, the result is considered to be indicative of potential minor adverse effects.

Because AETs, which form the basis for the chemical criteria of SMS, are derived from toxicity tests of sediment samples with a mixture of chemicals from various locations in Puget Sound, and an exceedance of those criteria is not always an accurate predictor of adverse effects, the SMS regulations and DMMP guidance state that site-specific toxicity tests supersede site-specific chemistry data from the same location. For example, if the concentration of a chemical was greater than the CSL chemical criterion at a location, but the sample did not exceed SQS in the biological testing, then the location would not be classified as a CSL exceedance. This weight-of-evidence approach, which combines multiple biological and chemical endpoints, is considered in the characterization of risks to the benthic community in the EW.

The application of SMS criteria to assess the potential for 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 on benthic invertebrates in the EW was performed using Thiessen polygons, a method commonly used to illustrate spatial variability in sampling intensity and to extrapolate results from smaller 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. The assumption that there is spatial homogeneity within polygons introduces uncertainty in all areal percentages discussed in this section. In areas with lower sample density, the uncertainty in this assumption increases (i.e., larger polygons are associated with lower sampling density; all points within a larger polygon are less likely to have the same characteristics as the point that defined the polygon). The uncertainty section includes a discussion of the spatial analysis using Thiessen polygons relative to an alternative interpolation approach.

The following subsections present the magnitude and spatial extent of SMS exceedances based on sediment chemistry, the results of the biological SMS characterization based on toxicity tests, and the overall SMS characterization, which combines the chemical and biological results. Chemicals were identified as COCs if they were detected in at least one surface sediment sample at a concentration that exceeds the SQS.[32] Uncertainties in these approaches are then presented, followed by a discussion of the risk conclusions.

Sediment Chemistry

Table A.6-1 presents a summary of the surface sediment COPC concentrations relative to SMS criteria (or DMMP guidelines for total DDTs). Surface sediment samples were collected from 243 locations within the EW; of those, 167 locations had one or more exceedances of the SQS or SL. All 30 COPCs exceeded the SQS in at least one location. Total PCBs most frequently (65%) exceeded its SQS criterion, followed by mercury (19%), and 1,4-dichlorobenzene (13%). All other COPCs exceeded their respective criteria in < 10% of the locations. Maps A.6-1, A.6-2, and A.6-3 present the distribution and magnitude of concentrations for total PCBs, mercury, and 1,4-dichlorobenzene.

[32] For benthic invertebrates, the same criteria were used to identify COCs and COPCs for sediment exposure. Therefore, all sediment COCs were also COPCs for the assessment of benthic invertebrates.

Table A.6-1. Frequencies of detected surface sediment COPC concentrations greater than risk-based chemical criteria

COPC	Detection Frequency		Frequency of Detected Concentrations > SQS/SL			Maximum Detected SQS/SL EF	Frequency of Detected Concentrations > SQS/SL and > CSL/ML			Maximum Detected CSL/ML EF
	No. of Samples[a]	Percent	No. of Samples[b]	Percent	No. of Samples with RL > SQS/SL[c]		No. of Samples[d]	Percent	No. of Samples with RL > CSL/ML[e]	
Metals										
Arsenic	162/231	70	2/231	0.87	0	4.2	2/231	0.87	0	2.6
Cadmium	155/231	67	2/231	0.87	0	1.3	1/231	0.43	0	1.0
Mercury	233/239	97	46/239	19	0	2.6	10/239	4.2	0	1.8
Zinc	231/231	100	5/231	2.2	0	3.0	1/231	0.43	0	1.3
PAHs										
2-Methylnaphthalene	87/240	36	1/240	0.42	0	2.2	1/240	0.42	0	1.3
Acenaphthene	126/240	53	16/240	6.7	0	14.0	6/240	2.5	0	4.0
Anthracene	209/240	87	1/240	0.42	0	1.0	0/240	0	0	ne
Benzo(a)anthracene	226/240	94	7/240	2.9	0	3.2	1/240	0.42	0	1.3
Benzo(a)pyrene	225/240	94	7/240	2.9	0	2.4	1/240	0.42	0	1.1
Benzo(g,h,i)perylene	212/240	88	4/240	1.7	0	1.8	0/240	0	0	ne
Total benzofluoranthenes	228/240	95	7/240	2.9	0	4.0	1/240	0.42	0	2.0
Chrysene	230/240	96	8/240	3.3	0	10	1/240	0.42	0	2.4
Dibenzo(a,h)anthracene	156/240	65	4/240	1.7	0	1.8	0/240	0	0	ne
Dibenzofuran	107/240	45	8/240	3.3	0	11	2/240	0.83	0	2.8
Fluoranthene	233/240	97	14/240	5.8	0	40	2/240	0.83	0	5.3
Fluorene	144/240	60	12/240	5.0	0	9.6	3/240	1.3	0	2.8
Indeno(1,2,3-cd)pyrene	210/240	88	6/240	2.5	0	1.7	0/240	0	0	ne
Phenanthrene	230/240	96	15/240	6.3	0	7.8	3/240	1.3	0	1.6
Pyrene	235/240	98	1/240	0.42	0	3.5	1/240	0.42	0	2.5
Total HPAHs	237/240	99	9/240	3.8	0	13	1/240	0.42	0	2.4
Total LPAHs	230/240	96	8/240	3.3	0	3.6	3/240	1.3	0	1.7

Table A.6-1. Frequencies of detected surface sediment COPC concentrations greater than risk-based chemical criteria (cont.)

COPC	Detection Frequency		Frequency of Detected Concentrations > SQS/SL			Maximum Detected SQS/SL EF	Frequency of Detected Concentrations > SQS/SL and > CSL/ML			Maximum Detected CSL/ML EF
	No. of Samples[a]	Percent	No. of Samples[b]	Percent	No. of Samples with RL > SQS/SL[c]		No. of Samples[d]	Percent	No. of Samples with RL > CSL/ML[e]	
Phthalates										
Bis(2-ethylhexyl) phthalate	207/231	90	9/231	3.9	2	40	5/231	2.2	1	24
Butyl benzyl phthalate	101/231	44	9/231	3.9	6	2.9	0/231	0	0	ne
Di-n-butyl phthalate	32/231	14	1/231	0.43	0	12	1/231	0.43	0	1.5
Other SVOCs										
1,4-Dichlorobenzene	146/231	63	29/231	13	2	350	9/231	3.9	0	120
2,4-Dimethylphenol	14/231	6.1	1/231	0.43	36	17	1/231	0.43	36	17
n-Nitrosodiphenylamine	2/231	0.87	1/231	0.43	0	6.4	1/231	0.43	0	4.5
Phenol	94/231	41	5/231	2.2	0	1.5	0/231	0	0	ne
PCBs										
Total PCBs	227/240	95	157/240	65	0	70	23/240	9.6	0	13
Pesticides										
Total DDTs	8/143	5.6	2/143	1.4	70	4.6	0/143	0	4	ne

[a] Represents the number of detects per total number of samples.

[b] Represents the number of detects > SQS/SL per total number of samples. If any individual sample had a TOC content > 4% or < 0.5%, that sample was considered to be greater than the SQS/SL if the dry-weight concentration was greater than the LAET. The number of detected concentrations > SQS/SL includes the number > CSL/ML.

[c] Represents the number of RLs greater than the SQS/SL. The number of samples with RLs > SQS/SL includes the number > CSL/ML. These chemicals are discussed in the uncertainty analysis.

[d] Represents the number of detects > the CSL/ML per the total number of samples. If any individual sample had a TOC content > 4% or < 0.5%, the sample was considered to be greater than the CSL/ML if the dry-weight concentration was greater than the 2LAET.

[e] Represents the number of RLs greater than the CSL/ML. These chemicals are discussed in the uncertainty analysis.

COPC – chemical of potential concern
CSL – cleanup screening level
DDT – dichlorodiphenyltrichloroethane
EF – exceedance factor
HPAH – low-molecular-weight polycyclic aromatic hydrocarbon
LPAH – high-molecular-weight polycyclic aromatic hydrocarbon
LAET – lowest-apparent-effect threshold
2LAET – second-lowest-apparent-effect threshold
ML – maximum level
ne – no exceedance
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
RL – reporting limit
SL – screening level
SQS – sediment quality standard
SVOC – semivolatile organic compound
TOC – total organic carbon

Twenty-three COPCs exceeded their respective CSL in at least one location, with total PCBs being the most frequently detected above its CSL criterion (23 of 240 locations, or 9.6%) followed by mercury (10 of 239 locations, or 4.2%); all other chemicals were detected above their respective CSL criterion in < 4% of the locations. Chemicals that were not detected in sediment but have reporting limits greater than the SMS chemical criteria are discussed in the uncertainty analysis.

Toxicity Tests

The toxicity results from the 51 locations tested as part of this SRI and in earlier studies accepted for use in the risk assessment were presented in Table A.3-8. The table also presents the final biological SMS classification for each location based on combined results from the three toxicity tests. Results for individual bioassays are shown on Map A.3-1, and the final classification for each sampling location under SMS rules for the interpretation of multiple toxicity tests is presented on Map A.6-4 (note that this map also shows chemistry results). Half of the bioassays were below the SQS and CSL criteria, about 12% exceeded the SQS criterion, and 38% exceeded the CSL criterion.

Final SMS Designation of Surface Sediment Data

Table A.6-2 identifies the locations where samples were collected for both chemical analysis and toxicity testing, the SMS designation for both chemistry and toxicity, and the final SMS designation based on site-specific toxicity test results that override the co-located chemistry results, per the SMS. At 36 of the total of 51 locations, the SMS designation based on toxicity test results differed from the SMS chemistry predictions; and at the remaining 15 locations, they were the same. Of those 36 samples for which the chemistry predictions differed from the toxicity test results, approximately 45% (n = 16) of the chemistry results underpredicted toxicity, while 55% (n = 20) of the chemistry results overpredicted toxicity. Map A.6-4 shows the SMS designations for both chemistry and toxicity test results for all locations.

Using the final SMS designation based on both sediment chemistry and toxicity test results, the percentage of the EW area not expected to result in adverse effects on the benthic community (i.e., ≤ SQS) was approximately 40%, and percentage of area in which minor adverse effects are expected (i.e. > CSL) was approximately 21%. Approximately 39% of the area falls between the SQS and the CSL (i.e., > SQS and ≤ CSL) and is generally interpreted as having the potential for minor adverse effects on the benthic community. Map A.6-5 shows the final designation of each area, as represented by Thiessen polygon, according to SMS rules.

Table A.6-2. Final SMS designations for locations with both chemical and site-specific toxicity test results

Location ID	Chemical SMS Designation	COPCs with Concentrations that Exceeded SMS Criteria	Toxicity Test SMS Designation	Designations Agree?	Final SMS Designation
EW-100	no exceedance	none	no exceedance	yes	no exceedance
EW-101	CSL	**mercury**	no exceedance	no	no exceedance
EW-102	no exceedance	none	no exceedance	yes	no exceedance
EW-103	SQS	mercury, fluoranthene, fluorene, phenanthrene, total PCBs	CSL	no	CSL
EW-104	SQS	mercury, benzo(a)anthracene, benzo(a)pyrene, total benzofluoranthenes, chrysene, fluoranthene, indeno(1,2,3-c,d)pyrene, total HPAHs, total PCBs	no exceedance	no	no exceedance
EW-105	SQS	total PCBs	no exceedance	no	no exceedance
EW-106	SQS	total PCBs	no exceedance	no	no exceedance
EW-107	SQS	total PCBs	no exceedance	no	no exceedance
EW-108	no exceedance	none	no exceedance	yes	no exceedance
EW-109	CSL	**total PCBs**	CSL	yes	CSL
EW-110	CSL	**arsenic**, zinc	CSL	yes	CSL
EW-111	SQS	total PCBs	no exceedance	no	no exceedance
EW-112	CSL	**total PCBs**	no exceedance	no	no exceedance
EW-113	SQS	mercury, total PCBs	SQS	yes	SQS
EW-114	CSL	**total PCBs**, 1,4-dichlorobenzene	no exceedance	no	no exceedance
EW-115	SQS	total PCBs	no exceedance	no	no exceedance
EW-116	CSL	**total PCBs**, butylbenzyl phthalate	CSL	yes	CSL
EW-117	SQS	total PCBs	CSL	no	CSL
EW-118	CSL	**total PCBs**	no exceedance	no	no exceedance
EW-119	SQS	mercury, total PCBs	no exceedance	no	no exceedance
EW-120	SQS	mercury, 1,4-dichlorobenzene, total PCBs	no exceedance	no	no exceedance
EW-124	SQS	Total PCBs	CSL[a]	no	CSL
EW-125	SQS	Total PCBs	CSL[a]	no	CSL
EW-126	no exceedance	none	SQS	no	SQS
EW-128	CSL	**total PCBs**	no exceedance	no	no exceedance

Table A.6-2. Final SMS designations for locations with both chemical and site-specific toxicity test results (cont.)

Location ID	Chemical SMS Designation	COPCs with Concentrations that Exceeded SMS Criteria	Toxicity Test SMS Designation	Designations Agree?	Final SMS Designation
EW-130	no exceedance	none	SQS[a]	no	SQS
EW-132	SQS	total PCBs	CSL	no	CSL
EW-133	CSL	**arsenic**, zinc, acenaphthene, anthracene, benzo(a)anthracene, benzo(a)pyrene, benzo(g,h,i)perylene, total benzofluoranthenes, chrysene, dibenzo(a,h)anthracene, fluoranthene, fluorene, indeno(1,2,3-c,d)pyrene, phenanthrene, total HPAHs, total LPAHs, dibenzofuran, total PCBs	CSL	yes	CSL
EW-134	SQS	total PCBs	CSL	no	CSL
EW-135	SQS	mercury, bis(2-ethylhexyl) phthalate, total PCBs	CSL	no	CSL
EW-136	SQS	mercury, total PCBs	CSL	no	CSL
EW-138	SQS	mercury, total PCBs	CSL	no	CSL
EW-141	CSL	**mercury**, total PCBs	SQS	no	SQS
EW-142	SQS	mercury, total PCBs	no exceedance	no	no exceedance
PDM-01	SQS	total PCBs	CSL	no	CSL
PDM-03	SQS	total PCBs	SQS	yes	SQS
PDM-06	CSL	**mercury**, total PCBs, total DDTs	CSL	yes	CSL
PDM-08	SQS	mercury	CSL	no	CSL
PDM-10	SQS	mercury	CSL	no	CSL
PDM-15	CSL	**acenaphthene**, fluorene, phenanthrene, total LPAHs, dibenzofuran, total PCBs	CSL	yes	CSL
EW09-SS-005	SQS	fluoranthene	no exceedance	no	no exceedance
EW09-SS-030	SQS	mercury	SQS	yes	SQS
EW09-SS-032	SQS	mercury	no exceedance	no	no exceedance
EW09-SS-033	SQS	mercury	CSL	no	CSL
EW09-SS-034	SQS	mercury	no exceedance	no	no exceedance
EW09-SS-035	SQS	mercury	CSL	no	CSL
EW09-SS-217	no exceedance	none	no exceedance	yes	no exceedance
EW09-SS-218	no exceedance	none	no exceedance	yes	no exceedance

Table A.6-2. Final SMS designations for locations with both chemical and site-specific toxicity test results (cont.)

Location ID	Chemical SMS Designation	COPCs with Concentrations that Exceeded SMS Criteria	Toxicity Test SMS Designation	Designations Agree?	Final SMS Designation
EW09-SS-220	SQS	acenaphthene, benzo(g,h,i)perylene, dibenzo(a,h)anthracene, fluoranthene, fluorene, indeno(1,2,3,-c,d)pyrene, phenanthrene, total LPAHs	no exceedance	no	no exceedance
EW09-SS-015	SQS	phenanthrene	SQS	yes	SQS
EW09-SS-215	CSL	**mercury**, zinc	no exceedance	no	no exceedance

[a] These locations had chemistry and bioassay results and were subsequently resampled for chemistry only. The chemistry results were consistent for the two samples and the bioassay result was retained from the initial analysis.

CSL – cleanup screening level

DDT – dichlorodiphenyltrichloroethane

EW – East Waterway

ID – identification

LPAH – low-molecular-weight polycyclic aromatic hydrocarbon

PCB – polychlorinated biphenyl

SMS – Washington State Sediment Management Standards

SQS – sediment quality standards

Bold identifies COPCs with detected concentrations greater than the CSL in that sample. COPCs not bolded had detected concentrations greater than the SQS but less than the CSL.

Uncertainties Associated with Sediment Risk Estimates

This section presents the 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 was selected as an ROC because the community encompasses infaunal and epibenthic benthic invertebrates as a functional group. This ROC addresses effects at the community level, reflecting the ecological functions that are achieved with diverse benthic invertebrate assemblages. This receptor group, the assessment endpoints (survival, growth, and reproduction), and the sediment regulatory framework (SMS criteria and DMMP guidelines) are aimed at ensuring the overall health of the community, and will likely be protective of most species.

AETs, which form the basis for SMS criteria and most of the DMMP guidelines, exist for 51 of the 63 chemicals that have been detected in EW surface sediments. Therefore, chemicals without such criteria, guidelines, or other relevant toxicity information or chemicals with guidelines not derived on the basis of sediment toxicity were not identified as COPCs during the problem formulation. There is some uncertainty regarding the risk to the benthic invertebrate community from these chemicals that could not be evaluated because of lack of sufficient toxicity information.

Exposure Assessment

Uncertainties in the sediment exposure assessment for the benthic invertebrate community were associated with the following factors.

Depth of biologically active zone – According to Ecology guidance for characterizing surface sediment under the SMS, the exposure potential and sediment unit of concern is the “biologically active zone” (often the top 10 cm). Past studies in Puget Sound have demonstrated that the majority of benthic macroinvertebrates are generally found within the uppermost 10 cm of the sediment (Ecology 2008). Although some species may be present at deeper depths below the sediment surface, 10 cm is generally assumed to represent a reasonable estimate of the sediment column where benthic organisms can be exposed to sediment contaminants. SPI data was used to provide site-specific information on the vertical distribution of benthic macroinvertebrates or the depth to anoxic sediments. Results from the recent SPI survey in the EW (Windward 2009a) indicate that the top 10 cm is a reasonable sediment horizon for assessing benthic invertebrate exposure in the EW.

Large clams (e.g., butter clams and geoducks) and some benthic invertebrate species will burrow deeper than 10 cm, which, as discussed above, is the depth to which the biologically active zone was assigned in the ERA. A risk characterization for these organisms could have a different outcome if concentrations in sediments between 0 and

10 cm were markedly different than those in the deeper sediments at which some clams and other benthic invertebrates have been found in the EW. However, the majority of these deeper organisms feed at or near the sediment surface or in the overlying water column, so the top 10 cm of the sediment column likely represent the depth at which these organisms would be most highly exposed to COPCs. In addition, water column exposures have been included as part of the surface water evaluation of risk to benthic invertebrates.

Relationships among sediment chemistry, toxicity, and actual *in situ* effects –The use of chemical criteria to assess *in situ* effects is uncertain. Factors such as site-specific chemical bioavailability, variable mixtures of chemicals, chemicals without criteria, and differing species-specific sensitivities may contribute to this uncertainty. Moreover, the SMS provide chemical-specific criteria to assess the risks from individual chemicals. Although these criteria were developed from field data in which mixtures of chemicals were common, the chemical-specific criteria are not intended to assess the cumulative risks to benthic invertebrates from exposure to multiple chemicals with potentially synergistic or antagonistic effects. The use of site-specific toxicity tests helps to reduce these uncertainties.

COPC screen – Eight chemicals were identified as COIs in sediment because they had no SMS criteria or DMMP guidelines (TBT, dibutyltin, monobutyltin, cobalt, molybdenum, vanadium, carbazole, and dioxins and furans) but were detected in more than 5% of the surface sediment samples. These chemicals were not selected as COPCs in sediment because of the lack of criteria or screening guidelines with which to evaluate the risk associated with sediment exposure. However, risk to benthic invertebrates from TBT exposure was considered using a tissue-residue evaluation, and risks from exposure to cobalt and molybdenum were considered using a surface water evaluation. The risk associated with exposure to these chemicals in surface sediment is uncertain.

RLs greater than criteria or guidelines – Nineteen chemicals had non-detected results with RLs greater than the corresponding SMS chemical criterion, or DMMP guideline, in at least one sediment sample in the baseline surface sediment dataset. Of these 19 chemicals, 4 were identified as benthic COPCs on the basis of the detected sediment concentrations. The RLs in each non-detect sample for each of these 19 chemicals were compared with the corresponding SQS/SL and CSL/ML chemical criteria and guidelines, and the results are presented in Table A.6-3. These RLs were evaluated relative to the SQS/SL and CSL/ML but were not classified as exceedances in the risk conclusions which could result in an underestimation of risk if the chemical is present at concentrations above the SQS/CSL and below the RL.

Table A.6-3. Summary of RLs that exceeded risk-based criteria

Chemicals with RLs that Exceeded SQS/SL	Detection Frequency	Maximum RL SQS/SL EF	No. of RL SQS/SL Exceedances	Maximum RL CSL/ML EF	No. of RL CSL/ML Exceedances
Phthalates					
Bis(2-ethylhexyl) phthalate	***207/231***	***2.8***	***2***	***1.7***	***1***
Butyl benzyl phthalate	101/231	1.9	6	ne	0
Other SVOCs					
1,2,4-Trichlorobenzene	7/231	9.3	41	4.2	13
1,2-Dichlorobenzene	2/231	3.3	11	3.3	11
1,3-Dichlorobenzene	2/214	1.1	1	ne	0
1,4-Dichlorobenzene	***146/231***	***1.3***	***2***	***ne***	***0***
2,4-Dimethylphenol	***14/231***	***17***	***36***	***17***	***36***
2-Methylphenol	6/231	3.0	3	3.0	3
Benzoic acid	3/231	3.1	10	3.1	10
Benzyl alcohol	2/231	3.3	8	2.6	3
Hexachlorobenzene	0/231	20	62	3.3	8
Hexachlorobutadiene	0/231	1.9	1	1.2	1
Pentachlorophenol	10/231	2.8	10	1.4	2
Pesticides					
Total DDTs	***8/143***	***39***	***70***	***3.9***	***4***
Aldrin	1/91	27	8	ne	0
Dieldrin	0/91	11	25	ne	0
gamma-BHC	0/91	2.0	5	ne	0
Total chlordane	1/91	10	14	ne	0
Heptachlor	0/91	2.0	5	ne	0

BHC – benzene hexachloride
CSL – cleanup screening level
DDT – dichlorodiphenyltrichloroethane
EF – exceedance factor
ML – maximum level
ne – no exceedance
RL – reporting limit
SL – screening level
SQS – sediment quality standard
SVOC – semivolatile organic compound

Bold and italics identifies COPCs for the benthic invertebrate community.

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 reflect the greater degree of analytical dilution required for quantification of other analytes, such as SVOC compounds (i.e., 2,4-dimethylphenol) in samples with elevated PAH concentrations. 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, and phenols are all more chemically reactive than the other SVOCs analyzed by EPA

Method 8270 (2003b). More-reactive compounds can be difficult to extract and often degrade during analysis.

The spatial distributions within EW sediment of the five chemicals with RLs that most often exceeded the SMS sediment chemical criteria or DMMP guidelines (i.e., total DDTs, hexachlorobenzene, 1,2,4-trichlorobenzene, 2,4-dimethyl phenol, and dieldrin) are presented on Maps A.6-6, A.6-7, A.6-8, A.6-9, and A.6-10. For all five chemicals, samples with RLs greater than the criteria or guidelines occurred throughout the EW with no spatial relationship between the detected results that were greater than the SQS/SL or CSL/ML chemical criteria and the RLs that were greater than the SQS/SL chemical criteria. Most of these chemicals were infrequently (< 7%) detected; dieldrin and hexachlorobenzene were never detected. There are 137 locations where an RL exceeded a criteria or guideline; at 112 (82%) of these locations there was a detected concentration that exceeded an SMS criteria or DMMP guideline. The remaining 25 locations had only RL exceedances, which were predominately associated with 1,2,4-trichlorobenzene (13 locations) and total DDTs (12 locations).

Spatial analysis using Thiessen polygons – Thiessen polygons were used in the ERA to estimate the areas of sediment within the EW that are potentially affected by exceedances of SMS criteria or DMMP guidelines. The area of each polygon is interpolated from a single data point from each sampling location. There is uncertainty associated with methods for the interpolation of point values to area values, including the Thiessen polygon method. Thiessen polygon boundaries are defined such that any arbitrary location within a polygon is closer to its associated sampling location than to any other sampling location. The chemical concentration within each Thiessen polygon is assumed to be uniformly distributed across the area of the polygon, so that the entire polygon has a single concentration for a given chemical. Given the spatial heterogeneity of anthropogenic chemicals in an aquatic environment, it is highly unlikely that this is the case; there is no way to know if the polygon concentration overestimates or underestimates actual conditions without additional sampling.

Effects Assessment

The uncertainty in the effects assessment for the benthic invertebrate community was associated with the use of SMS chemical and bioassay criteria or DMMP guidelines to assess the potential for a biological effect. These uncertainties are discussed below.

The likelihood of adverse effects on benthic organisms from chemicals detected in sediment was assessed using two approaches. In the first approach, surface sediment chemical concentrations were compared with risk-based chemical criteria, principally the SMS. In the second approach, site-specific sediment toxicity test results were compared with SMS biological effects criteria.

The SMS chemical criteria used in the first approach were developed from toxicity tests and community surveys using test species that represent a small but sensitive portion of the diverse benthic invertebrate community present in the EW. The toxicity test species included crustaceans and bivalve larvae, which are considered to represent taxonomic groups most sensitive to chemical exposure (Hyland et al. 1999). In addition, the benthic invertebrate community AETs, which address one of the endpoints of the SMS criteria and DMMP guidelines, were developed from benthic community metrics using Puget Sound data and incorporate the responses of invertebrates with wide ranges of feeding strategies and habitat requirements and therefore represent COPC concentrations likely to be protective of the benthic invertebrate community as a whole. However, there is some uncertainty associated with the benthic invertebrate community AETs because these values were based on the total abundances of several major benthic infaunal taxa (i.e., molluscs, crustaceans, and 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 EW benthic species may not be addressed by the benthic community AET criteria and guidelines. Consequently, there is some uncertainty associated with the risk estimates. It should be noted that SMS chemical criteria were developed for specific chemicals based on AETs empirically derived from a dataset of Puget Sound field-collected sediment samples that contained diverse chemical mixtures and 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 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[33] between 50 and 96% for benthic community, 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. The site-specific paired chemistry and bioassay data for EW sediment appear to have a lower reliability than those reported in the Puget Sound dataset in that the rate of disagreement between chemical and biological results was relatively high—approximately 69% of the paired sample designations did not agree (i.e., approximately 28% [n = 13] of the chemistry results underpredicted toxicity, and 41% [n = 20] of the chemistry results overpredicted toxicity).

Risk Conclusions for Sediment

In summary, 29 chemicals or groups of chemicals had at least one concentration in sediment that exceeded its respective SQS or SL and were therefore identified as COCs for the benthic invertebrate community. These chemicals include 4 metals, 16 individual PAHs or groups of PAHs, 3 phthalates, 4 other SVOCs, total PCBs, and total DDTs (Table A.6-1). The results of a spatial analysis using Thiessen polygons and the combination of chemistry and toxicity test results indicated that:

[33] 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.

- No adverse effects on benthic invertebrates living in intertidal and subtidal sediments were predicted for approximately 40% of the EW (i.e., the area in which the final SMS designation was less than or equal to the SQS/SL based on a combination of sediment chemistry and biological effects).
- There is a higher likelihood for adverse effects in approximately 21% of the EW (i.e., the area in which the final SMS designation was greater than the CSL/ML based on a combination of sediment chemistry and biological effects).
- The remaining 39% of the EW had a final SMS designation between the SQS and CSL, indicating the potential for minor adverse effects.

There is some uncertainty associated with these estimates because the areas were interpolated from individual points at which sediments were sampled within each polygon. The spatial extent of individual samples with exceedances of sediment 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.

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 and detected chemicals with exceedances of SQS criteria, the uncertainty associated with non-detected chemicals with RLs that exceeded their respective SQS chemical criteria is low.

A.6.1.1.2 Tissue-residue

This section presents risk estimates, uncertainties, and risk conclusions for the tissue-residue evaluation for the benthic invertebrate community. To characterize risk, hazard quotients (HQs) were calculated using the following equation:

$$HQ_{tissue} = \frac{EPC_{tissue}}{TRV_{tissue}}$$ **Equation 6-2**

Where:

HQ_{tissue} = hazard quotient for tissue residue
EPC_{tissue} = tissue–residue exposure point concentration (mg/kg)
TRV_{tissue} = tissue-residue toxicity reference value (mg/kg)

HQs were calculated based on both the NOAEL and LOAEL TRVs. A LOAEL HQ $\geq$ 1.0 is generally regarded as an indication of the potential for adverse effects because the benchmark is the effects concentration at which adverse effects were observed. The potential for adverse effects associated with a NOAEL HQ > 1.0 and a LOAEL HQ < 1.0 is considered low and uncertain because the true threshold for effects occurs at a concentration somewhere between the NOAEL and LOAEL. The potential for adverse effects is considered unlikely when the NOAEL HQ < 1.0.

Following the presentation of risks estimated by calculating HQs, the uncertainties in the exposure and effects data that could affect risk estimates for each of the COPCs are discussed. The risk conclusion section integrates risk estimates with associated uncertainties and identifies chemicals as COCs if LOAEL HQs were ≥ 1.0.

Risk Estimates

TBT and total PCBs were identified as COPCs for benthic invertebrates using the tissue-residue evaluation in the COPC screening process in Section A.2.5.1.2. The EPCs used for calculating HQs were concentrations in individual composite samples from each of the 13 sampling areas that included most of the EW (Map A.2-6).

LOAEL HQs were < 1.0 for all sampling areas for TBT and total PCBs, with the exception of Areas 3N and 5 for TBT (Tables A.6-4 and A.6-5). The LOAEL HQs > 1.0 for two areas (1.2 in Area 3N and 3.3 in Area 5) indicate a risk for the benthic invertebrate community, specifically gastropods, from exposure to TBT in those areas. However, all of the benthic invertebrate tissue TBT concentrations were below the tissue TBT concentration associated with reduced growth in polychaetes (0.54 mg/kg ww) (Meador and Rice 2001). For total PCBs, NOAEL HQs were > 1.0, and LOAEL HQs were < 1.0 in 10 of 13 sampling areas, with NOAEL HQs in those areas ranging from 1.4 to 3.5. These results indicate low and uncertain risks for the benthic invertebrate community from exposures to total PCBs in those areas.

Table A.6-4. HQ calculations for benthic invertebrates for the TBT tissue-residue evaluation

Sampling Area	EPC (mg/kg ww)	NOAEL HQ (NOAEL = 0.024 mg/kg ww)	LOAEL HQ (LOAEL = 0.12 mg/kg ww)
Area 2W	0.020	0.8	0.17
Area 3N	0.140	**5.8**	**1.2**
Area 3S	0.089	**3.7**	0.74
Area 4N	0.100	**4.2**	0.83
Area 4S	0.090	**3.8**	0.75
Area 5	0.390	**16**	**3.3**
Area 6	0.091	**3.8**	0.76
Area 8N	0.100	**4.2**	0.83
Area 8S	0.092	**3.8**	0.77
Area 9	0.088	**3.7**	0.73
Area 10N	0.057	**2.4**	0.48
Area 10S	0.098	**4.1**	0.82

EPC – exposure point concentration
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
NOAEL – no-observed-adverse-effect level
TBT – tributyltin
ww – wet weight

Bold and underline identify HQs > 1.0.

Table A.6-5. HQ calculations for benthic invertebrates for the PCB tissue-residue evaluation

Sampling Area	EPC (mg/kg ww)	NOAEL HQ (NOAEL = 0.11 mg/kg ww)	LOAEL HQ (LOAEL = 1.1 mg/kg ww)
Area 1	0.110	1.0	0.1
Area 2E	0.380	**<u>3.5</u>**	0.35
Area 2W	0.093	0.85	0.085
Area 3	0.240	**<u>2.2</u>**	0.22
Area 4	0.150 J	**<u>1.4</u>**	0.14
Area 5	0.290 J	**<u>2.6</u>**	0.26
Area 6	0.210	**<u>1.9</u>**	0.19
Area 7	0.260 J	**<u>2.4</u>**	0.24
Area 8	0.164	**<u>1.5</u>**	0.15
Area 9	0.230	**<u>2.1</u>**	0.21
Area 10	0.250	**<u>2.3</u>**	0.23
Area 11	0.180	**<u>1.6</u>**	0.16
Area 12	0.115	1.0	0.10

EPC – exposure point concentration
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
NOAEL – no-observed-adverse-effect level
PCB – polychlorinated biphenyl
ww – wet weight

<u>Bold and underline</u> identify NOAEL HQs > 1.0.

Uncertainties Associated with Tissue-residue Risk Estimates

This section discusses uncertainties associated with the exposure and effects assessment for the tissue-residue evaluation for benthic invertebrates. The primary uncertainties pertain to the type of tissue used in the evaluation and the tissue-based TRVs derived from literature toxicity data.

Benthic Invertebrate Tissue Data

The benthic invertebrate tissue samples were composites that contained multiple infaunal and epibenthic benthic invertebrate species (e.g., all organisms retained on a 1-mm mesh sieve excluding clams > 2 cm). Data from these samples thus represented the bioaccumulation of COPCs over a variety of species. In order to specifically address risks to clams from bioaccumulation of total PCBs and TBT, data on COPC concentrations in clams were compared with the benthic invertebrate tissue-residue TRVs, which were derived from a variety of invertebrates, including polychaetes, gastropods, mussels, oysters, clams, shrimp, and amphipods (Tables A.3-9 and A.3-10). EPCs were represented by the TBT or total PCB concentrations in each of the clam composite samples. Clams were composited by species and included butter clams (seven samples), Eastern soft-shell (two samples), little neck clams (two samples), cockles (two samples), and geoducks (four samples).[34] EPCs for clams (Table A.6-6) were generally lower than those for benthic invertebrate composite samples (Table A.6-4). One Eastern soft-shell clam tissue sample exceeded the TBT LOAEL TRV with an HQ of 1.2. However, the imposex endpoint for the TBT LOAEL is not directly relevant for clams. All of the clam tissue TBT concentrations were below the tissue TBT concentration associated with reduced growth in polychaetes (0.54 mg/kg ww) (Meador and Rice 2001). None of PCB NOAEL and LOAEL HQs for clams were > 1.0. These data indicate that the use of benthic invertebrate tissue data in the risk characterization approach is protective of clam species in the EW.

Table A.6-6. HQ calculations for clams based on the tissue-residue evaluation

COPC	EPC (mg/kg ww)[a]		NOAEL TRV (mg/kg ww)	LOAEL TRV (mg/kg ww)	NOAEL HQ		LOAEL HQ	
	Min	Max			Min	Max	Min	Max
TBT	0.0015	0.14	0.024	0.12	0.06	5.8	0.013	1.2
Total PCBs	0.0047	0.082	0.11	1.1	0.043	0.75	0.0043	0.075

[a] EPCs were calculated using data for butter clams, littleneck clams, Eastern soft-shell clams, cockles , and geoducks

COPC – chemical of potential concern
EPC – exposure point concentration
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
NOAEL – no-observed-adverse-effect level
PCB – polychlorinated biphenyl
TBT – tributyltin
TRV – toxicity reference value
ww – wet weight

TRVs

There is general uncertainty associated with the use of laboratory toxicity studies that used single-chemical exposures to estimate risk to organisms in the natural environment. Laboratory studies are conducted under controlled exposure environments using single contaminants. Effects associated with multi-chemical exposures and environmental stressors present in the environment were not factored into these studies. Exposure to mixtures of chemicals may result in the interaction of those chemicals. Exposure to mixtures of chemicals may result in antagonistic, synergistic, or additive effects. It is generally believed that the joint action of many complex mixtures is additive (Broderius 1991; Logan and Wilson 1995). The relevance of laboratory exposures to single chemicals to the exposure to mixtures of chemicals in the field is uncertain. Risk may be overestimated or underestimated.

Additional uncertainties associated with laboratory studies include how well the test species or life stage represents that of the benthic invertebrate community (which may underestimate or overestimate risks); the lack of endpoints other than reproduction,

[34] Intertidal clam samples and geoduck samples were collected for use in the HHRA. The geoduck data used in this uncertainty evaluation were for four samples for which whole-body concentrations were calculated based on separate data for edible meat and gutball. Details regarding these samples are presented in B.2.1.2 and Attachment 1 of the HHRA.

growth, or survival that could also result in adverse effects on the population (which may underestimate risks); and potentially large dosing intervals that may not accurately identify the true chemical threshold of effects (which may underestimate risks). Furthermore, laboratory studies typically use organisms that are more tolerant of non-chemical stressors, reproduce quickly and have short life spans and therefore may not represent the sensitivity or contaminant exposure duration of many field species.

The LOAEL and NOAEL TRVs for TBT are based on the imposex endpoint for gastropods and thus are not necessarily applicable to the remainder of the benthic invertebrate community. Therefore, there is uncertainty in the risks calculated for the benthic invertebrate community from TBT exposure (i.e., may overestimate risks for species other than gastropods).

There is some uncertainty in the NOAEL for PCBs because there was no NOAEL available below the lowest LOAEL, and an uncertainty factor of 10 was used to estimate the NOAEL. In addition, the sensitivity of shrimp (the species on which the lowest LOAEL and estimated NOAEL were based) appears to vary widely among the studies, with NOAELs for survival ranging from 1.3 to 18 mg/kg ww (Table A.3-10). Total PCB concentrations in EW benthic invertebrate tissue, which ranged from 0.093 to 0.380 mg/kg ww, did not exceed these higher NOAELs. An additional uncertainty is that the selected LOAEL TRV was based on a study in which shrimp were exposed to Aroclor 1016, an Aroclor that has not been detected in benthic invertebrate tissue or sediment collected from the EW.

Risk Conclusions

LOAEL HQs for the tissue-residue evaluation for benthic invertebrates were < 1.0 in all of the sampling areas for total PCBs. NOAEL HQs in most sampling areas were > 1.0 for PCBs. When clam tissue data were substituted for benthic invertebrate tissue data, all NOAEL and LOAEL HQs for total PCBs were < 1.0. There is some uncertainty associated with the NOAEL TRV for total PCBs. Risks to benthic invertebrates from total PCBs are low and uncertain in most of the sampling areas because NOAEL HQs were > 1.0. Because there were no LOAEL HQs > 1.0, total PCBs was not identified as a COC for the benthic invertebrate community based on the tissue-residue evaluation.

LOAEL TBT HQs for the tissue-residue evaluation were < 1.0 in all sampling areas except two (Areas 3N and 5). The TRV for TBT was based on imposex, which is an endpoint that specifically affects gastropods. Therefore, there is some uncertainty in the use of these toxicity data to characterize risks to the benthic invertebrate community. In conclusion, the risk to benthic invertebrates from TBT is low and uncertain in most areas of the EW, where NOAEL HQs were > 1.0 and LOAEL HQs were < 1.0. In Areas 3N and 5, where LOAEL HQs were > 1.0, there is a potential for risk to the benthic invertebrate community; therefore, TBT was identified as a COC based on the tissue-residue evaluation

A.6.1.1.3 Surface water

This section presents risk estimates, uncertainties, and risk conclusions for the surface water evaluation for the benthic invertebrate community. To characterize risk, HQs were calculated using the following equation:

$$HQ_{water} = \frac{EPC_{water}}{TRV_{water}} \qquad \textbf{Equation 6-3}$$

Where:

HQ_{water}	=	hazard quotient for surface water
EPC_{water}	=	surface water exposure point concentration (µg/L)
TRV_{water}	=	surface water toxicity reference value (µg/L)

The following subsections present of the risks estimated using the above equation and discuss the uncertainties in the exposure and effects data that could affect risk estimates for each of the COPCs. The risk conclusion section integrates risk estimates with associated uncertainties and identifies chemicals as COCs if HQs were > 1.0.

Risk Estimates

Cadmium, mercury, and TBT were identified as surface water COPCs for benthic invertebrates based on the COPC screen (Section A.2.5.1.3). Risks were estimated by calculating HQs using surface water EPCs that represented the bottom of the water column and chronic WQC that were identified as TRVs. EPCs were calculated on both a location-specific basis and on an individual sample basis. For the location-specific surface water evaluation, detected cadmium and TBT concentrations exceeded their respective TRVs at one location each (Table A.6-7; Map A.6-11), with HQs of 4.1 and 1.4, respectively. Location-specific EPCs for mercury never exceeded the TRV. TBT was not detected at any of the other five locations; however, the RLs at those locations exceeded the TRV for TBT. The detected cadmium concentration at location EW-SW-1 (37.8 µg/L) that exceeded the cadmium TRV of 9.3 µg/L is an anomalous value and is highly uncertain, as discussed below in the uncertainty section.

Table A.6-7. HQs for the benthic invertebrate community based on location specific surface water EPCs that represent the bottom of the water column

COPC	Location	Surface Water EPC (µg/L)[a]	TRV[b]	HQ
Cadmium (dissolved)	EW-SW-1	37.8	9.3	**<u>4.1</u>**
	EW-SW-2	0.073		0.0079
	EW-SW-3	0.079		0.0085
	EW-SW-4	0.044		0.0095
	EW-SW-5	0.091		0.0098
	EW-SW-6	0.074		0.0080
	HNF/E	0.0757		0.0081
	HNF/W	0.0751		0.0081
Mercury (dissolved)	EW-SW-1	0.0011	0.94	0.0012
	EW-SW-2	0.00034		0.00036
	EW-SW-3	0.00044		0.00047
	EW-SW-4	0.00021		0.00044
	EW-SW-5	0.00021		0.00044
	EW-SW-6	0.00021		0.00044
TBT	EW-SW-1	0.010[c]	0.0074	**<u>1.4</u>**[c]
	EW-SW-2	0.010		**<u>1.4</u>**
	EW-SW-3	0.010[c]		**<u>1.4</u>**[c]
	EW-SW-4	0.010[c]		**<u>1.4</u>**[c]
	EW-SW-5	0.010[c]		**<u>1.4</u>**[c]
	EW-SW-6	0.010[c]		**<u>1.4</u>**[c]

a Location-specific EPCs were equal to maximum detected concentrations when fewer than six samples were available (see Table A.3-3).

b TRV for cadmium (dissolved fraction) was based on the Washington State marine chronic WQC; TRVs for mercury (dissolved fraction) and TBT were based on federal marine chronic WQC.

c TBT was not detected in any samples at these locations, so EPCs and HQs for TBT at these locations were based on maximum RLs.

COPC – chemical of potential concern
EPC – exposure point concentration
HQ – hazard quotient
RL – reporting limit
TBT – tributyltin
TRV – toxicity reference value
WQC – water quality criteria
<u>Bold and underline</u> identify HQs > 1.0.

As noted in the exposure assessment, EPCs based on detected COPC concentrations in individual water samples collected from the bottom of the water column were used to represent conditions at that location at the time of sampling. Cadmium and TBT each had an EPC that exceeded the chronic WQC in one surface water sample; the HQs were 4.1 and 1.4, respectively (Table A.6-8; Map A.6-11). No other dissolved cadmium concentrations and none of the dissolved mercury concentrations in the individual samples exceeded WQC. The single detected dissolved cadmium concentration that exceeded the chronic WQC was 37.8 µg/L. As noted above, this cadmium detection is an anomalous value and is highly uncertain, as discussed below in the uncertainty section. The single detected TBT concentration that exceeded the chronic WQC was 0.01 µg/L. RLs for TBT in all other samples exceeded the chronic WQC.

Table A.6-8. HQs for the benthic invertebrate community based on EPCs for individual water samples that represent the bottom of the water column

COPC	Range of EPCs (µg/L)	TRV (µg/L)[a]	Range of HQs	Number of HQs > 1.0
Cadmium (dissolved)	0.055 – 37.8	9.3	0.00059 – <u>**4.1**</u>	1
Mercury (dissolved)	0.0001 – 0.0277	0.94	0.00010 – 0.029	0
TBT	0.01[b]	0.0074	<u>**1.4**</u>	1

a TRV for cadmium (dissolved fraction) was based on the Washington State marine chronic WQC; TRVs for mercury (dissolved fraction) and TBT were based on federal marine chronic WQC.

b Only one value is shown for TBT because there was only one detected concentration; thus, the EPC is equal to the single detected concentration.

COPC – chemical of potential concern
EPC – exposure point concentration
HQ – hazard quotient
TBT – tributyltin
TRV – toxicity reference value
WQC – water quality criteria
<u>**Bold and underline**</u> identify HQs > 1.0.

Uncertainties Associated with Surface Water Risk Estimates

This section discusses uncertainties associated with the COPC selection and exposure and effects assessment for the surface water evaluation for benthic invertebrates. The primary uncertainties are the COPC screen for naphthalene and the cadmium surface water data.

COPC Screen for Naphthalene

The COPC surface water screen for the benthic invertebrate community (Section A.2.5.2.3) used the Tier II values developed by Suter and Tsao (1996) for naphthalene because there are no state or federal WQC. Concentrations of naphthalene in surface water did not exceed the Tier II values so naphthalene was not selected as a COPC. However, concentrations of naphthalene in porewater did exceed the Tier II values, so a TRV was derived for porewater as described in Section A.3.2.4. This section compares the surface water data to the TRV derived for porewater and also discusses uncertainty associated with the Tier II values.

Tier II values were developed by Suter and Tsao (1996) for chemicals that lack the minimum species diversity requirements for calculating federal WQC. To derive Tier II values, GMAVs were calculated as for the federal WQC method (see Section A.3.2.3.1). The secondary acute value (SAV) was then calculated from the lowest GMAV divided by the final acute value factor (FAVF), which varied depending on the number of minimum species diversity requirements met. The SAV was then divided by the secondary acute-chronic ratio (SACR) to calculate the secondary chronic value (SCV), which was the Tier II chronic criterion. The SACR was based on the geometric mean of at least three ACRs. If three empirical ACRs were not available, a generic ACR of 17.9 was used until the total number of ACRs is three. For example, for naphthalene, one empirical ACR of 12.8 was available, so the final SACR of 16.0 was calculated as the geometric mean of 12.8, 17.9, and 17.9. Based on this process, the chronic and acute Tier II values for naphthalene were calculated as 12 and 190 µg/L, respectively. These values were calculated using data from only four studies with four different species of aquatic organisms (two daphnia species, rainbow trout, and fathead minnow). There is some uncertainty in using the Tier II values for the benthic invertebrate community because of the small dataset for the toxicity of naphthalene to aquatic organisms and because none of these organisms were benthic invertebrates.

The NOEC and LOEC TRVs derived from toxicity data found in the ECOTOX database were 0.16 and 8 µg/L, respectively, as described in the porewater effects assessment in Section A.3.2.4. Only one of the 59 surface water samples analyzed for naphthalene had a detected concentration (12 µg/L) that exceeded the LOEC TRV. This sample and one additional sample had detected concentrations (12 and 2.4 µg/L) that exceeded the NOEC TRV. All of the remaining 57 samples had detected concentrations or reporting limits less than or equal to the NOEC TRV. The LOEC TRV for naphthalene was selected for evaluating risk to the benthic invertebrate community because it is the lowest LOEC within in a large dataset that contains toxicity data for 20 aquatic invertebrates. The LOEC TRV of 8 µg/L is at least two orders of magnitude lower than other effect concentrations for aquatic invertebrates, which ranged from 800 to 100,000 µg/L.

There is also uncertainty in the use of the naphthalene NOEC TRV inasmuch as it was estimated using a generic uncertainty factor of 50 to convert the acute LOEC to a NOEC. This generic factor of 50 is likely conservative based on information presented in Suter and Tsao (1996). In that document, one ACR of 12.77 was available for naphthalene in surface water based on a study with fathead minnow. In addition, a default value of 17.9 is used as the ACR for aquatic organisms when no species-specific data are available. These ACRs of 12.77 and 17.9 indicate that the generic factor of 50 used to derive the NOEC from the LOEC is high and therefore may result in an overestimate of risk.

In conclusion, despite the uncertainty in the Tier II values for naphthalene, the comparison of concentrations in surface water with the NOEC and LOEC TRVs indicate a low risk to the EW benthic invertebrate community from exposure to naphthalene in surface water.

Cadmium Data

One dissolved cadmium concentration (37.8 µg/L) in the surface water dataset exceeded the WQC value of 9.3 µg/L. This sample was collected as part of the SRI in September 2008 at location SW-1 at the southern end of the EW. The dissolved cadmium concentration in this sample was substantially higher than that in any of the other samples collected as part of the SRI in 2008 or collected by King County in 1996 and 1997, all of which had concentrations ≤ 0.091 µg/L. This concentration of 37.8 µg/L appears to be an anomalous value for two reasons. First, the total cadmium concentration of 1.45 µg/L in the same sample was more than an order of magnitude lower than the dissolved concentration of 37.8 µg/L. The majority of the other dissolved cadmium concentrations in the ERA surface water dataset were less than the total concentrations in the same sample; in the few cases in which the dissolved concentrations were greater, the difference was within the range of analytical variance for cadmium. Second, a field duplicate that was collected at the same location on the same date had dissolved and total cadmium concentrations of 0.076 and 0.074 µg/L, respectively. These data indicate that there are analytical concerns with the dissolved cadmium concentration of 37.8 µg/L and that this value is anomalous and unlikely representative of conditions in EW.

TBT Reporting Limits

Only 1 of the 59 surface water samples analyzed for TBT had a detected concentration. The single detected concentration of 0.01 µg/L slightly exceeded the WQC of 0.0074 µg/L. The reporting limits for the non-detected results for TBT ranged from 0.008 to 0.01 µg/L, concentrations that are all slightly greater than the WQC. Therefore, there is uncertainty in the TBT risk characterization for the 58 samples in which TBT was not detected because the reporting limits exceed the WQC.

Risk Conclusions

In summary, potential risks to the benthic invertebrate community from surface water exposures were evaluated based on exceedances of surface water TRVs for the COPCs cadmium, mercury, and TBT:

- Dissolved cadmium concentrations for one location-specific EPC and for one individual sample EPC representing the bottom of the water column exceeded the surface water TRV (i.e., the Washington State chronic WQC). There is high uncertainty associated with the HQ of 4.1 for both the location-specific and individual sample EPCs because this dissolved cadmium concentration is considered anomalous.
- None of the location-specific or individual sample EPCs for dissolved mercury that represent the bottom of the water column exceeded the surface water TRV (i.e., the federal chronic WQC).

- TBT was detected in only 1 of 31 surface water samples from the bottom of the water column. One location-specific and one individual sample EPC exceeded the surface water TRV (i.e., the federal chronic WQC), each with an HQ of 1.4; TBT was undetected in the remaining 30 samples at reporting limits that slightly exceeded the WQC.

The single dissolved cadmium concentration that exceeded the Washington State marine chronic WQC was an anomalously high value based the following information: 1) the concentration was substantially higher than concentrations any of the other 130 samples, 2) the field duplicate sample collected at the same location on the same date had a substantially lower concentration, and 3), the total cadmium concentration in the same sample was more than an order of magnitude lower than the dissolved cadmium concentration. Dissolved cadmium concentrations in all other 130 surface water samples were lower than the WQC, indicating that the EW benthic invertebrate community will not be adversely affected from exposure to cadmium in surface water. Therefore, cadmium was not identified as a COC for the benthic invertebrate community based on the surface water evaluation.

Mercury was not identified as a COC for the benthic invertebrate community for the surface water pathway because it did not exceed the federal chronic WQC in any surface water samples collected from the bottom of the water column.

For TBT, the one detected concentration and reporting limits for the 30 undetected concentrations were slightly greater than the federal marine chronic WQC. This WQC of 0.0074 µg/L is based on protection of sensitive gastropod species that are susceptible to imposex and very low TBT concentrations. TBT was rarely detected in surface water, which suggests a low risk. However, the fact that the RLs were higher than criteria values resulted in uncertainty with respect to the surface water TBT concentrations. Because of this uncertainty, TBT was identified as a COC for the benthic invertebrate community for the surface water evaluation.

A.6.1.1.4 Porewater

This section presents risk estimates, uncertainties, and risk conclusions for the porewater evaluation for benthic invertebrates. To characterize risk, HQs were calculated using the following equation:

$$HQ_{porewater} = \frac{EPC_{porewater}}{TRV_{porewater}} \qquad \textbf{Equation 6-4}$$

Where:

$HQ_{porewater}$ = hazard quotient for porewater
$EPC_{porewater}$ = porewater exposure point concentration (mg/kg)
$TRV_{porewater}$ = porewater toxicity reference value (mg/kg)

HQs were calculated based on both the NOEC and LOEC TRVs. A LOEC HQ ≥ 1.0 is generally regarded as an indication of the potential for adverse effects, because the benchmark is an effects concentration at which adverse effects were observed. The potential for adverse effects associated with a NOEC HQ > 1.0 and a LOEC HQ < 1.0 is considered low and uncertain because the true threshold for effects occurs at a concentration somewhere between the NOEC and LOEC.

Following the presentation of risks estimated by calculating HQs, uncertainties in the exposure and effects data that could affect risk estimates for each of the COPCs are discussed. The risk conclusion section integrates risk estimates with associated uncertainties and identifies chemicals as COCs if LOEC HQs were ≥ 1.0.

Risk Estimates

Naphthalene was the only porewater COPC identified for benthic invertebrates based on the COPC screen (Section A.2.5.1.4). Naphthalene was detected in two porewater samples; both of these samples were collected from Area 4 (Map A.2-7). The two detected concentrations were 3.4 and 48 µg/L, as compared with the NOEC and LOEC TRVs of 0.16 and 8 µg/L, respectively. The two detected concentrations both exceeded the NOEC TRV, with HQs of 21 and 300 (Table A.6-9). The LOEC TRV was exceeded in one of these samples, with an HQ of 6; the location of this sample is shown on Map A.6-11.

Table A.6-9. Summary statistics for TRV exceedances in individual porewater samples

COPC	Number of Samples	Number of Detects	NOEC TRV		LOEC TRV	
			Number of Exceedances	HQs	Number of Exceedances	HQ
Naphthalene	12	2	2	**21 and 300**	1	**6**

LOEC – lowest-observed-effect concentration
na – not applicable
NOEC – no-observed-effect concentration
TRV – toxicity reference value
Bold and underline identify a NOEC or LOEC HQ > 1.0.

Uncertainties Associated with Porewater Risk Estimates for Naphthalene

As discussed in the previous section for surface water, the available data indicate that the NOEC TRV of 0.16 µg/L is a conservative value that likely overestimates risk to the benthic invertebrate community because it was derived from a generic uncertainty factor of 50, and other studies indicate a smaller difference between LOECs and NOECs of the same study (e.g., Suter and Tsao (1996) show an ACR of 12.77 for naphthalene).

In addition, the sediment data for naphthalene indicate that this chemical is not likely to pose a wide-spread risk to the benthic invertebrate community in the EW. Naphthalene was analyzed in porewater as a VOC. VOCs were the only class of chemicals analyzed in porewater because of their volatility and low affinity for adsorption onto sediment particles; exposure to other analytes was considered sufficiently represented by concentrations in sediment. Naphthalene is a low-molecular-weight PAH with a greater affinity for sediment than most VOC chemicals, and SQS and CSL values for naphthalene are available for comparison with sediment data. Out of the 240 sediment samples analyzed for naphthalene in the EW there were 118 samples in which naphthalene was detected. None of the detected concentrations exceeded the SQS. Naphthalene was not detected in the two sediment samples collected closest to porewater sampling Area 4 (EW09-SS-002 and EW09-SS-004). The sediment data indicate that naphthalene is not likely to pose a wide-spread risk to the benthic invertebrate community in the EW.

Risk Conclusions for Porewater

The LOEC HQ of 6 for naphthalene at one of these porewater sampling locations indicates a localized risk to benthic invertebrates. Naphthalene was identified as a COC for benthic invertebrates for the porewater evaluation because the LOEC HQ was exceeded at one porewater location and the benthic invertebrate assessment evaluated risk on a smaller scale then site-wide.

A.6.1.1.5 Summary of risk conclusions for the benthic Invertebrate community

Risks to the benthic invertebrate community were evaluated through four different approaches: sediment, tissue-residue, surface water, and porewater. The results of the sediment risk characterization, which used Thiessen polygons and a combination of chemistry and toxicity test results, indicated that:

- In approximately 40.4% of the EW (i.e., the area for which the final SMS designation was less than or equal to the SQS/SL based on a combination of sediment chemistry and biological effects), no adverse effects from exposure to contaminated sediment were predicted for benthic invertebrates living in intertidal and subtidal sediment.
- There is a higher likelihood for adverse effects from sediment exposure for benthic invertebrates in approximately 21.2% of the EW (i.e., the area for which the final SMS designation was greater than the CSL/ML based on a combination of sediment chemistry and biological effects).
- The remaining 38.4% of the EW had a final SMS designation between the SQS/SL and CSL/ML, indicating the potential for minor adverse effects from sediment exposure for benthic invertebrates.
- Twenty-nine chemicals were identified as COCs for the benthic invertebrate community based on the surface sediment chemistry evaluation

In addition, TBT was identified as a COC based on the tissue-residue evaluation; and TBT and naphthalene were identified as COCs based on the surface water and porewater evaluations, respectively (Table A.6-10). Risks to the benthic community from exposures to TBT in surface water throughout EW are uncertain because only one detected concentration exceeded WQC and the RLs were exceeded for the other samples. Only one localized area is expected to pose a risk to the benthic invertebrate community from exposure to naphthalene based on the porewater evaluation as well as the sediment and surface water evaluations.

Table A.6-10. Chemicals identified as COCs for the benthic invertebrate community

Chemical	Type of Evaluation			
	Surface Sediment	Tissue Residue	Surface Water	Porewater
Metals				
Arsenic	X			
Cadmium	X			
Mercury	X			
Zinc	X			
Organometals				
TBT		X	X	
PAHs				
2-Methylnaphthalene	X			
Acenaphthene	X			
Benz(a)anthracene	X			
Benzo(a)pyrene	X			
Benzo(g,h,i)perylene	X			
Chrysene	X			
Dibenzo (a,h)anthracene	X			
Dibenzofuran	X			
Fluoranthene	X			
Fluorene	X			
Indeno (1,2,3,-c,d)pyrene	X			
Naphthalene				X
Phenanthrene	X			
Pyrene	X			
Total benzofluoranthenes	X			
Total HPAH	X			
Total LPAH	X			
Other SVOCs				
Bis(2-ethylhexyl) phthalate	X			
Butyl benzyl phthalate	X			
Di-n-butyl phthalate	X			

Table A.6-10. Chemicals identified as COCs for the benthic invertebrate community (cont.)

Chemical	Type of Evaluation			
	Surface Sediment	Tissue Residue	Surface Water	Porewater
1,4-Dichlorobenzene	X			
2,4-Dimethylphenol	X			
n-Nitrosodiphenylamine	X			
Phenol	X			
PCBs				
Total PCBs	X			
Organochlorine Pesticides				
Total DDTs	X			

COPC – chemical of potential concern'
DDT – dichlorodiphenyltrichloroethane
HPAH – high-molecular-weight polycyclic aromatic hydrocarbon
LPAH – low-molecular-weight polycyclic aromatic hydrocarbon
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
SVOC – semivolatile organic compound
TBT – tributyltin
VOC – volatile organic compound

A.6.1.2 Crab

The risk characterization results for the tissue-residue and surface water evaluations are provided in the following subsections, each of which presents a risk estimate and an uncertainty analysis. Each subsection also discusses risk conclusions based on an application of the uncertainty to the risk estimate.

A.6.1.2.1 Tissue-residue

This section presents risk estimates, uncertainties, and risk conclusions for the tissue-residue evaluation for crab. To characterize risk, HQs were calculated using the following equation:

$$HQ_{tissue} = \frac{EPC_{tissue}}{TRV_{tissue}} \quad \textbf{Equation 6-5}$$

Where:

HQ_{tissue} = hazard quotient for tissue residue
EPC_{tissue} = tissue–residue exposure point concentration (mg/kg)
TRV_{tissue} = tissue-residue toxicity reference value (mg/kg)

HQs were calculated based on both the NOAEL and LOAEL TRVs. A LOAEL HQ ≥ 1.0 is generally regarded as an indication of the potential for adverse effects because the benchmark is the effects concentration at which adverse effects were observed. The potential for adverse effects associated with a NOAEL HQ > 1.0 and a LOAEL HQ < 1.0 is considered low and uncertain because the true threshold for effects occurs at a concentration somewhere between the NOAEL and LOAEL. For COPCs with NOAEL HQs < 1.0, effects are considered unlikely.

The following subsections present the risks estimated using the above equation and discuss uncertainties in the exposure and effects data that could affect risk estimates for each of the COPCs. The risk conclusion sections integrate risk estimates with associated uncertainties and identifies chemicals as COCs if LOAEL HQs were ≥ 1.0.

Risk Estimates

Based on the COPC screen, arsenic, cadmium, copper, zinc, and total PCBs were identified as COPCs for crab using the tissue-residue evaluation (Section A.2.5.1.5). The EPCs used for calculating HQs for crab were the 95% UCLs calculated using all nine crab composite samples collected throughout the EW. All NOAEL HQs were > 1.0 (Table A.6-11). LOAEL HQs for cadmium, copper, and zinc were ≥ 1.0 (1.4, 1.1 and 1.5, respectively); the LOAEL HQs for arsenic and total PCBs were < 1.0.

Table A.6-11. HQs for the tissue-residue evaluation for crab

COPC by Crab Species	EPC (mg/kg ww)	TRV (mg/kg ww)		HQ	
		NOAEL	LOAEL	NOAEL	LOAEL
Arsenic	5.19	1.28	21	**4.1**	0.25
Cadmium	3.61	0.6	2.6	**6.0**	**1.4**
Copper	29.1	2.6	26	**11.2**	**1.1**
Zinc	53.4	12.7	35.2	**4.2**	**1.5**
Total PCBs	0.45	0.11	1.1	**4.1**	0.41

COPC – chemical of potential concern
EPC – exposure point concentration
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
NOAEL – no-observed-adverse-effect level
PCB – polychlorinated biphenyl
TRV – toxicity reference value
ww – wet weight

Bold identifies NOAEL HQs > 1.0 and LOAELs ≥ 1.0.

Uncertainties Associated with Crab Tissue-Residue Risk Estimates

This section discusses uncertainties associated with the identification of COPCs and exposure and effects assessment for the tissue-residue evaluation for crab. The two primary uncertainties are the lack of toxicity data for some chemicals as part of the COPC screen and the TRVs derived for COPCs from toxicity data in the scientific literature.

COPC Screen

TRVs were not available for cobalt, lead, molybdenum, nickel, dibutyltin, selenium, dibenzofuran, dieldrin, phenol, and 17 individual PAHs for the COPC screen for the tissue-residue evaluation for crab. Therefore, risks to crab from exposure to these chemicals could not be evaluated because of the absence of toxicity data.

Exposure Concentrations

Whole-body crab concentrations were calculated from the edible meat and hepatopancreas concentrations, as described in Attachment 1. The uncertainty associated with the calculation is expected to be low. The accuracy of the measurements was evaluated and determined to be acceptable, and therefore the accuracy of the final value should be acceptable. The only portion of the crab that was not analyzed was the shell.

TRVs

As discussed in the benthic invertebrate tissue-residue evaluation uncertainty analysis, there is general uncertainty associated with the use of laboratory toxicity data to estimate risk to organisms in the natural environment. Laboratory studies are conducted under controlled exposure environments using single contaminants. Effects associated with multi-chemical exposures and environmental stressors present in the environment were not factored into these studies. There is uncertainty associated with TRVs based on single-chemical exposures and effects may be overestimated or underestimated based on potential combined effects of chemicals on crab in the EW.

Some other uncertainties associated with laboratory studies include how well the test species or its life stage represents that of crab (which may underestimate or overestimate risks); the lack of endpoints other than reproduction, growth, or survival that could also result in adverse effects on the crab population (which may underestimate risks); and potentially large dosing intervals that may not capture the actual chemical threshold of effects (which may underestimate risks). Furthermore, laboratory studies typically use organisms that are more tolerant of non-chemical stressors, reproduce quickly and have short life spans and therefore may not represent the sensitivity or contaminant exposure duration of many field species.

The tissue-residue TRVs selected for crab were assigned a level of uncertainty (i.e., low, medium, or high) based on the number of studies, types of endpoints, and species evaluated (Table A.6-12). The TRVs for arsenic and zinc are considered highly uncertain because of the small number of studies that included only the survival endpoint and used decapod species other than crab. Information on the sensitivity of crab to arsenic and zinc compared to the sensitivity of other decapods was not available. TRVs for cadmium, copper, and total PCBs have less uncertainty than do those for arsenic and zinc because a greater number of studies were conducted with several decapod species; although, with the exception of one study for copper that evaluated growth effects in prawn, only the survival endpoint was measured.

Table A.6-12. Level of uncertainty associated with tissue-residue TRVs for crab

COPC	No. of TRV Studies	Level of Uncertainty in TRV[a]	Rationale for Level of Uncertainty
Arsenic	2	high	The number of studies was small, and the studies included only the survival endpoint; neither study used crab as test species so data for other decapods were used.
Cadmium	9	medium	There were nine studies, but these studies had only survival endpoints; data were available for three crab species.
Copper	4	medium	Four studies had growth and survival endpoints; no data were available for crab species so data for other decapods were used; NOAEL was estimated using an uncertainty factor
Zinc	1	high	Only one study was available; the single study evaluated only the survival endpoint and did not use crab as a test species.
Total PCBs	5	medium	Five studies included only the survival endpoint; data were available for two crab species; NOAEL was estimated using an uncertainty factor

COPC – chemical of potential concern
PCB – polychlorinated biphenyl
TRV – toxicity reference value

A TRV was available for cadmium in crab muscle tissue, as presented in Section A.3.4.1.2 (Table A.3-14). Because edible meat tissue data are available for EW crab, these exposure data were compared with the muscle tissue NOAEL and LOAEL TRVs of 4.9 and 9.5 mg/kg ww, respectively, which were derived from Jennings and Rainbow (1979). Cadmium was detected in all nine composite crab edible tissue samples at concentrations ranging from 0.09 to 0.98 mg/kg ww. These concentrations are substantially lower than the muscle tissue NOAEL and LOAEL TRVs.

Risk Conclusions

Results of the risk characterization and uncertainty analysis for crab using the tissue-residue evaluation are summarized in Table A.6-13. Arsenic NOAEL HQs were > 1.0, and LOAEL HQs were ≤ 1.0, indicating that adverse effects on crab from exposure to arsenic are low and uncertain because the risk threshold lies between the NOAEL and LOAEL TRVs. There is also high uncertainty associated with the TRVs; it is unknown if this uncertainty is associated with an overestimate or underestimate of risks to crab in the EW. Based on this analysis, arsenic is not a COC for crab because LOAEL HQs were not > 1.0.

Table A.6-13. Summary of risk characterization for crab

COPC	HQ NOAEL	HQ LOAEL	Uncertainty	Selected as a COC?
Arsenic	4.1	0.25	There is high uncertainty in the TRVs because only survival was evaluated, and no test organisms were crab.	no
Cadmium	6.0	**<u>1.4</u>**	There is medium uncertainty in the TRVs because data were available for crab, but survival was the only endpoint evaluated.	yes
Copper	11.2	**<u>1.1</u>**	There is medium uncertainty in the TRVs because data were available for growth and survival but not for crab.	yes
Zinc	4.2	**<u>1.5</u>**	There is high uncertainty in the TRVs because only one study was available, and it evaluated survival in crayfish.	yes
Total PCBs	4.1	0.41	There is medium uncertainty in the TRVs because data were available for crab, but survival was the only endpoint evaluated.	no

COPC – chemical of potential concern
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
NOAEL – no-observed-adverse-effect level
PCB – polychlorinated biphenyl
TRV – toxicity reference value
<u>Bold and underline</u> identify LOAEL HQs > 1.0.

Cadmium, copper, and zinc NOAEL and LOAEL HQs were ≥ 1.0, indicating a potential for adverse effects. There is some uncertainty in the TRVs for cadmium and copper and high uncertainty in the TRV for zinc, but the effects on risk conclusions are unknown. Based on this analysis, cadmium, copper, and zinc were identified as a COCs for crab.

For total PCBs, the NOAEL HQ was > 1.0 and the LOAEL HQ was < 1.0, indicating a low potential for adverse effects, with some uncertainty associated with the risk threshold, which is between the NOAEL and LOAEL. There is also some uncertainty associated with the tissue-residue TRV for total PCBs, but it is not known how this uncertainty affects risk conclusions. Total PCBs is not identified a COC for crab because the LOAEL HQ did not exceed 1.0.

A.6.1.2.2 Surface water

This section presents risk estimates, uncertainties, and risk conclusions for the surface water evaluation for crab. To characterize risk, HQs were calculated using the following equation:

$$HQ_{water} = \frac{EPC_{water}}{TRV_{water}} \qquad \textbf{Equation 6-6}$$

Where:

HQ_{water} = hazard quotient for surface water
EPC_{water} = surface water exposure point concentration (µg/L)
TRV_{water} = surface water toxicity reference value (µg/L)

The following subsections present of the risks estimated using the above equation and discuss the uncertainties in the exposure and effects data that could affect risk estimates for each of the COPCs. The risk conclusion section integrates risk estimates with associated uncertainties and identifies chemicals as COCs if HQs were > 1.0.

Risk Estimates

Cadmium, mercury, and TBT were identified as surface water COPCs for crab based on the COPC screen (Section A.2.5.1.3). HQs were calculated using surface water EPCs that represent the bottom of the water column and surface water TRVs. EPCs were calculated on both a site-wide basis and on an individual sample basis. For the site-wide surface water evaluation, none of the HQs were > 1.0 (Table A.6-14).

Table A.6-14. HQs for crab based on site-wide surface water EPCs that represent the bottom of the water column

COPC	Site-Wide Surface Water EPC (µg/L)	TRV (µg/L)[a]	HQ
Cadmium (dissolved)	3.1	9.3	0.33
Mercury (dissolved)	0.00040	0.94	0.00043
TBT	0.01	0.066	0.15

[a] TRV for cadmium (dissolved) was based on the Washington State marine chronic WQC, the TRV for mercury (dissolved) was based on federal marine chronic WQC, and the TRV for TBT was based on the federal marine FCV.

COPC – chemical of potential concern
EPC – exposure point concentration
FCV – final chronic value
HQ – hazard quotient
TBT – tributyltin
TRV – toxicity reference value

As noted in the exposure assessment, EPCs based on detected COPC concentrations in individual water samples collected from the bottom of the water column were used to represent conditions at each location at the time of sampling. Cadmium was the only COPC with an EPC that exceeded the TRV; only one detected cadmium concentration (37.8 µg/L) exceeded the chronic WQC (Table A.6-15; Map A.6-11). As noted above, this cadmium detection is an anomalous value and is highly uncertain, as discussed below in the uncertainty section.

Table A.6-15. HQs for crab based on EPCs for individual water samples that represent the bottom of the water column

COPC	Range of EPCs (µg/L)	TRV (µg/L)[a]	Range of HQs	Number of HQs > 1.0
Cadmium (dissolved)	0.055 – 37.8	9.3	0.00059 – <u>**4.1**</u>	1
Mercury (dissolved)	0.0001 – 0.0277	0.94	0.00010 – 0.029	0
TBT	0.01[b]	0.066	0.15	0

a TRV for cadmium (dissolved fraction) was based on the Washington State marine chronic WQC, TRV for mercury (dissolved fraction) was based on federal marine chronic WQC, and the TRV for TBT was based on the federal marine FCV.

b Only one value is presented for TBT because there was only one detected concentration; thus, the EPC is equal to the single detected concentration.

COPC – chemical of potential concern
EPC – exposure point concentration
FCV – final chronic value
HQ – hazard quotient
TBT – tributyltin
TRV – toxicity reference value
UCL – upper confidence limit on the mean
WQC – water quality criteria
WQS – water quality standards

<u>**Bold and underline**</u> identify HQs > 1.0.

Uncertainty

Primary uncertainties associated with the surface water COPC selection and exposure and effects assessment for crab are the COPC screen for naphthalene and the cadmium surface water data. These uncertainties as the same as those for the surface water evaluation for benthic invertebrates and were discussed in detail in Section A.6.1.2.1.

Risk Conclusions

In summary, potential risks to crab from surface water exposures were evaluated based on exceedances of surface water TRVs for the COPCs cadmium, mercury, and TBT:

- Dissolved cadmium concentrations for one location-specific EPC and for one individual sample EPC representing the bottom of the water column exceeded the surface water TRV (i.e., the Washington State chronic WQC). There is high uncertainty associated with the HQ of 4.1 for both the location-specific and individual sample EPCs because this dissolved cadmium concentration is considered anomalous.
- Site-wide and individual sample EPCs for dissolved mercury that represent the bottom of the water column did not exceed the surface water TRV (i.e., the federal chronic WQC).
- Site-wide and individual sample EPCs for TBT that represent the bottom of the water column did not exceed the surface water TRV (i.e., the federal FCV).

The single dissolved cadmium concentration that exceeded the Washington State marine chronic WQC was an anomalously high value based the following information: 1) the concentration was substantially higher than concentrations any of the other 130 samples, 2) the field duplicate sample collected at the same location on the same date had a substantially lower concentration, and 3), the total cadmium concentration in the same sample was more than an order of magnitude lower than the dissolved cadmium concentration. Dissolved cadmium concentrations in all other 130 surface water samples were lower than the WQC, indicating that the EW benthic invertebrate community will not be adversely affected from exposure to cadmium in surface water. Therefore, cadmium was not identified as a COC for the benthic invertebrate community based on the surface water evaluation.

Neither mercury nor TBT were not identified as COCs for the benthic invertebrate community for the surface water evaluation because they did not exceed their surface water TRVs in any bottom water samples.

A.6.1.2.3 Summary of risk conclusions for crab

Risks to crab were evaluated based on the tissue-residue and surface water exposure. Based on the tissue-residue evaluation, cadmium, copper, and zinc were identified as COCs for crab. No chemicals were identified as COCs for crab based on the surface water evaluation.

A.6.2 FISH

This section presents the risk characterization and uncertainty analysis for each of the three fish ROCs for the EW: juvenile Chinook salmon, English sole, and brown rockfish. To characterize risk, HQs were calculated using the following generic equation for each type of evaluation (i.e., tissue-residue, dietary, and surface water):

$$HQ = \frac{EPC}{TRV}$$ **Equation 6-7**

Where:

HQ = hazard quotient
EPC = exposure point concentration (mg/kg)
TRV = toxicity reference value (mg/kg)

For the dietary and tissue evaluations, HQs are calculated based on both the NOAEL and LOAEL TRVs. A LOAEL HQ ≥ 1.0 is generally regarded as an indication of the potential for adverse effects because the benchmark is the effects concentration at which adverse effects are observed. The potential for adverse effects associated with a NOAEL HQ > 1.0 and a LOAEL HQ < 1.0 is considered low and uncertain because the true threshold for effects occurs at a concentration somewhere between the NOAEL and LOAEL. A NOAEL HQ ≤ 1.0 indicates that adverse effects are unlikely.

For the surface water evaluation, the TRV was based on a chronic value. An HQ > 1.0 indicates the potential for adverse effects based on chronic exposure, whereas an HQ ≤ 1.0 indicates that adverse effects are unlikely.

The following subsections present the risks estimated using the above equations to calculate HQs and discuss uncertainties in the exposure and effects data that could affect risk estimates for each of the COPCs. Finally, risk conclusions that integrate risk estimates with associated uncertainties are presented for each ROC, resulting in a determination of which chemicals are considered COCs for fish.

A.6.2.1 Juvenile Chinook salmon

This section presents risk estimates, uncertainties, and risk conclusions for juvenile Chinook salmon. Juvenile Chinook salmon were evaluated as an ROC to represent all migratory juvenile salmonids in the EW; they were also selected because they have been listed as threatened under the ESA (Section A.2.2.1). Because they are a listed species, adverse effects are evaluated for individuals rather than the population. Therefore, to protect individual juvenile Chinook salmon, chemicals were identified as dietary COCs in the risk conclusion section if NOAEL HQs were > 1.0, which is more protective than identifying chemicals as COCs if LOAEL HQs were ≥ 1.0. In addition, chemicals were selected as COCs for the surface water evaluation based on individual samples if the HQs were ≥ 1.0 based on chronic TRVs. Risks to juvenile Chinook salmon were addressed by considering the effects of reduced growth and survival.

A.6.2.1.1 Risk estimates

Arsenic, cadmium, chromium, copper, and vanadium were identified as COPCs in Section A.2.5.2.2 for juvenile Chinook salmon based on the dietary evaluation. No COPCs were identified for this ROC as a result of the tissue-residue evaluation. The surface water COPC screen identified cadmium, mercury, and TBT as COPCs for the surface water exposure of juvenile Chinook salmon (Section A.2.5.1.3).

Two types of data were used for the dietary evaluation: COPC concentrations in a single composite sample of juvenile Chinook salmon stomach contents, and COPC concentrations in juvenile Chinook salmon prey (i.e., the 95% UCL of all benthic invertebrate composite samples collected throughout the EW).

NOAEL and LOAEL HQs calculated using stomach contents data were ≤ 1.0 for all COPCs except cadmium (Table A.6-16). For cadmium, the NOAEL and LOAEL HQs were 4.9 and 1.0, respectively.

NOAEL and LOAEL HQs calculated using benthic invertebrate prey data were > 1.0 for all COPCs except arsenic. NOAEL HQs ranged from 1.0 to 12, and LOAEL HQs ranged from 0.67 to 2.4; the highest HQs were for cadmium. A LOAEL HQ could not be calculated for chromium because no LOAEL TRV was available.

Table A.6-16. HQ calculations for juvenile Chinook salmon

Type of Dietary Data	COPC	EPC (mg/kg dw)	TRV (mg/kg dw)		HQ	
			NOAEL	LOAEL	NOAEL	LOAEL
Stomach contents	arsenic	3.55	20	30	0.18	0.12
	cadmium	0.488	0.1	0.5	**<u>4.9</u>**	**<u>1.0</u>**
	chromium	1.59	9.42	na	0.17	na
	copper	17.3	50	100	0.35	0.17
	vanadium	1.47	2.04	10.2	0.72	0.14
Benthic invertebrate tissue	arsenic	20	20	30	1.0	0.67
	cadmium	1.2	0.1	0.5	**<u>12</u>**	**<u>2.4</u>**
	chromium	29	9.42	na	**<u>3.1</u>**	na
	copper	110	50	100	**<u>2.2</u>**	**<u>1.1</u>**
	vanadium	19	2.04	10.2	**<u>9.3</u>**	**<u>1.9</u>**

COPC – chemical of potential concern
EPC – exposure point concentration
dw – dry weight
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
na – not available (no LOAEL was available from the scientific literature for chromium)
NOAEL – no-observed-adverse-effect level
TRV – toxicity reference value
ww – wet weight
<u>Bold and underline</u> identify NOAEL HQs ≥ 1.0 and LOAEL HQs ≥ 1.0.

For the site-wide surface water evaluation, none of the site-wide EPCs exceeded the surface water TRVs; thus, all HQs were < 1.0 (Table A.6-17). Results of the surface water evaluation using individual samples, which was conducted as a more conservative analysis to represent conditions at each location at the time of sampling, are presented in Table A.6-18 and on Map A.6-11. Only one COPC concentration in an individual sample exceeded a surface water TRV; the single detected dissolved cadmium concentration (37.8 µg/L) exceeded the marine chronic WQC of 9.3 µg/L. This cadmium detection is an anomalous value and is highly uncertain, as discussed below in the uncertainty section.

Table A.6-17 HQs for juvenile Chinook salmon based on site-wide surface water EPCs

COPC	Site-Wide Surface Water EPC (µg/L)	TRV (µg/L)[a]	HQ
Cadmium (dissolved)	0.94	9.3	0.10
Mercury (dissolved)	0.00039	0.94	0.00041
TBT	0.01[b]	0.066	0.15

a TRV for cadmium (dissolved fraction) was based on the Washington State marine chronic WQC, the TRV for mercury (dissolved fraction) was based on federal marine chronic WQC, and the TRV for TBT was based on the federal marine FCV.

b A 95% UCL was not calculated for TBT because there was only one detected concentration; thus, the EPC is equal to the single detected concentration.

COPC – chemical of potential concern

EPC – exposure point concentration

FCV – final chronic value

HQ – hazard quotient

TBT – tributyltin

TRV – toxicity reference value

Table A.6-18. HQs for fish based on EPCs for individual water samples

COPC	Range of EPCs (µg/L)	TRV (µg/L)[a]	Range of HQs	Number of HQs > 1.0
Cadmium (dissolved)	0.009 – 37.8	9.3	0.00097 – **<u>4.1</u>**	1
Mercury (dissolved)	0.00013 – 0.00146	0.94	0.00014 – 0.0016	0
TBT	0.01[b]	0.066	0.15	0

a TRV for cadmium (dissolved fraction) was based on the Washington State marine chronic WQC, the TRV for mercury (dissolved fraction) was based on federal marine chronic WQC, and the TRV for TBT was based on the federal marine FCV.

b Only one value is shown for TBT because there was only one detected concentration; thus, the EPC is equal to the single detected concentration.

COPC – chemical of potential concern

EPC – exposure point concentration

FCV – final chronic value

HQ – hazard quotient

TBT – tributyltin

TRV – toxicity reference value

<u>Bold and underline</u> identify HQs > 1.0.

A.6.2.1.2 Uncertainty analysis

This section presents a discussion of the uncertainty associated with specific components of the problem formulation, the exposure and effects assessments, and the risk characterization for juvenile Chinook salmon.

COPC Screen

Eleven metals, eighteen individual PAH compounds, and dibenzofuran were identified as COIs as a result of the COI screen for fish. TRVs were available for all of the metals except cobalt and nickel. For the PAHs, TRVs were available for benzo(a)pyrene (Kim et al. 2008) and for a PAH mixture containing 21 PAH compounds (Meador et al. 2006); of the 18 PAHs listed as fish COIs, 12 were included in the PAH mixture and 6 were not. No TRV was found for dibenzofuran.

Because of the lack of TRVs for cobalt, nickel, and dibenzofuran, risks to juvenile Chinook salmon from these COIs could not be evaluated. The risks from PAHs other than benzo(a)pyrene are uncertain because six of the PAHs identified as COIs were not included in the PAH mixture in Meador et al. (2006). However, there are no data to indicate that the toxicity of those six PAHs to fish would be substantially higher than that of the other PAH compounds included in the mixture. In addition, the maximum dietary EPC for total PAHs (16 mg/kg dw) is an order of magnitude less than the NOAEL TRV for the PAH mixture (324 mg/kg dw [Table A.2-35]). Therefore, it is estimated that uncertainty associated with the COPC screen for individual PAHs is low.

Dietary exposure calculations

Dietary exposures for juvenile Chinook salmon were evaluated using composite samples of benthic invertebrates and salmon stomach contents as two lines of evidence. For the stomach contents data, only cadmium had a NOAEL HQ > 1.0; whereas for the benthic invertebrate data, four metals(cadmium, chromium, copper, and vanadium) had NOAEL HQs > 1.0 and three (cadmium, copper and vanadium) had LOAEL HQs> 1.0. The arsenic NOAEL HQ was equal to one. The uncertainties associated with both lines of evidence are discussed below.

There is uncertainty associated with the use of benthic invertebrate prey data to represent COPC concentrations in the juvenile Chinook salmon diet. Juvenile Chinook salmon also ingest water column organisms such as zooplankton, larval fish, and terrestrial insects that drift in the current (Cordell et al. 1996; 1997; 1999), in addition to infaunal benthic invertebrates. Water column prey are less closely associated with sediment than are benthic invertebrates and are less likely to have contaminant body burdens that reflect sediment exposure.

Another uncertainty associated with using the benthic invertebrate data to estimate dietary concentrations of COPCs for juvenile Chinook salmon is that the samples were composites of specimens collected from subtidal areas (mostly in deep-water areas) within the EW, and juvenile Chinook salmon generally do not forage in deep-water habitats (Tabor et al. 2004). The use of composite benthic invertebrate tissue samples is uncertain because preferential feeding in one area or a subset of the organisms that were included in the composites could result in exposures that could be either overestimated or underestimated by the 95th UCL of the composite samples.

Since Chinook salmon are a federally listed species, it is relevant to consider adverse effects at the level of the individual as well as the population. In this risk evaluation, HQ's were determined using the 95% UCL as the EPC for the benthic invertebrate tissue samples. The most conservative estimate of the exposure of an individual would be the maximum measured concentration [35]. Use of the maximum value would change the HQ exceedance conclusions only for arsenic since the EPCs based on the 95th UCL concentrations for the other trace metals already exceed their respective LOAELs. However, the use of the maximum concentration is based on the unlikely assumption that the entire diet is based on the consumption of subtidal, infaunal benthic invertebrates in one portion of the waterway.

Use of the stomach contents data is uncertain as this represents the diet of juvenile Chinook salmon in the EW at one point in time. Only one composite sample of stomach contents could be analyzed because of the limited sample mass collected. Thus, the measured stomach contents concentrations do not necessarily reflect the maximum concentration to which an individual could be exposed. However, this sample does provide a good representation of the average stomach contents for fish in the EW at the time of the collection event because of the large number of fish (n = 146[36]) whose stomach contents were composited in this sample.

The arsenic EPC is uncertain because arsenic speciation as organic or inorganic arsenic was not measured. The inorganic form of arsenic is the more toxic form of arsenic. The TRV was based on dietary dose of inorganic arsenic and the EPC is based on total arsenic. Therefore, the EPC most likely overestimates the dose of inorganic arsenic."

Surface Water Data

There is uncertainty in one dissolved cadmium concentration of 37.8 µg/L, as described in detail in the uncertainty section for the benthic invertebrate community (Section A.6.1.1.3). This concentration appears to be an anomalous value for two reasons: 1) the total cadmium concentration in the sample was more than an order of magnitude lower than the dissolved concentration, and 2) the field duplicate had dissolved and total concentrations of 0.076 and 0.074 µg/L, respectively. Excluding this anomalous dissolved cadmium concentration, the next highest concentration was 0.091 µg/L, which is below the Washington State marine chronic WQC of 0.94 µg/L.

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[35] For all five metals, the maximum measured concentration was greater than the 95th UCL, reflecting the low variability within the benthic invertebrate tissue dataset. The coefficient of variation (CV) values for all the benthic invertebrate tissue metals concentrations except chromium were less than 0.5 (arsenic CV=0.32, cadmium CV= 0.40, copper CV= 0.26 and vanadium CV = 0.30). The CV for chromium was 0.66

[36] A total of 165 fish were sampled for stomach contents. Nineteen fish had no measurable stomach contents and 146 fish contributed mass to the composite sample. The individual stomach content masses ranged from 0.01 – 0.502g with an average of 0.05g of stomach contents per fish.

TRVs

Uncertainty associated with toxicity studies available for fish may affect the risk conclusions. There is general uncertainty associated with the use of laboratory toxicity studies that used single-chemical exposures to estimate risks to organisms in the natural environment. Laboratory studies are conducted under controlled exposure environments using single contaminants. Effects associated with multi-chemical exposures and environmental stressors present in the environment were not factored into these studies. Exposure to mixtures of chemicals may result in the interaction of those chemicals. Exposure to mixtures of chemicals may result in antagonistic, synergistic, or additive effects. It is generally believed that the joint action of many complex mixtures is additive (Broderius 1991; Logan and Wilson 1995). The relevance of laboratory exposures to single chemicals to the exposure to mixtures of chemicals in the field is uncertain. Risk may be overestimated or underestimated.

Some other uncertainties associated with laboratory studies include how well the test species or life stage represents that of juvenile Chinook salmon (which may underestimate or overestimate risks); the lack of endpoints other than growth or survival that could also result in adverse effects on the fish population (which may underestimate risks); and potentially large dosing intervals that may not capture the actual chemical threshold of effects (which may underestimate risks). Furthermore, laboratory studies typically use organisms that are more tolerant of non-chemical stressors, reproduce quickly and have short life spans and therefore may not represent the sensitivity or contaminant exposure duration of many field species.

In addition, TRVs are considered less certain if there are a small number of studies, if endpoints are subchronic, or if data quality is questionable. The relative uncertainty in the selected TRVs for juvenile Chinook salmon are discussed below and summarized in Table A.6-19.

NOAEL and LOAEL TRVs for arsenic were selected based on a review of six studies, several of which reported similar LOAELs. Most studies were conducted with rainbow trout, so uncertainty is likely low for juvenile Chinook salmon because of the species similarity.

NOAEL and LOAEL TRVs for cadmium were selected based on a review of nine studies. Although a relatively large 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 that reported 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 and are more relevant in the evaluation of risk to juvenile Chinook salmon, as discussed in greater detail in the following subsection.

Table A.6-19. Level of uncertainty associated with dietary TRVs for juvenile Chinook salmon

COPC	No. of TRV Studies	Level of Uncertainty in TRV[a]	Rationale for Level of Uncertainty
Arsenic	6	low	The lowest LOAEL was based on a study conducted using a salmonid species (rainbow trout), and five of the six studies were chronic exposures conducted using rainbow trout
Cadmium	9	high	The lowest LOAEL was based on a study conducted using a non-salmonid species (juvenile rockfish) and was two orders of magnitude lower than those in the other six studies, five of which were conducted using salmonid species.
Chromium	1	high	Only one study was available, and no effects were observed at any of the dietary concentrations (i.e., no LOAEL).
Copper	15	medium	Large dataset included chronic studies with salmonids; selected LOAEL TRV was for a study conducted using channel catfish and was lower than the lowest LOAEL for any study that was conducted using a salmonid species (Atlantic salmon).
Vanadium	1	high	Only one study was available.

[a] Level of uncertainty in TRV was based in general on size of dataset, number of endpoints, type of study (i.e., acute or chronic), or species; other chemical-specific uncertainties were also considered.

COPC – chemical of potential concern

LOAEL – lowest observed adverse effect level

NOAEL – no observed adverse effect level

TRV – toxicity reference value

For chromium, only one study was available and only a NOAEL was reported; no effects were observed in this study. There is significant uncertainty associated with forming risk conclusions based on a NOAEL with no associated LOAEL (i.e., an unbounded NOAEL).

NOAEL and LOAEL TRVs for copper were selected based on a review of 15 studies, which is a relatively large number of studies. The lowest LOAEL reported for the growth of channel catfish (8 mg/kg dw) (Murai et al. 1981) was lower than the NOAELs for channel catfish growth reported in two other studies (Gatlin and Wilson 1986; Erickson et al. 2003), so Murai et al. (1981) was not selected for the derivation of the TRV. The available toxicity data suggest that the likely dietary threshold for toxic effects in salmonids (700 mg/kg dw) is higher than the selected dietary LOAEL (100 mg/kg dw), as discussed in greater detail in the following subsection.

The NOAEL and LOAEL TRVs for vanadium are highly uncertain because they were based on only one study. The effect of this uncertainty on the risk conclusions is unknown.

Selection of Salmon-Specific TRVs for Cadmium and Copper

As discussed in Section A.4.2.2.2, the selected LOAEL and NOAEL TRVs for cadmium and copper were based on reduced growth of brown rockfish; however, adverse effects in salmonids (i.e., rainbow trout) have only been observed at much higher dietary concentrations. These data indicate that the selected TRVs may result in the over-prediction of risk for juvenile Chinook salmon.

The only salmonid-specific LOAEL reported for cadmium was 1,395 mg/kg dw for reduced survival of rainbow trout fry exposed to dietary cadmium for 30 days (Szebedinsky et al. 2001) (Table A.6-20). Salmonid-specific dietary NOAELs were reported in five studies and ranged 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 786 mg/kg dw for reduced survival of rainbow trout fry exposed to dietary cadmium for 30 days (Szebedinsky et al. 2001).

Table A.6-20. Cadmium dietary toxicity studies for salmonids

Chemical	Test Species	NOAEL (mg/kg dw)[a]	LOAEL (mg/kg dw)[a]	Exposure Duration	Effect	Source
Cadmium chloride	rainbow trout fry	55	na	60 days	no effect on body weight, length, or survival	Mount et al. (1994)
Cadmium	Atlantic salmon	250[b]	na	4 weeks	no effect on growth rate (body weight)	Lundebye et al. (1999)
Cadmium chloride	juvenile rainbow trout	294[b]	na	15 – 30 days	no effect on growth rate or survival	Baldisserotto et al. (2005)
Cadmium chloride	juvenile rainbow trout	471	na	28 days	no effect on growth rate or survival	Franklin et al. (2005)
Cadmium nitrate	juvenile rainbow trout	786[b]	1,395[b]	30 days	57% survival	Szebedinsky et al. (2001)

[a] Concentrations are for elemental cadmium.

[b] Dietary dose was not reported as wet weight or dry weight and was assumed to be a dry-weight concentration.

na – not available

NOAEL – no-observed-adverse-effect level

Because five of the studies that were available for cadmium evaluated the dietary toxicity to salmonids, and in these studies no adverse effects on growth or survival were observed at dietary concentrations ranging from 55 to 786 mg/kg dw, it is unlikely that adverse effects would be observed in salmonids at lower concentrations. Assuming a NOAEL of 786 mg/kg dw (the highest NOAEL below the LOAEL), the NOAEL HQs were < 0.005 calculated using either exposure evaluation (benthic invertebrate or stomach contents data). Based on this analysis, risk to juvenile Chinook salmon from cadmium is likely to be very low.

For copper, salmonid-specific LOAELs were reported in four studies and ranged from 700 to 868 mg/kg dw for reduced growth in Atlantic salmon fry exposed to copper for 3 months (Berntssen et al. 1999a; Lundebye et al. 1999) (Table A.6-21). Salmonid-specific dietary NOAELs were reported in nine studies and ranged from 200 mg/kg dw for the survival of rainbow trout exposed to copper for 32 days (Handy 1992) to 1,042 mg/kg dw for the survival of rainbow trout fry exposed to copper for 28 days (Kamunde et al. 2001) (Table A.6-21). These data indicate that the effects of dietary copper occur at a substantially higher concentration for salmonids (700 mg/kg dw) than for brown rockfish (100 mg/kg dw). Using the highest NOAEL below the LOAEL (691.3 mg/kg dw), the HQs were 0.2 and 0.02 based on the benthic invertebrate and stomach contents data, respectively. This salmonid-specific analysis indicates that risk to juvenile Chinook salmon from copper exposure in the diet is likely to be very low.

Table A.6-21. Copper dietary toxicity studies for salmonids

Chemical	Test Species	NOAEL (mg/kg dw)[a]	LOAEL (mg/kg dw)[a]	Exposure Duration	Effect	Source
Copper sulfate	rainbow trout (138 g)	200	na	32 days	no effect on survival	Handy (1992)
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 growth	Berntssen et al. (1999b)
Copper sulfate pentahydrate	Atlantic salmon fry	500	700	3 months	reduced growth	Lundebye et al. (1999)
Copper sulfate pentahydrate	juvenile rainbow trout	287	730	8 weeks	reduced growth	Lanno et al. (1985b)
Copper sulfate pentahydrate	juvenile rainbow trout	730	na	8 weeks	no effect on survival	Lanno et al. (1985b)
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)
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 growth	Mount et al. (1994)
Copper sulfate pentahydrate	juvenile rainbow trout	1,042	na	28 days	no effect on survival or growth	Kamunde et al. (2001)

[a] Concentrations are for elemental copper.

dw – dry weight

LOAEL – lowest-observed-adverse-effect level

na – not available

NOAEL – no-observed-adverse-effect level

Summary of Uncertainties

Uncertainties in the problem formulation, the effects and exposure assessments, and risk characterization for juvenile Chinook salmon are summarized as follows:

- There is uncertainty associated with the use of benthic invertebrate tissue data to estimate dietary COPC concentrations. These data are uncertain because water column prey (e.g., zooplankton and larvae) may be a significant component in the diet of juvenile salmonids in EW. Water column prey are less closely associated with sediment than are benthic invertebrates and are less likely to have contaminant body burdens that reflect sediment exposure.
- The stomach contents data have uncertainty because they represent the diet of juvenile Chinook salmon in the EW during a single collection event, and thus may not represent a typical diet. Additionally, because of the sample mass limitations, the stomach contents from a large number of fish (n =146) were combined prior to analysis and therefore the stomach contents data represents the average exposure of the individual fish.
- The arsenic EPC calculated from the benthic invertebrate tissue data is uncertain because the speciation of the arsenic as organic or inorganic arsenic was not measured. Inorganic arsenic is the more toxic form of arsenic. The TRV was based on a dietary dose of inorganic arsenic and the EPC is based on total arsenic. Therefore, the EPC most likely overestimates the dose of inorganic arsenic.
- The maximum measured benthic invertebrate tissue metals concentrations exceeded the 95th UCL for all five metals that were identified as COPCs due to the relatively low variance in the tissue concentrations of these metals.
- There is uncertainty associated with the highest detected surface water concentration for dissolved cadmium; a comparison of total and dissolved concentrations in this sample, as well as comparison of the original sample with the field duplicate samples, indicates that this value is an anomaly. The next highest concentration was below the Washington state marine chronic WQC.
- Risks to juvenile Chinook salmon from exposure to cobalt, nickel, and dibenzofuran could not be evaluated because TRVs were not available. Therefore, it is not known whether these COPCs pose a risk to juvenile Chinook salmon.
- There are uncertainties associated with using laboratory effects data to estimate risk to fish in the EW, resulting in either overestimates or underestimates of risk.
- Estimated risks from dietary chromium exposure are highly uncertain because they were based on a NOAEL from a study in which no effects were observed; no other chromium toxicity studies were available, so a LOAEL could not be derived.

- The cadmium LOAEL TRV is highly uncertain because it is substantially lower than LOAELs reported in eight other studies, and the observed growth effect was partially attributed to food avoidance rather than toxicity. Salmonid-specific studies indicated that the NOAEL and LOAEL TRVs for salmonids are substantially higher that the selected LOAEL TRV. Using the NOAEL from salmonid studies resulted in a NOAEL HQ of < 0.05 for both benthic invertebrate and stomach contents data, indicating that risk to juvenile Chinook salmon from cadmium dietary exposure is unlikely.
- There is uncertainty in the LOAEL TRV selected for copper because the toxicity study results indicated that the LOAEL may be substantially higher for salmonid species. Using the NOAEL from salmonid studies resulted in NOAEL HQs of 0.2 and 0.02 based on benthic invertebrate data and stomach contents data, respectively, indicating that risk to juvenile Chinook salmon from copper dietary exposure is unlikely.
- The vanadium TRV was highly uncertain because only one study was found, and it is not known whether risks to juvenile Chinook salmon were overestimated or underestimated because of this uncertainty.
- There are uncertainties associated with the use of cadmium and TBT WQC to evaluate risk to fish because the values were derived with an emphasis on the protection of snails (i.e., for TBT) and other marine invertebrates.

A.6.2.1.3 Risk conclusions

The results of the risk characterization for juvenile Chinook salmon based on the dietary evaluation are summarized in Table A.6-22. The primary uncertainty associated with all NOAEL and LOAEL HQs is the use of benthic invertebrate data to estimate dietary COPC concentrations. Stomach contents data are considered to be more representative of the COPCs ingested by juvenile Chinook salmon than are benthic invertebrate data.

NOAEL HQs calculated using stomach contents data were all < 1.0 for all COPCs except cadmium. When the NOAEL HQs for chromium, copper, and vanadium were calculated using the benthic invertebrate data, the HQs were all > 1.0 (Table A.6-22). However, the dietary evaluation based on benthic invertebrate tissue concentrations is less certain than that based on the stomach contents data. In addition, because of uncertainty in the application of the selected fish TRVs to salmonids, NOAEL HQs based on stomach contents data for cadmium and copper are expected to be lower than those presented in Table A.6-22. It is not known how uncertainties associated with the TRVs for chromium and vanadium would affect the NOAEL HQs, but results from the stomach contents analysis indicated that risks to juvenile Chinook salmon are unlikely. Therefore, based on the available data and the uncertainty evaluation, risks to juvenile Chinook salmon from exposure to arsenic, chromium, copper, and vanadium in the diet are unlikely. None of these four chemicals were identified as COCs because none of the NOAEL HQs were > 1.0 based on the stomach contents data.

Table A.6-22. Summary of risk characterization for juvenile Chinook salmon for the dietary evaluation

COPC	HQ Based on Benthic Invertebrate Data		HQ Based on Stomach Contents Data		Primary Uncertainty	Selected as a COC?
	NOAEL	LOAEL	NOAEL	LOAEL		
Arsenic	**1.0**	0.67	0.18	0.12	Benthic invertebrate data do not represent the water column portion of the juvenile Chinook salmon diet which may have lower contaminant concentrations.	no
Cadmium	**<u>12</u>**[a]	**<u>2.4</u>**	**<u>4.9</u>**[a]	**<u>1.0</u>**	Benthic invertebrate data do not represent the water column portion of the juvenile Chinook salmon diet which may have lower contaminant concentrations; high uncertainty in TRV for cadmium; data suggest higher TRV for salmonids	**yes**
Chromium	**<u>3.1</u>**	na	0.17	na	Benthic invertebrate data do not represent the water column portion of the juvenile Chinook salmon diet which may have lower contaminant concentrations; no effects were observed in the one chromium study, so the NOAEL for chromium is highly uncertain	no
Copper	**<u>2.2</u>**	**<u>1.1</u>**	0.35	0.17	Benthic invertebrate data do not represent the water column portion of the juvenile Chinook salmon diet which may have lower contaminant concentrations; data suggest higher TRV for salmonids	no
Vanadium	**<u>9.3</u>**	**<u>1.9</u>**	0.72	0.14	Benthic invertebrate data do not represent the water column portion of the juvenile Chinook salmon diet which may have lower contaminant concentrations; high uncertainty in TRV for vanadium because only one study was available	no

[a] The NOAEL HQs calculated using an alternative salmonid NOAEL of 55 mg/kg dw were < 0.05 for both benthic invertebrate data and stomach contents data.

COC – chemical of concern
COPC – chemical of potential concern
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
NOAEL – no-observed-adverse-effect level
TRV – toxicity reference value
<u>Bold and underline</u> identify NOAEL HQs ≥ 1.0 and LOAEL HQs ≥ 1.0.

The NOAEL HQ for cadmium calculated using stomach contents data and based on a NOAEL that was extrapolated from the LOAEL TRV (0.1 mg/kg dw) was 4.9. Therefore, cadmium was identified as a COC for juvenile Chinook salmon. However, there is high uncertainty associated with the cadmium LOAEL TRV (0.5 mg/kg dw) because it is substantially lower than concentrations associated with observed adverse effects in the other eight studies, and the observed growth effect was partially attributed to food avoidance rather than toxicity. Using the lowest reported NOAEL (55 mg/kg dw), which was from a salmonid study, the NOAEL HQ calculated using the stomach contents data was 0.01. These data indicate there is a very low risk to juvenile Chinook salmon from exposure to cadmium in the diet.

The results of the risk characterization for juvenile Chinook salmon for the surface water evaluation based on both site-wide and individual EPCs are summarized in Table A.6-23. HQs were < 1.0 based on site-wide EPCs, and only the cadmium HQ was > 1.0 based on individual EPCs. However, cadmium was not identified as a COC because the concentration resulting in the exceedance was an anomaly, and it is highly unlikely that it represents the actual concentration of cadmium in the sample in which it was reported.

Table A.6-23. Summary of the risk characterization for juvenile Chinook salmon for the surface water evaluation

COPC	HQ Based on Site-Wide EPC	HQ Based on Maximum Individual EPC	Primary Uncertainty	Selected as a COC?
Cadmium	0.10[a]	**<u>4.1</u>**[a]	Concentration in the single sample that exceeded the chronic WQC was an anomaly.	no
Mercury	0.0041[a]	0.0016	Relatively low uncertainty.	no
TBT	0.15	0.15	WQC is driven by risk to snails and other marine invertebrates; therefore the FCV was used to evaluate risk to fish.	no

[a] HQ was calculated using dissolved surface water concentrations and dissolved TRV.

COC – chemical of concern

COPC – chemical of potential concern

EPC – exposure point concentration

FCV – final chronic value

HQ – hazard quotient

TBT – tributyltin

WQC – water quality criteria

<u>Bold and underline</u> identify HQs > 1.0.

A.6.2.2 English sole

This section presents the risk estimates, uncertainties, and risk conclusions for English sole. Chemicals were identified as COCs in the risk conclusions if LOAEL HQs for the tissue-residue and dietary evaluations and the chronic HQs for the surface water evaluation were ≥ 1.0.

A.6.2.2.1 Risk estimates

This section presents the HQ calculations for English sole. COPCs evaluated for English sole were as follows:

- TBT and total PCBs for the tissue-residue evaluation
- Arsenic, cadmium, chromium, copper, vanadium, and benzo(a)pyrene for the dietary evaluation
- Cadmium, mercury, and TBT for the surface water evaluation

The LOAEL HQ for TBT in tissue was < 1.0 (Table A.6-24). For PCBs in tissue, a range of LOAEL TRVs was used because of the uncertainty in the study from which the TRVs were derived (as discussed in both the effects section [Section A.4.2.1.3] and uncertainty section [Section A.6.2.2.2]). The LOAEL HQs for PCBs were >1.0, ranging from 1.6 to 7.9. For the dietary COPCs, LOAEL HQs for arsenic and benzo(a) pyrene were < 1.0 (Table A.6-24). LOAEL HQs for cadmium, copper, and vanadium were > 1.0, ranging from 1.1 to 2.4. A LOAEL HQ for chromium could not be calculated because there was no TRV, and the NOAEL HQ for chromium (3.1) was highly uncertain, as discussed in the uncertainty analysis (Section A.6.2.2.2).

Table A.6-24. HQ calculations for English sole

Type of Evaluation	COPC	EPC (mg/kg)[a]	NOAEL TRV	LOAEL TRV	NOAEL HQ	LOAEL HQ
Tissue residue	TBT	0.030	0.029	0.29	1.0	0.10
	total PCBs	4.1	0.104 – 0.52[b]	0.52 – 2.64[b]	**7.9 – 39**[b]	**1.6 – 7.9**[b]
Dietary	arsenic	20	20	30	1.0	0.66
	cadmium	1.2	0.1	0.5	**12**	**2.4**
	chromium	29	9.42	na	**3.1**	na
	copper	110	50	100	**2.2**	**1.1**
	vanadium	19	2.04	10.2	**9.5**	**1.9**
	benzo(a)pyrene	0.68	1.5	2.0	0.45	0.34

[a] Tissue-residue concentrations are wet weight; dietary concentrations are dry weight.

[b] Because of the uncertainty in the LOAEL, LOAEL HQs were calculated from a range of effect concentrations reported in Hugla and Thome (1999). The NOAEL TRV range was estimated by dividing the LOAEL TRV range by an uncertainty factor of 5.

COPC – chemical of potential concern
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
NOAEL – no-observed-adverse-effect level
PCB – polychlorinated biphenyl
TRV – toxicity reference value
Bold and underline identify NOAEL HQs > 1.0 and LOAEL HQs ≥ 1.0.

The HQs for the surface water evaluation for English sole were the same as those for juvenile Chinook salmon presented in Tables A.6-17 and A.6-18. None of the COPCs had HQs > 1.0 based on site-wide EPCs. On an individual sample basis, the dissolved cadmium concentration exceeded the marine chronic WQC in one sample, with an HQ of 4.1. However, this cadmium detection is an anomalous value and is highly uncertain, as discussed in Section A.6.2.1.2 (the uncertainty analysis for juvenile Chinook salmon).

A.6.2.2.2 Uncertainty analysis

This section presents a discussion of the uncertainty associated with specific components of the problem formulation, the exposure and effects assessments, and the risk characterization for English sole.

ROC Selection

English sole are benthic fish that live in close contact with sediment 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 sediment (e.g., shiner surfperch). As part of the EW SRI, shiner surfperch were collected and analyzed to represent prey for various fish and wildlife ROCs. For total PCBs, the 95% UCL calculated for shiner surfperch tissue was 2.3 mg/kg ww, which is lower than the 95% UCL calculated for English sole tissue. This PCB EPC resulted in a lower HQ for shiner surfperch than for English sole. For TBT, the 95% UCL calculated for shiner surfperch tissue was 0.0656 mg/kg ww, which was higher than the 95% UCL of 0.030 mg/kg ww calculated for English sole tissue. Thus, the TBT LOAEL HQ (0.24) for shiner surfperch was slightly higher than the TBT LOAEL HQ for English sole; both HQs were < 1.0, indicating low risk to benthic fish.

COPC Screen

For the dietary evaluation, TRVs were not available for cobalt, nickel, dibenzofuran, and 18 individual PAHs. As discussed in the uncertainty section for juvenile Chinook salmon (Section A.6.2.1.2), uncertainty associated with the COPC screen for individual PAHs is low because a TRV was available for a PAH mixture that included most of the individual PAH COIs. Risks to English sole from dietary exposure to cobalt, nickel, and dibenzofuran could not be evaluated because of the lack of acceptable toxicity data for those COIs.

For the tissue-residue evaluation, TRVs were not available for the following COIs: alpha-BHC, individual DDT compounds (4,4′-DDD, 4,4′-DDE, and 4,4′-DDT), individual chlordane components (alpha-chlordane, beta-chlordane, cis-nonachlor, and trans-nonachlor), beta-endosulfan, and heptachlor epoxide. As discussed in Section A.2.5.2.1, surrogate TRVs for related chemicals were used in the screening process for all of the COIs listed above except alpha-BHC. Maximum concentrations of these COIs (with the exception of beta-endosulfan) in fish tissue ranged from approximately 2 to 3 orders of magnitude lower than the surrogate TRVs (Table A.2-31), so these surrogate

TRVs are assumed to be sufficient for the screening process for the related COIs without TRVs. The risk to fish from exposure to alpha-BHC could not be evaluated because a surrogate TRV was not available.

There is uncertainty associated in the selected NOAEL TRV for 2,3,7,8-TCDD used to screen for dioxin-like PCB congeners and dioxins and furans. Although the selected NOAEL from Fisk et al. (1997) was the lowest NOAEL from a study that reported whole-body tissue concentrations, there is evidence that effects may occur at lower tissue concentrations. Giesy et al. (2002) reported significant effects on adult survival in adult female rainbow trout exposed to 2,3,7,8-TCDD in the diet for up to 320 days; fillet tissue concentrations of 0.44 ng/kg ww were reported at day 200 (Giesy et al. 2002; Jones et al. 2001). Whole-body tissue data were not reported. The authors estimated that the fillet concentration reported at day 200 was a 28-fold underestimate of the fillet concentration when mortality became significant (Giesy 2006). Using this factor of 28, the fillet concentration associated with adverse effects would be approximately 12 ng/kg ww. A NOAEL lower than the LOAEL was not available, so a NOAEL of 2.4 ng/kg ww was calculated for fillets assuming the NOAEL was equal to the LOAEL divided by 5. Three composite English sole fillet samples were analyzed for PCB and dioxin/furan congeners for the EW HHRA; the maximum total TEQ calculated using both PCB and dioxin/furan data was 1.35 ng/kg ww. Because the maximum total TEQ in English sole fillets was lower than the NOAEL TRV for fillets, the use of this alternate value as the TRV would not have changed the screening results for English sole.

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 assumed percentage of sediment ingestion because it was based on subjective observations by experienced fish biologists and not 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, was evaluated assuming 0 and 10% sediment ingestion to bracket the 1% estimate.

Assumptions of 0 and 10% sediment ingestion resulted in slightly different HQs compared with assumptions of 1% ingestion (Tables A.6-25). A change in sediment ingestion rate did not change the HQ from < 1.0 to > 1.0, or vice versa, for any individual COPC, except arsenic, for which the NOAEL changed from 1.0 to 1.1. These calculations indicate that the sediment ingestion rate has a very small effect on the HQs for English sole and that the use of different rates would not change any risk conclusions.

Table A.6-25. Dietary HQs for English sole as a function of sediment ingestion

COPC	0% Sediment Consumption		1% Sediment Consumption		10% Sediment Consumption	
	NOAEL HQ	LOAEL HQ	NOAEL HQ	LOAEL HQ	NOAEL HQ	LOAEL HQ
Arsenic	1.0	0.66	1.0	0.66	**<u>1.1</u>**	0.70
Cadmium	**<u>12</u>**	**<u>2.4</u>**	**<u>12</u>**	**<u>2.4</u>**	**<u>13</u>**	**<u>2.5</u>**
Chromium	**3.0**	na	**<u>3.1</u>**	na	**<u>3.4</u>**	na
Copper	**<u>2.2</u>**	**<u>1.1</u>**	**<u>2.2</u>**	**<u>1.1</u>**	**<u>2.3</u>**	**<u>1.2</u>**
Vanadium	**<u>9.2</u>**	**<u>1.8</u>**	**<u>9.5</u>**	**<u>1.9</u>**	**<u>12</u>**	**<u>2.4</u>**
Benzo(a)pyrene	0.45	0.34	0.45	0.34	0.48	0.36

COPC – chemical of potential concern
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
na – not available (no LOAEL TRV was available for chromium)
NOAEL – no-observed-adverse-effect level
<u>Bold and underline</u> identify NOAEL HQs > 1.0 and LOAEL HQs ≥ 1.0.

Foraging Range

English sole were assumed to forage exclusively in the EW even though it is known that they migrate seasonally to spawn in Puget Sound. The assumption that they forage only within the EW may potentially overestimate or underestimate exposure depending on the relative magnitude and extent of contamination in other foraging areas.

Surface Water Data

There are uncertainties in the cadmium surface water data because the highest detected dissolved cadmium concentration is an anomaly, as discussed in detail for juvenile Chinook salmon in Section A.6.2.1.2. The next highest concentration was 0.091 µg/L, which is below the Washington state marine chronic WQC of 0.94 µg/L.

TRVs

Uncertainties associated with deriving TRVs from laboratory studies are the same as those discussed for juvenile Chinook salmon in Section A.6.2.1.2. There is 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 of congeners in the food web, resulting in exposures to PCB congener mixtures that are potentially more or less biologically active than commercial Aroclor mixtures. Because laboratory toxicity tests evaluated for TRV selection generally were conducted with a single Aroclor mixture, the potency of weathered PCB mixtures in EW fish relative to those in fish from laboratory toxicity studies is uncertain.

In addition, TRVs are considered less certain if there are a small number of studies, if endpoints are subchronic, or if data quality is questionable. The relative uncertainty in the selected TRVs for English sole are presented in Table A.6-26, and the uncertainties associated with TBT, total PCBs, and cadmium are discussed in more detail below.

Table A.6-26. Level of uncertainty associated with selected dietary and tissue-residue TRVs for English sole

COPC	No. of TRV Studies	Level of Uncertainty in TRV[a]	Rationale for Level of Uncertainty
Tissue Residue			
TBT	4	high	Dataset for whole-body fish was small (two studies); data for two egg-residue studies were also available, and the concentrations were converted to whole-body estimates. The four studies consisted of chronic growth and reproductive studies with three species; no studies were conducted with English sole or other flatfish.
Total PCBs	17	medium to high	Dataset was large and consisted of chronic growth, reproduction, and survival endpoints with 12 species, not including English sole; high uncertainty in the lowest LOAEL derived from the Hugla and Thome (1999) fecundity endpoint.
Dietary			
Arsenic	6	medium	Dataset was medium-sized and consisted of survival endpoints for two species, not including English sole.
Cadmium	9	high	Lowest LOAEL was two orders of magnitude lower than LOAELs in the other eight studies; the growth effect in the TRV-derived study may have been related to food avoidance; no studies were conducted with English sole.
Chromium	1	high	Only one study was available, and no effects were observed at any of the dietary concentrations.
Copper	15	medium	Dataset was large and consisted of chronic growth and survival studies with five species, not including English sole.
Vanadium	1	high	Only one study was available.
Benzo(a)pyrene	5	low	Dataset was medium-sized and consisted of chronic studies with English sole.

[a] Level of uncertainty in TRV was based, in general, on size of dataset, number of endpoints, type of study (i.e., acute or chronic), species; other chemical-specific uncertainties were also considered.

COPC – chemical of potential concern

LOAEL – lowest observed effect level

PCBs – polychlorinated biphenyls

TBT – tributyltin

TRV – toxicity reference value

TBT TRV

As noted in Section A.4.2.1.2, there are uncertainties associated with the development of the TBT TRV for fish tissue. The selected TRV (0.29 mg/kg ww) was based on the Triebskorn et al. (1994)study, which reported the lowest whole-body tissue TBT concentrations associated with reduced growth in rainbow trout.

A lower LOAEL (0.159 mg TBT/kg ww) for reduced body weight in Japanese flounder larvae following 65 days of dietary exposure to TBT was identified (Shimasaki et al. 2003) but did not meet the TRV study selection criteria because the reported tissue concentrations were whole-body samples minus intestines, livers, kidneys, and gall bladders. Additional uncertainties associated with this study included:

- Parental fish were experimentally manipulated to produce only female offspring, which were subsequently used for the TBT toxicity experiments.
- High mortality was observed in both the control and TBT-exposed groups (survival was 42% in the control group and 57% in the LOAEL group). In standardized fish toxicity tests, control survival less than 90% generally invalidates the test (e.g., ASTM 1996).

Another study with a lower LOAEL (0.047 mg TBT/kg ww) for zebrafish masculization (Santos et al. 2006) was not included because the reproductive significance of the increased proportion of males in the TBT-exposed population (82%of test fish were male) relative to the control population (63%of control fish were male) was uncertain. Both zebrafish and the Japanese flounder used in Shimasaki et al. (2003) are fish species that are known to undergo sex reversal in response to environmental and chemical stressors.

PCB TRV

As noted in Section A.4.2.1.1, there are a number of uncertainties associated with the Hugla and Thome (1999) study, which reported the lowest effects concentrations. Because of these uncertainties, a range of LOAELs (0.52 and 2.64 mg/kg ww for the fecundity endpoint 2.64 mg/kg ww for the spawning and egg hatchability endpoints) was selected to represent the lowest exposure levels over which adverse reproductive effects may occur. This section presents a detailed evaluation of the 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 estimate 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 Hugla and Thome (1999), it would appear that the fish analyzed as whole-body concentrations at 50 days must have been males (if any females had been analyzed at 50 days, fewer than six females would have been available for the analysis of ovaries 1 year later). However, based on personal correspondence (Leroy 2007), the authors have stated that both male and female fish were included in whole-fish tissue analyses conducted at 50 days. 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) stated that a one-way analysis of variance (ANOVA) was used to analyze the data, and ANOVA assumes independence of observations. The authors reported that three fish were spawned two times each (Leroy 2007). 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 three (exposed fish) rather than a sample size of six (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. Although 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 in 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 personal correspondence (Leroy 2007), the authors 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). 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 than the typical minimum spawning age of 6 years and are smaller size. Because temperature was also used to affect the barbel reproductive cycle in this study, it is uncertain whether these manipulations may also have 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.

If the next higher LOAEL of 9.3 mg/kg ww for sheepshead minnow based on exposures to maternal fish and effects on egg and larval survival (Hansen et al. 1974a) had been selected as the LOAEL TRV, the LOAEL HQ for English sole would have been 0.44. The highest NOAEL below this LOAEL was 1.9 mg/kg ww from the same study. If this NOAEL had been selected as the NOAEL TRV, the NOAEL HQ would have been 2.2. Uncertainties in the Hansen et al. (1974a) study include elevated PCB concentrations of 0.52 to 0.64 mg/kg ww in control fish, which are above the lowest LOAEL from Hugla and Thome (1999) and the enhancement of egg production through the injection of 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 in eggs and embryos ranged from 0.857 to 77.9 mg/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). 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 ratios of PCB concentrations in fertilized eggs to those in maternal adults would likely range from 0.90 to 2.35. This range is uncertain because the data represented 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 0.71 to 183 mg/kg ww, which is comparable to the range of measured whole-body LOAELs from studies with adults (0.52 to 429 mg/kg ww). Although there are additional uncertainties associated with these studies, based on the likely range of adult tissue concentrations extrapolated from these studies, the use of egg LOAELs would not have resulted in risk conclusions different from those based on whole-body LOAELs.

Cadmium TRVs

The lowest cadmium dietary LOAEL and NOAEL TRVs (0.5 and 0.1 mg/kg dw, respectively) for growth of juvenile brown rockfish were more than an order of magnitude lower than the other cadmium dietary TRVs, and the observed growth 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 the selected TRV and the next highest TRV were based on 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 the dietary toxicity data reported for cadmium.

HQs calculated using the next higher LOAEL (800 mg/kg dw) and its associated NOAEL TRV (500 mg/kg dw) (Hatakeyama and Yasuno 1987) would have been substantially < 1.0 for English sole. Thus, risks from cadmium are uncertain for English sole.

Summary of Uncertainties

Uncertainties in the problem formulation, the effects and exposure assessments, and risk characterization for English sole are summarized as follows:

- The selection of shiner surfperch instead of English sole as an ROC would not affect risk conclusions for the tissue-residue evaluation for TBT and PCBs.
- Risks to English sole from exposure to cobalt, nickel, dibenzofuran, and alpha-BHC could not be evaluated because there were no TRVs for these COIs. Surrogate TRVs were used for individual DDT compounds, individual chlordane compounds, and heptachlor epoxide. It is unlikely that the use of TRVs for cobalt, nickel, dibenzofuran, and alpha-BHC, had they been available, would have changed the results of the COPC screen.
- The use of an alternative TRV for 2,3,7,8-TCDD in the COPC screen, which was lower but had greater uncertainty, would not have affected the outcome of the screen for dioxin-like PCBs or dioxins and furans.
- Uncertainty in the sediment ingestion rate had a small effect on HQ calculations but would not change any risk conclusions for English sole.
- There is uncertainty associated with the highest detected cadmium concentration. A comparison of total and dissolved concentrations in this sample, as well as a comparison of the original sample with the field duplicate samples, indicates that this value was an anomaly. No other dissolved cadmium concentrations exceeded the Washington State marine chronic WQC.
- There are uncertainties associated with using laboratory effects data to estimate risk to fish in the EW, resulting in either overestimates or underestimates of risks.

- There is uncertainty associated with both ends of the HQ range (1.6 to 7.9) associated with risks to English sole from total PCBs. The HQ range of 1.6 to 7.9 came from a single paper; the lower HQ (based on the higher LOAEL) was associated with complete reproductive impairment (i.e., no spawning), and the higher HQ (based on the lower LOAEL) was associated with a fecundity endpoint.
- Effects data for cadmium are highly variable, with most studies showing lower sensitivity of fish to cadmium than the study from which the TRV was selected. There is also uncertainty associated with the selected TRV because effects were partially attributed to reduced feeding. The LOAEL HQ would decrease from 2.4 to 0.002 if the next higher LOAEL were used.
- Estimated risks from chromium exposure are highly uncertain because they were based on a NOAEL from a study in which no effects were observed; no other chromium toxicity studies were available.
- The vanadium TRV was highly uncertain because only one study was found, and it is not known whether risks to English sole were overestimated or underestimated because of this uncertainty.

A.6.2.2.3 Risk conclusions

English sole was evaluated as the ROC to represent benthivorous fish and to be protective of planktivorous fish in the EW. English sole are exposed to sediment-associated chemicals because of their close proximity to the sediment and their diet of benthic invertebrates. Results of the risk characterization for English sole are summarized in Table A.6-27.

For the tissue-residue evaluation, the estimated risk to English sole from exposure to TBT was low, based on the LOAEL HQ of 0.10. For PCBs in tissue, LOAEL HQs calculated from the two LOAEL TRVs reported in Hugla and Thome (1999) were 1.6 and 7.9. Total PCBs was identified as a COC for English sole for the tissue-residue evaluation; TBT was not identified as a COC.

The highest LOAEL HQs calculated using the dietary evaluation were for cadmium and vanadium (2.4 and 1.9, respectively). These HQs indicate a potential risk to English sole from dietary exposure, although there was a high level of uncertainty associated with each of the TRVs for these COPCs. Cadmium and vanadium were identified as COCs for English sole for the dietary evaluation.

For chromium exposure based on the dietary evaluation, a LOAEL HQ could not be calculated because a LOAEL TRV was not available. There is significant uncertainty associated with forming risk conclusions using a NOAEL TRV when no effects data are available. Therefore, the risk to English sole from chromium exposure in the diet is unknown, and chromium was not identified as a COC for English sole for the dietary evaluation.

Table A.6-27. Summary of risk characterization for English sole

COPC	LOAEL or Chronic HQ	Primary Uncertainty	Selected as COC?
Tissue-Residue Evaluation			
TBT	0.10	high uncertainty in TBT TRV	no
Total PCBs	**1.6 – 7.9**[a]	high uncertainty in the low range of the selected TRVs for PCBs[b]	yes
Dietary Evaluation			
Arsenic	0.66	medium uncertainty in arsenic TRV	no
Cadmium	**2.4**	high uncertainty in cadmium TRV because effect may have been a result of food avoidance	yes
Chromium	na	high uncertainty in chromium TRV because a LOAEL was not available	no
Copper	**1.1**	medium uncertainty in copper TRV	yes
Vanadium	**1.9**	high uncertainty in vanadium TRV because of paucity of data	yes
Benzo(a)pyrene	0.34	low uncertainty	no
Surface Water Evaluation			
Cadmium	0.00097 – **4.1**[c]	concentration in the single sample that exceeded the chronic WQC was an anomaly	no
Mercury	0.00014 – 0.0016[c]	relatively low uncertainty	no
TBT	0.15[c]	WQC is driven by risk to snails and other marine invertebrates and therefore the FCV was used to evaluate risk to fish	no

a Because of uncertainty in the LOAEL, LOAEL HQs were calculated from a range of effects concentrations reported in Hugla and Thome (1999).

b Results from the studies reporting the lowest LOAELs were uncertain because of a lack of dose response in the fecundity endpoint and uncertainties in the statistical significance of the fecundity endpoint for the low dose, number of fish used in the experiment, and fish handling and maintenance protocols.

c A range of chronic HQs is presented based on both site-wide and individual-sample calculations.

COC – chemical of concern
COPC – chemical of potential concern
FCV – final chronic value
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
NOAEL – no-observed-adverse-effect level
TBT – tributyltin
TRV – toxicity reference value

Bold and underline identify LOAEL HQs ≥ 1.0.

The LOAEL HQ of 1.1 for copper indicates there may be a risk to English sole because the estimated concentration of copper in the diet is slightly above the concentration associated with reduced growth in juvenile rockfish. Copper was identified as a COC for English sole.

LOAEL HQs were < 1.0 for arsenic and benzo(a)pyrene, with relatively low uncertainty, indicating a low risk to English sole from dietary exposure to these COPCs. Arsenic and benzo(a)pyrene were not identified as COCs.

No COCs were identified based on the surface water evaluation for English sole.

A.6.2.3 Brown rockfish

This section presents risk estimates, uncertainties, and risk conclusions for brown rockfish. Chemicals were identified as COCs in the risk conclusions if LOAEL HQs for the tissue-residue (based on site-wide EPCs) and dietary evaluations were > 1.0 and if chronic HQs for the surface water evaluation were ≥ 1.0. In addition, if tissue-residue LOAEL HQs were >1.0 based on individual samples, the COPC was considered for selection as a COC after sediment data had been evaluated to determine whether or not the individual tissue concentrations were consistent with elevated COPC sediment concentrations in the vicinity of the individual rockfish.

A.6.2.3.1 Risk estimates

This section presents the HQ calculations for brown rockfish. COPCs evaluated for brown rockfish were as follows:

- Mercury, TBT, beta-endosulfan, and total PCBs for the tissue-residue evaluation
- Arsenic, cadmium, chromium, copper, vanadium, and benzo(a)pyrene for the dietary evaluation
- Cadmium, mercury, and TBT for the surface water evaluation

HQs for brown rockfish were evaluated on a site-wide basis, using EPCs calculated with data from throughout the EW, and on an individual basis, using sample-specific data associated with each brown rockfish sample.

For mercury, TBT, and beta-endosulfan in tissue of brown rockfish evaluated on a site-wide basis, LOAEL HQs were < 1.0 (Table A.6-28). For PCBs in tissue, a range of LOAEL TRVs was used because of uncertainty in the values, as mentioned in the uncertainty analysis Section A.6.2.3.2 and discussed in more detail in Section A.6.2.1.2 for juvenile Chinook salmon. The LOAEL HQs for PCBs were > 1.0, ranging from 1.5 to 7.7 (Table A.6-28). For the dietary COPCs for brown rockfish evaluated on a site-wide basis, LOAEL HQs were < 1.0 for all COPCs except cadmium, which had a LOAEL HQ of 2.5. A LOAEL HQ could not be calculated for chromium because there was no LOAEL TRV. The NOAEL HQ for chromium was < 1.0 but is uncertain, as discussed in the uncertainty analysis (Section A.6.2.3.2).

Table A.6-28. Site-wide HQ calculations for brown rockfish for both the tissue-residue and dietary evaluations

Type of Evaluation	COPC	EPC (mg/kg)[a]	NOAEL TRV	LOAEL TRV	NOAEL HQ	LOAEL HQ
Tissue residue	TBT	0.22	0.029	0.29	**<u>7.6</u>**	0.76
	mercury	0.21	0.23	0.39	0.91	0.54
	total PCBs	4.0	0.104 – 0.52[b]	0.52 – 2.64[b]	**<u>7.7 – 38</u>**[b]	**<u>1.5 – 7.7</u>**[b]
	beta-Endosulfan	0.0056	0.0031	0.031	**1.8**	0.18
Dietary	arsenic	14	20	30	0.68	0.45
	cadmium	1.3	0.1	0.5	**<u>13</u>**	**<u>2.5</u>**

Table A.6-28. Site-wide HQ calculations for brown rockfish for both the tissue-residue and dietary evaluations (cont.)

Type of Evaluation	COPC	EPC (mg/kg)[a]	NOAEL TRV	LOAEL TRV	NOAEL HQ	LOAEL HQ
	chromium	4.4	9.42	na	0.47	na
	copper	73	50	100	**<u>1.5</u>**	0.73
	vanadium	3.3	2.04	10.2	**<u>1.6</u>**	0.32
	benzo(a)pyrene	0.19	1.5	2.0	0.13	0.095

[a] Tissue-residue concentrations are wet weight; dietary concentrations are dry weight.

[b] Because of the uncertainty in the LOAEL, LOAEL 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
EPC – exposure point concentration
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
na – not available
NOAEL – no-observed-adverse-effect level
PCB – polychlorinated biphenyl
TRV – toxicity reference value

<u>Bold and underline</u> identify NOAEL HQs > 1.0 and LOAEL HQs ≥ 1.0.

For the tissue-residue evaluation for individual brown rockfish, LOAEL HQs were < 1.0 for beta-endosulfan (Table A.6-29 and Map A.6-12). For TBT, 3 of the 13 individual brown rockfish samples had LOAEL HQs > 1.0, with the highest HQ equal to 1.4. For mercury, 1 of the 15 individual brown rockfish samples had a LOAEL HQ > 1.0, with a LOAEL HQ of 1.1. For risk from total PCBs estimated with the lower LOAEL TRV of 0.52 mg/kg ww, LOAEL HQs were > 1.0 in 13 of the 15 individual brown rockfish samples, with a maximum HQ of 12. Using the second-lowest LOAEL TRV of 2.64 mg/kg ww, LOAEL HQs were >1.0 in 6 of the 15 individual brown rockfish, with a maximum of 2.3 (Table A.6-29 and Map A.6-12).

Table A.6-29. HQ calculations for individual brown rockfish for the tissue-residue evaluation

COPC	EPC (mg/kg ww)		NOAEL TRV (mg/kg ww)	LOAEL TRV (mg/kg ww)	NOAEL HQ		LOAEL HQ	
	Min	Max			Min	Max	Min	Max
TBT	0.038	0.42	0.029	0.29	**<u>1.3</u>**	**<u>14</u>**	0.13	**<u>1.4</u>**
Mercury	0.04	0.418	0.23	0.39	0.17	**<u>1.8</u>**	0.10	**<u>1.1</u>**
Total PCBs	0.4	6.2	0.104[a]	0.52[a]	**<u>3.8</u>**	**<u>60</u>**	0.77	**<u>12</u>**
			0.52[a]	2.64[a]	0.77	**<u>12</u>**	0.15	**<u>2.3</u>**
beta-Endosulfan	0.00045	0.013	0.0031	0.031	0.15	**4.2**	0.015	0.42

[a] Because of the uncertainty in the LOAEL, LOAEL HQs were calculated from a range of effect 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
EPC – exposure point concentration
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
na – not available
NOAEL – no-observed-adverse-effect level
PCB – polychlorinated biphenyl
TRV – toxicity reference value
ww – wet weight

<u>Bold and underline</u> identify NOAEL HQs > 1.0 and LOAEL HQs ≥ 1.0.

For dietary COPCs evaluated for individual brown rockfish, LOAEL HQs were < 1.0 for all COPCs except cadmium (Table A.6-30 and Map A.6-12). LOAEL HQs for cadmium were >1.0 in all brown rockfish samples, with a maximum of 7.5. LOAEL HQs could not be calculated for chromium because there is no LOAEL TRV for chromium. The NOAEL HQs for chromium were < 1.0.

Table A.6-30. HQ calculations for individual brown rockfish for the dietary evaluation

COPC	EPC (mg/kg dw)		NOAEL TRV (mg/kg dw)	LOAEL TRV (mg/kg dw)	NOAEL HQ		LOAEL HQ	
	Low	High			Low	High	Low	High
Arsenic	12.6	13.7	20	30	0.63	0.68	0.42	0.46
Cadmium	1.2	1.4	0.1	0.5	**<u>12</u>**	**<u>114</u>**	**<u>2.5</u>**	**<u>2.7</u>**
Chromium	2.4	6.3	9.42	na	0.25	0.67	na	na
Copper	69.7	77.6	50	100	**<u>1.4</u>**	**<u>1.6</u>**	0.70	0.78
Vanadium	2.5	4.8	2.04	10.2	**<u>1.2</u>**	**<u>2.3</u>**	0.24	0.47
Benzo(a)pyrene	0.134	0.25	1.5	2.0	0.089	0.17	0.067	0.13

COPC – chemical of potential concern
dw – dry weight
EPC – exposure point concentration
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
na – not available
NOAEL – no-observed-adverse-effect level
PCB – polychlorinated biphenyl
TRV – toxicity reference value
ww – wet weight

<u>Bold and underline</u> identify NOAEL HQs > 1.0 and LOAEL HQs ≥ 1.0.

The HQs for the surface water evaluation for brown rockfish were the same as those for juvenile Chinook salmon, as presented in Tables A.6-17 and A.6-18. None of the COPCs had HQs > 1.0 based on site-wide EPCs. On an individual sample basis, the dissolved cadmium concentration exceeded the marine chronic WQC in one sample, with an HQ of 4.1. However, this cadmium detection is an anomalous value and is highly uncertain, as discussed in Section A.6.2.1.2 (the uncertainty analysis for juvenile Chinook salmon)

A.6.2.3.2 Uncertainty analysis

This section presents a discussion of the uncertainty associated with specific components of the problem formulation, the exposure and effects assessments, or risk characterization for brown rockfish.

ROC Selection

The use of brown rockfish to represent upper-trophic-level fish in the EW is slightly uncertain. However, to evaluate another upper-tropic-level fish, concentrations of COPCs in sand sole were compared with those in brown rockfish. Unlike brown rockfish, sand sole have a foraging range that is larger than the EW. The sand sole samples were collected in 2005 following the completion of the Phase 1 removal action (Windward 2006b). Data were available for mercury and PCBs in whole-body sand sole tissue, but no data were available for TBT and beta-endosulfan. The sand sole EPCs for mercury (0.087 mg/kg ww; 95% UCL) and total PCBs (1.31 mg/kg ww; maximum detect instead of the 95% UCL because there were only five detected concentrations) were considerably lower than the brown rockfish EPCs for mercury (0.21 mg/kg ww; 95% UCL) and total PCBs (4.0 mg/kg ww; 95% UCL). These data indicate that exposures of brown rockfish to mercury and total PCBs in the EW are higher than exposures of sand sole, and that the protection of brown rockfish will also result in the protection of sand sole.

COPC Screen

Uncertainties in the COPC screen are the same as those discussed for English sole in Section A.6.2.2.2. In summary, risks to brown rockfish from cobalt, nickel, and dibenzofuran based on the dietary evaluation and from alpha-BHC based on the tissue-residue evaluation could not be evaluated because toxicity data were not available for those COIs. There is some uncertainty associated with TRVs for individual DDT compounds (4,4′-DDD, 4,4′-DDE, and 4,4′-DDT), individual chlordane components (alpha-chlordane, beta-chlordane, cis-nonachlor, and trans-nonachlor), and heptachlor epoxide. However, surrogate TRVs for related chemicals were determined to be acceptable for the use in the screening process for these COIs without TRVs.

There is also uncertainty in the selected NOAEL TRV for 2,3,7,8-TCDD (as discussed in Section A.6.2.2.2). If the alternative and less certain NOAEL TRV of 2. 4 ng/kg ww had been compared with the maximum total TEQ (i.e., calculated using PCB and dioxin/furan congeners) of 6.73 ng/kg ww for brown rockfish, then dioxin-like PCBs and/or dioxins/ furans would have screened in as a COPC. As a COPC, the 95% UCL for total TEQ would have been compared with both the NOAEL and LOAEL TRVs of 2.4 and 12 ng/kg ww, respectively. The total TEQ 95% UCL for brown rockfish was 5.33 ng/kg ww, resulting in a LOAEL HQ of 0.44. This evaluation indicates that the risk to brown rockfish from exposure to dioxin-like PCBs and dioxins/furans is low with some uncertainty and that total TEQ would not have been selected as a COC.

Dietary Composition

Concentrations of COPCs in the diet of brown rockfish are a function of the types of prey consumed and their COPC concentrations. The relative percentages of fish, shrimp, crab, and benthic invertebrates in the brown rockfish diet were from one study conducted in central Puget Sound by Wingert et al. (1979). Because there is likely some variability in the diet of brown rockfish, there is uncertainty in the HQs calculated using dietary assumptions from only one study. COPC concentrations are generally higher in benthic invertebrates and crab. Therefore, if the percentage of benthic invertebrates in the brown rockfish diet had been underestimated, then risks to brown rockfish would also be underestimated. Likewise, an overestimate in the percentage of benthic invertebrates in the diet would have resulted in an overestimate of risk. It is not possible to determine whether risks have been overestimated or underestimated without additional data on the fraction of benthic invertebrates in the brown rockfish diet.

Incidental Sediment Ingestion

The exposure assessment for brown rockfish 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 was 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) for bottom-feeding fish such as English sole and Pacific staghorn sculpin. Therefore, uncertainty in dietary exposure estimates calculated using Equation 4-1, as described in Section A.4.1.2 were evaluated assuming 0 and 10% sediment ingestion to bracket the 1% estimate.

Assumptions of 0 and 10% sediment ingestion resulted in slightly different HQs compared with the assumption of 1% ingestion (Table A.6-31). None of the HQs changed from < 1.0 to > 1.0, or vice versa, based on a change in the sediment ingestion rate. These calculations indicate that the sediment ingestion rate has a very small effect on the HQs for brown rockfish and would not change any risk conclusions.

Table A.6-31. Dietary HQs for brown rockfish as a function of sediment ingestion

COPC	0% Sediment Consumption		1% Sediment Consumption		10% Sediment Consumption	
	NOAEL HQ	LOAEL HQ	NOAEL HQ	LOAEL HQ	NOAEL HQ	LOAEL HQ
Arsenic	0.67	0.45	0.68	0.45	0.73	0.49
Cadmium	**<u>13</u>**	**<u>2.5</u>**	**<u>13</u>**	**<u>2.5</u>**	**<u>13</u>**	**<u>2.7</u>**
Chromium	0.43	na	0.47	na	0.74	na
Copper	**<u>1.5</u>**	0.73	**<u>1.5</u>**	0.73	**<u>1.6</u>**	0.79
Vanadium	**<u>1.3</u>**	0.27	**<u>1.6</u>**	0.32	**<u>4.2</u>**	0.84
Benzo(a)pyrene	0.12	0.092	0.13	0.095	0.16	0.12

COPC – chemical of potential concern
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
na – not available (no LOAEL TRV was available for chromium)
NOAEL – no-observed-adverse-effect level
<u>Bold and underline</u> identify NOAEL HQs > 1.0 and LOAEL HQs ≥ 1.0.

Shrimp Data for Individual Rockfish

There is some uncertainty in the dietary EPCs calculated for individual rockfish because shrimp were a relatively large component of the rockfish diet (47%) in these calculations, and only one composite shrimp sample, collected site-wide, was available from the EW. If shrimp do not forage over a large area and if, on a smaller scale, there is more variability in the COPC concentrations, the dietary exposure concentrations for individual rockfish may differ from those in the risk characterization analysis. The only other invertebrate for which location-specific samples were collected was mussels. Although mussels may bioaccumulate COPCs to a greater or lesser extent than do shrimp, the use of mussels in place of shrimp allows for a different location-specific analysis to be conducted. COPC concentrations in mussels were lower than those in shrimp for all dietary COPCs except cadmium and vanadium; likewise, the HQs for all dietary COPCs except cadmium and vanadium were lower using the mussel data instead of the shrimp data. The dietary HQs for individual rockfish were recalculated for cadmium and vanadium using the mussel data. To calculate the dietary concentration for each individual rockfish, the site-wide shrimp EPC was replaced with the COPC concentration in the mussel sample collected closest to the individual rockfish sample. When these mussel data were used, the range of individual cadmium LOAEL HQs increased from 2.5 to 2.7 to 3.9 to 7.5, and the range of individual vanadium LOAEL HQs increased from 0.067 to 0.13 to 0.03 to 0.16. The overall risk conclusions did not change because the LOAEL HQs for cadmium were still > 1.0 and the LOAEL HQs for vanadium were still < 1.0.

Surface Water Data

There are uncertainties in the cadmium surface water data because the highest detected dissolved cadmium concentration is an anomaly, as discussed in detail for juvenile Chinook salmon in Section A.6.2.1.2. The next highest concentration was 0.091 µg/L, which is below the Washington State marine chronic WQC of 0.94 µg/L.

TRVs

Uncertainties associated with deriving TRVs from laboratory studies were discussed in the uncertainty sections for juvenile Chinook salmon and English sole (Sections A.6.2.1.1 and A.6.2.1.2). The relative uncertainty in the selected TRVs for evaluating risk to brown rockfish is presented in Table A.6-32.

Table A.6-32. Level of uncertainty associated with selected dietary and tissue-residue TRVs for brown rockfish

COPC	No. of TRV Studies	Level of Uncertainty in TRV	Rationale for Level of Uncertainty
Tissue Residue			
TBT	4	high	Dataset for whole-body fish was small (two studies); data for two egg-residue studies were also available, and the concentrations were converted to whole-body estimates. The four studies consisted of chronic growth and reproductive studies with three species; no studies were conducted with rockfish species.
Total PCBs	17	medium to high	Dataset was large dataset and consisted of chronic growth, reproduction, and survival endpoints with 12 species, not including brown rockfish; high uncertainty in the lowest LOAEL derived from the Hugla and Thome (1999) fecundity endpoint.
Mercury	16	medium	Dataset was large and consisted of chronic growth, reproduction, and survival studies with eight species; no studies were conducted with rockfish species.
beta-Endosulfan	3	high	Only three studies were available; these studies reported LC50 tissue concentrations for three species, not including rockfish.

Table A.6-32. Level of uncertainty associated with selected dietary and tissue-residue TRVs for brown rockfish (cont.)

COPC	No. of TRV Studies	Level of Uncertainty in TRV	Rationale for Level of Uncertainty
Dietary			
Arsenic	6	medium	Dataset was medium-sized and consisted of survival endpoints for two species, not including rockfish.
Cadmium	9	high	Lowest LOAEL was two orders of magnitude lower than the LOAELs in the other eight studies; the growth effect in the TRV-derived study may have been related to food avoidance; no growth endpoint for rockfish
Chromium	1	high	Only one study was available, and no effects were observed at any of the dietary concentrations.
Copper	15	low	Dataset was large dataset and consisted of chronic growth and survival studies with five species, including rockfish; the selected TRV was based on the rockfish study.
Vanadium	1	high	Only one study was available.
Benzo(a)pyrene	5	low	Dataset was medium-sized and consisted of chronic studies with rockfish; the selected TRV was based on a rockfish study.

COPC – chemical of potential concern
LC50 – concentration that is lethal to 50% of an exposed population
LOAEL – lowest observed effect level
PCB – polychlorinated biphenyl
TBT – tributyltin
TRV – toxicity reference value

As discussed in Section A.6.2.2.2, there are a number of uncertainties in the PCB TRVs from the Hugla and Thome (1999) study that reported the lowest effects concentrations. If the next highest LOAEL TRV of 9.3 mg/kg ww from Hansen et al. (1974a) had been used in the risk calculations, the LOAEL HQ for brown rockfish would have been 0.43.

There is also uncertainty in the cadmium LOAEL TRV (also discussed in Section A.6.2.2.2). The LOAEL HQ calculated using the lowest LOAEL TRV was 2.5; the LOAEL HQ calculated using the second-lowest LOAEL TRV was substantially lower, at 0.002. This range indicates that there is high uncertainty in the cadmium risk estimate.

Only one relevant toxicity study was available for chromium, and only a NOAEL was reported; no effects were observed in this study so a LOAEL could not be determined. There is significant uncertainty associated with forming risk conclusions based on a NOAEL with no associated LOAEL (i.e., an unbounded NOAEL).

Summary of Uncertainties

Uncertainties in the problem formulation, the effects and exposure assessments, and risk characterization for brown rockfish are summarized as follows:

- The selection of another upper-trophic-level species (i.e., sand sole) instead of brown rockfish as an upper-trophic-level ROC would not have resulted in different risk conclusions.
- Risks to brown rockfish from exposure to cobalt, nickel, dibenzofuran, and alpha-BHC could not be evaluated because there were no TRVs for these COIs. Surrogate TRVs were used for individual DDT compounds, individual chlordane compounds, beta- endosulfan, and heptachlor epoxide. It is unlikely that the use of TRVs for cobalt, nickel, dibenzofuran, and alpha-BHC, had they been available, would have changed the results of the COPC screen.
- The percentage of benthic invertebrates and crab in the brown rockfish diet is likely variable and could affect the HQs. Data documenting the brown rockfish diet were only available from one study. It is not known whether the dietary percentages of benthic invertebrates and crab from this study, which were used in the HQ calculations, resulted in an overestimate or underestimate of risk to brown rockfish.
- Use of an alternative but less certain NOAEL TRV for 2,3,7,8-TCDD in the COPC screen would have resulted in the identification of dioxin-like PCBs and/or dioxins and furans as COPCs. However, use of the alternative but less certain LOAEL TRV in the risk characterization would have resulted in a LOAEL HQ of 0.44, indicating that there is a low risk to brown rockfish from exposure to dioxin-like PCBs or dioxins and furans.
- The use of a range of sediment ingestion rate did not change any risk conclusions for brown rockfish.
- There is uncertainty associated with the highest detected dissolved cadmium concentration in surface water; a comparison of total and dissolved concentrations in this sample, as well as a comparison of the original sample to the field duplicate samples, indicates that this value is an anomaly. No other cadmium concentrations exceeded the WQC.
- There are uncertainties associated with using laboratory effects data to estimate risk to fish in the EW, which could result in either overestimates or underestimates of risks.
- Because of numerous uncertainties in the study that reported the lowest PCB effects concentrations (Hugla and Thome 1999), risks to brown rockfish from PCBs are uncertain. LOAEL HQs based on toxicity data reported in this study ranged from 1.5 to 7.7.

- Effects data for cadmium are highly variable, with most studies showing lower sensitivity than those in the study from which the TRV was selected. There is also uncertainty associated with the selected TRV because effects were partially attributed to reduced feeding. The LOAEL HQ decreased from 2.5 to 0.002 when the second-lowest LOAEL was used.
- Estimated risks from chromium exposure are highly uncertain because they were based on a NOAEL from a study in which no effects were observed; no other chromium toxicity studies were available.

A.6.2.3.3 Risk conclusions

Brown rockfish, a benthic omnivorous fish, was selected to represent upper-trophic-level fish in the LDW. The HQs for individual brown rockfish are presented for informational purposes and were used to determine the potential for risk when evaluated on a localized basis but were not used to identify COCs. Results of the risk characterization for brown rockfish are summarized in Table A.6-33.

Because LOAEL HQs for mercury, beta-endosulfan, arsenic, copper, vanadium, and benzo(a)pyrene were < 1.0 (with the exception of one individual rockfish sample for mercury), there is low risk to brown rockfish in the EW from exposure to these chemicals. There is some uncertainty in the TRVs used to estimate risk for these COPCs. It is not known if this uncertainty results in an overestimate or underestimate of risk to brown rockfish. None of these COPCs were identified as COCs for brown rockfish based on the tissue-residue or dietary evaluations.

For TBT, the site-wide LOAEL HQ is < 1.0, indicating there is a low risk to brown rockfish. However, three individual samples had LOAEL HQs > 1.0, indicating that there is some risk to individual brown rockfish from TBT exposure. The rockfish with tissue TBT concentrations above the LOAEL were collected from the west side of the EW along T-18, an area with elevated sediment TBT concentrations; therefore TBT was identified as a COC for brown rockfish based on the tissue-residue evaluation.

For PCBs, LOAEL HQs > 1.0 on site-wide and individual basis indicate that there is some risk to brown rockfish from PCB exposure as measured in tissue. There is high uncertainty associated with the use of the lowest LOAEL TRV. When the more certain LOAEL TRV was used in risk calculations, the site-wide LOAEL HQ was 1.5, and LOAEL HQs for six individual rockfish were > 1.0. Total PCBs was identified as a COC for brown rockfish based on the tissue-residue evaluation.

For cadmium, LOAEL HQs > 1.0 on site-wide and individual bases indicate some risk to brown rockfish from cadmium exposure as measured in the diet. However, there is uncertainty associated with the cadmium TRV because the growth effects on fish observed in the study from which the TRV was derived may have been caused by food avoidance. When the second-lowest LOAEL TRV was used to estimate risks, the LOAEL HQ was 0.002. Because the site-wide LOAEL HQ was >1.0, cadmium was identified as a COC for brown rockfish based on the tissue-residue evaluation.

Table A.6-33. Summary of risk characterization for brown rockfish

COPC	LOAEL or Chronic HQ: Site-Wide	LOAEL or Chronic HQ: Individual Fish (No. of Fish with HQs > 1.0)[a]	Primary Uncertainty	Selected as COC?
Tissue Residue Evaluation				
TBT	0.76	0.13 – **<u>1.4</u>** (3)	high uncertainty in TBT TRV	yes
Mercury	0.4	0.1 – **<u>1.1</u>**	medium uncertainty in mercury TRV	no
Total PCBs – lowest TRV[b]	**<u>7.7</u>**	0.77 – **<u>12</u>** (13)	high uncertainty because of a number of uncertainties associated with the lowest LOAEL TRV for PCBs[c]	yes
Total PCBs – second-lowest TRV[b]	**<u>1.5</u>**	0.15 – **<u>2.3</u>** (6)	medium uncertainty in the second-lowest LOAEL TRV for PCBs[d]	yes
beta- Endosulfan	0.18	0.015 – 0.42	high uncertainty in beta-endosulfan TRV	no
Dietary Evaluation				
Arsenic	0.45	0.42 – 0.46	medium uncertainty in arsenic TRV	no
Cadmium	**<u>2.5</u>**	**<u>2.5 – 2.7</u>** (13)	high uncertainty in cadmium TRV because effect may have been a result of food avoidance	yes
Chromium	na	na	risk could not be evaluated because a LOAEL was not available	no
Copper	0.73	0.70 – 0.78	medium uncertainty in copper TRV	no
Vanadium	0.32	0.24 – 0.47	high uncertainty in vanadium TRV because of paucity of data	no
Benzo(a)pyrene	0.095	0.067 – 0.13	low uncertainty	no
Surface Water Evaluation				
Cadmium	0.00097 – **<u>4.1</u>**[e]		the concentration in the single sample that exceeded the chronic WQC was an anomaly	no
Mercury	0.00014 – 0.0016[e]		relatively low uncertainty	no
TBT	0.15[e]		WQC is driven by risk to snails and other marine invertebrates and therefore the FCV was used to evaluate risk to fish	no

[a] Thirteen individual rockfish samples were analyzed. The number in parentheses indicates the number of individual rockfish samples with LOAEL HQs > 1.0.

[b] Because of uncertainty in the LOAEL, LOAEL HQs were calculated from a range of effects concentrations reported in Hugla and Thome (1999).

[c] The lowest LOAEL was uncertain because of a lack of dose response in the fecundity endpoint and uncertainties in the statistical significance of the fecundity endpoint for the low dose, number of fish used in the experiment, and fish handling and maintenance protocols.

[d] The second-lowest LOAEL from Hugla and Thome (1999) had some uncertainties associated with fish handling and maintenance protocols.

[e] A range of chronic HQs is presented based on both site-wide and individual-sample calculations.

COC – chemical of potential concern
COPC – chemical of potential concern
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
NOAEL – no-observed-adverse-effect level
TBT – tributyltin
TRV – toxicity reference value

<u>Bold and underline</u> identify NOAEL HQs > 1.0 and LOAEL HQs ≥ 1.0.

For chromium, a LOAEL HQ could not be calculated because no LOAEL TRV for fish was available. Therefore, risk to brown rockfish from chromium exposure in the diet is unknown. Chromium was not identified as COC for brown rockfish based on the tissue-residue evaluation.

No COCs were identified for brown rockfish based on the surface water evaluation.

A.6.2.4 Summary of risk conclusions for fish

Four chemicals (i.e., cadmium, copper, vanadium, and total PCBs) were identified as COCs for fish based on the dietary or tissue-residue evaluations (Table A.6-34). No chemicals were identified as COCs based on the surface water evaluation. Cadmium was identified as a COC for all three fish ROCs based on the dietary evaluation. Copper and vanadium were identified as COCs for English sole based on the dietary evaluation. Based on the tissue-residue evaluation, total PCBs was identified as a COC for both English sole and brown rockfish, and TBT was identified as a COC for brown rockfish.

Table A.6-34. Chemicals identified as COCs for fish ROCs

Type of Evaluation	Chemical	Juvenile Chinook Salmon	English Sole	Brown Rockfish
Dietary	arsenic	no	no	no
	cadmium	**yes**	**yes**	**yes**
	chromium	no	no	no
	copper	no	**yes**	no
	vanadium	no	**yes**	no
	benzo(a)pyrene	ne	no	no
Tissue residue	mercury	ne	ne	no
	TBT	ne	no	**yes**
	total PCBs	ne	**yes**	**yes**
	beta-endosulfan	ne	ne	no
Surface water	cadmium	no	no	no
	mercury	no	no	no
	TBT	no	no	no

COC – chemical of concern
ne – not evaluated (chemicals were not selected as COPCs)
ROC – receptor of concern
TBT – tributyltin

A.6.3 WILDLIFE

This section presents the risk characterization and uncertainty analysis for each of the four aquatic-dependent wildlife ROCs (pigeon guillemot, osprey, river otter, harbor seal). To characterize risk, HQs were calculated based on estimated ingested doses of COPCs (as described in Section A.5.1) and dose-based TRVs (as presented in Section A.5.2) using the following equation:

$$HQ = \frac{Dose}{TRV} \qquad \textbf{Equation 6-8}$$

Where:

HQ	=	hazard quotient
Dose	=	dietary dose (mg/kg bw/day)
TRV	=	dietary dose toxicity reference value (mg/kg bw/day)

HQs are calculated using both the NOAEL and LOAEL TRVs. A LOAEL HQ ≥ 1.0 is generally regarded as an indication of the potential for adverse effects because the benchmark is the effects concentration at which adverse effects are observed. The potential for adverse effects associated with a NOAEL HQ > 1.0 and a LOAEL HQ < 1.0 is considered low and uncertain because the true threshold for effects occurs at a concentration somewhere between the NOAEL and LOAEL. A NOAEL HQ < 1.0 indicates that risk is unlikely.

The following subsections present the calculation of HQs and discuss the uncertainties in the exposure and effects data that may result in overestimates or underestimates of risk for each of the COPCs and ROCs. Finally, risk conclusions that integrate risk estimates with associated uncertainties are presented for each ROC, resulting in a determination of which chemicals are considered to be COCs.

A.6.3.1 Pigeon guillemot

This section presents the risk estimates, uncertainties, and risk conclusions for pigeon guillemot. Chemicals were identified as COCs in the risk conclusion if LOAEL HQs were ≥ 1.0.

A.6.3.1.1 Risk estimates

Mercury, total PCBs, PCB TEQ, and total TEQ were identified as COPCs for pigeon guillemot based on the screening presented in Section A.2.5.3. Both NOAEL and LOAEL HQs were < 1.0 for this ROC for all COPCs (Table A.6-35).

Table A.6-35. HQs for pigeon guillemot

COPC	Dose (mg/kg bw/day)	TRV (mg/kg bw/day)		HQ	
		NOAEL	LOAEL	NOAEL	LOAEL
Mercury	0.0062	0.0146	0.073	0.43	0.085
Total PCBs	0.16	0.49	1.4	0.33	0.12
PCB TEQ[a]	4.2×10^{-6}	1.4×10^{-5}	1.4×10^{-4}	0.30	0.030
Total TEQ	4.7×10^{-6}	1.4×10^{-5}	1.4×10^{-4}	0.33	0.033

[a] TEQ was calculated using TEFs for birds from Van den Berg et al. (1998).

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
TEF – toxic equivalency factor
TEQ – toxic equivalent
TRV – toxicity reference value

A.6.3.1.2 Uncertainty assessment

This section presents a discussion of the uncertainty associated with the problem formulation and the exposure and effects assessment for pigeon guillemot.

Problem Formulation

ROC Selection

Pigeon guillemot was selected as an ROC to represent EW birds that consume benthic organisms, bottom-dwelling fish, and pelagic fish. Other birds with similar dietary exposures include mergansers, spotted sandpiper, bufflehead, and goldeneye. There is some uncertainty associated with how well the exposure of pigeon guillemot to chemicals in the EW represents the exposures of these other species. As discussed in Section A.2.3.3.2, mergansers, bufflehead, and goldeneye are less exposed than pigeon guillemot because they either: 1) obtain less of their prey from the EW because of their larger foraging areas, 2) are lower-trophic-level-consumers and are thus less exposed to bioaccumulative chemicals, or 3) do not breed in the vicinity of the EW and are not exposed during sensitive life stages.

An additional uncertainty with the selection of pigeon guillemot is its sensitivity to chemicals in the EW relative to the sensitivity of other species, which is unknown because of the absence of laboratory toxicological data for many wildlife species.

COPC Screen

Fifty-four chemicals or chemical groups were identified as COIs for birds. For 20 of these COIs, effects data were available for each chemical (or chemical group). For another 28 chemicals (20 individual PAHs and 8 pesticides), effects data were values for related, surrogate chemicals because effects data were not available for the COI, so there is some uncertainty regarding the COPC screen for these chemicals. For the remaining six COIs (silver, monobutyltin, dibutyltin, 1,4-dichlorobenzene, phenol, and mirex), risks to pigeon guillemot could not be evaluated because effects data were not available. No further evaluation could be conducted for these COIs.

Exposure Assessment

Uncertainty in the exposure assessment for pigeon guillemot is associated with the following:

- Direct sediment contact
- Incidental sediment ingestion rate
- Dietary composition
- Site use
- TEQ approach
- Surface water data for total TEQ

These uncertainties are discussed in detail below.

Direct Sediment Contact

Risks to wildlife from direct contact with sediment are considered to be insignificant relative to risks from incidental sediment ingestion (EPA 2000b). However, the exclusion of this pathway adds only a small amount of uncertainty to the risk estimate for pigeon guillemot because this species only contacts the sediment briefly when it captures prey. In addition, feathers provide a barrier that reduces the potential for direct contact with the sediment.

Incidental Sediment Ingestion Rate

To address uncertainties in the amount of sediment incidentally ingested by pigeon guillemot while foraging, ingested doses of COPCs were calculated assuming the sediment ingestion rate was 10% of the food ingestion rate versus 2% assumed in Section 5.1.2.2. This conservative assumption did not result in an increase in any of the HQs by more than 0.1 and did not change any risk conclusions.

Dietary Composition

There is uncertainty in the dietary composition of pigeon guillemot in the EW. To address this uncertainty, the dietary exposure of pigeon guillemot was calculated using the conservative assumption that the diet consisted of only the fish or invertebrate prey species that had the maximum EPC (i.e., the maximum of the species consumed by pigeon guillemot presented in Table A.5-4) for mercury, total PCBs, PCB TEQ, and total TEQ. Based on this conservative assumption, LOAEL HQs for these COPCs were well below 1.0 (Table A.6-36).

Table A.6-36. HQs for pigeon guillemot assuming maximum EPC in diet

COPC	Maximum EPC (mg/kg ww)[a]	Exposure Dose (mg/kg bw/day)	HQ	
			NOAEL	LOAEL
Mercury	0.21	0.021	1.4	0.29
Total PCBs	4.1	0.41	0.84	0.29
PCB TEQ[b]	7.25×10^{-5}	7.3×10^{-6}	0.52	0.052
Total TEQ[b]	7.90×10^{-5}	4.0×10^{-6}	0.29	0.029

[a] Maximum EPC for total PCBs was for English sole, and maximum EPCs for mercury, PCB TEQ, and total TEQ were in brown rockfish.

[b] TEQ was calculated using TEFs for birds from Van den Berg et al. (1998).

bw – body weight
COPC – chemical of potential concern
EPC – exposure point concentration
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
NOAEL – no-observed-adverse-effect level
PCB – polychlorinated biphenyl
TEF – toxic equivalency factor
TEQ – toxic equivalent
ww – wet weight

Site Use

To address uncertainty regarding the site use factor of 0.5, ingested doses of COPCs were calculated assuming that pigeon guillemot feed only in the EW, with a site use factor of 1.0. This increased the HQs in Table A.6-35 by a factor of 2, but the highest NOAEL and LOAEL HQs remained less than 1.0.

TEQ 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 Section A.2.4.2.4. The rationale for the use of TEFs was based on evidence that there is a common mechanism of toxicity for certain dioxins, furans, and PCB congeners, which involves binding of the congeners to the aryl hydrocarbon (Ah) receptor as an initial step. Data on the relative binding affinity of particular PCB congeners as compared with 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 produce dioxin-specific biochemical and toxic responses.

An uncertainty in the TEQ approach is related to the derivation of TEF values. Limitations in the underlying data used to derive TEFs, such as the relevance of the endpoints in the studies and a lack of information on interspecies variability, contribute to the uncertainty. Although these uncertainties were identified by Van den Berg et al. (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 2003c). According to EPA (EPA 2003c), the TEQ method is technically appropriate for evaluating risks to birds and mammals, and uncertainties associated with the TEQ method are not greater than other uncertainties in the ERA process (EPA 2003c).

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 from < 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; TEFs were highly variable, ranging from 0.001 to 0.5. For PCBs 126 and 169, data are available from only one study (*in ovo* with chickens). These TEFs, which were derived using either EROD induction or *in ovo* endpoints, are most accurate for the assessment of effects based on concentrations in whole embryos (EPA 2003c). 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 EW results in additional uncertainty in the risk estimates for birds. It is not known whether the uncertainties discussed above would overestimate or underestimate risk.

Surface Water Data for Total TEQ

Surface water samples were not analyzed for dioxins and furans, so the surface water EPC for PCB TEQ was also used for total TEQ. There is some uncertainty in this assumption because potential toxicity from dioxins and furans was not included in the surface water exposure estimate. However, this uncertainty is expected to be very low. For example, the amount of PCB TEQ ingested in water by pigeon guillemot is 8.5×10^{-11}, which is only 0.002% of the entire dose of 4.2×10^{-6}. The relative contribution of dioxin/furan TEQ to the total TEQ is expected to be similar to the contribution from PCB TEQ. Therefore, the use of the PCB TEQ as the surface water EPC for total TEQ has a negligible effect on the risk calculations.

Effects Assessment

Uncertainty associated with available toxicity benchmarks for birds may affect risk estimates. Exposure to mixtures of chemicals may result in interactions of those chemicals. These joint chemical actions may result in antagonistic, synergistic, or additive effects. It is generally believed that the joint action of many complex mixtures is additive (Broderius 1991; Logan and Wilson 1995). Therefore, there is uncertainty associated with risk estimates for pigeon guillemot based on single chemical laboratory exposures.

Some other uncertainties associated with laboratory studies include how well the test species or life stage represents that of the pigeon guillemot (which may underestimate or overestimate risks); the lack of endpoints other than reproduction, growth, or survival that could also result in adverse effects on the population (which may underestimate risks); and potentially large dosing intervals that may not capture the actual chemical threshold of effects (which may underestimate risks).

TRVs are considered to be less certain if there were only a small number of relevant studies, if endpoints were subchronic, or if data quality was questionable. The relative uncertainty in the selected TRVs for birds and its potential effect on risk estimates are summarized in Table A.6-37.

Table A.6-37. Level of uncertainty associated with TRVs for birds

COPC	No. of TRV Studies	Uncertainty in TRV[a]	Rationale
Mercury	8	medium	Selected TRVs were based on a chronic reproduction endpoint; NOAEL was extrapolated from the LOAEL based on an uncertainty factor of 5.
Total PCBs	13	medium	Selected TRVs were based on a chronic reproduction endpoint.
PCB TEQ and total TEQ	2	high	No dietary studies were available; only two relevant studies were available; selected TRVs were based on a study with acute high-level weekly exposure via IP injection.

[a] Level of uncertainty key:
Low = large dataset that includes chronic studies
Medium = medium-sized dataset that includes chronic studies
High = small dataset that includes 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
TEQ – toxic equivalent
TRV – toxicity reference value

Summary of Uncertainties

Uncertainties in the problem formulation and exposure and effects assessment are summarized as follows:

- Uncertainties in the ROC selection, direct sediment contact, sediment ingestion rate, dietary composition, and site use are expected to have minimal or no effect on risk conclusions.
- There were no toxicity data to screen six COIs
- It is not known whether the uncertainty in the TEQ approach associated with bird TEFs would result in an overestimate or underestimate of risk to pigeon guillemot.
- There are uncertainties associated with using laboratory effects data to estimate risks to pigeon guillemot in the EW, which could result in either overestimates or underestimates of risks.

A.6.3.1.3 *Risk conclusions*

Risks to pigeon guillemot were evaluated by comparing estimated dietary doses with dietary TRVs. Mercury, total PCBs, PCB TEQ, and total TEQ were identified as COPCs for pigeon guillemot as a result of the COPC screen. Uncertainties in the problem formulation and the exposure and effects assessment for pigeon guillemot were evaluated with the conclusion that the potential for adverse effects from exposure to these COPCs is unlikely because the NOAEL and LOAEL HQs were all < 1.0. Based on the results of the risk characterization, none of the four COPCs for pigeon guillemot were identified as COCs.

A.6.3.2 Osprey

This section presents the risk estimates, uncertainties, and risk conclusions for osprey. Chemicals were identified as COCs in the risk conclusion if LOAEL HQs were ≥ 1.0.

A.6.3.2.1 *Risk estimates*

Total PCBs was the only chemical identified as a COPC for osprey based on the chemical screening presented in Section A.2.5.3. The NOAEL and LOAEL HQs for total PCBs were less than 1.0 (Table A.6-38).

Table A.6-38. HQs for osprey

COPC	Dose (mg/kg bw/day)	TRV (mg/kg bw/day)		HQ	
		NOAEL	LOAEL	NOAEL	LOAEL
Total PCBs	0.10	0.49	1.4	0.21	0.074

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
TRV – toxicity reference value

A.6.3.2.2 *Uncertainty assessment*

This section presents a discussion of the uncertainty associated with the problem formulation and the exposure and effects assessment for osprey.

Problem Formulation

ROC Selection

Osprey were selected to represent bird species in the EW that consume primarily fish. Other birds with similar dietary exposures include western grebe and bald eagle. There is some uncertainty associated with how well the exposure of osprey to chemicals in the EW represents the exposure of these other species. As discussed in Section A.2.3.3.1, western grebes are less exposed to contaminants in the EW than are osprey because they are present in the vicinity of the EW only during the winter months and not during their breeding season. Bald eagles are less exposed than are osprey based on their larger foraging range and lower ingestion rates normalized for body weight. Bald eagles may prey on higher-trophic-level organisms (e.g., piscivorous birds), which can have higher concentrations of bioaccumulative chemicals in their tissue compared with those of osprey prey. However, piscivorous birds are likely to be only a small portion of the bald eagle diet in the EW; Watson et al. (1991) found that birds were 7% of the bald eagle diet in the Columbia River estuary, and many of those bird species fed on organisms that were lower in the food chain. The low percentage of piscivorous birds in the bald eagle's diet combined with the larger foraging range for bald eagle are expected to result in lower exposures when compared with osprey.

An additional uncertainty with the selection of osprey is its sensitivity to chemicals in the EW relative to other avian species, which is unknown because of the absence of laboratory toxicological data for many avian species.

COPC Screen

Fifty-four chemicals or chemical groups were identified as COIs for birds. For 20 of these COIs, effects data were available for the specific chemical (or chemical group). For another 28 chemicals (20 individual PAHs and 8 pesticides), effects data were values for related, surrogate chemicals because effects data were not available for the COI, so there is some uncertainty regarding the COPC screen for these chemicals. For the remaining six COIs (silver, monobutyltin, dibutyltin, 1,4-dichlorobenzene, phenol, and mirex), risks to osprey could not be evaluated because effects data were not available. No further evaluation could be conducted for these COIs.

Exposure Assessment

Uncertainty in the exposure assessment for osprey is associated with the following:

- Direct sediment contact
- Incidental sediment ingestion rate
- Dietary composition
- Site use

These uncertainties are discussed in detail below.

Direct Sediment Contact

Risks to wildlife from direct contact with sediment are considered to be 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 osprey because osprey rarely contact the sediment when capturing fish prey (i.e., they forage from approximately the top 1 m of water column). In addition, feathers provide a barrier that reduces the potential for direct contact with sediment.

Incidental Sediment Ingestion Rate

To address uncertainties in the amount of sediment incidentally ingested by osprey while foraging, ingested doses of total PCBs were calculated assuming the sediment ingestion rate was 10% of the food ingestion rate versus 2% assumed in Section 5.1.2.2. This conservative assumption would result in an increase of the NOAEL and LOAEL HQs by no more than 0.02 and would not change the risk conclusions.

Dietary Composition

There is uncertainty in the dietary composition of osprey in the EW. To address this uncertainty, the dietary exposure of osprey was calculated using the conservative assumption that the diet consisted of only the prey species for osprey (i.e., shiner surfperch or juvenile Chinook salmon) that had the maximum EPC for total PCBs (i.e., maximum of values presented in Table A.5-4). Based on this conservative assumption, NOAEL and LOAEL HQs for total PCBs were still below 1.0 (Table A.6-39).

Table A.6-39. HQs for osprey assuming maximum EPC in diet

COPC	Maximum EPC (mg/kg ww [a])	Exposure Dose (mg/kg bw/day)	HQ	
			NOAEL	LOAEL
Total PCBs	2.3	0.20	0.41	0.14

[a] The maximum EPC for total PCBs was for shiner surfperch.

bw – body weight
COPC – chemical of potential concern
EPC – exposure point concentration
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
NOAEL – no-observed-adverse-effect level
PCB – polychlorinated biphenyl
TEF – toxic equivalency factor
TEQ – toxic equivalent
ww – wet weight

Site Use

To address uncertainty in the site use factor of 0.41, ingested doses of total PCBs were calculated assuming that osprey feed only in the EW, resulting in a site use factor of 1.0. This increases the exposure doses and HQs by a factor of 2.4, but the NOAEL and LOAEL HQs remained less than 1.0.

Effects Assessment

There are uncertainties associated with available toxicity benchmarks for birds, which were discussed in Section A.6.3.1.2.

An alternative method to the dietary approach used in this ERA for evaluating risk to osprey from exposure to total PCBs (the only COPC identified for osprey) is to use data for PCB concentrations in osprey eggs collected by the USGS from nests near the EW (Johnson et al. 2009) and compare these concentrations to NOAEL and LOAEL TRVs for total PCB concentrations in bird eggs. This alternative approach is presented below.

The USGS collected 11 osprey eggs from seven nests along the LDW in 2006 and 2007, and 7 eggs from four nests along the LDW in 2003 (Johnson et al. 2009). These eggs were analyzed for PCB Aroclors and a subset of PCB congeners.[37] Total PCB concentrations in the 18 eggs collected between 2003 and 2007, based on the sum of congeners, ranged from 0.30 to 3.7 mg/kg ww. Total PCB concentrations in the 18 eggs, based on the sum of PCB Aroclors, were higher, ranging from 0.61 to 7.3 mg/kg ww. Total PCB concentrations in eggs collected from nests closest to the EW (at T-18 and T-104) ranged from 0.69 to 1.2 mg/kg ww for the sum of congeners and from 1.5 to 2.3 mg/kg ww for the sum of Aroclors. Because the foraging range of osprey is larger than the EW or the LDW, and for the purpose of conducting a conservative evaluation, the maximum total PCB concentration in eggs (7.3 mg/kg ww, based on the sum of Aroclors in an egg collected near river mile 4.4 in the LDW) was used in this analysis.

Five studies that related PCB concentrations in bird eggs to adverse effects were available (Table A.6-40).[38] 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 that ranged 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 and thus was not associated with reduced hatchability. The next lowest LOAEL (16 mg/kg ww [Aroclor 1254]), which resulted in reduced hatching success in the second generation of ring 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 reproductive effects in the screech owl (McLane and Hughes 1980). This egg concentration was selected as the NOAEL TRV.

Table A.6-40. Toxicity studies of 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; egg hatchability was not affected	Lowe and Stendell (1991)
Aroclor 1254	ring dove	na	**16**	two generations	reduced hatchability and embryo survival	Peakall et al. (1972); Peakall and Peakall (1973)

[37] 50 of the 209 PCB congeners were analyzed in the eggs.
[38] For PCBs, reproductive studies with chickens were not considered, as discussed in Section A.5.2.1.2.

Chemical	Test Species	NOAEL (mg/kg ww egg)	LOAEL (mg/kg ww egg)	Exposure Duration	Effect	Source
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
PCB – polychlorinated biphenyl
ww – wet weight
Bold and underline identify the NOAEL and LOAEL selected as the TRVs.

The NOAEL and LOAEL HQs for total PCBs were calculated using the maximum osprey egg concentration of total PCBs and the NOAEL and LOAEL TRVs, resulting in NOAEL and LOAEL HQs of 1.0 and 0.46, respectively (Table A.6-41). These HQs indicate that adverse effects on osprey reproduction from exposure to total PCBs are unlikely.

Table A.6-41. HQs for osprey for total PCBs based on egg data

COPC	Maximum Concentration in Eggs (mg/kg ww)	NOAEL TRV (mg/kg ww)	LOAEL TRV (mg/kg ww)	NOAEL HQ	LOAEL HQ
Total PCBs	7.3	7.1	16	1.0	0.46

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

It should also be noted that four nests near the EW (near Terminal 5 (T-5), T-18, T-104, and T-105) did not produce eggs in all years they were monitored (2003, 2006, and 2007), whereas the other five nests along the LDW produced eggs in each year they were monitored. The nest locations and years in which no young were produced are as follows: T-104 in 2003; T-5, T-18, T-104, and T-105 in 2006; and T_18 and T-104 in 2007.

Johnson et al. (2009) noted that egg concentrations of total PCBs reported in their study were less than recognized effect level concentrations and were not correlated with productivity. They listed possible explanations for nest failures as follows: 1) poorly condition individuals, 2) inexperienced breeders, 3) embryo death from inconsistent incubation behaviors attributed to disturbance or other factors, 4) limited prey abundance during a critical time period for egg production, 5) other contaminants not analyzed in the study, 6) exposure to contaminants *in ovo*, which may have affected adult reproductive potential, or 7) additive or synergistic relationships among contaminants. Johnson et al. (2009) did note that the average number of young produced per active nest in the Duwamish River (including those along EW), was higher than the level necessary to maintain a stable population.

Summary of Uncertainties

Uncertainties in the problem formulation and exposure and effects assessment are summarized as follows:

- Uncertainties in the ROC selection, direct sediment contact, sediment ingestion rate, dietary composition, and site use are expected to have minimal or no effect on the risk conclusions.
- There were no toxicity data to screen six COIs.
- There are uncertainties associated with using laboratory effects data to estimate risks to osprey in the EW, which could result in either overestimates or underestimates of risks.
- An analysis of risk using egg data indicated that risk to osprey is unlikely at the concentrations of total PCBs observed in eggs collected by the USGS from osprey nests in the vicinity of the EW.

A.6.3.2.3 Risk conclusions

Risks to osprey were evaluated by comparing estimated dietary doses with dietary TRVs. Total PCBs was identified as the only COPC for osprey based on the COPC screen. Uncertainties in the problem formulation and the exposure and effects assessment for osprey were evaluated with the conclusion that risks from exposure to total PCBs from the EW are unlikely because both the NOAEL and LOAEL HQs were less than 1.0. Total PCBs was not selected as a COC for osprey as a result of this risk characterization.

A.6.3.3 River otter

This section presents the risk estimates, uncertainties, and risk conclusions for river otter. Chemicals were identified as COCs in the risk conclusion section if LOAEL HQs were ≥ 1.0.

A.6.3.3.1 Risk estimates

Mercury, selenium, total PCBs, PCB TEQ, and total TEQ were identified as COPCs for river otter based on the chemical screening presented in Section A.2.5.3. The LOAEL HQs for all COPCs and the NOAEL HQ for selenium were less than 1.0 (Table A.6-42).

The NOAEL HQs for mercury, total PCBs, PCB TEQ, and total TEQ ranged from 1.0 to 1.5.

Table A.6-42. HQs for river otter

COPC	Exposure Dose (mg/kg bw/day)	TRV (mg/kg bw/day)		HQ	
		NOAEL	LOAEL	NOAEL	LOAEL
Mercury	0.0023	0.0017	0.0084	1.3	0.27
Selenium	0.018	0.055	0.080	0.33	0.23
Total PCBs	0.069	0.045	0.089	1.5	0.78
PCB TEQ[a]	7.4×10^{-7}	6.5×10^{-7}	4.9×10^{-6}	1.1	0.15
Total TEQ	7.8×10^{-7}	6.5×10^{-7}	4.9×10^{-6}	1.2	0.16

[a] PCB TEQ and total TEQ were calculated using TEFs for mammals from Van den Berg et al.(2006).

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
TEF – toxic equivalency factor
TEQ – toxic equivalent
TRV – toxicity reference value

A.6.3.3.2 Uncertainty assessment

This section presents a discussion of the uncertainty associated with the problem formulation, the exposure and effects assessment, and the risk characterization for river otter.

Problem Formulation

ROC Selection

River otter were selected as an ROC to represent semi-aquatic mammals (e.g., excluding marine mammals) that could forage in the EW. The other two semi-aquatic mammal species, muskrats and raccoons, are less exposed than river otter to site-related contaminants because muskrats feed on plants, which are not abundant in the EW, and raccoons feed on a greater proportion of terrestrial prey. Therefore, risk estimates for river otter should provide conservative risk estimates for muskrats and raccoons.

COPC Screen

Fifty-four chemicals or chemical groups were identified as COIs for mammals. For 23 of these COIs, effects data were available for the specific chemical (or chemical group). For another 11 chemicals (all pesticides), effects data were values for related, surrogate chemicals because effects data were not available for the COI, so there is some uncertainty regarding the COPC screen for these chemicals. For the remaining 20 COIs (15 individual PAHs, silver, monobutyltin, 1,4-dichlorobenzene, pentachlorophenol, and mirex), risks to river otter could not be evaluated because effects data were not available. No further evaluation could be conducted for these COIs.

Exposure Assessment

Uncertainty in the exposure assessment for river otter is associated with the following:

- Direct sediment contact
- Incidental sediment ingestion rate
- Dietary composition
- Site use
- TEQ approach
- Surface water data for total TEQ

These uncertainties are discussed in detail below.

Direct Sediment Contact

Risks to wildlife from direct contact with sediment are considered insignificant relative to risks from the incidental sediment ingestion (EPA 2000b). However, the exclusion of this pathway adds only a small amount of uncertainty to the risk estimate for river otter because river otter fur is expected to provide a barrier that reduces the potential for direct contact with sediment.

Incidental Sediment Ingestion Rate

To address uncertainties in the amount of sediment incidentally ingested by river otter while foraging, ingested doses of COPCs were calculated assuming the sediment ingestion rate was 10% of the food ingestion rate versus the 2% assumed for HQs presented in Table A.6-42. This conservative assumption resulted in an increase of the NOAEL and LOAEL HQs by no more than 0.02 and did not change the risk conclusions.

Dietary Composition

There is uncertainty in the dietary composition of river otter in the EW. To address this uncertainty, exposure was calculated using the conservative assumption that the river otter's diet consisted of only the fish and invertebrate species that had the maximum EPC (i.e., maximum of species consumed by river otter presented in Table A.5-4) for each of the COPCs. Based on this conservative assumption, LOAEL HQs were below 1.0 for all COPCs except total PCBs; the LOAEL HQ for total PCBs was 1.4 (Table A.6-43). The NOAEL HQs for mercury, total PCBs, PCB TEQ, and total TEQ were > 1.0, ranging from 1.8 to 3.7. These data indicate a potential for adverse effects to river otter from exposure to total PCBs (i.e., LOAEL HQ > 1.0) and low and uncertain risks from exposure to mercury, PCB TEQ, and total TEQ (i.e., NOAEL HQ > 1.0 and LOAEL HQ < 1.0) based on uncertainty in the dietary composition. Although these calculations are conservative, and it is unlikely that river otter would consume only one fish species, this assessment illustrates that the estimated risk from total PCBs can change depending on dietary composition.

Table A.6-43. HQs for river otter assuming maximum EPC in diet

COPC	Maximum EPC (mg/kg ww)[a]	Exposure Dose (mg/kg bw/day)	HQ	
			NOAEL	LOAEL
Mercury	0.21	0.0062	3.7	0.74
Selenium	1.2	0.35	0.64	0.44
Total PCBs	4.1	0.12	2.7	1.4
PCB TEQ[b]	4.01×10^{-5}	1.2×10^{-6}	1.8	0.24
Total TEQ[b]	7.90×10^{-5}	2.3×10^{-6}	3.6	0.47

a Maximum EPCs for mercury, PCB TEQ, and total TEQ were for brown rockfish, maximum EPC for selenium was in crab, and maximum EPC for total PCBs was in English sole.

b PCB TEQ and total TEQ were calculated using TEFs for mammals from Van den Berg et al.(2006).

bw – body weight
COPC – chemical of potential concern
EPC – exposure point concentration
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
NOAEL – no-observed-adverse-effect level
PCB – polychlorinated biphenyl
TEF – toxic equivalency factor
TEQ – toxic equivalent
ww – wet weight

To further investigate the uncertainty in dietary composition in a more realistic manner, HQs were calculated with the assumption that crab, brown rockfish, shiner surfperch, and English sole are consumed in equal proportions, and that salmon are not consumed because they are not present in the EW throughout the year. Clam and mussel dietary percentages were kept at 1% each, and the remaining 98% of the diet was equally apportioned to brown rockfish, shiner surfperch, English sole, and crab (24.5% each). For each prey type, the EPC used in the original risk calculations (as presented in Table A.5-4) were used. For selenium and PCB TEQ, the NOAEL HQs were ≤ 1.0, indicating that risks are unlikely for these COPCs (Table A.6-44). For mercury, total PCBs, and total TEQ, the NOAEL HQs were > 1.0, and the LOAEL HQs were < 1.0, indicating that risks are low and uncertain for these COPCs. These conclusions are the same as those based on the original HQs (presented in Table A.6-36), except for PCB TEQ, which had risks considered low and uncertain as a result of the original HQ calculations.

Table A.6-44. HQs for river otter based on alternate dietary assumptions

COPC	Exposure Dose (mg/kg bw/day)	HQ	
		NOAEL	LOAEL
Mercury	0.0026	1.5	0.31
Selenium	0.0223	0.4	0.28
Total PCBs	0.0783	1.7	0.88
PCB TEQ[s]	6.4×10^{-7}	1.0	0.13
Total TEQ[s]	6.8×10^{-7}	1.1	0.14

[a] PCB TEQ and total TEQ were calculated using TEFs for mammals from Van den Berg et al.(2006).

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
TEF – toxic equivalency factor
TEQ – toxic equivalent
ww – wet weight

Site Use

To address uncertainty in the site use factor of 0.23, ingested doses of COPCs were calculated using the conservative assumption that river otter obtain half of their food from the EW, resulting in a site use factor of 0.5. This increased the exposure doses and HQs by a factor of 2.2. The LOAEL HQs for mercury, selenium, and PCB TEQ remained below 1.0, and the LOAEL HQ for total PCBs increased from 0.78 to 1.7. These calculations indicate the potential for adverse risks to river otter from exposure to total PCBs based on uncertainty associated with site use, although it is unlikely that river otter would consume half of their diet from the EW.

TEQ 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* over *in vitro* toxicity data. Despite the numerous biological variables such as species, strain, gender, and age included in these studies, the TEF values for a given congener generally fell within a range of about an order of magnitude for mammals (Sanderson and Van den Berg 1999). It is not known whether the uncertainty in these TEFs would underestimate or overestimate risks.

Surface Water Data for Total TEQ

Surface water samples were not analyzed for dioxins and furans, so the surface water EPC for PCB TEQ was also used for total TEQ. There is some uncertainty in this assumption because potential toxicity from dioxins and furans is not included in the surface water exposure estimate. However, this uncertainty is expected to be very low. For example, the dose of PCB TEQ ingested by river otter via the water pathway only is 4.8×10^{-11} mg/kg bw/day, which is only 0.006% of the entire dietary dose of 7.4×10^{-7} mg/kg bw/day. The relative contribution of dioxin/furan TEQ to the total TEQ is expected to be similar to the contribution from PCB TEQ. Therefore, the use of the PCB TEQ as the surface water EPC for total TEQ has a negligible effect on the risk calculations.

Effects Assessment

Uncertainty associated with available toxicity data for mammals may affect risk estimates. The general uncertainty associated with toxicity studies are the same as those discussed in Section A.6.3.1.2 for pigeon guillemot. Specific uncertainties associated with toxicity studies for the mammalian COPCs are presented in Table A.6-45. It should be noted that for all COPCs except selenium, studies have been conducted with mink, a closely related species, which reduces the uncertainty in evaluating toxicity thresholds for river otter. In particular, there are a large number of mink PCB studies, indicating that uncertainty associated with the total PCB TRV is lower than that associated with the TRVs for other COPCs.

Table A.6-45. Level of uncertainty associated with TRVs for mammals

COPC	Number of TRV Studies	Uncertainty in TRV[a]	Rationale
Mercury	3	high	Selected TRVs were based on a chronic growth endpoint; NOAEL extrapolated from the LOAEL based on the use of an uncertainty factor.
Selenium	4	medium	Selected TRVs were based on a subchronic growth endpoint.
Total PCBs	10	medium	Selected TRVs were based on a chronic reproduction endpoint; NOAEL extrapolated from the LOAEL based on the use of an uncertainty factor.
PCB TEQ and total TEQ	7	medium	Selected TRVs were based on a subchronic growth endpoint.

[a] Level of uncertainty key:
Low = large dataset that includes chronic studies with species (e.g., mink) taxonomically similar to the ROC
Medium = medium-sized dataset that includes chronic studies
High = small dataset that includes only subchronic studies, unbounded NOAELs/LOAELs, or data with questionable data quality

COPC – chemical of potential concern
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

Summary of Uncertainties

Uncertainties in the problem formulation and exposure and effects assessment for river otter are summarized as follows:

- Uncertainties in the ROC selection, direct sediment contact, and sediment ingestion rate are expected to have minimal effect on risk conclusions.
- There is a potential for adverse effects from exposure to total PCBs if the site use is higher than estimated or if the dietary composition of river otter consists of only the prey species with the highest total PCB concentrations, although these scenarios are unlikely.
- It is not known whether the uncertainties in the TEQ approach associated with mammalian TEFs would result in an overestimate or underestimate of risk to river otter.
- No toxicity data were available to screen 20 COIs
- Gaps in toxicity data (i.e., limited number of studies for some COPCs, extrapolation of NOAELs from LOAELs) resulted in uncertainty in the risk estimates.
- There are uncertainties associated with using laboratory effects data to estimate risk to river otter in the EW, which could result in either overestimates or underestimates of risks.

A.6.3.3.3 Risk conclusions

Risks to river otter were evaluated by comparing estimated dietary doses with dietary TRVs. Mercury, selenium, and PCBs were identified as COPCs for river otter. Uncertainties in the problem formulation and the exposure and effects assessment for river otter were evaluated, with the following risk conclusions:

- **Mercury –** The NOAEL HQ for mercury was slightly greater than 1.0, but the LOAEL HQ was less than 1.0, indicating that there is a low and uncertain risk from mercury exposure.
- **Selenium –** Risks from dietary exposure to selenium are unlikely because both the NOAEL and LOAEL HQs were less than 1.0.
- **Total PCBs –** The NOAEL HQ for total PCBs was 1.5, but the LOAEL HQ was less than 1.0, indicating that there is a low and uncertain risk from PCB exposure. Uncertainty analyses also indicate that risk is low and uncertain.
- **PCB TEQ and total TEQ** – NOAEL HQs were 1.1 and 1.2 for PCB TEQ and total TEQ, respectively, but LOAEL HQs were < 1.0. Therefore, risks are low and uncertain for PCB TEQ and total TEQ.

None of the COPCs were selected as COCs because the LOAEL HQs were < 1.0.

A.6.3.4 Harbor seal

This section presents the risk estimates, uncertainties, and risk conclusions for harbor seal. Chemicals were identified as COCs in the risk conclusion section if LOAEL HQs were ≥ 1.0.

A.6.3.4.1 Risk estimates

Mercury, total PCBs, PCB TEQ, and total TEQ were identified as COPCs for harbor seal based on the screening presented in Section A.2.5.3. The NOAEL and LOAEL HQs for all COPCs were less than 1.0 (Table A.6-46).

Table A.6-46. HQs for harbor seal

COPC	Dose (mg/kg bw/day)	TRV (mg/kg bw/day)		HQ	
		NOAEL	LOAEL	NOAEL	LOAEL
Mercury	0.00025	0.0017	0.0084	0.14	0.029
Total PCBs	0.00813	0.045	0.089	0.18	0.091
PCB TEQ[a]	9.5×10^{-8}	6.5×10^{-7}	4.9×10^{-6}	0.15	0.019
Total TEQ[a]	1.0×10^{-7}	6.5×10^{-7}	4.9×10^{-6}	0.15	0.021

[a] PCB TEQ and total TEQ were calculated using TEFs for mammals from Van den Berg et al.(2006).

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
TEF – toxic equivalency factor
TEQ – toxic equivalent
TRV – toxicity reference value

A.6.3.4.2 Uncertainty assessment

This section presents a discussion of the uncertainty associated with the problem formulation and the exposure and effects assessment for harbor seal.

Problem Formulation

ROC selection

Harbor seal was selected to represent marine mammals that may use the EW. All three species that could potentially forage in the EW (i.e., harbor seal, sea lion, and harbor porpoise) are opportunistic feeders, primarily feeding on fish and some invertebrates. Therefore, harbor seals should have an exposure similar to that of sea lions and harbor porpoises. There is some uncertainty regarding the relative site use among the three species, which is addressed below in the exposure assessment summary of uncertainties. An additional uncertainty with the selection of harbor seal is its sensitivity to chemicals in the EW relative to that of the other two marine mammal species, which is unknown because of the unavailability of laboratory toxicological data for many wildlife species.

COPC Screen

Fifty-four chemicals or chemical groups were identified as COIs for mammals. For 23 of these COIs, effects data were available for the specific chemical (or chemical group). For another 11 chemicals (all of which were pesticides), effects data were only available for related chemicals, so there is some uncertainty regarding the COPC screen for these chemicals. For the remaining 20 COIs (15 individual PAHs, silver, monobutyltin, 1,4-dichlorobenzene, pentachlorophenol, and mirex), risks to harbor seal could not be evaluated because effects data were not available. No further evaluation could be conducted for these COIs.

Exposure Assessment

Uncertainty in the exposure assessment for harbor seal is associated with the following:

- Direct sediment contact
- Incidental sediment ingestion rate
- Dietary composition
- Site use
- TEQ approach
- Surface water data for total TEQ

These uncertainties are discussed in detail below, except for the uncertainties associated with the TEQ approach, which are the same as those for river otter and were discussed previously in Section A.6.3.3.2.

Direct Sediment Contact

Risks to wildlife from direct contact with sediment were considered insignificant relative to risks from incidental sediment ingestion (EPA 2000b). However, the exclusion of this pathway adds only a small amount of uncertainty to the risk estimate for harbor seal because harbor seal fur is expected to provide a barrier that reduces the potential for direct contact with sediment.

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 sediment ingestion rate was 10% of the food ingestion rate versus the 2% assumed in Section 5.1.2.2. This conservative assumption did not result in an increase in any of the HQs by more than 0.1 and did not change risk conclusions.

Dietary Composition

There is uncertainty in the dietary composition of harbor seal in the EW. To address this uncertainty, exposure was calculated using the conservative assumption that the harbor seal's diet consists of only fish and invertebrate species that had the maximum EPC (i.e., the maximum value for harbor seal prey presented in Table A.5-4) for each of the COPCs. Based on this conservative assumption, NOAEL and LOAEL HQs were below 1.0 for all COPCs (Table A.6-47).

Table A.6-47. HQs for harbor seal assuming maximum EPC in diet

COPC	Maximum EPC (mg/kg ww)[a]	Exposure Dose (mg/kg bw/day)	HQ	
			NOAEL	LOAEL
Mercury	0.21	0.00066	0.39	0.078
Total PCBs	4.1	0.013	0.28	0.14
PCB TEQ[b]	4.01×10^{-5}	1.2×10^{-7}	0.19	0.026
Total TEQ[b]	4.25×10^{-5}	1.3×10^{-7}	0.20	0.027

a The maximum EPCs for mercury, PCB TEQ, and total TEQ were in brown rockfish, and the maximum EPC for total PCBs was in English sole.

b TEQ was calculated using TEFs for mammals from Van den Berg et al.(2006).

bw – body weight
COPC – chemical of potential concern
EPC – exposure point concentration
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
NOAEL – no-observed-adverse-effect level
PCB – polychlorinated biphenyl
TEF – toxic equivalency factor
TEQ – toxic equivalent
ww – wet weight

Site Use

To address uncertainty in the site use factor of 0.1, ingested doses of COPCs were calculated assuming harbor seal obtain one-quarter of their food from in the EW, resulting in a site use factor of 0.25. This conservative assumption increased the exposure doses and HQs by a factor of 2.5. The NOAEL and LOAEL HQs for all of the COPCs remained below 1.0.

Surface Water Data for Total TEQ

Surface water samples were not analyzed for dioxins and furans, so the surface water EPC for PCB TEQ was also used for total TEQ. There is some uncertainty in this assumption because potential toxicity from dioxins and furans is not included in the surface water exposure estimate. However, this uncertainty is expected to be very low. For example, the amount of PCB TEQ ingested by harbor seal via the water pathway only is 3.8×10^{-11} mg/kg bw/day, which is only 0.04% of the entire dietary dose of 9.5×10^{-8} mg/kg bw/day. The relative contribution of dioxin/furan TEQ to the total TEQ is expected to be similar to the contribution from PCB TEQ. Therefore, the use of the PCB TEQ as the surface water EPC for total TEQ has negligible effects on the risk calculations.

Effects Assessment

Uncertainties associated with available toxicity data for mammals are discussed in Section A.6.3.3.2. In addition, there were no toxicity studies available for marine mammals, unlike river otter, which had available data for mink, a closely related species.

Summary of Uncertainties

Uncertainties in the problem formulation and exposure and effects assessment for harbor seal are summarized as follows:

- Uncertainties in the ROC selection, direct sediment contact, sediment ingestion rate, dietary composition, and site use are expected to have minimal effect on risk conclusions.
- It is not known whether the uncertainties in the TEQ approach associated with mammalian TEFs would result in an overestimate or underestimate of risk to river otter.
- No toxicity data were available to screen 20 COIs.
- There are uncertainties associated with gaps in toxicity data and in using laboratory effects data to estimate risks to harbor seal in the EW, which could result in either overestimates or underestimates of risks. Uncertainty in the potential for additive effects of mixtures of chemicals in the EW may result in an underestimate of risks.

A.6.3.4.3 Risk conclusions

Risks to harbor seal were evaluated by comparing estimated dietary doses with dietary TRVs. Mercury, total PCBs, PCB TEQ, and total TEQ were identified as COPCs for harbor seal in the COPC screen. Uncertainties in the problem formulation and the exposure and effects assessment for harbor seal were evaluated with the conclusion that the potential for adverse effects from exposure to each of the four COPCs is low because the NOAEL and LOAEL HQs were all < 1.0. Therefore, none of these COPCs was identified as a COC for harbor seal.

A.6.3.5 Summary of risk conclusions for wildlife

In summary, none of the five COPCs for wildlife was identified as a COC based on the wildlife risk characterization (Table A.6-48). Risks were either unlikely because NOAEL HQs were < 1.0, or risks were low and uncertain because NOAEL HQs were > 1.0, but LOAEL HQs were <1.0.

Table A.6-48. Chemicals identified as COCs for wildlife ROCs

COPC	Pigeon Guillemot	Osprey	River Otter	Harbor Seal
Mercury	no	not evaluated	no	no
Selenium	not evaluated	not evaluated	no	not evaluated
Total PCBs	no	no	no	no
PCB TEQ	no	not evaluated	no	no
Total TEQ	no	not evaluated	no	no

COC – chemical of concern
COPC – chemical of potential concern
PCB – polychlorinated biphenyl
ROC – receptor of concern
TEQ – toxic equivalent

A.7 Selection of Ecological Risk Drivers

This section presents the rationale for the identification of chemicals as risk drivers for EW based on estimated risks to ecological receptors. The risk drivers from both this ERA and the HHRA will be the focus of the remedial analyses in the FS.

In this ERA, ecological risks from chemicals were assessed consistent with CERCLA (EPA 1998). Based on this assessment, chemicals detected in surface sediment, sediment porewater, surface water, and tissue samples collected from the EW were grouped as follows: 1) chemicals for which there is no cause for concern, 2) COPCs, and 3) COCs. Chemicals considered to be risk drivers for ecological receptors are a subset of the COCs. The following factors were considered in identifying risk drivers. These considerations are consistent with those used for the LDW ERA and 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
- Magnitude of exposure concentrations compared to TRVs
- Comparison of concentrations in EW sediment with regional background concentrations in sediment

The risk drivers will be the focus of detailed analyses presented in the FS for all remedial alternatives. COCs not selected as risk drivers in the EW ERA will be evaluated qualitatively in the EW FS. All COCs will be mapped and discussed in the RI (although the RI will provide greater detail for the risk drivers). In consultation with EPA and consistent with the evaluation of non-risk drivers in the LDW, COCs not selected as risk drivers in the EW ERA will be evaluated qualitatively in the EW FS. This evaluation will include a follow-up check for the non-risk-driver COCs to ensure that sediment with elevated levels of these COCs will be included in the remedial footprint of the remedial alternatives evaluated in the FS. Furthermore, all COCs will be included in the long-term monitoring plan for the EW.

The subsections that follow provide more discussion on the risk drivers selected based on the benthic invertebrate, crab, and fish COCs; there were no COCs, and hence no risk drivers, for wildlife ROCs.

A.7.1 Risk Driver Evaluation for Benthic Invertebrates and Crab

Table A.7-1 provides a summary of the risk driver evaluation for all benthic invertebrate and crab COCs. Twenty-eight chemicals were identified as risk drivers for the benthic invertebrate community (Table A.7-1). These were selected because the detected concentrations of those chemicals in the EW 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. In addition, TBT was identified as a risk driver based on concentrations that exceeded the TBT tissue TRV in two composite benthic invertebrate tissue samples.

Table A.7-1. Benthic invertebrate and crab risk driver evaluation

ROC	Exposure Pathway	COCs	Maximum NOAEL-Based HQ	Maximum LOAEL-Based HQ	Rationale	Risk Driver
Benthic invertebrate community	sediment	28 SMS chemicals[a]	range of values	range of values	**uncertainty in exposure data –** low uncertainty in the exposure dataset **uncertainty in effects data –** based on SMS **uncertainty in risk characterization –** each of these 28 chemicals had at least one detected exceedance of SQS in the baseline surface sediment dataset , 6 chemicals only exceeded the SQS and 22 chemicals exceeded both the SQS and the CSL of the WA state SMS	yes
		total DDTs	na	1.4	**uncertainty in exposure data –**uncertainty in sediment dataset due to low detection frequency (5.6%) and known analytical uncertainties in historical data due to analytical interferences from PCBs **uncertainty in effects data –** The SL value is uncertain because not based on AET but instead based on ML divided by 10. **uncertainty in risk characterization** – Two samples had concentrations above the SL and no detected results exceeded the ML	no
Benthic invertebrate community	tissue residue	TBT	16	3.3	**uncertainty in exposure data –**low uncertainty in exposure dataset **uncertainty in effects data –** TRV is applicable to gastropods, and therefore, there is uncertainty regarding the relevance of the TRV to the entire benthic invertebrate community primarily because the imposex endpoint is specific to gastropods. **uncertainty in risk characterization –** two composite tissue samples representing two areas of EW exceeded the LOAEL TRV	yes
Benthic invertebrate community	surface water	TBT	na	1.4	**uncertainty in exposure data –** high uncertainty in the surface water dataset (only one detected value in surface water dataset); the reporting limits associated with the non-detected results exceeded the WQC **uncertainty in effects data –** low uncertainty in toxicity dataset	no

Table A.7-1. Benthic invertebrate and crab risk driver evaluation (cont.)

ROC	Exposure Pathway	COCs	Maximum NOAEL-Based HQ	Maximum LOAEL-Based HQ	Rationale	Risk Driver
Benthic invertebrate community	porewater	naphthalene	300	6	**uncertainty in exposure data –**low uncertainty in the exposure dataset **uncertainty in effects data –** high uncertainty in the lowest LOEC (the lowest LOEC within in a large dataset that contained toxicity data for 20 aquatic invertebrates); the LOEC TRV of 8 µg/L was at least two orders of magnitude lower than other effect concentrations for aquatic invertebrates **uncertainty in risk characterization –** only one porewater sample exceeded the LOEC; naphthalene did not exceed SMS in any sediment sample	no
Crab	tissue residue	cadmium	6.0	1.4	**uncertainty in effects data –**uncertainty in effects data because toxicity data were available for survival endpoints only and there were no data for crab; TRV based on crayfish toxicity **uncertainty in risk characterization** – maximum exceedance of LOAEL is less than 2 **comparison to background–** exposure concentration in EW sediment (SWAC of 0.66 mg/kg dw) was less than PSAMP rural Puget Sound concentration (0.73 mg/kg dw [90th percentile])	no
		copper	11	1.1	**uncertainty in effects data –**uncertainty in effects data because toxicity data were available for survival endpoints only and there were no data for crab; TRV based on shrimp toxicity **uncertainty in risk characterization** – maximum exceedance of LOAEL is less than 2 **comparison to background –** exposure concentration in EW sediment (SWAC of 62 mg/kg dw) was similar to PSAMP rural Puget Sound concentration (50 mg/kg dw [90th percentile])	no
		zinc	4.2	1.5	**uncertainty in effects data –** high uncertainty in the lowest LOAEL; only one study was available and it evaluated only the survival endpoint and did not use crab as a test species; TRV based on crayfish toxicity **uncertainty in risk characterization** – maximum exceedance of LOAEL is less than 2	no

[a] Arsenic, cadmium, mercury, zinc, acenaphthene, benzo(a)anthracene, benzo(a)pyrene, benzo(g,h,i)perylene, chrysene, dibenzo (a,h)anthracene, fluoranthene, fluorene, indeno(1,2,3,-c,d)pyrene, phenanthrene, pyrene, total benzofluoranthenes, HPAH, LPAH, bis(2-ethylhexyl) phthalate, butyl benzyl phthalate, di-n-butyl phthalate, 1,4-dichlorobenzene, 2-methylnaphthalene, 2,4-dimethylphenol, dibenzofuran, n-nitrosodiphenylamine, phenol, and total PCBs.

Table A.7-1. Benthic invertebrate and crab risk driver evaluation (cont.)

COC – chemical of concern
dw – dry weight
ERA – ecological risk assessment
EW – East Waterway
HPAH – high-molecular-weight polycyclic aromatic hydrocarbon
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
LOEC – lowest observed effect concentration
LPAH – low-molecular-weight polycyclic aromatic hydrocarbon
ML – maximum level
NOAEL – no-observed-adverse-effect level
PSAMP – Puget Sound Ambient Monitoring Program
PCB – polychlorinated biphenyl
ROC – receptor of concern
SMS – Washington State Sediment Management Standards
SQS – sediment quality standard
SWAC – spatially weighted average concentration
TRV – toxicity reference value

The other COC based on sediment chemistry for benthic invertebrates is total DDTs. Total DDTs does not have an SQS value but does have an exceedance factor greater than 1.0 for the benthic invertebrate community based on the DMMP SL guideline. However, total DDTs was not selected as a risk driver, primarily because DDTs were detected in relatively few samples (8 detections in 143 samples), and only two detected concentrations were above the SL. Also, there is uncertainty associated with the DMMP SL guideline for total DDTs. DMMP guidelines are not state standards for in-place sediment, and the SL value for total DDTs is based on an uncertainty factor of 10 applied to the ML value, which is based on an AET value. There were no detected concentrations of total DDTs greater than the ML.

TBT was identified as a COC for the benthic invertebrate community based on surface water exposure. TBT was not identified as a risk driver because of the high uncertainty in the surface water TBT dataset. TBT was detected in only one out of 59 surface water samples, and the reporting limits for the non-detected concentrations were above the federal marine chronic WQC. Therefore, there is uncertainty as to whether TBT is present at levels above the WQC.

Naphthalene was identified as a COC for the benthic invertebrate community based on porewater exposure, but was not selected as a risk driver because naphthalene concentrations exceeded the LOAEL TRV in only one porewater sample, and naphthalene did not exceed SMS in any sediment sample. In addition, there was uncertainty in the lowest LOAEL. The LOAEL TRV of 8 µg/L is at least two orders of magnitude lower than other effect concentrations for aquatic invertebrates in a large dataset for more than 20 species of aquatic invertebrates.

Cadmium, copper, and zinc were identified as COCs for crab based on the tissue residue evaluation; there were no COCs based on water chemistry data. Cadmium and copper were not identified as risk drivers because of uncertainty in the effects dataset (only survival endpoints were evaluated, and crayfish and shrimp were the test species), relatively low risk estimates (LOAEL HQs of 1.4 for cadmium and 1.1 for copper), and spatially weighted average concentrations (SWACs) in sediment were less than or similar to concentrations in the PSAMP dataset from rural Puget Sound (NOAA and Ecology 2000) (Table A.7-1). Background information is important because the CERCLA program generally does not require cleanup to concentrations below natural or anthropogenic background levels (EPA 2002). Zinc was not identified as a risk driver because of the low level of risk associated with the maximum LOAEL HQ of 1.5 and high uncertainty in the TRV. Only one study was available to develop the LOAEL; the single study evaluated only the survival endpoint and crayfish as the test species. COCs not selected as risk drivers will be evaluated qualitatively in the EW FS.

A.7.2 RISK DRIVER EVALUATION FOR FISH

Table A.7-2 provides a summary of the risk driver evaluation for fish ROCs. Total PCBs was selected as a risk driver for English sole and brown rockfish because PCBs in tissue residues exceeded the higher LOAEL TRV that was associated with significant effects and uncertainties are low in the exposure data. While there are uncertainties in the study, particularly associated with the lower LOAEL, the effects were significant and more certain at the higher LOAEL. COCs that were not selected as risk drivers for fish included cadmium (juvenile Chinook salmon, English sole, and brown rockfish), copper (English sole), and vanadium (English sole) and TBT (rockfish).

Cadmium was not selected as a risk driver for juvenile Chinook salmon, English sole, or brown rockfish, 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). In addition, the SWAC for cadmium in EW sediment was less than sediment cadmium concentrations in the PSAMP dataset from rural Puget Sound (NOAA and Ecology 2000) (Table A.7-2).

Copper was not selected as a risk driver for English sole primarily because the risk estimate was low, with a LOAEL HQ of 1.1. There is uncertainty associated with the dietary TRV which was driven by reduced growth in juvenile brown rockfish and the SWAC for copper in EW sediment was similar to sediment copper concentrations in the PSAMP dataset from rural Puget Sound (NOAA and Ecology 2000) (Table A.7-2).

Vanadium was not selected as a risk driver for English sole because of high uncertainty in the effects dataset for this chemical (one toxicity study) and because the vanadium sediment SWAC in the EW was 56.7 mg/kg dw, which is less than the 90th percentile vanadium concentration (64 mg/kg dw) in PSAMP rural Puget Sound sediment (NOAA and Ecology 2000). Finally, the risk estimate was low (LOAEL-based HQ of 1.9).TBT was not selected as a risk driver for brown rockfish because concentrations of TBT in tissue exceeded the LOAEL TRV in only three individual fish. In addition, there was uncertainty in the selected TRV and the risk estimates were low (LOAEL-based HQs of 1.1, 1.1, and 1.4 in the three individual brown rockfish).

Table A.7-2 Fish risk driver evaluation

ROC	Exposure Pathway	COC	Maximum NOAEL-Based HQ	Maximum LOAEL-Based HQ	Rationale	Risk Driver
English sole	tissue residue	total PCBs	7.9 – 39[a]	1.6–7.9[a]	**uncertainty in exposure data –** low uncertainty in tissue residue concentrations **uncertainty in effects data –**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; however, significant effects observed in higher LOAEL TRV **uncertainty in risk characterization** – EPC exceeded both LOAELs	yes
Brown rockfish	tissue residue	total PCBs	12 – 60[a]	2.3 – 12[a]	**uncertainty in exposure data –** low uncertainty in tissue residue 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; however, significant effects observed in higher LOAEL TRV **uncertainty in risk characterization** – EPC representing population of rock fish exceeded both LOAELs	yes
Brown rockfish	tissue residue	TBT	14	1.4	**uncertainty in exposure data –** low uncertainty in tissue residue concentrations **uncertainty in effects data –** high uncertainty in toxicity dataset **uncertainty in risk characterization** – EPC representing population of rock fish did not exceed LOAEL; three of 13 individual rockfish concentrations exceed the LOAEL; the maximum exceedance of LOAEL is low	no

Table A.7-2 Fish risk driver evaluation (cont.)

ROC	Exposure Pathway	COC	Maximum NOAEL-Based HQ	Maximum LOAEL-Based HQ	Rationale	Risk Driver
Juvenile Chinook salmon	dietary	cadmium	4.9	1.0	**uncertainty in exposure data –** low uncertainty in stomach content chemical concentrations; uncertainty in use of benthic invertebrate data as prey when compared to stomach content chemical concentrations **uncertainty in effects data –** high uncertainty in lowest LOAEL; TRV was two orders of magnitude lower than LOAELs in the other eight studies, and the growth effect in the TRV-derived study may have been related to food avoidance All salmonid studies resulted in higher LOAELs (i.e., lower sensitivities compared with selected TRV test species). **comparison to background –** exposure concentration in EW sediment (SWAC of 0.66 mg/kg dw) was less than PSAMP rural Puget Sound concentration (0.73 mg/kg dw [90th percentile])	no
English sole	dietary	cadmium	12	2.4	**uncertainty in exposure data –** low uncertainty in benthic invertebrate tissue concentrations **uncertainty in effects data –** high uncertainty in lowest LOAEL; TRV was two orders of magnitude lower than LOAELs in the other eight studies, and the growth effect in the TRV-derived study may have been related to food avoidance **comparison to background –** exposure concentration in EW sediment (SWAC of 0.66 mg/kg dw) was less than PSAMP rural Puget Sound concentration (0.73 mg/kg dw [90th percentile])	no
	dietary	copper	2.2	1.1	**uncertainty in exposure data –** low uncertainty in benthic invertebrate tissue concentrations **uncertainty in effects data –** medium uncertainty in toxicity dataset **uncertainty in risk characterization** –exceedance of LOAEL is low **comparison to background –** exposure concentration in EW sediment (SWAC of 62 mg/kg dw) was slightly higher than the PSAMP rural Puget Sound concentration (50 mg/kg dw [90th percentile])	no

Table A.7-2 Fish risk driver evaluation (cont.)

ROC	Exposure Pathway	COC	Maximum NOAEL-Based HQ	Maximum LOAEL-Based HQ	Rationale	Risk Driver
	dietary	vanadium	9.5	1.9	**uncertainty in exposure data –** low uncertainty in benthic invertebrate tissue **uncertainty in effects data –** high uncertainty in the lowest LOAEL; only one toxicity study was available **uncertainty in risk characterization** –exceedance of LOAEL is low **comparison to background –** exposure concentration in EW sediment (SWAC of 56.7 mg/kg dw) was less than PSAMP rural Puget Sound concentration (64 mg/kg dw [90^{th} percentile])	no
Brown rockfish	dietary	cadmium	13	2.5	**uncertainty in exposure data –** low uncertainty in prey tissue concentrations **uncertainty in effects data –** high uncertainty in lowest LOAEL; TRV was two orders of magnitude lower than LOAELs in the other eight studies, and the growth effect in the TRV-derived study may have been related to food avoidance **comparison to background –** exposure concentration in EW sediment (SWAC of 0.66 mg/kg dw) was less than PSAMP rural Puget Sound concentration (0.73 mg/kg dw [90^{th} percentile])	no

[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.

dw – dry weight
ERA – ecological risk assessment
EW – East Waterway
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
NOAEL – no-observed-adverse-effect level
PSAMP – Puget Sound Ambient Monitoring Program
PCB – polychlorinated biphenyl
ROC – receptor of concern
SWAC – spatially weighted average concentration
TRV – toxicity reference value

A.8 Conclusions

Baseline risks in this ERA were calculated based on chemical concentrations in surface sediment, tissue, surface water, and porewater samples collected from the EW to estimate the chemical exposure of benthic invertebrates, crabs, fish, birds, and mammals that may reside or forage in the EW for at least a portion of their lives. 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 EW from COPCs are summarized below, including a summary of the COPCs that were identified as COCs. Following the summary of risk conclusions, this section summarizes the selection of risk driver chemicals.

A.8.1 Summary of Risk Conclusions

A.8.1.1 Benthic invertebrate community

Risks to the benthic invertebrate community were evaluated through four different approaches: surface sediment, tissue-residue, surface water, and sediment porewater. 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 approximately 40% of the EW area (i.e., the area in which chemical concentrations were less than or equal to SQS chemical criteria and/or sediments were non-toxic according to SQS biological effects criteria). There is a higher likelihood for adverse effects in approximately 21% of the EW area, which had chemical concentrations or biological effects in excess of the CSL values. The remaining 39% of the EW area had chemical concentrations or biological effects between the SQS and CSL values, indicating the potential for minor adverse effects. Some uncertainty is associated with these area estimates because areas were calculated using Thiessen polygons by interpolating from individual points at which sediments were sampled. Twenty-nine chemicals or groups of chemicals had at least one concentration that exceeded its respective SQS or SL and were therefore identified as COCs for the benthic invertebrate community. These chemicals include 4 metals, 16 individual PAHs or group of PAHs, 3 phthalates, 4 other SVOCs, total PCBs, and total DDTs.

TBT was identified as a COC based on the tissue-residue evaluation because the LOAEL TRV was exceeded in composite benthic tissue samples from 2 of the 13 areas that were evaluated. For total PCBs, risk was predicted to be low and uncertain for because tissue concentrations were below LOAEL TRVs but greater than NOAEL TRVs in some areas.

One detected TBT concentration in surface water exceeded the marine chronic WQC for TBT. However, reporting levels associated with the undetected results also exceeded the WQC. Therefore, it was concluded that risks are low and uncertain for the exposure of the benthic invertebrate community to TBT in surface water and TBT was identified as a COC.

Risks to the benthic invertebrate community from all VOCs detected in sediment porewater were unlikely, except for naphthalene, which exceeded the LOEC at only one location. Naphthalene was identified as a COC based on the porewater evaluation.

A.8.1.2 Crabs

Risks to crab were evaluated through two different approaches: tissue-residue and surface water. Cadmium, copper, and zinc were identified as COCs because concentrations of these COPCs in crab tissue were greater than LOAEL TRVs, indicating a potential for adverse effects on crab. Arsenic and total PCBs, the two remaining COPCs, had concentrations in crab tissue below LOAEL TRVs but above NOAEL TRVs indicating low but uncertain risks to crab. These two COPCs were not identified as COCs because concentrations were below LOAEL TRVs. Based on the surface water evaluation, no COCs were identified for crab and risks are unlikely based on surface water exposures.

A.8.1.3 Fish

Risks to the three fish ROCs (juvenile Chinook salmon, English sole, and brown rockfish) were evaluated through tissue-residue or dietary approaches (depending on the chemical) and exposure to surface water. Five chemicals (cadmium, copper, vanadium, TBT, and total PCBs) were identified as COCs for fish, based on the tissue-residue or dietary evaluations, which indicated a potential for risks; no COCs were identified through surface water evaluation indicating risks are unlikely from surface water exposures. Cadmium was identified as a COC for all three fish ROCs for the dietary evaluation. Copper and vanadium were identified as COCs for English sole for the dietary evaluation. Total PCBs was identified as a COC for English sole and brown rockfish and TBT for brown rockfish based on the tissue-residue evaluation. Risks were considered low and uncertain or unlikely for the remaining COPCs.

A.8.1.4 Wildlife

Risks to wildlife ROCs were evaluated based on ingested doses of aquatic prey, surface water, and sediment. Risks were evaluated for two bird ROCs (pigeon guillemot and osprey) and two mammal ROCs (river otter and harbor seal). Exposures were below NOAELs for all COPCs and therefore risks to birds and harbor seals from chemicals in EW are considered to be unlikely. For river otter, risks associated with all COPCs except total PCBs are unlikely because exposures were below NOAELs. The potential for adverse effects was considered low and uncertain for river otters exposed total PCBs because the NOAEL TRV was exceeded, but the LOAEL TRV was not exceeded. No COCs were identified for the bird or mammal ROCs.

A.8.2 SUMMARY OF RISK DRIVER CHEMICALS

COCs that were identified as risk drivers for ecological receptors were based on the risk estimates, uncertainties discussed in this ERA, and background concentrations in accordance with EPA guidance (1992, 1997a, b, 1998) and consistent with the LDW ERA. The risk drivers from both this ERA and the HHRA will be the focus of remedial analyses in the FS. COCs not selected as risk drivers in the EW ERA will be evaluated qualitatively in the EW FS.

COCs that were identified as risk drivers are noted in Table A.8-1. Twenty-eight COCs were selected as risk drivers in sediment for the benthic invertebrate community because the concentrations of these 28 chemicals exceeded SMS in one or more locations. TBT was identified as a risk driver for the benthic invertebrate community based on a tissue-residue evaluation. Total PCBs was identified as a risk driver for English sole and brown rockfish because tissue concentrations for these fish were greater than both LOAEL TRVs, and uncertainties in these risk estimates were relatively low. Other COCs were not selected as risk drivers because of uncertainties in exposure or effects data, magnitude of exposure concentrations compared to TRVs, or comparison to sediment concentrations in regional background data.

Table A.8-1. COCs and risk drivers identified for ERA receptors

Receptor	Evaluation Type	COPCs	COCs	Risk Driver
Benthic Invertebrates	sediment	29 chemicals including metals, PAHs, PCBs, phthalates, other SVOCs, and total DDTs	29 COPCs[a]	28 SMS chemicals
	tissue residue	TBT, total PCBs	TBT	TBT
	surface water	cadmium, mercury, TBT	TBT	none
	porewater	naphthalene	naphthalene	none
Crab	tissue residue	arsenic, cadmium, copper, zinc, and total PCBs	cadmium, copper, zinc	none
	surface water	cadmium, mercury, TBT	none	none
Fish	dietary	arsenic, cadmium, chromium, copper, vanadium, benzo(a)pyrene	cadmium, copper, vanadium	none
	tissue residue	beta-endosulfan, mercury, total PCBs, TBT	total PCBs, TBT	total PCBs
	surface water	cadmium, mercury, TBT	none	none
Birds	dietary dose	mercury, total PCBs, PCB TEQ	none	none
Mammals	dietary dose	mercury, selenium, total PCBs, PCB TEQ	none	none

[a] Arsenic, cadmium, mercury, zinc, acenaphthene, benzo(a)anthracene, benzo(a)pyrene, benzo(g,h,i)perylene, chrysene, dibenzo (a,h)anthracene, fluoranthene, fluorene, indeno(1,2,3,-c,d)pyrene, phenanthrene, pyrene, total benzofluoranthenes, HPAH, LPAH, bis(2-ethylhexyl) phthalate, butyl benzyl phthalate, di-n-butyl phthalate, 1,4-dichlorobenzene, 2-methylnaphthalene, 2,4-dimethylphenol, dibenzofuran, n-nitrosodiphenylamine, phenol, total PCBs, and total DDTs. All COCs have exceedances of SMS chemical criteria except total DDTs, which was based on exceedance of DMMP guideline.

COC – chemical of potential concern
COPC – chemical of potential concern
DDT – dichlorodiphenyltrichloroethane
DMMP – dredged material management program
PAH – polycyclic aromatic hydrocarbon
HPAH – high-molecular-weight polycyclic aromatic hydrocarbon
LPAH – low-molecular-weight polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
SMS – Washington State Sediment Management Standards
SVOC – semivolatile organic compound
TBT – tributyltin
TEQ – toxic equivalent

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