LWG
LOWER WILLAMETTE GROUP

PORTLAND HARBOR RI/FS

APPENDIX G
BASELINE ECOLOGICAL RISK ASSESSMENT

DRAFT FINAL

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July 1, 2011

Prepared for
The Lower Willamette Group

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WE-09-0001

RECOMMENDED FOR INCLUSION IN ADMINISTRATIVE RECORD

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LIST OF ACRONYMS

Acronym	Definition
2,4-D	2,4-dichlorophenoxyacetic acid
2,4-DB	4-(2,4-dichlorophenoxy)butyric acid
2,4,5-T	2,4,5-trichlorophenoxyacetic acid
ACR	acute-to-chronic ratio
AET	apparent effects threshold
ANOVA	analysis of variance
AOC	area of concern
AOPC	area of potential concern
aRPD	apparent redox potential discontinuity
ASTM	American Society for Testing and Materials
AWQC	ambient water quality criteria
BAF	bioaccumulation factor
BCF	bioconcentration factor
BEHP	bis(2-ethylhexyl) phthalate
BERA	baseline ecological risk assessment
BEST	Biomonitoring of Environmental Status and Trends (protocol)
BHHRA	baseline human health risk assessment
BMR	biomagnification regression
BSAF	biota-sediment accumulation factor
BSAR	biota-sediment accumulation regression
bw or BW	body weight
CERCLA	Comprehensive Environmental Response, Compensation, and Liability Act
CFD	cumulative frequency distribution
cfs	cubic feet per second
COC	contaminant of concern
COI	contaminant of interest
COPC	contaminant of potential concern
CRD	Columbia River Datum
CSM	conceptual site model

Acronym	Definition
DDD	dichlorodiphenyldichloroethane
DDE	dichlorodiphenyldichloroethylene
DDT	dichlorodiphenyltrichloroethane
DL	detection limit
dw	dry weight
EC50	concentration that causes a non-lethal effect in 50% of an exposed population
Eco-SSL	ecological soil screening level
EE/CA	engineering evaluation/cost analysis
EPA	US Environmental Protection Agency
EPC	exposure point concentration
ERA	ecological risk assessment
ERM	effects range – median
EROD	ethoxyresorufin-O-deethylase
ESA	Endangered Species Act
ESB	equilibrium partitioning sediment benchmark
FAV	final acute value
FCV	final chronic value
FPM	floating percentile model
FIR	food ingestion rate
FN	false negative
FP	false positive
FS	feasibility study
FSP	field sampling plan
FWM	food web model
GIS	geographic information system
GMAV	genus mean acute value
GWPA	groundwater pathway assessment
HCH	hexachlorocyclohexane
HHRA	human health risk assessment
HPAH	high-molecular-weight polycyclic aromatic hydrocarbon
HQ	hazard quotient

Acronym	Definition
ID	identification
IWC	integrated water column
J-qualifier	estimated concentration
L0	Level 0 (non-toxic)
L1	Level 1 (low toxicity)
L2	Level 2 (moderate toxicity)
L3	Level 3 (high toxicity)
LC50	concentration that is lethal to 50% of an exposed population
LC10	concentration that is lethal to 10% of an exposed population
LCV	lowest chronic value
LD50	dose that is lethal to 50% of an exposed population
LOAEL	lowest-observed-adverse-effect level
LOE	line of evidence
LPAH	low-molecular-weight polycyclic aromatic hydrocarbon
LRM	logistic regression model
LWG	Lower Willamette Group
LWR	Lower Willamette River
MATC	maximum acceptable toxicant concentration
MCPA	2-methyl-4-chlorophenoxyacetic acid
MCPP	methylchlorophenoxypropionic acid
MQ	mean quotient
N-qualifier	presumptive evidence of a compound
NAVD	North American Vertical Datum
NMFS	National Marine Fisheries Service
NN	natural neighbors
NOAA	National Oceanic and Atmospheric Administration
NOAEL	no-observed-adverse-effect level
O&M	operation and maintenance
OC	organic carbon
ODEQ	Oregon Department of Environmental Quality
ODFW	Oregon Department of Fish and Wildlife
PABAK	prevalence- and bias-adjusted (Cohen’s) kappa

Acronym	Definition
PAH	polycyclic aromatic hydrocarbon
PCB	polychlorinated biphenyl
PEC	probable effects concentration
PEL	probable effects level
PIT	passive integrated transponder
pMax	maximum probability of toxicity
PRE	preliminary risk evaluation
PRG	preliminary remediation goal
QAPP	quality assurance project plan
REV	reference envelope value
RI	remedial investigation
RM	river mile
RSET	Regional Sediment Evaluation Team
SCRA	site characterization and risk assessment
SIR	sediment ingestion rate
SL	screening level
SL1	screening level 1
SL2	screening level 2
SLERA	screening-level ecological risk assessment
SMDP	scientific/management decision point
SOW	scope of work
SPI	sediment profile imaging
SQG	sediment quality guideline
SQV	sediment quality value
SSD	species sensitivity distribution
SUF	site use factor
SVOC	semivolatile organic compound
SWAC	spatially weighted average concentration
T-qualifier	value calculated or selected from multiple results
TBT	tributyltin
TCDD	tetrachlorodibenzo-*p*-dioxin
TEF	toxic equivalency factor

Acronym	Definition
TEQ	toxic equivalent
TOC	total organic carbon
total DDx	sum of all six DDT isomers (2,4′-DDD; 4,4′-DDD; 2,4′-DDE; 4,4′-DDE; 2,4′-DDT; and 4,4′-DDT)
TPH	total petroleum hydrocarbons
TRV	toxicity reference value
TSC	threshold sediment concentration
TTC	threshold tissue concentration
TU	toxicity unit
TZW	transition zone water
UCL	upper confidence limit on the mean
UF	uncertainty factor
UPL	upper prediction limit
USACE	US Army Corps of Engineers
USFWS	US Fish and Wildlife Service
USGS	US Geological Survey
UV	ultraviolet
VOC	volatile organic compound
WDFW	Washington State Department of Fish and Wildlife
WDG	Washington Department of Game
WOE	weight of evidence
WQS	water quality standards
ww	wet weight
XAD	Infiltrex™ 300 system with an XAD-2 resin column

GLOSSARY

Term	Definition
acute	occurring within a short period of time, typically an hour to a day in ecotoxicology
acute-to-chronic ratio	the ratio of the concentration at which acute effects occur to that at which chronic effects occur
ambient water quality criterion	contaminant concentration considered to be protective of aquatic biota
ammocoete	filter-feeding larval life stage of the lamprey
anadromous	describes fish species that migrate to saltwater and then return to freshwater rivers and lakes to breed
apparent redox potential discontinuity depth	an estimation of the depth at which the oxygenated surface sediment layer transitions to anoxic conditions; used as a measure of community succession in the sediment profile imaging analysis
assessment endpoint	the explicit expression of the ecological entity to be evaluated in an ecological risk assessment
benthic	relating to or characteristic of the bottom of an aquatic body or the organisms and plants that live there
benthopelagic	living and feeding (on benthic as well as free-swimming organisms) on the bottom as well as throughout the water column
benthos	organisms that live in or on the sediment or other bottom substrates in a water body
bioaccumulation	the accumulation of a substance in an organism
bioconcentration factor	the concentration of a contaminant in the tissues of an organism divided by the concentration in water
biomagnification	the increase in concentration of a substance in the tissue of an organism within each successive increase of trophic level
biomagnification regression	a mathematical equation that attempts to describe the relationship between the concentration of a chemical in prey tissue and the concentration of the chemical in predator/consumer tissue using co-located data pairs
biota-sediment accumulation factor	the ratio of the concentration of a contaminant in the tissue of an organism to the concentration in sediment

Term	Definition
biota-sediment accumulation regression	a mathematical equation that attempts to describe the relationship between the concentration of a contaminant in the tissue of an organism and the concentration of the contaminant in sediment using co-located data pairs
bioturbation	the disturbance of sediment by the actions of organisms living on or in the bottom
contaminant of concern (COC)	the subset of contaminants posing potentially unacceptable risk that are necessary and sufficient to develop and evaluate remedial alternatives that are protective of ecological resources
contaminant of interest (COI)	contaminant detected in the Study Area in any exposure medium (i.e., surface water, transition zone water, sediment, and tissue)
contaminant of potential concern (COPC)	the subset of contaminants of interest with maximum detected concentrations that are greater than screening-level effect thresholds
Contaminant posing potentially unacceptable risk	The subset of contaminants of potential concern exceeding toxicity reference values in the final step of the risk characterization (i.e., considering ecologically relevant diets and exposure area sizes)
chironomid	small non-biting midges (in the fly family) with an aquatic larval stage during which they significantly contribute to the benthic biomass of an ecosystem
chronic	occurring over a longer period of time relative to an organism's life
community	a group of interacting organisms (multiple species) that share a common environment in both space and time
composite sample	an analytical sample created by mixing together two or more individual samples; tissue composite samples are composed of two or more individual organisms, and sediment composite samples are composed of two or more individual sediment grab samples
conceptual site model	a description of the links and relationships between contaminant sources, routes of release or transport, exposure pathways, and the ecological receptors at a site
congener	a specific chemical within a group of structurally related chemicals (e.g., PCB congeners)
crustacean	an invertebrate with several pairs of jointed legs, a hard protective outer shell, two pairs of antennae, and eyes at the end of stalks (e.g., crayfish, beach fleas, and sand hoppers)

Term	Definition
decapod	a group of crustaceans with an external skeleton and five pairs of walking legs (e.g., crayfish and prawns)
detritivore	an organism that eats detritus (e.g., Pacific lamprey ammocoetes)
detritus	loose, unconsolidated material, primarily composed of tiny organic fragments (e.g., remains of plants and animals, bacteria, fungi)
ecological risk assessment	a process to evaluate the likelihood that adverse ecological effects might occur or are occurring as a result of exposure to one or more contaminants
dose	the quantity of an contaminant taken in or absorbed at any one time, expressed on a body weight-specific basis; units are generally expressed as mg/kg bw/day
effects assessment	the part of a risk assessment that describes the relationship between exposure to a contaminant and effects on ecological receptors
effect threshold	a level of contaminant exposure of a receptor above which a particular effect is expected to occur or below which no effect is expected to occur
empirical data	data quantified in a laboratory
epibenthic	bottom-dwelling aquatic organisms that live on the sediment or other hard surface
equilibrium partitioning sediment benchmark	sediment concentration derived using the equilibrium partitioning approach to assess the likelihood of significant adverse effects to benthic organisms
equilibrium partitioning approach	based on a theory stating that a nonionic chemical in sediment partitions between sediment organic carbon, porewater, and benthic organisms; at equilibrium, if the concentration in any one phase is known, the concentration in the others can be predicted
exposure assessment	the part of a risk assessment that characterizes the contaminant exposure of a receptor
exposure pathway	physical route by which an contaminant moves from a source to a biological receptor
exposure point	the location or circumstances at which an organism is assumed to contact a contaminant
exposure point concentration	the concentration of an contaminant at the exposure point

Term	Definition
exposure scale	size of the area throughout which a receptor might come in contact with an contaminant as determined by home range or foraging habits
hazard quotient	the quotient of the concentration of a contaminant in an environmental medium divided by the effect threshold
herbivores	organisms that eat primarily plants
home range	area over which an individual organism conducts activities throughout its lifespan
infauna	bottom-dwelling aquatic organisms that burrow within a soft substrate
invertivore	organism that eats primarily insects or other invertebrates
line of evidence	one method for evaluating risks to a particular ecological receptor; is generally specific to an exposure pathway and/or medium
lipid-normalized concentration	a chemical concentration in biota tissue adjusted for lipid concentration
lowest-observed-adverse-effect level	the lowest level of exposure to a contaminant that causes a measured response that negatively affects an organism
macroinvertebrate	invertebrate large enough to be seen by the naked eye
macropthalmia	lamprey juvenile (life-stage following ammocoete)
measurement endpoint	the exposure and/or effect measure used to evaluate the assessment endpoint in an ecological risk assessment
meiofauna	very small benthic invertebrates that live among the sand grains below the sediment surface; typically too small to be seen by the naked eye
no-observed-adverse-effect level	the highest level of exposure to a contaminant that does not cause a measured negative response of an organism
organic carbon-normalized concentration	a chemical concentration in sediment adjusted for organic carbon content
oligochaete	a type of segmented worm that is widely distributed in both sediment and soil
omnivore	an organism that eats both animal and plant matter
pelagic	pertaining to, living in, or occurring in an open water body

Term	Definition
periphyton	algae, bacteria, microorganisms (along with organic material) attached to hard substrates (e.g., rock, roots, etc.) that occur in a water body
piscivore	an organism that eats primarily fish
population	a group of organisms belonging to the same species
porewater	water that fills the spaces between grains of sediment
predicted data	data not quantified in a laboratory but estimated using a model
reference threshold	a lower level response (survival or growth) in toxicity tests from a reference area representing the limit of the normal or expected responses in the absence of exposure to site-specific sediment contamination
regression	the statistical relationship between a random variable and one or more independent variables
remediation goal	contaminant-specific requirements that establish acceptable exposure levels for each exposure pathway; may be used as cleanup criteria in a remedial action
riparian	situated or living along the bank of a river or stream
risk	the chance that a specific ecological component experiences a particular adverse effect from exposure to contaminants from a hazardous waste site; the severity of risk increases if the severity of the adverse effect increases or if the chance of the adverse effect occurring increases
risk characterization	a part of the risk assessment process in which exposure and effects data are integrated in order to evaluate the likelihood of associated adverse effects
risk threshold	a level of contaminant exposure of a receptor above which a particular effect is expected to occur or below which no effect is expected to occur
screening level risk assessment	a part of the risk assessment in which contaminants of potential concern are identified by comparing maximum contaminant concentrations to screening level effect thresholds
sediment quality guideline	a published sediment concentration used to evaluate sediment quality based on effects to aquatic organisms

Term	Definition
site use factor	the fraction of time that a receptor spends foraging at the site relative to the entire home range and based on consideration of seasonal use
special status species	ecological organisms that are protected by federal and/or state regulations or otherwise deemed culturally significant
species	related individuals that share common characteristics and are capable of breeding among themselves and producing fertile offspring
species sensitivity distribution	a mathematical model that attempts to compile effect thresholds for a related set of species
Study Area	the portion of the Lower Willamette River that extends from River Mile 1.9 to River Mile 11.8
threshold sediment concentration	a sediment concentration above which a particular effect is expected to occur or below which no effect is expected to occur
threshold tissue concentration	a tissue concentration above which a particular effect is expected to occur or below which no effect is expected to occur
toxicity threshold	used to define the onset of specific level of adverse effect
trophic level	a feeding level within an ecosystem at which energy is transferred (e.g., herbivores, carnivores)
toxic equivalency factor	numerical values developed by the World Health Organization that quantify the toxicity of dioxin, furan, and dioxin-like PCB congeners relative to 2,3,7,8-tetrachlorodibenzodioxin
toxicity reference value	a toxicity threshold that has been used in a risk assessment
transition zone water	porewater associated with the upper layer of the sediment column; may contain both groundwater and surface water
upper confidence limit on the mean	a conservative high-end statistical measure of central tendency

EXECUTIVE SUMMARY

A draft final baseline ecological risk assessment (BERA) has been prepared following the ecological risk assessment (ERA) approach presented in the Portland Harbor Remedial Investigation/Feasibility Study (RI/FS) Programmatic Work Plan (Integral et al. 2004b) and direction in the US Environmental Protection Agency (EPA)-prepared February 2008 Problem Formulation document (EPA 2008j) (included in the BERA as Attachment 2). The approach was prepared by the Lower Willamette Group (LWG) based on the requirements of the scope of work and Administrative Order on Consent (EPA 2001) entered into with EPA for conducting the RI/FS. The approach is also consistent with EPA guidance for conducting ERAs (EPA 1997, 1998).

The overall purpose of the BERA is to determine if deleterious ecological effects from exposure to uncontrolled releases of hazardous substances to the Lower Willamette River (LWR) might be occurring in the Study Area under baseline conditions.[1] If so, then the BERA provides information to risk managers to support decisions about preliminary remediation goals (PRGs), areas of potential concern, and methods to analyze remedial action alternatives for the protection of ecological receptors in the FS.

Incorporating the results of the BERA and the baseline human health risk assessment, these PRGs will provide preliminary estimates of the long-term goals to be achieved by any cleanup actions in Portland Harbor. During the FS process, PRGs will be refined based on background sediment quality, technical feasibility, and other risk management decisions. EPA will identify the final sediment remediation goals for the site in the Record of Decision, following the completion of the FS. Given that the Portland Harbor Superfund Site is located in an urban and industrialized area, the regional land uses and physical and chemical baseline conditions will play a role in risk management decisions. For most ecological receptors, the draft final BERA assumed that the entire Study Area represents potential habitat; further evaluation of specific habitat areas should be another key component considered when making future risk management decisions.

The evaluation of potentially unacceptable risks to ecological receptors at the Portland Harbor Superfund Site has been an ongoing and iterative process involving both the LWG and EPA, with oversight and direction from EPA. This process has been documented by numerous reports and technical memoranda over the past several years. Data from the Study Area were collected by LWG during three sampling rounds (Rounds 1, 2, and 3) concurrent with the production of documents that refined the assessment and delineation of risks. Data from all LWG sampling rounds as well as other relevant and acceptable sources combined with a series of exposure assumptions and effects thresholds form the basis of the risk estimates in this draft final BERA. The risk estimates evaluate ecological receptors under worst-case exposure scenarios

[1] Baseline conditions are the conditions represented by the BERA dataset, which was collected between June 2002 and November 2007. The BERA dataset is found in Attachment 4.

(e.g., assuming that organisms get 100% of their food from the Study Area and using organism-level measurement endpoints).

Benthic invertebrates, fish, birds, mammals, amphibians/reptiles, and aquatic plants were identified as ecological receptors in the conceptual site model (CSM). These receptors were evaluated in the risk assessment using multiple lines of evidence (LOEs). The assessment endpoints for all receptors are based on the protection and maintenance of their populations and communities, except that organism health was designated by EPA as the assessment endpoint for juvenile Chinook salmon, Pacific lamprey ammocoetes, and bald eagle.

This document identifies contaminants whose measured or predicted concentration exceeds a defined adverse effects threshold, typically a toxicity reference value (TRV). These contaminants are termed contaminants of potential concern (COPCs). COPCs are drawn from a longer list of contaminants of interest (COIs). Risk estimates are stated as hazard quotients (HQs), which are calculated as the concentration at the point of exposure divided by the adverse effects threshold. Any COPC with an HQ $\geq$ 1 in the final step of the risk characterization (i.e., considering ecologically relevant diets and exposure area sizes) for at least one LOE, in any location in the Study Area is identified as a contaminant posing potentially unacceptable risk.[2] The results of the BERA will be used in the FS to identify contaminants of concern (COCs), areas of concern (AOCs), and receptors of concern. Section 12 presents the LWG's ecological risk management recommendations regarding COCs, AOCs and receptors of concern for developing and evaluating remedial alternatives that are protective of ecological resources.

Key findings of the BERA include the following:

- In total, 89 contaminants (as individual chemicals, sums, or totals) with HQ $\geq$ 1 pose potentially unacceptable risk.[3]
- The primary risk of ecologically significant adverse effects on ecological receptors in the Study Area is from four groups of chemical mixtures:polychlorinated biphenyls (PCBs), dioxins and furans, polycyclic aromatic hydrocarbons (PAHs), and total DDx (all isomers of dichlorodiphenyltrichloroethane [DDT] [2,4′-dichlorodiphenyldichloroethane (DDD), 4,4′-DDD, 2,4′-dichlorodiphenyldichloroethylene (DDE), 4,4′-DDE, 2,4′-DDT, and 4,4′-DDT]).
- The identification of the primary contributors to risk is consistent with EPA risk assessment guidance (EPA 1997) and is not intended to suggest that other

[2] This has the same meaning as the term "potentially posing unacceptable risk" in the draft final Portland Harbor RI/FS baseline human health risk assessment (Kennedy/Jenks 2011).

[3] The likelihood and ecological significance of the potentially unacceptable risk varies across COPCs and LOEs from very low to high. Therefore, the potentially unacceptable risks range from negligible to significant.

contaminants in those areas, and generally in the Study Area, do not also present potentially unacceptable risk.

- HQs ≥ 1 occur for PCBs throughout the Study Area for mink (HQs range from 19 to 33) and for river otter (HQs range from 21 to 31), indicating possible population-level effects expressed as reduced reproductive success.
- Reproductive success in spotted sandpiper, bald eagle, and osprey might also be reduced because of PCB exposure, as indicated by HQs ≥ 1 throughout the Study Area for spotted sandpiper (max HQ = 12) and bald eagle (max HQ = 3.9) and, over a smaller area, for osprey (max HQ = 4.4).
- Overall, a greater degree of uncertainty is associated with PCB risk estimates for birds than for mammals because of uncertainty about both exposure and the effects data. Uncertainty is higher for otter than for mink because otter-specific effects data are not available.
- The combined toxicity of dioxins/furans and dioxin-like PCBs, expressed as total toxic equivalent (TEQ), poses the potential risk of reduced reproductive success in mink (max HQ = 12), river otter (max HQ = 2.3), sandpiper (max HQ = 20), bald eagle (max HQ = 53), and osprey (max HQ = 3.8). The PCB TEQ fraction of total TEQ is responsible for the majority of total TEQ exposure, but the total dioxin/furan TEQ fraction also exceeds its TRV in some locations of the Study Area. As for total PCBs, a greater degree of uncertainty is associated with total TEQ risk estimates for birds and otter than for mink.
- The potential for adverse effects in fish due to exposure to total PCBs is low: HQs are < 1 for fish and benthic invertebrates via the surface water LOE, tissue-residue HQs ≥ 1 occur over only a moderate spatial extent (or, for large-home-range fish, in relatively few samples), and uncertainty in the tissue-residue TRV is more likely to over- than underpredict risk.
- DDx compounds pose low to negligible risk of reduced reproductive success to individual bald eagles and Pacific lamprey ammocoetes and only within limited portions of the Study Area.
- For certain LOEs, DDx HQs are ≥ 1 for sculpin and spotted sandpiper, although risk to their populations was assessed to be negligible. This conclusion is based on the weight of evidence (WOE). Surface water HQs are < 1 for fish and benthic invertebrates. Tissue-residue HQs ≥ 1 for sculpin occur at low magnitude and over a limited spatial area; HQs ≥ 1 for sculpin in transition zone water (TZW) (porewater associated with the upper layer of the sediment column) are also spatially limited and likely overestimate risk. For the spotted sandpiper, dietary HQs ≥ 1 occur over a limited spatial area, with effects uncertainties likely to result in overestimated risk.
- Contaminant concentrations in TZW were compared with surface water effects thresholds to predict risk to benthic invertebrates, fish, amphibians, and aquatic plants. TZW risks were evaluated in a focused study of only nine locations in the

Study Area with known or likely pathways for discharge of contaminated upland groundwater to the Study Area. Fifty-eight COPCs measured in TZW have baseline HQs ≥ 1 (14 metals, 16 PAHs, 3 semivolatile organic compounds [SVOCs], the pesticides 4,4′-DDT and total DDx, 16 volatile organic compounds [VOCs], 5 gasoline-range hydrocarbon fractions, and the conventionals cyanide and perchlorate).

- TZW exceedances (i.e., HQs ≥ 1) are localized, indicating that none of the TZW COPCs is likely to pose risk to Study Area benthic invertebrate communities or fish populations. Risks to amphibians and plants are even lower because the species in the Study Area are unlikely to use the habitats where contaminated groundwater discharges.
- Thirty-eight TZW COPCs,[4] including 6 metals, 16 PAHs, 2 SVOCs, 2 pesticides, 10 VOCs, cyanide, and perchlorate, pose potentially unacceptable risk to Pacific lamprey ammocoetes in localized areas. However, compared to other aquatic species, lamprey ammocoetes have average or lower sensitivity to chemicals that cause toxicity across several different modes of action (Andersen et al. 2010); the water TRVs are thus conservative for lamprey ammocoetes. Given their feeding habits and the low oxygen levels at the depths represented by the TZW samples, lamprey ammocoetes have relatively low exposure to TZW compared with surface water in the hyporheic zone; thus, the exposure estimates, too, are conservative.
- For many COIs, the available exposure and effects data preclude a quantitative risk evaluation. These contaminants have nonetheless been identified as posing potentially unacceptable risk. Risk to fish could not be evaluated for 17 tissue-residue COIs, 11 dietary COIs, 5 surface water COIs, and 9 TZW COIs. Risk to birds and mammals could not be evaluated for 19 dietary COIs. Risk to amphibians and aquatic plants could not be evaluated for 19 surface water COIs and 16 TZW COIs.
- COPCs occur at concentrations that are projected to pose unacceptable benthic risks for about 7% of the Study Area. Unlike other ecological receptors, for which risk was evaluated on a chemical-specific basis, risk to the benthic invertebrate community was evaluated in large part by considering exposure to the mixture of chemicals present in the Study Area sediments.
- The benthic community risk evaluation relied primarily on toxicity tests and multivariate predictive models based on the toxicity test and sediment chemistry results. Following a point-by-point assessment of potential effects on benthic organisms, based on measured or predicted toxicity and benthic tissue-residue data, multiple COPCs were identified: metals, tributyltin (TBT), PAHs, several

[4] Though not a CERCLA COPC, gasoline-range aliphatic hydrocarbons (C10-C12) were evaluated as an uncertainty, have HQ > 10, and also may pose risk to individual lamprey organisms.

SVOCs, two phenolic compounds, dibutyl phthalate, total PCBs, total DDx, and other pesticides.

- The COPCs in sediment that are spatially associated with locations of potentially unacceptable risk to the benthic community or populations are PAHs, PCBs, and DDx compounds.

These findings can be used in the FS as the framework for analyzing and comparing alternatives to remediate areas of elevated risk to ecological receptors.

By design, risk assessments are conservative in the face of uncertainty. In this context, conserverative means efforts made to miniminze the chances of underestimating exposure or risk. Consistent with the methods of EPA's Problem Formulation (EPA 2008j), receptor-contaminant pairs posing potentially unacceptable risk were identified using conservative methods and assumptions. Examples of conservatism include assumptions that contaminants are bioavailable and assumptions that yielded such low effect thresholds (TRVs) that, in the case of essential metals, they had to be adjusted upward because they were below nutritional requirements for some, but not all, fish species.

Not all uncertainties create a conservative bias. Some can lead to underestimation of risk, for example unavailability of exposure or effects data, thresholds that do not account for untested sensitive species, synergistic interactions among the multiple chemicals present at the site, and metabolic processes that increase the toxicity of accumulated chemicals.

The following sections outline the Problem Formulation that provides a framework for the draft final BERA, and highlight overall conclusions for risks to individual receptor groups (benthic invertebrates, fish, birds, mammals, amphibians/reptiles, and aquatic plants).

ES.1 PROBLEM FORMULATION

Elements of the Problem Formulation were provided as part of Appendix B of the Programmatic Work Plan (Integral et al. 2004b), in the draft *Portland Harbor RI/FS, Ecological Preliminary Risk Evaluation* (Windward 2005a), and in Appendix G of the *Portland Harbor RI/FS Comprehensive Round 2 Site Characterization Summary and Data Gaps Analysis Report* (Integral et al. 2007). EPA developed and directed the LWG to use a Problem Formulation document (EPA 2008j), which provides the methods for completing the BERA and accounts for data and information collected to date. The Problem Formulation document is included as Attachment 2 to the BERA.

ES.1.1 Identification of COPCs

The BERA follows the steps and procedures laid out in the Problem Formulation document for defining ecological COPCs. From chemical data for biological tissue, surface sediment, surface water, and TZW, over 100 contaminants, including metals and various organic compounds, were identified as COPCs:

- **Surface sediment –** Sixty-seven COPCs were identified. Surface sediment COPCs were evaluated as part of the benthic invertebrate risk assessment.
- **Tissue –** Seventeen COPCs were identified. Tissue COPCs were evaluated in the benthic invertebrate and fish risk assessments as part of the tissue-residue LOE.
- **Diet –** Eight dietary COPCs were identified for fish, and 24 dietary COPCs were identified for wildlife (birds and mammals). Dietary COPCs, which were identified from both tissue and sediment data, were evaluated as part of the fish and wildlife assessments**.**
- **Surface water –** Fourteen COPCs were identified. Surface water COPCs were evaluated as part of the benthic invertebrate, fish, amphibian, and aquatic plant risk assessments.
- **TZW –** Fifty-eight COPCs were identified. TZW COPCs were evaluated as part of the benthic invertebrate, fish, amphibian, and aquatic plant risk assessments.

ES.1.2 Refined Conceptual Site Model

The CSM describes relationships between contaminants and the resources potentially affected by their release. The following ecological receptors were selected for assessment:

- **Benthic invertebrate community**[5] – benthic macroinvertebrate community as a whole, bivalves (clams), and decapods (e.g., crayfish)
- **Omnivorous fish populations –** largescale sucker, carp, and pre-breeding white sturgeon
- **Invertivorous fish populations –** sculpin, peamouth, and juvenile Chinook salmon[6]
- **Piscivorous fish populations –** smallmouth bass and northern pikeminnow
- **Detritivorous fish individuals –** Pacific lamprey ammocoetes
- **Sediment-probing invertivorous bird populations –** spotted sandpiper
- **Omnivorous bird populations**[7] **–** hooded merganser
- **Piscivorous bird populations**[8] **–** osprey and bald eagle

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[5] Clams and crayfish are members of the benthic macroinvertebrate community, but were also evaluated separately to satisfy EPA's request for population-level assessments.

[6] Juvenile Chinook salmon were evaluated at the organism level; the other invertivorous fish receptor species were evaluated at the population level.

[7] Belted kingfisher were evaluated in the uncertainty assessment.

[8] Bald eagles were evaluated at the organism level.

- **Aquatic-dependent carnivorous mammal populations –** mink and river otter
- **Amphibian and reptile populations –** amphibians (e.g., frog and salamander species)
- **Aquatic plant community –** aquatic plant community (e.g., phytoplankton, periphyton, macrophyte species)

ES.1.3 Analysis Plan

The major components described in the draft final BERA analysis plan are an assessment of exposure, an assessment of effects, and a characterization of risk reflecting the integration of exposure and effects. An analysis of uncertainties is included in the third component.

Exposure Assessment

As stipulated in the Problem Formulation (EPA 2008j), all COPCs were first evaluated on a sample-by-sample basis. Because a sample-by-sample scale is not ecologically relevant for most of the receptors evaluated in the BERA, COPCs were next evaluated at an exposure scale that is ecologically relevant for each specific receptor. For dietary risks to fish and wildlife, exposure estimates were also determined for a diet consisting of multiple prey species, using prey portions reported in the literature. Exposure concentrations are based both on contaminant concentrations quantified in the analytical laboratory (empirical concentrations), and, for some LOEs, on predicted values (i.e.,for the tissue-residue LOE, the dietary LOE for shorebirds, and the bird egg LOE). Exposure of benthic invertebrates was assessed based on contaminant concentrations in individual samples of sediment, water, and TZW.

Effects Assessment

The effects assessment involves two general approaches. For most ecological receptors, the effects of COPCs were assessed by comparing contaminant concentrations in each environmental medium to chemical- and medium-specific TRVs or site-specific sediment quality values (SQVs). EPA (2008f) specified the TRVs that were used in the BERA. However, some TRVs selected by EPA are associated with significant uncertainty; these TRVs were further evaluated as part of the risk characterization process. Consistent with the Problem Formulation, for all receptors and receptor groups evaluated at the community or population level, the lowest-observed-adverse-effect level (LOAEL) was used. The no-observed-adverse-effect level (NOAEL) was used for receptors evaluated at the organism level (bald eagle and Pacific lamprey ammocoete).

The second effects assessment approach uses sediment toxicity bioassays as a direct measure of the effects of sediment contaminant mixtures on the survival and biomass of benthic invertebrates in the laboratory. As directed by EPA in its Problem Formulation (EPA 2008j), two models were evaluated for the development of site-specific SQVs; several published sets of sediment quality guidelines (SQGs) also were evaluated to predict unacceptable risks to the benthic community. All sets of SQVs and SQGs were tested to establish their reliability as predictors of benthic toxicity in the Study Area.

Risk Characterization and Uncertainty Analysis

Risk was characterized primarily as an HQ, calculated as the receptor- and medium-specific exposure concentration divided by the respective effect threshold. The risk characterization integrates information on contaminant exposure and effects to identify contaminants posing potentially unacceptable risk.

Potentially unacceptable risks were identified through an iterative process of analyzing exposure and effects data for the various contaminants and ecological receptors, with increasing realism at each step in the process. For most receptors, multiple LOEs were evaluated. For each LOE, risk characterization began with a screening-level ERA (see Section 3.1 and Attachment 5) and progressed iteratively through the final step in the risk characterization. Throughout the process receptor-contaminant pairs that showed the potential for adverse effects were further analyzed and those that did not were screened out. The final step in the process reflects the most realistic risk estimates. Each receptor-LOE-COPC combination with HQ ≥ 1 based on the final step in the risk characterization constitutes evidence of potentially unacceptable risk. In total, 89 ecological contaminants (as individual contaminants, sums, or totals) were identified in this BERA as posing potentially unacceptable risk.

Additional information was used to characterize the numerical risk estimates relative to assessment endpoints at the level of the population and community. Uncertainties in the exposure and effects data, spatial extent of HQs ≥1, magnitude of HQs, level of effect represented by the TRV, and WOE (e.g., consistency across multiple LOEs) were considered in formulating risk conclusions.

ES.2 BENTHIC INVERTEBRATE RISK ASSESSMENT

Fifty-five contaminants (individual chemicals, sums, or totals) were identified as posing potentially unacceptable risk to benthic invertebrates because they exceeded a site-specific SQV, tissue TRV, or surface water TRV. These COPCs include eight metals (arsenic, cadmium, chromium, copper, lead, mercury, silver, and zinc), TBT, monobutyltin, 20 individual PAHs and PAH sums, bis(2-ethylhexyl) phthalate (BEHP), dibutyl phthalate, phenol, 4-methylphenol, three SVOCs (dibenzofuran, benzyl alcohol, and carbazole), total PCBs, two VOCs (trichloroethene and ethylbenzene), and 15 pesticides (various DDx forms, dieldrin, endrin, endrin ketone, endosulfan, chlordane, and beta- and delta-hexachlorocyclohexane).

TZW COPCs with HQs ≥ 1 include 14 metals (barium, beryllium, cadmium, cobalt, copper, iron, lead, magnesium, manganese, nickel, potassium, sodium, vanadium, zinc), 16 individual PAHs (2-methylnaphthalene, acenaphthene, anthracene, benzo(a)anthracene, benzo(a)pyrene, benzo(b)fluoranthene, benzo(g,h,i)perylene, benzo(k)fluoranthene, chrysene, dibenzo(a,h)anthracene, fluoranthene, fluorene, indeno(1,2,3-cd)pyrene, naphthalene, phenanthrene, pyrene), three SVOCs (1,2-dichlorobenzene, 1,4-dichlorobenzene, dibenzofuran), pesticides (4,4′-DDT, total DDx), 16 VOCs (1,1-dichloroethene, 1,2,4-trimethylbenzene, 1,3,5-trimethylbenzene, benzene, carbon disulfide, chlorobenzene, chloroethane, chloroform, cis-1,2-dichloroethene, ethylbenzene, isopropylbenzene, m,p-xylene, o-xylene, toluene, total xylenes, trichloroethene), gasoline-range hydrocarbons, cyanide, and perchlorate.

Sediment profile imaging (SPI) data were examined to determine whether locations associated with sediment toxicity also tended to have a less mature benthic community structure than would be expected for the physical characteristics of the location. Once individual locations were reclassified, the evaluation focused on the locations where community response was worse than expected (potentially due to non-physical factors). Specifically, the evaluation focused on locations where immature (Stage 1) and transitional (Stage 2) communities were present in fine-grained depositional environments. An attempt was made to identify other factors influencing the community, including bottom slopes and sediment chemistry. The SPI data analysis was used as corroborative evidence in the BERA. The analysis suggested that the physical environment in the Study Area can explain the condition of the benthic community throughout this area of the river. In over 90% of the images evaluated, the successional stage matched the expected community based on the physical regime, when slope was included as a habitat characteristic. Of the 31 cases where the community stage was not as might be predicted by the physical environment, 19 appear to be related to sediment toxicity. These qualitative results suggest that the benthic community is typical of a large river system that is predominantly influenced by physical processes. Impacts from sediment contamination appear to be primarily associated with depositional areas that have received historical releases of contamination.

The potential for benthic risk was determined as follows. First, bioassay results were mapped for four test endpoints (*Chironomus dilutus* survival, *Chironomus dilutus* biomass, and the same for *Hyalella azteca*). Next, sediment COPC concentrations were compared with SQVs based on both the floating percentile model and the logistic regression model to predict several different levels of benthic toxicity in locations without bioassay data. Next, these sediment chemistry data were used (with bioaccumulation models[9]) to predict locations where organisms might accumulate COPCs to concentrations above tissue TRVs. Locations where estimated sediment concentrations exceed SQVs or where empirical or predicted tissue concentrations exceed TRVs were identified as potential benthic risk areas.[10] Empirical tissue-residue data were mapped, allowing a visual assessment of concordance across LOEs. Water (both surface water and TZW) TRV exceedances were considered along with sediment SQV and tissue TRV exceedances. Water TRV exceedances were found to co-occur with areas of SQV exceedances. Based on the spatial analysis, PAHs, PCBs, and DDx were found to pose potentially unacceptable risks to the benthic invertebrate community. The spatial

[9] The analysis used the same bioaccumulation models as those created for developing the PRGs to be used in the FS.

[10] The qualifier “potential” is used because because EPA has not yet selected benthic areas of concern, and the measurement endpoints used to delineate the risk are organism-level endpoints but the assessment endpoints are the benthic community or populations of benthic species.

evaluation indicates that approximately 7% of the Study Area poses potentially unacceptable risk to the benthic community.

ES.3 FISH RISK ASSESSMENT

Four primary quantitative LOEs were used to characterize risks to fish: the tissue-residue LOE, the dietary-dose LOE, the surface water LOE, and the TZW LOE. Benthic fish exposure to PAHs in sediment was also evaluated as a qualitative LOE per EPA's Problem Formulation (EPA 2008j) and included an assessment of the apparent health of pre-breeding sturgeon; this LOE was inconclusive.

Fifty-nine contaminants were identified as posing potentially unacceptable risk to at least one fish receptor based on the tissue, dietary, surface water, and TZW LOEs. Of these, 44 COPCs had HQs ≥ 1 only for the TZW LOE. The exposure and effects data are insufficient to evaluate risk to fish from 17 tissue-residue COIs, 11 dietary COIs, 5 surface water COIs, and 9 TZW COIs.

Risk conclusions for each fish receptor were reached by evaluating the risk estimates and the reliability of each LOE. Total PCBs were found to pose low risk to populations of piscivorous fish (smallmouth bass and northern pikeminnow) and the small-home-range invertivorous fish sculpin. The potential for adverse effects on fish assessment endpoints from total PCBs as established by the tissue-residue LOE was assessed to be low: the other LOE used to evaluate PCB risks—surface water—results in HQs < 1, tissue-residue HQs ≥ 1 occur over only a moderate spatial extent (or in relatively few samples for large-home-range fish), and uncertainty in the tissue-residue TRV is more likely to over- than underpredict risk.

Thirty eight[11] TZW COPCs, including six metals (barium, iron, manganese, sodium, vanadium, and zinc),[12] 16 PAHs (2-methylnaphthalene, acenaphthene, anthracene, benzo(a)anthracene, benzo(a)pyrene, benzo(b)fluoranthene, benzo(g,h,i)perylene, benzo(k)fluoranthtene, chrysene, dibenzo(a,h)anthracene, fluoranthene, fluorene, indeno(1,2,3-cd)pyrene, naphthalene, phenanthrene, pyrene), two SVOCs (1,2-dichlorobenzene, 1,4-dichlorobenzene), the pesticides 4,4′-DDT and total DDx, 10 VOCs (benzene, carbon disulfide, chlorobenzene, chloroform, cis-1,2dichloroethene, ethylbenzene, o-xylene, toluene, total xylenes, and trichloroethene), cyanide, and perchlorate, pose potentially unacceptable risk to Pacific lamprey ammocoetes in localized areas associated with contaminated groundwater discharges to the river. However, the water TRVs were derived to be protective of highly sensitive species and

[11] Because petroleum compounds are not CERCLA contaminants, gasoline-range aliphatic hydrocarbons (C10 - C12) have been excluded from the count even though they may be contributing to potentially unacceptable risk. These contaminants were evaluated as an uncertainty, have HQ > 10, and may pose risk to individual lamprey.

[12] There is substantial uncertainty as to whether the source of barium, iron, and manganese in TZW is anthropogenic.

probably overpredict potentially unacceptable risk to lamprey ammocoetes, whose sensitivity to chemical toxicity across several modes of action was average or lower lower than that of most aquatic species (Andersen et al. 2010) Also, given their feeding habits and the low oxygen levels at the depths represented by the TZW samples, lamprey ammocoetes have relatively lower exposure to TZW than to surface water. The exposure assessment conservatively assumed that lamprey ammocoetes are exposed to undiluted TZW.

Risks to fish from other COPCs with HQs ≥ 1 in the final step of the risk characterization were found not likely to result in ecologically significant adverse effects at the population level. In some cases, the selected TRV probably underestimates the threshold for ecologically significant adverse effects, and in other cases the great majority of samples used to estimate exposure results in HQs < 1. Furthermore, TZW exposure assumptions probably overestimate risk.

ES.4 WILDLIFE RISK ASSESSMENT

Risks to wildlife receptors were evaluated using two LOEs. Dietary dose was used for all six wildlife receptors. Ingestion of prey tissue and incidental ingestion of sediment are reflected in dietary-dose estimates. Tissue residues in bird eggs was used as a second LOE for bald eagle and osprey.

Twelve contaminants (copper, lead, mercury, benzo[a]pyrene, dibutyl phthalate, total PCBs, PCB TEQ, total dioxin/furan TEQ, total TEQ, aldrin, 4,4′-DDE, and total DDx) were identified as posing potentially unacceptable risk for at least one bird receptor. Six contaminants (aluminum, lead, total PCBs, PCB TEQ, total dioxin/furan TEQ, and total TEQ) were identified as posing potentially unacceptable risk to mink or river otter.

Of these COPCs, total PCBs pose the primary risk.Calculated risk estimates indicate that populations of both mink and river otter in the Study Area might be experiencing reduced reproductive success because of exposure to PCBs. Reproductive success in spotted sandpiper, bald eagle, and osprey might also be reduced because of PCB exposure. A greater degree of uncertainty is associated with PCB risk estimates for birds than for mammals because of greater uncertainties about exposure and the effects data. Uncertainty is higher for otter than for mink because otter-specific effects data are lacking.

Total TEQ exposure also poses the potential for reduced reproductive success in mink, river otter, spotted sandpiper, bald eagle, and osprey. The PCB TEQ is responsible for the majority of total TEQ exposure, but the total dioxin/furan TEQ also exceeds itsTRVs in some locations of the Study Area. As is the case for total PCBs and for the same reasons, a greater degree of uncertainty is associated with risk estimates for birds and otter than for mink.

Osprey egg data (used as a surrogate for bald eagle) suggest that DDx compounds could pose low risk of reduced reproductive success to individual bald eagle organisms within limited portions of the Study Area. Concentrations of total DDx in egg tissue from two of five exposure areas exceeded the NOAEL for bald eagle, but not where the highest sediment DDx concentrations occurred. In all five exposure areas, egg concentrations were below both the bald eagle-specific LOAEL TRV and the dietary effect threshold. There is significant uncertainty about the source of DDx in the osprey eggs collected from the Study Area, given that adults nest and lay eggs shortly after returning to the LWR from their overwintering grounds in Mexico and Central America.

Risk to wildlife from other COPCs with HQs ≥ 1 in the final step of the risk characterization were found unlikely to result in ecologically significant adverse effects in the receptor populations: the HQs are of low magnitude and over a limited spatial extent, with uncertainties in exposure and effects likely to result in overestimated risk.

ES.5 AMPHIBIAN/REPTILE AND AQUATIC PLANT RISK ASSESSMENT

Two LOEs were used to characterize risks to amphibians and aquatic plants: surface water and TZW. The same exposure and effects data were used to assess risk to amphibians and aquatic plants, although several uncertainties apply to only one of the receptor groups. Thirty-three contaminants were identified as posing potentially unacceptable risk to amphibians and aquatic plants. Five of these COPCs have HQs ≥1 in both surface water and TZW (zinc, benzo(a)anthracene, benzo(a)pyrene, naphthalene, and total DDx), one is a COPC based on surface water only (BEHP), and 27 are COPCs based on TZW only. None of these COPCs is expected to have ecologically significant adverse effects on amphibian or aquatic plant populations in the Study Area. In general, surface water concentrations of COPCs are below algae- or amphibian-specific thresholds, exceed surface water thresholds by a low magnitude and at low frequency, or both. These conditions indicate low to negligible risk. Furthermore, surface water COPCs with HQs ≥ 1 occur in samples collected during the less sensitive non-reproductive periods (when amphibians may not be present in the Study Area), again indicating negligible risk to amphibian populations. For the TZW LOE, the great majority of samples result in HQs < 1, indicating a limited spatial extent of exceedance. Although highly uncertain, risk to amphibians and aquatic plants from TZW is likely negligible because of a low level of exposure to TZW. There is some uncertainty concerning the relevance of selected TRVs: aquatic plant-, algae-, or amphibian-specific thresholds for several COPCs are either limited or not available. Because amphibians had been selected as the surrogate for reptiles, risk conclusions for amphibians also apply to reptiles.

ES.6 RISK MANAGEMENT RECOMMENDATIONS

Section 12 of the draft final BERA identifies the COCs, receptors, and AOCs that the LWG considers necessary and sufficient to develop and evaluate remedial alternatives that are protective of ecological resources. The FS will also evaluate whether remedial alternatives for these COCs, receptors, and AOCs address the full list of contaminants posing potentially unacceptable risk. In summary, the risk management recommendations are as follows:

- For non-benthic receptors, total PCBs and total TEQ are the recommended COCs. Mink is the recommended receptor of concern. Most of the contaminants posing potentially unacceptable risk were not recommended as COCs for non-benthic receptors based on risk characterization considerations (magnitude, spatial extent, and ecological significance of HQs $\geq$ 1). This list includes all the metals, butyltin, phthalate, pesticide, and VOC COPCs.
- For aquatic receptors exposed via TZW, 4,4′-DDT, total DDx,[13] chlorobenzene, benzo(a)anthracene, benzo(a)pyrene, naphthalene, carbon disulfide, cyanide, cis-1,2-dichloroethene, and trichloroethene are the recommended COCs. These recommendations presume that contaminated groundwater source control measures will be implemented prior to sediment remedies. The Oregon Department of Environmental Quality is working with upland property owners to implement contaminated groundwater source control measures prior to sediment remedies.
- The benthic risk assessment methodologies are designed to address chemical mixtures and do not conclusively identify contaminants causing toxicity. For benthic organisms, methodologies for delineating benthic AOCs (rather than identifying COCs) and for evaluating remedial action alternatives are recommended. Recommended benthic AOCs were mapped by applying the comprehensive benthic approach based on EPA’s April 21, 2010, guidelines for assessing benthic risk in the FS (EPA 2010a). Those maps are preseented in Section 12 of the BERA.

[13] There is uncertainty associated with 4,4′-DDT and total DDx as COCs because HQs based on filtered samples are less than 100. This suggests that the risk from DDx compounds in TZW may be lower than indicated by the maximum concentrations in unfiltered samples because of the lower bioavailability of the particulate-bound fraction of the contaminant.

1.0 INTRODUCTION

This document presents the baseline ecological risk assessment (BERA) component of the Portland Harbor Superfund Site Remedial Investigation/Feasibility Study (RI/FS). The overall purpose of the baseline risk assessment is to determine if deleterious ecological effects may be occurring at the Study Area under current conditions and in the absence of any remedial actions. In the event that such unacceptable risks are predicted, the BERA provides information to risk managers on future approaches for protecting ecological receptors.

The BERA follows the ecological risk assessment (ERA) approach presented in the *Portland Harbor RI/FS Programmatic Work Plan, Appendix B: Ecological Risk Assessment* (Integral et al. 2004a). The approach was prepared by the Lower Willamette Group (LWG) based on the requirements of the scope of work (SOW) and Administrative Order on Consent (EPA 2001) entered into with the US Environmental Protection Agency (EPA) for conducting the RI/FS and subsequent direction by EPA. The approach is also consistent with EPA guidance for conducting ERAs (EPA 1997, 1998) and EPA's *Problem Formulation for the Baseline Ecological Risk Assessment at the Portland Harbor Site* (hereafter referred to as EPA's Problem Formulation), dated February 15, 2008, as subsequently amended and modified to include toxicity reference values (TRVs) for tissues and the reference envelope approach for evaluating benthic toxicity tests (EPA 2008j).

The evaluation of potentially unacceptable risks to ecological receptors at the Portland Harbor Superfund Site has been an ongoing and iterative process involving both the LWG and EPA, with oversight and direction from EPA. This process has been documented through numerous reports and technical memoranda over the last several years. Key documents include those listed above and the following:

- *Portland Harbor RI/FS Programmatic Work Plan* (Integral et al. 2004b), hereafter referred to as the Programmatic Work Plan
- *Portland Harbor Superfund Site Ecological Risk Assessment: Comprehensive Synopsis of Approaches and Methods* (Draft) (Windward 2004)
- *Portland Harbor Superfund Site Ecological Risk Assessment Interpretive Report: Estimating Risks to Benthic Organisms Using Predictive Models Based on Sediment Toxicity Tests* (Draft) (Windward et al. 2006)
- *Portland Harbor Superfund Site Proposed Ecological Risk Assessment Decision Framework* (Draft) (Windward 2006b)

Estimates of risk were made on the basis of preliminary datasets in two documents:

- *Portland Harbor RI/FS, Ecological Preliminary Risk Evaluation* (Windward 2005a), hereafter referred to as the Ecological Preliminary Risk Evaluation (PRE)

- *Portland Harbor RI/FS Comprehensive Round 2 Site Characterization Summary and Data Gaps Analysis Report* (Integral et al. 2007), hereafter referred to as the Comprehensive Round 2 Report

Data from the area of study were collected in three sampling rounds (Rounds 1, 2, and 3) concurrent with the production of documents describing the ERA process and evaluating ecological receptors. The initial sampling of the Study Area (Round 1 sampling) was conducted concurrent with the preparation of the Programmatic Work Plan from summer 2002 until spring 2004. The Ecological PRE evaluated preliminary risks to ecological receptors based on Round 1 data. Round 2 data were collected from summer 2004 until December 2005 to support the Comprehensive Round 2 Report and fill data gaps from Round 1 sampling. During the preparation and following the submittal of the Comprehensive Round 2 Report, EPA and LWG identified additional data gaps that were filled through a third round of sampling. Round 3 sampling was conducted from January 2006 until February 2008. Data from all LWG sampling rounds as well as relevant and acceptable non-LWG-collected data are evaluated in this BERA. The approach applied in this BERA and the risk results and conclusions supersede prior approaches and estimates of risk.

The relationship of the BERA to the overall RI/FS process for the Portland Harbor Superfund site is depicted in Figure 1-1.

The RI initially focused on the stretch of the Lower Willamette River (LWR) from River Mile (RM) 3.5 to RM 9.2 and adjacent areas associated with the in-water portion of this stretch of the river. The *Portland Harbor RI/FS Programmatic Work Plan* (Integral et al. 2004b) refers to that initial Study Area as the “ISA.” In the Programmatic Work Plan (Integral et al. 2007), the area of investigation was broadened to include areas of the river extending from approximately RM 1.9 to RM 11; this expanded area was termed the “Study Area.” For the BERA, the area of investigation was extended to RM 11.8. The term “Study Area” was retained for the BERA and includes the 10-mile stretch of the river between approximately RM 1.9 and RM 11.8.

The BERA has two broad objectives:

- Identify unacceptable risks posed by contaminants to aquatic and aquatic-dependent ecological receptors in the Study Area.
- In the event that unacceptable ecological risks are found and require remedial actions, provide information that risk managers can use to set cleanup levels protective of ecological receptors.

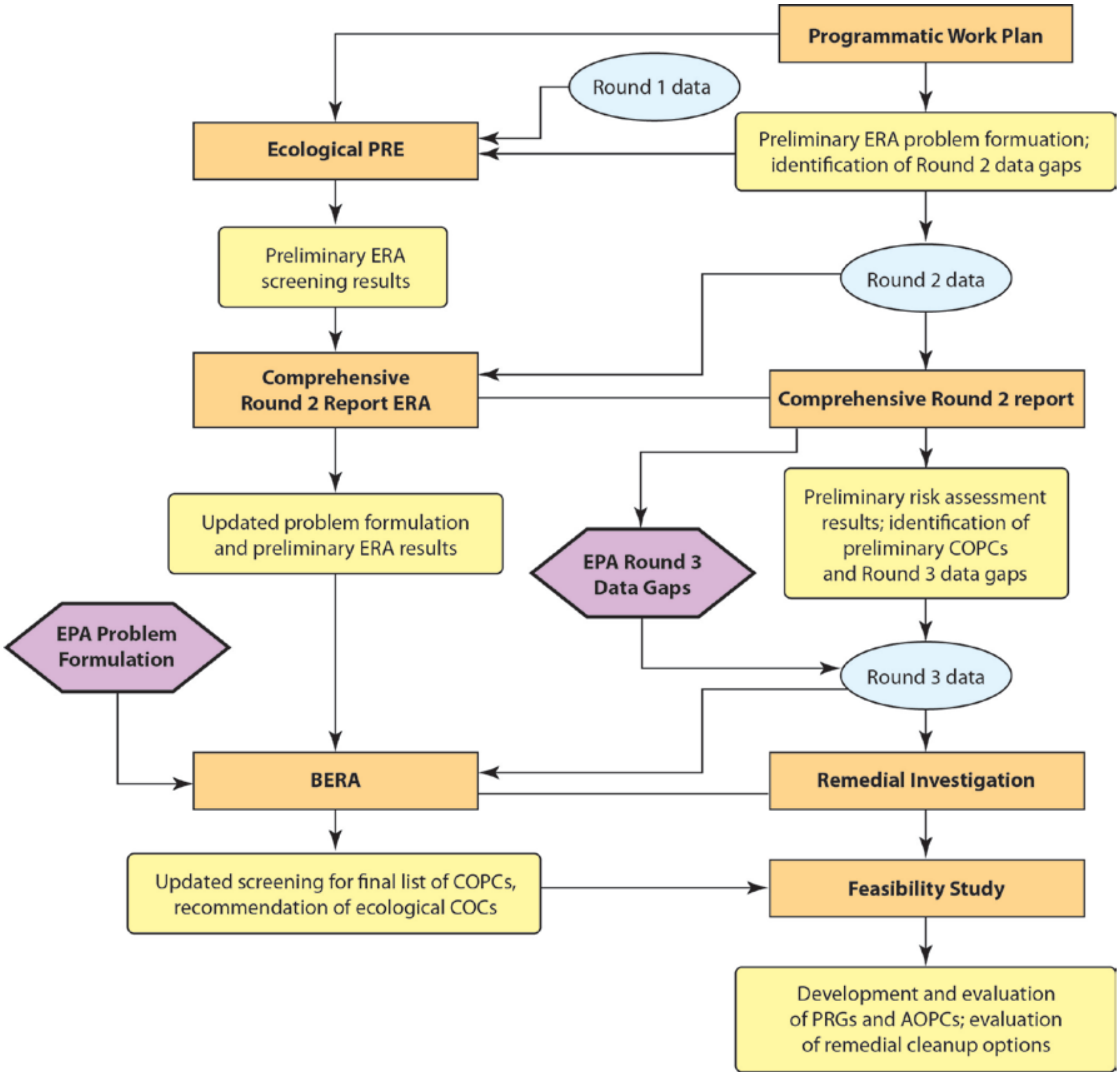


Figure 1-1. ERA Process as Part of the Portland Harbor RI/FS

This document identifies contaminants whose measured or predicted concentration exceeds a defined adverse effects threshold, typically a toxicity reference value (TRV). These contaminants are termed contaminants of potential concern (COPCs). COPCs are drawn from a longer list of contaminants of interest (COIs). Risk estimates are stated as hazard quotients (HQs), which are calculated as the concentration at the point of exposure divided by the adverse effects threshold. Any COPC with an HQ ≥ 1 in the final step of the risk characterization (i.e., considering ecologically relevant diets and exposure area sizes) for at least one LOE, in any location in the Study Area is identified as a contaminant posing potentially unacceptable risk.[14] The results of the BERA will be used

[14] This has the same meaning as the term "potentially posing unacceptable risk" in the draft final Portland Harbor RI/FS baseline human health risk assessment (Kennedy/Jenks 2011).

in the FS to identify contaminants of concern (COCs), areas of concern (AOCs), and receptors of concern. Section 12 presents the LWG's ecological risk management recommendations regarding COCs, AOCs and receptors of concern for developing and evaluating remedial alternatives that are protective of ecological resources.

The BERA will be used to support the development, in the FS, of contaminant thresholds defining preliminary remediation goals (PRGs) for sediment. The PRGs will provide preliminary estimates of the long-term goals to be achieved by sediment cleanup actions in Portland Harbor. During the FS process, the PRGs will be refined on the basis of background sediment quality, technical feasibility, and other risk management considerations. EPA will identify the final remediation goals for the site in the Record of Decision, following completion of the FS.

Each subsequent section of this document includes an introductory summary of its contents and organization. Text boxes highlight key elements of the overall ecological picture of the Study Area and key uncertainties in the analysis.

The remainder of this document is organized as follows:

- Section 2.0 – Site Description – This section presents general information about the ecological habitats and species present in the LWR.
- Section 3.0 – BERA Problem Formulation – This section summarizes the EPA's Problem Formulation (Attachment 2) that was used to conduct the BERA, including the selected assessment and measurement endpoints.
- Section 4.0 – BERA Data – This section presents a summary of the Study Area data used in the BERA.
- Section 5.0 – Identification of COPCs – This section summarizes the process used to identify ecological COPCs and presents the COPCs for each ecological receptor group.
- Section 6.0 – Benthic Invertebrate Risk Assessment – This section presents the exposure, effects, and risk characterization evaluation for the benthic invertebrate community.
- Section 7.0 – Fish Risk Assessment – This section presents the exposure, effects, and risk characterization evaluation for selected fish receptors.
- Section 8.0 – Wildlife Risk Assessment – This section presents the exposure, effects, and risk characterization evaluation for selected bird and mammal receptors.
- Section 9.0 – Amphibian Risk Assessment – This section presents the exposure, effects, and risk characterization evaluation for amphibians.
- Section 10.0 – Aquatic Plant Risk Assessment – This section presents the exposure, effects, and risk characterization evaluation for aquatic plants.

- Section 11.0 – Ecological Risk Conclusions – This section presents the overall risk conclusions of the BERA.
- Section 12.0 – Ecological Risk Management Recommendations – This section presents the LWG's ecological risk management recommendations for the FS.
- Section 13.0 – References – This section lists the references cited throughout the text.

The following attachments are also included as part of this BERA:

- Attachment 1 provides documentation of BERA-related EPA communication and decisions.
- Attachment 2 presents the BERA problem formulation per EPA (2008j).[15]
- Attachment 3 presents the data management and data calculation rules.
- Attachment 4 is an electronic attachment that presents summary statistics characterizing data by medium, raw data tables of all data used for the BERA, and results of the calculations of 95th upper confidence limit on the mean (UCL) concentration to represent exposure point concentrations (EPCs).
- Attachment 5 presents the screening-level ERA (SLERA) and refined screening process used to identify ecological COPCs.
- Attachment 6 presents the benthic modeling calculations and results.
- Attachment 7 presents the reliability of the generic sediment quality guidelines (SQGs) in predicting benthic toxicity.
- Attachment 8 describes the methods for and results of the biota-sediment accumulation regression (BSAR) analyses.
- Attachment 9 presents the revised process by which TRVs were derived for aquatic biological tissue (2008f) as well as the tissue TRVs selected.
- Attachment 10 summarizes the derivation of ecological thresholds for COPCs in surface water and transition zone water (TZW).
- Attachment 11 compares contaminant concentrations in sediment and surface water from background locations to those of the Study Area.
- Attachment 12 presents the sample-by-sample and individual prey component risk evaluation for fish.
- Attachment 13 details the exposure and effects assumptions used in the fish dietary line of evidence (LOE).

[15] Footnotes were added by LWG to the Problem Formulation to indicate actual implementation and/or any changes related to later agreements between EPA and LWG regarding the Problem Formulation.

- Attachment 14 summarizes the literature-derived fish dietary and wildlife dietary and tissue-residue (egg) TRVs.
- Attachment 15 provides the results of the lamprey toxicity tests.
- Attachment 16 details the exposure and effects assumptions used in the wildlife dietary LOE.
- Attachment 17 presents the sample-by-sample and individual prey component risk evaluation for wildlife.
- Attachment 18 presents the evaluation of future risks to benthic invertebrates from sediment.
- Attachment 19 presents maximum hazard quotients (HQs) and the number of individual samples resulting in HQ ≥ 1 for each receptor-LOE-contaminant combination posing potentially unacceptable risk.

2.0 SITE DESCRIPTION

Although the BERA focuses specifically on contaminant stressors (because it is part of the RI/FS), a physical description of the Study Area is pertinent to understanding the site's current ecological condition, and to developing a conceptual site model (CSM) on which the BERA analysis is based. This section presents a description of what is known about the physical conditions, aquatic and riparian habitats, and aquatic-dependent species that occur in the LWR.The Study Area is defined in the RI as the in-water portion below or equal to + 13 ft North American Vertical Datum (NAVD) from RM 1.9 to RM 11.8. Accordingly, the BERA is limited to in-water and riparian zone risks, and not upland ecological risks.

As discussed in the RI, Portland Harbor is a heavily industrialized reach of the LWR located immediately downstream of downtown Portland, Oregon and extending almost to the confluence with the Columbia River. Water levels on the Willamette River are cyclical and vary by season. Annual low water levels occur during the regional dry season from August to November. Winter (November to March) river stage is relatively high but variable because of short-term changes in precipitation in the Willamette Basin. Finally, a distinct and persistent period of relatively high water levels occurs from late May through June, when Willamette River flow into the Columbia is slowed by high-water stage flow in the Columbia River during the spring freshet in the Willamette River Basin and the much larger Columbia River Basin.

Tidal action also compounds the hydrology and interplay of the two rivers, affecting the Willamette River upstream as far as Portland Harbor and beyond. Tides along the North American West Coast are mixed semidiurnal (two unequal high tides and two unequal low tides daily), with spring and neap tides occurring every two weeks. Tides along the Washington and Oregon coast have an average tidal range of approximately 8 ft (but vary from about 5 to 12 ft). A high (i.e., flood) spring tide can influence Willamette River levels by up to 3 ft in Portland Harbor when the river is at a low stage. These tidal fluctuations can result in short-term flow reversal (i.e., upstream flow) in Portland Harbor during times of low river stage combined with large flood tides. As river stage rises, the tidal effect is gradually dampened and disappears at river levels around 10 ft Columbia River Datum (CRD).

The majority of the Study Area is industrialized, with modified shoreline and nearshore areas. Wharves and piers extend out toward the channel, and bulkheads and riprap revetments armor the riverbank. Active dredging has produced a uniform navigation channel with little habitat diversity. However, some segments of the Study Area are more complex, with small embayments, shallow water areas, gently sloped beaches, localized small wood accumulations, and less shoreline development, all of which provide habitat for a suite of local fauna. Because of the size of the river, the majority of the habitats are associated with the river bottom or water column (riparian and marsh habitats are mostly limited to relatively narrow strips along the shoreline, and constitute a much smaller area than the river itself). The benthic habitat characteristics generally reflect the energy regime of the riverbed at a given location, except where anthropogenic features and activities (e.g., prop wash, dredging) modify the sediment texture. The energy regime is primarily a function of river width and depth, although shoreline or channel alterations can modify water flow and sediment transport dynamics. In general, faster currents occur in the deeper portions of the river channel, and slower currents and eddies occur in the shallow nearshore areas. Typically, fine-grained sediments (i.e., silt, clay) dominate in relatively low-energy environments and coarse sediments (i.e., sand, gravel) indicate higher-energy environments.

The numerous organisms that use the LWR can be divided into the following general groups: invertebrates, fishes, birds, mammals, amphibians, reptiles, and aquatic plants. Each group makes an important contribution to the ecological function of the river based on its trophic level; abundance; and interaction with the physical, chemical, and biological environment. Riverine invertebrates are predominantly benthic, utilizing substrates such as fine-grained sediments, gravel and cobble, plant roots, and large woody debris. The benthic invertebrate community within the LWR is dominated by small organisms that live on or in the sediment, many of which are feeding on and processing organic material imported from upstream areas.

The LWR is an important migration corridor for anadromous fishes, such as salmon and lamprey, and provides habitat for numerous resident fish species (more than 40 species have been collected) that represent four major feeding guilds: omnivores/herbivores, benthopelagic/benthic invertivores, piscivores, and detritivores. Numerous aquatic-dependent bird species (more than 20 species commonly occur based on available information) use habitats within the LWR. The trophic representation of these birds is broad and includes herbivores, carnivores and omnivores, sediment-probing invertivores and omnivores, and piscivores. Six aquatic or semi-aquatic mammals use or may use the LWR, including opportunistic piscivores.

The LWR provides limited habitat for amphibians and reptiles. Amphibians prefer undisturbed areas that offer ephemeral wetlands with emergent vegetation and shallow waters (Sparling et al. 2000). Reptiles prefer shallow, quiescent aquatic areas and wet vegetated terrestrial habitats. High turbidity, riprap, and other bank modifications prevent the widespread development of dense submerged and emergent plant communities along the riverbanks. More detailed information on habitats and organisms using the LWR is presented in Section 2.1 and Section 2.2.

2.1 HABITAT TYPES IN THE LOWER WILLAMETTE RIVER

This section discusses the general types and quality of aquatic habitat available to species in the LWR.

2.1.1 Open-Water Habitat

The LWR is characterized by a navigation channel and an extensively developed shoreline. Most open-water habitat in the Study Area is in the main river channel, but there are also several shallower backwater sites (e.g., Willamette Cove, Swan Island Lagoon, Balch Creek Cove, Cottonwood Cove, individual slips). The deep open water provides foraging habitat for fish and wildlife that feed mainly in the water column. "Deep" in this context means a depth greater than 20 ft (6 m), with an average depth of 39 ft (12 m) ± 10 ft (3 m) and a maximum depth of 78 ft (24 m). Shallow-water habitats provide refuge for juvenile salmonids and other fishes, as well as foraging opportunities for birds and mammals. Aside from Willamette Cove and Swan Island Lagoon, shallow-water habitats are largely limited to the narrow strip between the shoreline and the navigation channel.

Three types of benthic habitats occur in the open-water areas of the LWR:

- Unconsolidated sediments (sands and silts) in the deeper water (greater than approximately 20 ft [6 m]) CRD) of the navigation channel and lower channel slopes
- Unconsolidated sediments (sands and silts) in shallow water in gently sloping nearshore areas (e.g., beaches and benches) and on the upper channel slopes (Figure 2-1)
- Developed shoreline areas (e.g., rock riprap, sheet pile, bulkheads) (Figure 2-2).

Figure 2-1. Nearshore In-Water Habitat

Figure 2-2. In-Water Bulkhead Structure

Benthic habitats are typically unvegetated, although benthic diatoms and periphyton are present on more stable surfaces. The navigation channel habitat is subject to variable (seasonal and annual) hydrodynamic forces, the effects of navigation, natural sediment deposition, bed load transport and erosion, and periodic navigational dredging. These forces vary spatially throughout the system, largely as a function of the channel cross-sectional area, resulting in both relatively stable and unstable sedimentary environments and patchy infaunal and epibenthic communities characteristic of the local physical regime. The physical sedimentary regimes are a function of hydrodynamic conditions caused by the local riverbank morphologies in nearshore areas, and overall channel characteristics in more open-water habitats. Areas away from frequent anthropogenic disturbance support infaunal invertebrate communities that are characteristic of large river systems. Conversely, exposed nearshore areas, particularly around active berths, docks, and boat ramps, tend to have more limited benthic communities because of their greater physical disturbance. The hard surfaces of the developed shoreline provide habitat for an epibenthic community.

2.1.2 Bank and Riparian Habitat

In 2007-2008, the City of Portland updated its natural resource inventory of the 12-mile reach of the Willamette River extending from the Broadway Bridge (RM 11.6) to the Columbia River (City of Portland 2008). The inventory qualitatively ranked riparian corridors and wildlife habitat areas based on connectivity to patches, connectivity to water, interior area, and patch size. Riparian corridor function was ranked according to six classes of attribute defined by the City of Portland: wildlife movement corridor, large wood/channel dynamics, organic inputs, nutrient cycling and food web,[16] stream flow

[16] Provides food for aquatic and terrestrial species and contributes to the ongoing physical and biological nutrient cycling system.

moderation/flood storage, microclimate/shade, and bank function/control of sediment nutrients and pollutants (City of Portland 2008).

The most common bank types occurring in the Study Area are riprap (Figure 2-3), sandy and rocky beach (Figures 2-4 and 2-5), unclassified fill, and seawall. In 2008, the City of Portland (2008) reported that the dominant bank types in the North Reach of the Willamette River (Broadway Bridge to the Columbia River) were vegetated[17] riprap (25%), unclassified fill (21%), and beach (23%) (Map 2-1). The classification was based on physical characteristics and not any specific elevation.

Figure 2-3. Riprap Bank

[17] Vegetation on riprap typically consists of Himalayan blackberry and other invasive species.

Figure 2-4. Intertidal Beach

Figure 2-5. Vegetated Bank

The riprap or rocky bank type is usually fairly steep with no or very little adjacent shallow-water habitat. These areas are usually exposed to heavy wave action and strong currents. The sandy bank type with no emergent vegetation is characterized by gently to steeply sloped beaches. This bank type is often adjacent to steep riprapped shorelines or developed uplands that are frequently exposed to heavy wave action and faster moving water. The rocky or vegetated sandy bank types are located in more protected areas in the Study Area, such as at the end of slips or in Swan Island Lagoon.

The type of riverbank present at a given location is expected to influence fish use at that location. Riverbanks with large woody debris that provide cover and create small shallow pools are more likely to be used by juvenile salmonids and other small fish species (Bjornn and Reiser 1991; Sedell and Froggatt 1984). Riprap and rocky substrate are the preferred habitats of sculpin and smallmouth bass (Farr and Ward 1992; SEA et al. 2003; Wydoski and Whitney 2003). Sculpin are found predominantly in the shallow-water habitats and smallmouth bass in areas with moderate current. The shallow backwater pools and slow-moving areas of the river provide habitat for juvenile largescale suckers (yearling and sub-yearling) and peamouth (Wydoski and Whitney 2003). The peamouth remain nearshore during winter months, moving to deeper waters in the summer months. The shallow waters with abundant plants and woody debris for cover are the preferred habitat of largemouth bass (Figure 2-6).

Figure 2-6. Shallow Nearshore Area

Numerous aquatic and shorebird species such as cormorants and spotted sandpipers use the habitats in the LWR. The upland environment near the LWR is primarily urban, with fragmented areas of riparian forest, wetlands, and associated upland forests. Historical development and filling of channels and wetlands has left only small strips or isolated pockets of riparian wildlife habitat, except for areas such as Harborton Wetlands, Oaks Bottom, Forest Park, and Powers Marine Park. Although isolated wildlife habitat areas exist along the LWR corridor, linkages to the larger landscape are limited to few areas, such as Forest Park.

In the City of Portland's earlier version of the Willamette River corridor natural resource inventory (Adolfson et al. 2000), 15 sites of significant habitat value for fish and wildlife were identified. These habitat sites are known to be used by numerous aquatic birds, aquatic mammals, and semi-aquatic mammals. Significant habitat sites in the Study Area include the South Rivergate corridor at the north end of the Study Area, the Harborton forest and wetlands, Willamette Cove, the railroad corridor, and the Swan Island beaches and lagoon on the southern end of the Study Area (Adolfson et al. 2000). The available wildlife habitat in the Study Area is shown on Map 2-2. Other important habitat sites identified in the general area were Kelley Point at the confluence of the Willamette and the Columbia Rivers, and the Ross Island and Oaks Bottom Complex near RM 16. The 2008 inventory identified 20 special habitat areas within the North Reach (Broadway Bridge to the Columbia River), including the Willamette River; portions of the Columbia Slough, Johnson Creek, and Tryon Creek; urban nesting sites such as bridges and chimney roosts; bluff areas; grasslands at Powell Butte; native oak assemblages; bottomland hardwood forests; and wetlands (City of Portland 2008).

2.2 SPECIES PRESENCE AND HABITAT USE

Although the ecological habitats of the LWR have been greatly modified by development, many invertebrate, fish, bird, mammal, amphibian, reptile, and plant species, including some protected by the Endangered Species Act (ESA), use habitats within and along the river. The following subsections present an overview of the various aquatic or river-dependent biological communities in the LWR.

2.2.1 Benthic Invertebrates

The distribution and composition of invertebrate communities in riverine systems are functions of physical, chemical, and biological interactions. These interactions affect the temporal stability of habitats, the amount of sunlight and oxygen available to organisms, the abundance and quality of food, and breeding opportunities. The diversity and abundance of invertebrates in rivers tend to be greatest where habitats are varied (i.e., spatially heterogeneous) with some moderate, predictable disturbances (e.g., seasonal flooding) (Thorp and Covich 2001).

Invertebrates in large river systems are predominantly benthic; those that burrow within a soft substrate are typically referred to as infauna, while those that live on the sediment or other hard surface are called epifauna. Benthic invertebrates may be large enough to be seen by the naked eye (macroinvertebrates, such as crayfish; Figure 2-7) or small enough to live among the sand grains below the sediment surface (e.g., meio- or microfauna). Benthic macroinvertebrate communities in large rivers are represented by a diverse array of species including arthropods (e.g., insects, mites, amphipods, crayfish), annelid worms, clams, snails, and nematodes. Many meiofauna (e.g., rotifers, early larval stages of many invertebrates, and nematodes) and other microorganisms (e.g., protozoans, bacteria) are also a significant part of the benthic community.

Figure 2-7. Crayfish in the Study Area

Benthic communities serve various functions in large river ecosystems. Infaunal and epifaunal invertebrates often make up a significant portion of the heterotrophic biomass in a river system (Jahn and Anderson 1986) and thus serve as an important food source for other invertebrates, fish, birds, and mammals. Benthic invertebrates control energy flow by acting as principal processors of organic matter (Merritt et al. 1984) and are also involved in nutrient cycling between the sediment and overlying water (particularly infauna and meiofauna).

Benthic invertebrates represent a spectrum of feeding types, including those that graze on periphyton and macrophytes (grazers), process large organic material often imported from terrestrial habitats (shredders), remove suspended particulate organic material from the water column (filter feeders or collectors), gather organic detritus from the sediment surface (gatherers), glean organic matter from sediment they consume (deposit feeders), prey on other invertebrates and small fish (predators such as some oligochaetes, chironomid midges, and crayfish), and parasitize other organisms (e.g., nematodes). Lifestyles are also diverse and reflect various strategies to adapt to changes in environmental conditions. The River Continuum Concept (see text box) predicts that invertebrate communities in deep rivers are typically dominated by organisms that forage for organic matter in or on the sediments and organisms that filter organic matter out of the water column (Cummins and Klug 1979) because suspended or newly settled fine organic material is the primary food resource available in large rivers. The benthic community in the LWR is dominated by organisms that filter feed (e.g., clams, amphipods,[18] and polychaetes) or deposit feed (e.g., tubificid worms and chironomid larvae) (Integral et al. 2004a).

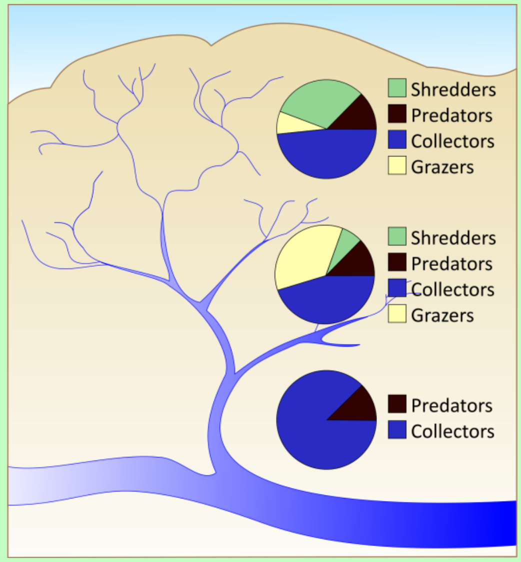
As discussed in Section 2.1, the Study Area is characterized by a navigation channel (which is maintained through active dredging) with a predominantly developed

[18] *Corophium* sp. is the dominant benthic amphipod in the LWR and can feed on both deposited and suspended organic material.

shoreline[19] (e.g., rock riprap, sheet pile, bulkheads). The channel habitat is uniform and consists of unconsolidated sediments (sands and silts) that are typically subject to periodic transport. Depending on the local physical regimen in the channel, the sediment may be seasonally stable or unstable, resulting in heterogeneous benthic communities. With some exceptions (e.g., parts of Willamette Cove and Balch Creek Cove), shallow-water habitats in the Study Area are largely limited to the narrow strip between the shoreline and the navigation channel. The benthic communities in the shallow-water habitats are controlled by physical characteristics such as slope, grain size, and the magnitude of disturbance events that may occur, with more well-developed communities and longer-lived species found in more stable areas, typified by fine-grained sediments.

[19] Pockets of riparian habitat occur throughout the Study Area in areas that have not been fully developed or where restoration activities have taken place.

What is the River Continuum Concept?



Contaminant inputs to the environment (such as those evaluated in this risk assessment) are but one of many ecological factors that affect benthic communities. Knowledge about those other factors can provide perspective on the condition of benthic communities in the Study Area. The River Continuum Concept is one of those factors.

Physical and biological characteristics of a river can change dramatically from its headwaters to its mouth. As a river widens and deepens in its course downstream, parts of the bottom are removed from the photic zone, reducing the influence of the riparian zone as a local source of food (including particulate carbon) and shading effects. As a result of these changes, benthic communities undergo marked shifts in composition, abundance, and feeding strategies. The River Continuum Concept is a holistic view that assigns sections of a river into one of three general classifications. The headwaters (upper reaches of the watershed) are usually very narrow and lined by dense riparian vegetation, which limits the penetration of sunlight and photosynthetic production of organic material in the water. The majority of the organic matter that does make its way into the system is in the form of plant material that falls or washes into the river. In the mid-reaches, the river becomes wider, allowing more sunlight to penetrate, and in-river structures such as large wood debris and rocks become important, either as suppliers of organic material or as substrate for primary producers such as periphyton. In the lower reaches, production through photosynthesis decreases because of an increase in water cloudiness and surface film of fine particulate organic matter imported from middle and upper reaches of the river.

The invertebrate community changes along the course of the river because of differences in the structure and, to some degree, the location (i.e., water column vs. bottom) of the organic material. In the upper reaches of a river, shredders (e.g., mayfly and stonefly larvae) and collectors (e.g., midge larvae and nematodes) make up a large percentage of the invertebrate community because of the large amount of coarse plant matter that collects on the bottom. Shredders rework coarse organic material, such as small sections of leaves. In the feeding process, the leaves are broken up into finer particulates, much of which is transported downstream as suspended material. Collectors utilize this fine suspended particulate organic matter and catch or filter particles using adapted appendages or behaviors. In the mid-reaches of the river, there is an increase in the proportion of grazers (e.g., snails, caddis fly larvae) feeding on the periphyton, which accumulates on larger structures such as stones, wood, and large aquatic plants. Shredders make up only a small percentage of the invertebrates in lower sections of rivers because of reduced supply of coarse organic matter. In the lower reaches, collectors that feed on the fine particulate organic matter and surface films are the most abundant invertebrates.

The infaunal invertebrate community within the Study Area is numerically dominated by oligochaete worms, many (~ 35) species of chironomid larvae (midges), amphipods (particularly *Corophium* spp.), and the introduced Asiatic clam, *Corbicula fluminea* (Integral et al. 2004a). Worms (nematodes and polychaetes) are also common infaunal invertebrates. The epifaunal community is dominated by mayfly (Ephemeroptera) larvae, caddisfly (Trichoptera) larvae, flatworms, decapods (primarily crayfish), and organisms similar to those that dominate the infauna, including midge larvae, amphipods, oligochaetes, and molluscs.

When Do Chironomids Inhabit the LWR?

These aquatic insects (i.e., midges) are an important component of the Study Area ecology, Among the most common and most abundant aquatic invertebrates, chironomids tolerate a wide range of environmental conditions, from swift-moving streams and deep slow-moving rivers to stagnant ditches, lakes, and ponds rich in decomposing organic matter. In the lower Willamette, chironomid species represent up to 50% of the benthic infauna at some locations (Integral et al. 2004a).

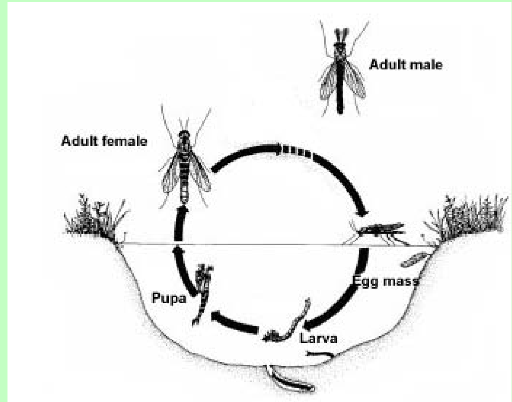


Figure courtesy of the University of Florida Entomology & Nematology Department

The life cycle of chironomids has four stages. Eggs are laid on the surface of the water as a gelatinous mass which, depending on the species, may contain up to 3,000 eggs. The eggs sink to the bottom and hatch within a week. After leaving the egg mass, the larvae burrow into the mud or construct small tubes in which they live. The larvae feed on suspended organic matter in the water and in the mud.

Larvae living in low-oxygen environments are commonly called "blood worms" because of their dark red color, which is caused by blood hemoglobin and allows the larvae to live in areas with low dissolved oxygen. Depending on water temperature, 2 to 7 weeks after leaving the egg mass, the larvae transform into pupae. After 3 days, the pupae swim to the surface, where they emerge as adults several hours later.

The timing of midge emergence is species-specific and can occur year-round; however, most adults emerge during the spring and summer. Adults mate in swarms soon after emerging. Adults live for only 3 to 5 days and do not feed. Midge larvae are eaten by a large variety of aquatic organisms, such as dragonfly nymphs, predaceous diving beetles, and a variety of fish species, particularly bottom-feeding fish such as carp.

Table 2-1 provides a full list of taxa collected during invertebrate surveys in the Study Area.

Table 2-1. Invertebrates Collected in the Study Area During Round 1 and Round 2

Phylum	Class or Order	Scientific Name	Common Name
Bryozoa			Moss animals
Cnidaria	Hydrozoa (Anthoathecatae)	*Hydra* sp.	Hydra
Platyhelminthes	Turbellaria		Flatworm
Nemertea	Enopla (Hoplonemertea)	*Prostoma* sp.	Ribbon worm
Nematoda			Roundworm
Annelida	Polychaeta	Aeolosomatidae	Worm
		Manayunkia speciosa	Sabellid worm
	Oligochaeta	Lumbriculidae sp.	Lumbricid worm
		Haplotaxidae sp.	Worm
		Enchytraeidae sp.	Enchytraeid worm
		Aulodrilus limnobius	Tubificid worm
		Aulodrilus pigueti	Tubificid worm
		Aulodrilus piqueti	Tubificid worm

Table 2-1. Invertebrates Collected in the Study Area During Round 1 and Round 2

Phylum	Class or Order	Scientific Name	Common Name
		Aulodrilus pluriseta	Tubificid worm
		Aulodrilus sp.	Tubificid worm
		Branchiura sowerbyi	Tubificid worm
		Limnodrilus hoffmeisteri	Tubificid worm
		Quistradrilus multisetosus	Tubificid worm
		Chaetogaster sp.	Naidid worm
		Dero digitata	Naidid worm
		Dero sp.	Naidid worm
		Nais barbata	Naidid worm
		Nais pardalis	Naidid worm
		Nais variabilis	Naidid worm
		Pristina aequiseta	Naidid worm
		Pristina leidyi	Naidid worm
		Pristina osborni	Naidid worm
		Pristinella sp.	Naidid worm
		Slavina appendiculata	Naidid worm
		Stylaria lacustris	Naidid worm
		Vejdovskyella sp.	Naidid worm
	Hirudinea	Unknown sp.	Leech
		Erpobdellidae sp.	Leech
Mollusca	Gastropoda	*Ferrissia* sp.	Limpet
		Menetus opercularis	Pulmonate snail
		Physa sp.	Bladder (or patch) snail
	Bivalvia (Unionoida)	*Anodonta nuttalliana*	Winged floater
		Corbicula fluminea	Asiatic clam
		Margaritifera falcata	Western pearlshell mussel
		Pisidium sp.	Fingernail clam
Arthropoda	Arachnida	*Arrenurus* sp.	Water mite
		Frontipoda sp.	Water mite
		Hygrobates sp.	Water mite
		Lebertia sp.	Water mite
		Limnesia sp.	Water mite
		Limnesiidae sp.	Water mite
		Torrenticola sp.	Water mite
		Unionicola sp.	Water mite

Table 2-1. Invertebrates Collected in the Study Area During Round 1 and Round 2

Phylum	Class or Order	Scientific Name	Common Name
	Crustacea (Isopoda)	*Caecidotea* sp.	Isopod
	Crustacea (Amphipoda)	*Anisogammarus* sp.	Amphipod
		Corophium sp.	Amphipod
		Corophium spinicorne	Amphipod
		Gammaridae sp.	Amphipod
		Hyalella sp.	Amphipod
	Ostracoda	Unknown sp.	Ostracod or seed shrimp
	Insecta (Ephemeroptera)	*Caenis* sp.	Mayfly
		Stenonema terminatum	Mayfly
	Insecta (Odonata)	Gomphidae	Dragonfly
		Stylurus sp.	Dragonfly
	Insecta (Trichoptera)	Hydroptilidae sp.	Caddisfly
		Hydroptila sp.	Caddisfly
		Orthotrichia sp.	Caddisfly
		Oecetis sp.	Caddisfly
		Polycentropodidae sp.	Caddisfly
		Polycentropus sp.	Caddisfly
	Insecta (Diptera)	*Ablabesmyia* sp.	Midge
		Brillia sp.	Midge
		Bryophaenocladius sp.	Midge
		Chironomini gr.	Midge
		Chironomus sp.	Midge
		Cladopelma sp.	Midge
		Cladotanytarsus sp.	Midge
		Corynoneura sp.	Midge
		Cricotopus bicinctus gr.	Midge
		Cricotopus sp.	Midge
		Cryptochironomus sp.	Midge
		Demeijerea sp.	Midge
		Dicrotendipes sp.	Midge
		Endochironomus sp.	Midge
		Eukiefferiella brevicalcar gr.	Midge
		Glyptotendipes sp.	Midge
		Harnischia sp.	Midge

Table 2-1. Invertebrates Collected in the Study Area During Round 1 and Round 2

Phylum	Class or Order	Scientific Name	Common Name
		Nanocladius sp.	Midge
		Orthocladius complex	Midge
		Parachironomus sp.	Midge
		Paracladopelma sp.	Midge
		Parakiefferiella sp.	Midge
		Paralauterborniella nigrohalteris	Midge
		Paraphaenocladius sp.	Midge
		Paratanytarsus sp.	Midge
		Phaenopsectra sp.	Midge
		Polypedilum sp.	Midge
		Procladius sp.	Midge
		Psectrocladius sp.	Midge
		Pseudochironomus sp.	Midge
		Rheotanytarsus sp.	Midge
		Stenochironomus sp.	Midge
		Tanytarsus sp.	Midge
		Thienemanniella sp.	Midge
		Xenochironomus xenolabis	Midge

Note: Invertebrates were collected using a variety of equipment, including Hester-Dendy multiplate samplers, van Veen grabs, and ponar grabs. More information is provided in Integral et al. (2004a).

Oligochaete worms feed on bacteria, diatoms, detritus, and other micro-organisms by ingesting large quantities of sediment and extracting organic material. Some oligochaete species live within an inch (approximately 1 to 3 cm) of the sediment surface, while others live in tubes attached to filamentous algae, submerged plants, and terrestrial debris (Brusca and Brusca 2003). Chironomids have an aquatic larval stage during which they feed on the sediment surface or from the water column. Depending on the species, the larvae of this diverse group can forage directly on plant or detrital material on or in the sediment, collect suspended material from the water column, or prey upon other invertebrates. Some chironomid species can cling to rocks, aquatic macrophytes, and other hard substrates; other species burrow into the sediment.

Amphipods have diverse feeding strategies and can consume various kinds of plant and animal material. The most common amphipod in the Study Area, *Corophium* spp., is a tube-building amphipod (McCabe et al. 1997) that often occurs in high densities in fine-grained sediment and feeds on bound organic material from both the sediment surface and the water column. The introduced Asiatic clam (*Corbicula fluminea*), the most abundant bivalve in the Study Area, feeds from the near-bottom water on zooplankton, phytoplankton, and organic detritus. They can live in water up to about 90 ft (30 m) deep but are predominantly found in water depths from 0 to 6 ft (2 m) characterized by stable sand and gravel substrates (Pennak 1978). Long-lived freshwater mussels found in the LWR are also filter feeders, although several species (specifically unionaceans) have a parasitic larval form that requires a specific fish host. Nematodes are free-living roundworms that are typically parasitic in natural freshwater habitats. Most specimens are confined to the top few inches (5 cm) of the substrate (Pennak 1978). Infaunal nematodes can also be direct deposit feeders (feeding on sediment), while others are detritivores or microscavengers that feed on the sediment surface (Brusca and Brusca 2003). Freshwater flatworms are primarily represented by turbellarians, which are typically free-living on hard substrates and actively forage for prey.

Why Were Asiatic Clams Used in the BERA?

The Asiatic clam (*Corbicula fluminea)*, a Southeast Asia native, was introduced to North America in the early 1900s. These small (1.5 in. [<4 cm]) clams are taxonomically and functionally related to the native freshwater mussels and fingernail clams found in the LWR. (Asiatic clams are placed in the same suborder, Corbiculacea, with fingernail clams.) These bivalves all share similar feeding strategies (filter and surface detrital feeding) and physiological mechanisms for exposure to and uptake of contaminants in sediment.

The Asiatic clam is tolerant of a variety of environmental conditions, preferring sand or gravel substrates in medium to larger rivers. Compared with less resilient native mussels and clams, Asiatic clams are more successful in drainage systems subject to periodic anthropogenic or natural disturbance. As with most bivalves, the Asiatic clam is sensitive to environmental stresses such as temperature extremes and hypoxia. However, other life history traits more than compensate for this sensitivity. The low age to maturity, high fecundity, and dispersal mechanism of this species allow it to recover quickly from disturbance (McMahon and Bogen 2001), unlike native bivalve populations.

In the LWR, Asiatic clams were the most numerous bivalve collected and often were among the three most abundant benthic invertebrates. Asiatic clams are considered an undesirable invasive species in Oregon, and harvesting them is prohibited by law. Nonetheless, where present, they play a significant role in the food chain because they are abundant and productive. Asiatic clams may affect the cycling of nutrients or compete with native mussels for food. The clams are consumed by many species, including shorebirds, diving ducks, amphibians, reptiles, crayfish, other invertebrate predators, and fish (e.g., carp, bluegill, and sturgeon) (Thorp and Covich 2001).

The widespread distribution and abundance of the Asiatic clam in the LWR makes it a useful species for environmental monitoring and investigations of environmental quality.

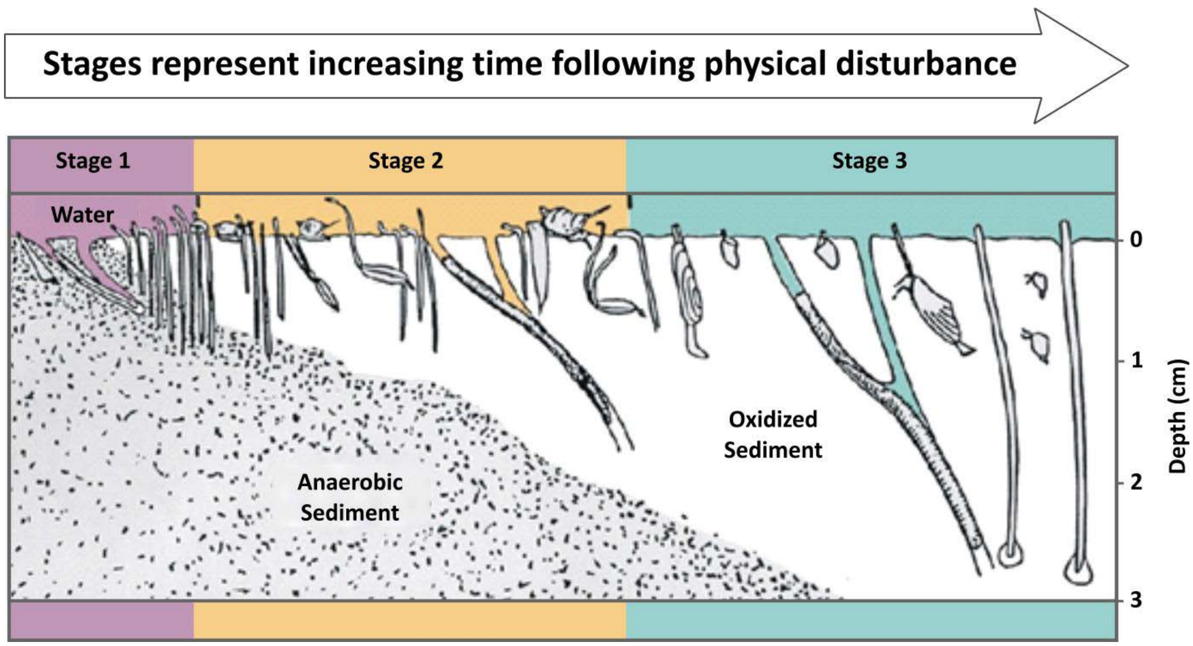
Mayflies and caddisflies, which can be found in many microhabitats in the LWR (e.g., burrowing in sediment, clinging to the undersides of rocks), use various feeding strategies, including grazing on algae and diatoms, filtering particles from the water column, and preying on other organisms. Crayfish are omnivores with a diet composed mainly of aquatic vegetation, but they will eat fish, aquatic insects, and detritus when aquatic vegetation is less available (Pennak 1978). Crayfish forage continuously, but feeding activities peak from dusk until dawn as a predator avoidance behavior (Thorp and Covich 2001).

Infaunal community samples were collected by LWG from 22 locations (Map 2-3) within the Study Area in the fall of 2002 to provide information on community structure, including relative abundance of taxa (Integral et al. 2004a). Infaunal community data were not collected in upstream or reference areas. The samples were primarily collected with a van Veen (0.1-m^2) grab sampler (two samples were subsampled from a 0.3-m^2 power grab) and sieved through a 0.5-mm screen, and the infauna were identified and enumerated in 21 of the 22 samples (a tar-like material in one sample reacted with sample preservatives, precluding the sorting of infauna).

The total number of taxa ranged from 6 to 21 per 0.1 m^2, and densities ranged from 7 to 590 per 0.1 m^2. Chironomid larvae, oligochaetes, and the Asiatic clam *Coribicula fluminea* were the most abundant taxa. Chironomid larvae were found at densities ranging from 1 to 326 per 0.1 m^2 at 18 of the 21 sampling locations, usually with two to five chironomid taxa at each location. Oligochaetes had a similar distribution with a total of 12 taxa found in the Study Area, usually with three to six taxa per sample. The oligochaete *Limnodrilus hoffmeisteri* was the most common taxon present at 19 of the 21 locations, with densities ranging from 17 to 316 per 0.1 m^2. Clams were represented by two taxa, *Corbicula fluminea* and *Pisidium* sp., at 19 of the 21 locations, with densities that ranged from 1 to 191 per 0.1 m^2 (*Corbicula fluminea*, the Asiatic clam, was the more abundant). Three groups of crustaceans (amphipods, isopods, and ostracods) were found during the survey, usually in low densities (i.e., fewer than 5 organisms per sample) with only *Corophium* spp., a small tube-dwelling amphipod, reported at higher densities (i.e., 10 to 148 per 0.1 m^2). As predicted by the River Continuum Concept, the composition of functional groups at the 21 sampling locations was dominated by filterers and gatherers (i.e., collectors) that feed on organic material suspended in the water column or newly settled on the sediment surface.

During the summers of 2002 and 2005, LWG conducted surveys of the epifaunal communities present in the Study Area by deploying artificial multiplate samplers in the water column. In 2002, multiplate samplers were deployed at 10 locations in the Study Area and at 2 locations between RM 9.0 and RM 13.0, which at that time were regarded as upstream reference areas. After 6 weeks, the multiplate samplers were retrieved, and the organisms that had colonized the substrate were identified. Chironomid larvae, oligochaetes, and *Corophium* spp. dominated the community collected on the multiplate samplers. Other common epifaunal invertebrates were sponges and bryozoans.

In 2002, the LWG surveyed the infaunal community structure throughout the Study Area using a sediment profile camera (SEA 2002) that evaluates physical and biological characteristics of a cross section of approximately the top 8 in. (20 cm) of the sediment column. This information is used to describe the benthic community characteristics, including community successional stage. The successional stage represents the response of the community to physical (e.g., flood, prop wash), chemical (exposure to contaminants), or biological (large-scale predation) disturbance events. Stage 1 communities are early colonizers composed of small, opportunistic species with short life cycles that typically dwell at the sediment-water interface. Stage 3 communities are usually composed of larger, longer-lived, deeper-dwelling organisms that reflect greater habitat stability and are most often found in fine-grained sediments. Stage 3 communities are considered climax or mature communities that can also indicate a longer time interval since the last disturbance event. Stage 2 communities are those transitioning from Stage 1 to Stage 3, and are characterized by organisms such as tube-dwelling amphipods or organisms that burrow or feed within the top few centimeters of the sediment column. Figure 2-8 depicts a conceptual model of benthic community maturation following a disturbance. The persistence of Stage 1 communities indicates continual perturbation of the benthic environment that can include physical (e.g., continual burial, erosion), chemical (contaminant exposure, high sulfides from decomposition of organic material in the absence of oxygen), or biological (e.g., predation) factors (SEA 2002).



Source: Rhoads and Germano (1986)

Figure 2-8. Conceptual Model of Benthic Community Response to Perturbation

Overall, the successional stages present in the LWR appeared to be closely associated with the sediment grain size of the substrate and its physical regime (Table 2-2). Late successional stage (Stage 3) communities were predominantly found only in fine-grained sediments (fine sands, silts, and clays), but not all areas of fine-grained sediment supported Stage 3 communities. Earlier successional stages were present in areas of fine-grained sediment that appeared likely to have experienced some type of periodic disturbance (prop wash along pier faces, within the navigation channel, or along channel slopes). Earlier community stages were also present in areas with very high rates of sediment deposition, most likely the result of ongoing burial. Stage 1 and 2 communities were typically found in areas of active transport of primarily coarse-grained sediment (Map 2-4).

Table 2-2. Distribution of Benthic Community Successional Stages by Physical Regime

Successional	Physical Regime

Stage	Highly Depositional	Depositional	Erosional/ Transport	Mixed	Unknown[a]	Total
Early	64	48	66	31	10	219
Transitional	4	5	8	5	0	22
Mature	68	87	19	28	3	205
Indeterminate[b]	6	2	29	4	36	77
Grand Total	**142**	**142**	**122**	**68**	**49**	**523**

[a] Almost all physical regimes classified as unknown were classified as such because debris was present.

[b] Indeterminate successional stages were associated primarily with coarse-grained sediments or debris fields that the SPI camera could not penetrate.

SPI – sediment profile imaging

What is Sediment Profile Imaging?

Given the importance of benthic communities to ecological analysis, all available tools to describe their condition should be considered. Sediment profile imaging (SPI) is one such tool. SPI is a technology to evaluate benthic community response to perturbations (chemical, physical, or biological) in soft-bottom habitats. This evaluation is based on the theory that organisms representing specific functional types occur in a predictable succession following a disturbance, from Stage 1 through Stage 3 communities. The profiles captured by SPI reflect the degree of maturation of community structure and function.

Stage 1 is characterized by very small, abundant organisms (typically opportunistic ostracods, tubificid oligochaetes, and some chironomids in freshwater systems (Soster and McCall 1990)) that can capitalize on the short-term availability of a habitat or a resource. These short-lived, early colonizers live within millimeters of the sediment-water interface (SWI), reflecting both their small size and the limited depth to which oxygen can enter sediment by molecular diffusion from the overlying water column alone (a deeper oxygenated layer depends primarily on biological activity of larger organisms).

Disturbed communities that have undergone some type of recovery will have infaunal members (e.g., tubicolous amphipods, small bivalves) that may burrow within the top few centimeters of the sediment column. These Stage 2 transitional communities are the first to rework deeper sediment and extend the oxygenated sediment zone to several centimeters or more. The presence of Stage 2 communities in freshwater environments is evidenced by dense amphipod tubes and/or the presence of shallow feeding voids.

Communities in stable environments with adequate food typically include larger, less abundant, longer-lived organisms (e.g., deposit-feeding oligochaetes, larger bivalves) that burrow or feed up to 20 cm below the sediment surface. These mature Stage 3 communities are responsible for the mixing of surface sediment with deeper underlying sediment in a process known as bioturbation. Organisms in these mature, relatively undisturbed communities can occur below the oxygenated zone by using physical or biological adaptations that give them access to the oxygenated water at the sediment surface or retard the influx of anoxic porewater into their burrows or tubes.

Image to the left shows a fine sand veneer over slightly sandy, gray silt. The surface layer of brown, sorted fine sand varies from 0.3 to 2.5 cm in thickness. There are subsurface methane pockets 16.8 cm below the SWI and extending to the bottom of the frame. Small tubes are present at the SWI. This sampling location is highly depositional and considered a Stage 1 community.

Image to the left shows gray, fine sandy silt with abundant amphipod tubes at the sediment-water interface. The dense assemblage of amphipod tubes, the sequestering of fine-grained sediment, and resultant colonization is a classic Stage 2 assemblage.

Image to the left shows soft, gray, slightly fine sandy silt with well-formed feeding voids from Stage 3 infauna, indicating that the subsurface sediment is being extensively reworked by the resident infauna.

The infaunal community structure in the upper segment of the Study Area (RM 7.0 to RM 10.0) was characterized by the widespread presence of Stage 3 infauna, both in nearshore areas (including Swan Island Lagoon) and main channel sediments (SEA 2002). In the middle segment of the Study Area, the sediments were coarse, indicative of higher current velocity. As might be expected, the infaunal community structure in the middle segment of the Study Area (RM 5.0 to RM 7.0) was dominated by Stage 1 infauna (SEA 2002). The fines that had been carried through the middle segment appeared to have been deposited in the lower reaches of the Study Area (RM 2.0 to RM 5.0), resulting in a fine-grained region. Again, as might be expected, the infaunal community structure in the lower segment of the Study Area was dominated by Stage 3 infauna, either by themselves or in association with Stage 1 infauna (SEA 2002). The presence of both Stage 1 and Stage 3 infauna likely indicates seasonal recruitment to an area with more mature communities.

Areas that were composed solely of Stage 1 infauna include the slips, the upstream portion of the segment from RM 2.0 to RM 5.0, and at the confluence of Multnomah Channel and the Study Area. The only areas that exhibited solely Stage 1 infauna coincided with regions of rapid deposition (the western main channel between RM 8.4 and RM 9.3, isolated nearshore locations that may be subject to physical or chemical perturbation, the northeast corner of Swan Island Lagoon, and the main channel of the river to the western shore between RM 7.1 and RM 7.5), suggesting that the habitat was too unstable or the depositional rate exceeded the ability of any longer-living Stage 3 infauna to colonize successfully and survive.

During the summer of 2002, LWG collected crayfish for contaminant analysis at 23 locations throughout the Study Area (SEA et al. 2003). The crayfish were not identified to species as part of the studies; however, only one crayfish species, the native western freshwater crayfish (*Pacifastacus leniusculus*), has been identified in the LWR (Boersma et al. 2006; Friesen 2005).

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

2.2.2 Fish

The diverse fish species that use habitats within the LWR include anadromous fish such as salmon and lamprey as well as numerous resident fish, including recreational species such as bass and sturgeon.

Piscivorous birds, aquatic mammals, and certain fish species rely on fish for food. Fish of all feeding guilds maintain the nutrient and energy cycles between aquatic primary producers and higher levels in both the aquatic and terrestrial food chains. Contaminants within the system can directly affect individual fish organisms, fish populations, and higher-trophic-level aquatic-dependant fauna that consume prey whose tissues contain contaminant residues.

Fish species identified as using habitat within the Study Area were grouped into the following feeding guilds:

- **Herbivores/omnivores –** fish that feed on vegetation, invertebrates, or both
- **Benthopelagic/benthic invertivores –** fish that feed primarily on invertebrates living either in the water column or on bottom substrates
- **Piscivores –** fish that feed primarily on fish
- **Detritivores –** fish that feed primarily on organic detritus

The following subsections provide detailed information on prey preferences, habitat preferences, and site use by fish species within these feeding guilds. The information is based on numerous studies, including Farr and Ward (1993), Fishman (1999), Beak (2000), and North et al. (2002), and a comprehensive compilation of published and unpublished literature on fish use of Portland Harbor (Ellis Ecological 2000). Fish species known to be present or to have been present in the LWR are listed in Table 2-3. It is likely that other species are present in the LWR but, not having been reported as such, are not included in Table 2-3. The species observed during the LWG sampling activities are indicated.

Table 2-3. Fish Known to be Present in the LWR

Species	Scientific Name	Resident or Migratory[a]
Herbivore/Omnivore		
Chiselmouth	*Acrocheilus alutaceus*	Resident
Mountain sucker	*Catostomus platyrhynchus*	Resident
Eulachon	*Thaleichthys pacificus*	Migratory
Bluegill[b]	*Lepomis macrochirus*	Resident
Common carp[b]	*Cyprinus carpio*	Resident
Pumpkinseed[b]	*Lepomis gibbosus*	Resident

Table 2-3. Fish Known to be Present in the LWR

Species	Scientific Name	Resident or Migratory[a]
Largescale sucker[b]	*Catostomus macrocheilus*	Resident
Brown bullhead[b]	*Ameiurus nebulosus*	Resident
Goldfish[b]	*Carassius auratus*	Resident
Green sturgeon[c]	*Acipenser medirostris*	Migratory
White sturgeon[b]	*Acipenser transmontanus*	Migratory
Yellow bullhead[b]	*Ameiurus natalis*	Resident
Invertivore		
American shad[b]	*Alosa sapidissima*	Migratory
Banded killifish[b]	*Fundulus diaphanus*	Resident
Chinook salmon[b, d]	*Oncorhynchus tshawytscha*	Migratory
Coastal cutthroat trout[e]	*Oncorhynchus clarki clarki*	Migratory
Coho salmon[f]	*Oncorhynchus kisutch*	Migratory
Mountain whitefish	*Prosopium williamsoni*	Resident
Peamouth[b]	*Mylocheilus caurinus*	Resident
Rainbow trout	*Oncorhynchus mykiss*	Resident
Redear sunfish[b]	*Lepomis microlophus*	Resident
Sockeye salmon	*Oncorhynchus nerka nerka*	Migratory
Steelhead[b, g]	*Oncorhynchus mykiss gairdneri*	Migratory
Prickly sculpin[b]	*Cottus asper*	Resident
Threespine stickleback[b]	*Gasterosteus aculeatus*	Both
Reticulate sculpin[b]	*Cottus perplexus*	Resident
Mottled sculpin	*Cottus bairdi*	Resident
Paiute sculpin	*Cottus beldingi*	Resident
Riffle sculpin	*Cottus gulosus*	Resident
Shorthead sculpin	*Cottus confuscus*	Resident
Starry flounder[b]	*Platichthys stellatus*	Migratory
Torrent sculpin	*Cottus rhotheus*	Resident
Warmouth[b]	*Lepomis gulosus*	Resident
Piscivore		
Black crappie[b]	*Pomoxis nigromaculatus*	Resident
Largemouth bass[b]	*Micropterus salmoides*	Resident

Table 2-3. Fish Known to be Present in the LWR

Species	Scientific Name	Resident or Migratory[a]
Northern pikeminnow[b]	*Ptychocheilus oregonensis*	Resident
Smallmouth bass[b]	*Micropterus dolomieui*	Resident
Walleye[b]	*Stizostedion vitreum*	Resident
White crappie	*Pomoxis annularis*	Resident
Yellow perch[b]	*Perca flavescens*	Resident
Detritivore		
Pacific brook lamprey	*Lampetra pacifica*	Resident
Pacific lamprey[b, h]	*Entosphenus tridentatus (formerly known as Lampetra tridentata)*	Migratory
River lamprey[i]	*Lampetra ayresi*	Migratory
Western brook lamprey[j]	*Lampetra richardsoni*	Resident

[a] Wydoski and Whitney (2003).
[b] Species observed during LWG sampling activities.
[c] Known to be present in the Columbia River; federally listed as threatened.
[d] Federally listed as threatened in Lower Columbia River and Upper Willamette River; state-listed as a critical species on ODFW sensitive species list.
[e] Federally listed as species of concern; state-listed as a vulnerable species on ODFW sensitive species list.
[f] Federally listed as threatened in Lower Columbia River; state-listed as endangered on the ODFW endangered species list.
[g] Federally listed as threatened in Lower Columbia River and Upper Willamette River; state-listed as a critical species on ODFW sensitive species list.
[h] Federally listed as species of concern; state-listed as a vulnerable species on ODFW sensitive species list.
[i] Federally listed as species of concern.
[j] State-listed as a vulnerable species on ODFW sensitive species list in the Columbia River systems.
LWR – Lower Willamette River
ODFW – Oregon Department of Fish and Wildlife

2.2.2.1 Herbivorous/Omnivorous Fish

Omnivorous and herbivorous fish in the LWR are exposed to contaminants primarily through their diet and incidental ingestion of sediment and water.

2.2.2.1.1 Herbivorous Fish

Only two herbivorous fish species are known to be common in the LWR: the chiselmouth (*Arocheilus alutaceus*) and the mountain sucker (*Catostomus platyrhynchus*) (Table 2-3). The chiselmouth and mountain sucker are benthic feeders and consume diatoms, algae, insects, and plants (Wydoski and Whitney 2003). Both species are resident and native to the region and have been captured in the LWR (Farr and Ward 1992; Beak 2000; Hughes and Gammon 1987; Tetra Tech 1995). Chiselmouth inhabit moderate-to-fast-moving pools, creeks, rivers, and lake margins over sandy or gravel substrate (Wydoski and Whitney 2003; Page and Burr 1991). Mountain sucker inhabit shallow waters of mountain streams over sandy to rocky substrate (Wydoski and Whitney 2003; Scott and Crossman 1973).

2.2.2.1.2 Omnivorous Fish

Ten omnivorous fish occur in the LWR (Table 2-3). Omnivores are predominantly bottom feeders that ingest sediment along with a variety of animal, plant, and detrital material. The largescale sucker (*Catostomus macrocheilus*) is a common native resident of the LWR (Beak 2000; Hughes and Gammon 1987; Tetra Tech 1995; Farr and Ward 1992). It consumes insect larvae as a juvenile; diatoms, detritus, crustaceans, and snails as an adult; and large amounts of sediment during feeding (CBFWA 1996). The species has a long life span (up to 15 years) and reaches reproductive maturity in 3 to 5 years for males and in 4 to 6 years for females (Wydoski and Whitney 2003). Largescale sucker generally inhabit shallow bottom areas of large riverine and estuarine waters.

Two omnivorous sturgeon species, white sturgeon (*Acipenser transmontana*) and green sturgeon (*Acipenser medirostris*), are found in the LWR, including Portland Harbor. Sturgeon rely on large, complex river systems for many of their life stages and can feed opportunistically on prey ranging from benthic invertebrates to large fish (Beamesderfer and Farr 1997). In addition, white sturgeon is a native species with a very long life span (e.g., some living more than 100 years) (Dees 1961). White sturgeon are highly valued by the Tribes, and they are also a popular sports fish. The annual commercial and sport harvest of sturgeon from the LWR has been estimated at 1,000 to 2,000 fish.

How Much of Their Lives Do White Sturgeon Spend in the LWR?

A long-lived, wide-ranging, omnivorous native fish of cultural importance to Tribes and sport anglers, white sturgeon (*Acipenser transmontanus*) is a receptor of great value.

Sturgeon identify prey on the bottom surface using their long barbels and then extend their mouths and suck up the prey item (USFWS 1961). Sturgeon may live up to 100 years of age. White sturgeon are known to be present in the Willamette River during their juvenile (pre-breeding) life stage. The average age of sturgeon collected from the Study Area of the LWR during 2007 sampling was 13 years old (ages ranged from 7 to 26 years). (Age analysis of juvenile sturgeon was determined by Ruth Farr and Michele Weaver at the Oregon Department of Fish and Wildlife (ODFW) using pectoral fin ray samples following ODFW protocols (Beamesderfer et al. 1998)).Wydoski and Whitney (2003) report that male sturgeon reach sexual maturity at 9 years of age and that females mature at 13 to 16 years. The median age at sexual maturity for white sturgeon in the lower Columbia River was reported as 24 years (DeVore et al. 1995).

Some studies suggest that sturgeon can show strong site fidelity (Veinott et al. 1999), while other studies indicate that individual sturgeon can have large ranges (DeVore and Grimes 1993). The home ranges of sturgeon are studied through the use of passive integrated transponder (PIT) tags or spaghetti wires, which are attached to sturgeon captured and released to track their movements. One juvenile white sturgeon collected from the Study Area during sampling in March 2007 was tagged with a spaghetti wire that had been placed by the Washington State Department of Fish and Wildlife (WDFW). The age of this tagged sturgeon based on a pectoral fin ray sample was 7 years old. Per WDFW (2007), the sturgeon was originally tagged in June 2006 at Rocky Point, which is located along the west shore of Grays Bay near the Pacific/Wahkiakum counties border on the Washington side of the Columbia River. The initial tagging location was approximately 72 miles from the location where the sturgeon was collected. The movement of this fish indicates a large home range for sturgeon, even during their pre-breeding life stage.

The common carp (*Cyprinus carpio*) is a long-lived (more than 20 years) exotic species resident to the LWR (Hughes and Gammon 1987). Adult fish are largely benthic feeders and consume copepods along with algae and plant fragments (Wydoski and Whitney 2003). The common carp has been found to be evenly distributed throughout the LWR, with population numbers increasing as the water temperature increases through the sampling season (Farr and Ward 1992; Tetra Tech 1995).

Brown bullhead (*Ameiurus nebulosus*) and yellow bullhead (*Ameiurus natalis*)—two introduced members of the Ictaluridae family—are resident in the LWR. These species are bottom feeders with similar life spans (i.e., approximately 5 years) and habitat preferences, although the preferred water depth for yellow bullhead (0 to 10 m) is shallower than that of brown bullhead (0 to 40 m) (Scott and Crossman 1973). In addition, brown bullhead tolerate low dissolved oxygen levels and high temperatures, whereas yellow bullhead prefer clear stream or pond water with aquatic vegetation (Wydoski and Whitney 2003). Yellow bullhead were captured frequently in several studies (Farr and Ward 1992; Beak 2000; Hughes and Gammon 1987) but infrequently in others (e.g., four fish in Beak (2000), one fish in Hughes and Gammon (1987)). Only a few brown bullhead were caught by Farr and Ward (1992).

Other omnivores possibly present in the LWR include several introduced species: the pumpkinseed sunfish (*Lepomis gibbosus*), bluegill (*Lepomis macrochirus*), and goldfish (*Carassius auratus*). Pumpkinseed prefer quiet vegetated pools in low-velocity areas of rivers (Wydoski and Whitney 2003). Habitat preferences of the bluegill are similar: low-gradient, low-velocity areas with abundant pools and aquatic vegetation (Stuber et al. 1982). Pumpkinseed and bluegill are benthopelagic species that have been caught in the LWR (Farr and Ward 1992; Beak 2000; Tetra Tech 1995). Goldfish are benthic feeders that prefer low-velocity, stagnant water of ponds, lakes, and slow-moving rivers (Wydoski and Whitney 2003).

2.2.2.2 Invertivorous Fish (Benthopelagic)

Three species of non-salmonid benthopelagic invertivorous fish (primarily feeding on invertebrates) may occur in the LWR: peamouth (*Mylocheilus caurinus*), American shad (*Alosa sapidissima*), and eulachon (*Thaleichthys pacificus*). The peamouth and eulachon are natives to the LWR; the American shad, a native to the East Coast, was introduced to the West Coast in the late 1800s.

The American shad is anadromous and a repeat spawner, migrating to fresh water after spending 2 to 6 years in the ocean (Stier and Crance 1985). Shad can live up to 11 years and reach reproductive maturity within 4 to 5 years (Stier and Crance 1985). Juvenile shad remain in fresh water for their first summer, moving to marine waters in the fall. Adult American shad prefer to spawn in broad flats or shallow water of large rivers (Stier and Crance 1985). While in fresh water, juvenile shad consume insects, crustaceans, zooplankton, and benthic invertebrates (Wydoski and Whitney 2003).

As adults, peamouth feed on benthic invertebrates, crustaceans, insects, and small fish and can live up to 13 years (Wydoski and Whitney 2003). Peamouth prefer shallow areas of lakes and slow-moving rivers, remaining nearshore during winter months and moving to deeper waters in the summer months (Wydoski and Whitney 2003).

The eulachon (*Thaleichthys pacificus*) is a native pelagic species that may be seasonally present in LWR. Eulachon inhabit predominantly marine waters, migrating to estuaries and coastal rivers to spawn. It is estimated that eulachon spend less than 6 weeks a year in fresh water.

2.2.2.3 Invertivorous Fish (Benthic)

Several non-salmonid benthic invertivores reside in the LWR, including seven sculpin species (*Cottus* spp.), starry flounder (*Platichthys stellatus*), warmouth (*Lepomis gulosus*), and threespine stickleback (*Gasterosteus aculeatus)* (Table 2-3). These species are native residents, except for warmouth, which is an introduced resident.

The prickly sculpin (*Cottus asper*) is one of seven members of the Cottidae family present in the LWR. The prickly sculpin lives approximately 4 to 5 years and reaches maturity within 2 to 4 years (Wydoski and Whitney 2003). The prickly sculpin is a benthic feeder as an adult and consumes crustaceans, aquatic insect larvae, fish, and molluscs (Wydoski and Whitney 2003). It prefers shallow water with sand, gravel, or rubble bottoms and abundant aquatic vegetation (Wydoski and Whitney 2003). It is also tolerant of salinity. Several studies suggested that the prickly sculpin is the most common sculpin in the LWR (Farr and Ward 1992; Hughes and Gammon 1987; Tetra Tech 1995). Other sculpin species reported to occur in the Willamette River are the reticulate sculpin (*Cottus perplexus*), mottled sculpin (*C. bairdi*), Paiute sculpin (*C. beldingi*), shorthead sculpin (*C. confuscus*), riffle sculpin (*C. gulosus*), and torrent sculpin (*C. rhotheus*) (Farr and Ward 1992; Hughes and Gammon 1987; Tetra Tech 1995). These species have similar life spans, and their adult diets consist of aquatic insects, crustaceans, snails, and fish eggs (Wydoski and Whitney 2003).

What Type of Habitat Do Sculpin Prefer?

Because sculpin (*Cottus* spp.) are abundant in the Study Area, dwell in and near the riverbed, forage over very small home ranges, and are likely to contact sediment-associated contaminants, they represent a key species for examination.

Seven sculpin species have been reported in the LWR: prickly, reticulate, riffle, mottled, Paiute, shorthead, and torrent sculpin. These sculpin species have a similar benthic habitat preference unique among fish living in the LWR. Sculpin generally prefer shallow water and tend to forage within a very small home range. Although some species may be found in sandy areas (i.e., prickly, reticulate, and riffle sculpin), sculpin are more commonly found in habitats with bottom substrates such as rubble, gravel, boulders, or rocks (Wydoski and Whitney 2003). Their small home ranges and benthic habitat use may result in higher exposures to sediment-associated contaminants.

In the Study Area, sculpin were most commonly collected from three general nearshore areas with coarse bottom substrates: riprapped areas, areas with unclassified fill (e.g., rocks, debris, concrete), and areas with man-made structures. Sculpin in the Study Area also were frequently observed in and collected from within locations with moderate cover. The length of sculpin collected from the Study Area ranged from approximately 3.5 to 7 in.

The threespine stickleback, a native to the LWR, can live in both freshwater and marine systems but spawn in freshwater habitats. It is a benthic feeder in fresh water, consuming small crustaceans, insects, and fish eggs (Wydoski and Whitney 2003). Threespine stickleback typically live up to 3 years and are found close to the bottom of streams and lakes near aquatic vegetation (Wydoski and Whitney 2003).

Starry flounder inhabit shallow to deep estuarine waters, although they can travel far upstream in rivers to forage. Starry flounder are benthic feeders, consuming crabs, molluscs, and small fish (Orcutt 1950).

The warmouth is exotic to the region. This species prefers backwater habitats with slow-moving water and dense vegetation and is known to be adversely affected by channelization (McMahon et al. 1984a). Juvenile warmouth feed on protozoa, bacteria, and zooplankton; adults feed on aquatic insect larvae, crayfish, and small fish (Wydoski and Whitney 2003).

2.2.2.4 Invertivorous Fish (Salmonids)

Seven species of salmonids are known to occur in this region: Chinook salmon (*Oncorhynchus tshawytscha)*, steelhead trout (*O. mykiss gairdneri)*, rainbow trout (*O. mykiss*), coho salmon (*O. kisutch*), sockeye salmon (*O. nerka nerka*), coastal cutthroat trout (*O. clarki clarki*), and mountain whitefish (*Prosopium williamsoni*) (Table 2-3). Many of these species are anadromous, hatching in fresh water, migrating to salt water, and returning to fresh water to spawn. Rainbow trout and mountain whitefish are resident species and are not anadromous. The larger salmon species are piscivorous as adults in the ocean, but are grouped with invertivores because juveniles prey primarily on invertebrates during their residence in rivers. Piscivorous adult salmon feed relatively little when returning upriver during their spring migrations.

Chinook salmon follow two life history patterns in the Willamette River, a stream type and an ocean type. Spring runs generally follow the stream-type pattern, spending 1 year or more in fresh water before migrating to the ocean. Summer and fall runs generally follow the ocean-type pattern, migrating to the ocean about 3 months after emergence (Healey 1991). Chinook salmon are semelparous, spawning only once then dying. Chinook spawn in gravel runs, and their eggs require high oxygen concentrations. Juveniles reside in marginal areas of rivers and find cover near woody debris and tree roots (Healey 1991). While in fresh water, juvenile Chinook salmon feed on aquatic insect larvae and terrestrial insects (Wydoski and Whitney 2003; Healey 1991).

Steelhead trout winter runs enter fresh water in March or April and spawn in May and June (NMFS 1996). The majority of steelhead in Washington and Oregon smolt after 2 years in fresh water; however, some juveniles can spend up to 7 years in fresh water before migrating to the ocean (NMFS 1996). Steelhead are iteroparous, being able to spawn multiple times, although most steelhead in this region spawn only once (NMFS 1996). Juvenile steelhead feed on aquatic insects and insect larvae while in fresh water (Wydoski and Whitney 2003).

Rainbow trout, the freshwater resident form of steelhead trout, have a lifespan ranging from 3 to 8 years. They consume aquatic insects, insect larvae, worms, and fish eggs as juveniles and aquatic insects and fish as adults (Wydoski and Whitney 2003; Raleigh et al. 1984). Rainbow trout inhabit the clear, cold water of stream riffles and pools with abundant vegetation present (Raleigh et al. 1984).

Coho salmon are also semelparous and anadromous (Sandercock 1991). Coho prefer to spawn in gravel located at the head of stream riffles (Wydoski and Whitney 2003; Sandercock 1991). After emergence, fry remain in freshwater habitat for 1 to 2 years before migrating to marine waters. Juvenile coho inhabit shallow waters, less than 20 ft deep, in backwater areas, side channels, and small creeks with overhanging vegetation (Sandercock 1991). Like other salmonid species, juvenile coho are insectivores and consume mostly insects, insect larvae, worms, and fish eggs (Wydoski and Whitney 2003).

What Salmonid Species Are Common in the LWR?

Salmon are iconic species of the Pacific Northwest. The LWR is considered critical habitat for several salmonid species including Chinook salmon, steelhead, and coho salmon. Chinook were the most prevalent species caught using both electrofishing and beach seine gear in a 2001 ODFW study. In the beach seine catch, sub-yearling Chinook were the highest catch overall (94.7%), followed by coho (0.6%), and unidentified salmonids (4.7%). Electrofishing catch was comprised of Chinook (47.1%), coho (11.5%), steelhead (3.0%), and unidentified salmonids (38.4%) (North et al. 2002). It appears that some seasonal variation in relative abundance occurs among these species. The relative abundance from most to least fish caught per unit effort by beach seine in the LWR was coho, Chinook, and steelhead in spring; Chinook, coho, and steelhead in summer; and Chinook, steelhead, and coho in fall (North et al. 2002). This information contrasts with the results from Portland General Electric out-migrant counts at Willamette Falls and at the Clackamas hydroelectric dam, which found these salmonid species to be abundant shorter periods of time (Domina 1997). This discrepancy is probably due to the different locations sampled and different observation methods.

Salmonids, both adult and juvenile, are common in the LWR during various times of the year. Timing of downstream migration of juvenile salmonids has been documented by monitoring yearling Chinook movement patterns downstream to Willamette Falls (Schreck et al. 1994b), seasonal fish trapping at Willamette Falls (Domina 1997; Massey 1967), and sequential seasonal sampling within the harbor (Beak 2000; Farr and Ward 1993; Fishman 1999; Ward and Farr 1989, 1990; Ward and Knutsen 1991; Ward et al. 1988; 1994). Juvenile salmon can be found in the LWR year-round (various life stages), but peak periods of downstream migration appear to be March through mid-June and November.

Based on telemetry data, juvenile Chinook salmon appear to have a longer residence time in Portland Harbor than steelhead or coho salmon (Ward et al. 1992; North et al. 2002). Average migration rates were 15.5 km/day for steelhead, 13.8 km/day for coho, 11.0 km/day for yearling Chinook, and 7.2 km/day for sub-yearling Chinook (North et al. 2002). Migration duration for juvenile Chinook salmon through the LWR from Willamette Falls to the mouth of the Columbia River ranged from 2 days to 2 months, based on calendar year 2001 ODFW studies (North et al. 2002). Beach seining data collected in 2001 showed that the migration duration of sub-yearling fall Chinook salmon is shorter than that of yearling spring Chinook salmon. Preliminary radio telemetry studies found that the range of residence times for sub-yearling fall Chinook was 1.2 to 6.8 days from RM 9.5 to RM 3.5 and 1.6 to 26.8 days from RM 18.5 to RM 3.5 (Ellis Ecological 2001). Residence time of smaller juvenile salmon (less than 108 mm) has not been measured and may vary from that reported here. Periods of adult salmonid migration through Portland Harbor are not as well documented as downstream movements (Ellis Ecological 2001).

Sockeye salmon spawn in gravel riffles of streams and tributaries to lakes. Upon emergence, juvenile sockeye spend 1 to 2 years in freshwater habitats, usually the pelagic zone of lakes (Wydoski and Whitney 2003). Juvenile sockeye consume zooplankton while in fresh water.

Coastal cutthroat trout have variable life history patterns. Some are anadromous, migrating to marine waters and returning to fresh water to spawn; some are potamodromous, spending most of their lives in streams and lakes and migrating to tributaries to spawn; and some are non-migratory, remaining in small streams and headwater tributaries (Trotter 1997). Coastal cutthroat trout are known to spawn in the smallest headwater streams (Wydoski and Whitney 2003). Upon emergence, juveniles prefer low-velocity backwater areas until large enough to move into riffles and overwinter in pools with logs and vegetation for cover (Trotter 1997). Anadromous juveniles remain in freshwater habitats for 2 to 4 years before migrating to marine waters. While in fresh water, juveniles are pelagic feeders and consume fish, insect larvae, and sand shrimp (Trotter 1997).

The mountain whitefish is a native salmonid and prefers riffle areas and large pools of cold streams (Wydoski and Whitney 2003). This species feeds on crustaceans, larval insects, and some fish (Wydoski and Whitney 2003).

2.2.2.5 Piscivorous Fish

Northern pikeminnow (*Ptychocheilus oregonensis*), smallmouth bass (*Micropterus dolomieui)*, largemouth bass (*Micropterus salmoides*), black crappie (*Pomoxis nigromaculatus*), white crappie (*Pomoxis annularis)*, walleye (*Stizostedion vitrem*), and yellow perch (*Perca flavescens*) are piscivorous fish species known to inhabit the region (Table 2-3). As high-trophic-level predators, all of these species play a key role in the dynamics of the aquatic community. Because of their high trophic status, these fish have a greater potential than many other species for biomagnifying contaminants such as polychlorinated biphenyls (PCBs), dichlorodiphenyltrichloroethanes (DDTs), and mercury. The diets of piscivorous fish in the LWR have been shown to be similar (Fishman 1999). Of the piscivores listed in Table 2-3, northern pikeminnow is the only native species.

The northern pikeminnow has a long life span, up to 19 years and reaches reproductive maturity at 3 years for males and 4 years for females (Wydoski and Whitney 2003). Northern pikeminnow are benthopelagic and inhabit large riverine systems, remaining nearshore in summer and occupying deeper waters in the winter (Wydoski and Whitney 2003). Of six common species in the LWR, northern pikeminnow was the most commonly caught fish (Ward and Nigro 1992).

Smallmouth bass, largemouth bass, black crappie, and white crappie are members of the Centrarchidae family and are piscivorous or semi-piscivorous. All four species are benthopelagic, consuming fish, crayfish, other crustaceans, molluscs, and worms as adults and insect larvae and zooplankton as juveniles (Wydoski and Whitney 2003; Turner 1966; George and Hadley 1979). None of the species is native. The largemouth bass has a longer life span (i.e., 12 to 16 years) than smallmouth bass (approximately 10 years) (Wydoski and Whitney 2003). Largemouth bass inhabit warm shallow waters with abundant plants and woody debris available for cover (Wydoski and Whitney 2003). Smallmouth bass prefer riverine systems with a moderate current and rocky substrate (Wydoski and Whitney 2003) and use riprap for cover (Farr and Ward 1992). Largemouth and smallmouth bass are reported to be common throughout the LWR (Farr and Ward 1992; Tetra Tech 1995; Ward and Nigro 1992; Beak 2000).

Both black and white crappie were introduced to the LWR. The black crappie has a relatively long life span (approximately 13 years); the white crappie lives 7 to 9 years (Wydoski and Whitney 2003). Black crappie prefer areas of low velocity and turbidity with abundant vegetative cover and nest in soft mud (Edwards et al. 1982a). White crappie inhabit low-gradient, low-turbidity, slow-moving riverine systems with abundant vegetative cover and shallow areas for nesting (Edwards et al. 1982b). Several studies have shown black and white crappie to be abundant centrarchid species in the LWR (Farr and Ward 1992; Beak 2000; Ward and Nigro 1992).

The walleye is another introduced resident to the LWR with a long life span (i.e., 17 years) (McMahon et al. 1984b). Walleye consume fish and crustaceans as adults. The species requires moderate-to-large riverine systems with abundant shallow vegetated areas for all life stages, and prefers to spawn in rocky areas in rivers or below falls (Scott and Crossman 1973; McMahon et al. 1984b; Wydoski and Whitney 2003). Walleye have been captured in the LWR as part of several studies (Beak 2000; Hughes and Gammon 1987; Tetra Tech 1995), and Farr and Ward (1992) suggested that walleye prefer less developed areas of the LWR.

The yellow perch is exotic to the Study Area but appears to be common throughout the LWR (Beak 2000; Hughes and Gammon 1987; Tetra Tech 1995). Yellow perch can live up to 10 years, but most live for 7 years (Krieger et al. 1983). The yellow perch prefer shoreline habitat with pools and vegetation in freshwater systems, although they can tolerate brackish water (Krieger et al. 1983). On the basis of tagging studies and reports from large lakes, yellow perch are reported to have small home ranges (Wydoski and Whitney 2003).

2.2.2.6 Detritivorous Fish

Four species of detritivorous lamprey are native to the Willamette River: the Pacific lamprey (*Entosphenus tridentatus,* formerly known as *Lampetra tridentata*), the river lamprey (*Lampetra ayresi),* the western brook lamprey (*Lampetra richardsoni),* and the Pacific brook lamprey (*Lampetra pacfica*) (Table 2-3). Juvenile lamprey (ammocoetes) are unique to the fish community in the Study Area because they live burrowed in the sediment where they filter algae, detritus, and other organic material from the near-bottom water column (Figure 2-9). Lamprey ammocoetes are the only detritivorous fish present in the LWR. This species resides in fresh water for up to 6 years (Figure 2-9).

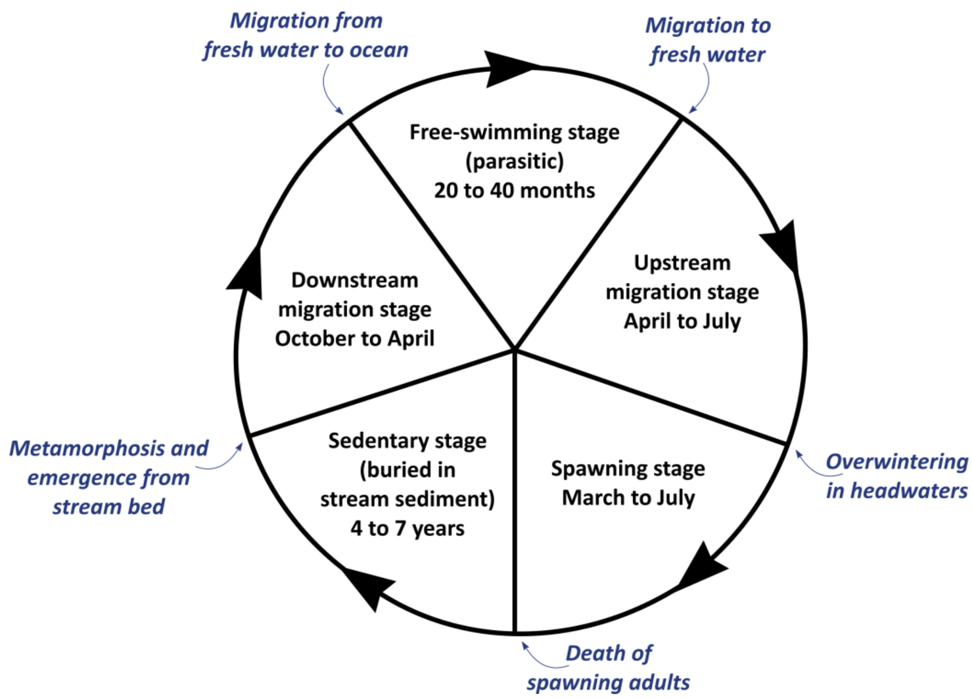


Figure 2-9. Typical Lifecycle of a Pacific Lamprey

The Pacific lamprey and river lamprey share many similar life history traits. Both are filter feeders as juveniles, consuming phytoplankton and detritus while burrowed in freshwater sediment (Kostow 2002a, b; Moore and Mallatt 1980). Pacific lamprey have a longer life span (up to 12 years) than do river lamprey (up to 8 years) and take longer to mature (4 to 7 years versus 4 to 6 years, respectively) (Kostow 2002a, b). Pacific lamprey is a species of concern under the ESA and an Oregon state sensitive species. Adult lamprey travel through the LWR while migrating to upstream spawning areas in the upper river; the amount of time spent in the Study Area is unknown. Growth in adult lamprey, which do not feed in fresh water, results primarily from parasitic feeding on other fish in the ocean or estuary. Farr and Ward (1992) and Beak (2000) reported collecting a few Pacific lamprey in the LWR. However, approximately 10,000 adult lamprey are harvested annually upstream at Willamette Falls (Kostow 2002a, b), and these fish must pass through the Study Area on their upstream migration. Because of declines in the number of returning lamprey, the current harvest is a dramatic reduction from the 1940s' and 1950s' annual harvests of 300,000 to 500,000 lamprey.

The two resident lamprey, western brook and Pacific brook, are similar to the anadromous species in that juveniles remain burrowed in mud until maturity, feeding on diatoms and detritus (Kostow 2002a, b). Both the western brook and Pacific brook lamprey live less than 6 years and reach maturity within 4 to 6 years. As adults, these two species remain in fresh water, migrating downstream from the spawning grounds. However, unlike the anadromous species, whose adults become ectoparasitic, the two resident species do not feed as adults (Kostow 2002a, b). As soon as they become adults, they spawn and die. Friesen and Ward (1996) reported collecting western brook lamprey in streams of the Tualatin Basin in the Willamette River basin.

When Do Pacific Lamprey Use the LWR?

Photo courtesy of the US Fish and Wildlife Service, Armon Barrows, illustrator

As the only detritus-eating fish in the LWR, the Pacific lamprey (*Entosphenus tridentatus)* fills an ecological niche not otherwise represented in this risk assessment. An anadromous species, Pacific lamprey spend their early life stages in fresh water, migrate to the ocean, and return as adults to spawn in fresh water. Adults are briefly present in the LWR from April to July during their upstream spawning migration. Their migration to headwater streams involves alternating periods of upstream swimming and resting while sucking rocks. When water temperatures in headwater streams reach 10 to 15°C the following spring, the lamprey deposit 10,000 to 100,000 very small eggs in gravel and sandy bottoms. The adults die within 4 days of spawning.

The young hatch in 2 to 3 weeks and swim to areas of low flow where they burrow into the sediment. During this stage, the larvae (ammocoetes) are blind and sedentary. They survive by filtering food particles such as detritus, diatoms, and algae. The juvenile lamprey remain burrowed in the mud for about 4 to 6 years, rarely moving to new areas (see Figure 2-9). Lamprey at this life stage are present in the LWR, but the duration of their residence in the LWR is unknown. Based on an extensive sampling effort, juvenile lamprey appear to be scarce in the Study Area (Windward 2006a). Transformation from the larval to juvenile life stages (metamorphosis) generally occurs during July through October. During metamorphosis the lamprey develop eyes, a mouth with teeth, and other physiological characteristics preparing them for a parasitic lifestyle in salt water. Metamorphosing lamprey are also present in the LWR. After a 2-month metamorphosis they emerge as adults about 10 to 13 cm long. In late winter or early spring, the new adults migrate to the ocean. While living in the ocean, lamprey are scavengers, predators, or parasites on larger prey such as salmon and marine mammals. After 2 to 3 years in the ocean they return to fresh water to spawn.

2.2.3 Wildlife

A diverse group of birds and a small number of aquatic or aquatic-dependent mammals are known to occupy habitat areas in the LWR. Birds that use the LWR represent various feeding guilds, each filling a distinct role in the ecosystem. Many of the bird species using LWR habitats migrate beyond the United States, and are protected under the Migratory Bird Treaty Act (16 USC 703-712). Mammals that use the LWR are predominantly piscivorous, although their diet may also include amphibians and aquatic invertebrates. Birds and mammals provide a pathway for the transfer of energy and nutrients from the aquatic to the terrestrial ecosystem; they also may serve as prey for other predators. Piscivorous birds and mammals are relatively high on the food chain and may be exposed to greater concentrations of contaminants that biomagnify at successive levels of the food chain. The presence of and habitat use by wildlife species in the Study Area is summarized in the following subsections.

2.2.3.1 Birds

Numerous aquatic-dependent bird species use habitats associated with the LWR. Of the sites along the LWR with significant habitat, as identified by Adolfson et al. (2000), the Oaks Bottom Complex supports the greatest abundance and diversity of birds. Within this area is the Oaks Bottom Wildlife Refuge, a facility offering diverse habitat closely associated with Ross Island, upstream of the Study Area (see Map 2-2). More than 200 bird species have been reported in this area, including nesting raptors and river birds such as green-backed heron, northern shoveler, pintail, mallard, wood duck, coot, wigeon, gulls, and cormorant (Adolfson et al. 2000).

Bird species identified as using habitat within the Study Area were grouped into the following feeding guilds and are discussed in the following subsections:

- **Herbivores –** birds that feed predominantly on plant material
- **Diving carnivores and omnivores –** birds that usually swim on the surface or dive to feed on invertebrates or a mix of invertebrates, fish, and occasionally plants from the sediment surface
- **Sediment-probing invertivores and omnivores –** birds that probe in sediments for invertebrates in shallow water along the shoreline
- **Piscivores –** birds that feed exclusively on fish

Table 2-4 lists the aquatic and semi-aquatic bird species that may breed along the LWR. Table 2-5 lists species that may occur seasonally or for which the LWR represents only part of their habitat.

Table 2-4. Resident Bird Species Potentially Breeding in the Study Area

Species	Scientific Name	Residency Status[a]
Herbivores		
Canada goose	*Branta canadensis*	Some residents; some winter guests
Mallard	*Anas platyrhynchos*	Year-round
Diving Carnivores and Omnivores		
American dipper	*Cinclus mexicanus*	Mostly year-round
Common merganser	*Mergus merganser*	Mostly year-round
Hooded merganser	*Lophodytes cucullatus*	Some year-round; some winter guests
Pied-billed grebe	*Podilyumbus podiceps*	Summer; many winter guests
American coot	*Fulica americana*	Year-round
Cinnamon teal	*Anas cyanoptera*	Summer
Wood duck	*Aix sponsa*	Some year-round
Sediment-Probing Invertivores and Omnivores		
Common snipe	*Gallinago gallinago*	Mostly year-round
Killdeer	*Charadrium vociferous*	Year-round; some winter guests
Spotted sandpiper	*Actitis macularia*	Some year-round
Sora	*Porzana carolina*	Mostly summer; some winter guests
Virginia rail	*Rallus limicola*	Some year-round

Table 2-4. Resident Bird Species Potentially Breeding in the Study Area

Species	Scientific Name	Residency Status[a]
Piscivores		
American bittern	*Botaurus lentiginosus*	Year-round
Bald eagle[b]	*Haliaeetus leucocephalus*	Mostly year-round; some winter guests
Belted kingfisher	*Ceryle alcyon*	Mostly year-round
Double-crested cormorant	*Phalacrocorax auritus*	Some summer; many winter guests
Great blue heron	*Ardea herodias*	Year-round
Green heron	*Butorides virescens*	Some year-round
Osprey	*Pandion halieatus*	Summer

[a] Puchy and Marshall (1993).
[b] Oregon state-listed as threatened species.

Table 2-5. Bird Species Seasonally or Minimally Associated with Aquatic Habitat in the Study Area

Species	Scientific Name
Herbivores	
Aleutian Canada goose	*Branta canadensis leucopareia*
American wigeon	*Anas americana*
Canvasback	*Aythya valisineria*
Gadwall	*Anas strepera*
Northern pintail	*Anas acuta*
Northern shoveler	*Anas clypeata*
Ring-necked duck	*Aythya collaris*
Tundra swan	*Cygnus columbianus*
Diving Carnivore and Omnivores	
American peregrine falcon[a]	*Falco peregrinus*
Barrow's goldeneye	*Bucephala islandica*
Bonaparte's gull	*Larus philadelphia*
Bufflehead	*Bucephala albeola*
Common goldeneye	*Bucephala clangula*
Common teal (green-winged teal)	*Anas carolinensis*
Eared grebe	*Podiceps nigricollis*
Greater scaup	*Aythya marila*
Harlequin duck[b]	*Histrionicus histrionicus*
Horned grebe	*Podiceps auritus*
Lesser scaup	*Aythya affinis*
Red-necked grebe[c]	*Podiceps grisegena*
Ruddy duck	*Oxyura jamaicensis*

Table 2-5. Bird Species Seasonally or Minimally Associated with Aquatic Habitat in the Study Area

Species	Scientific Name
Wilson's phalarope	*Phalaropus tricolor*
Ring-billed gull	*Larus delawarensis*
Sandhill crane	*Grus canadensis*
Tri-colored blackbird[b]	*Agelaius tricolor*
Sediment-Probing Invertivores and Omnivores	
California gull	*Larus californicus*
Dunlin	*Calidris alpine*
Greater yellowlegs	*Tringa melanoleuca*
Least sandpiper	*Calidris minutilla*
Lesser yellowlegs	*Tringa flavipes*
Long-billed curlew	*Numenius americanus*
Long-billed dowitcher	*Limnodromus scolopaceus*
Marbled godwit	*Limosa fedoa*
Mew gull	*Larus canus*
Semipalmated plover	*Charadrius semipalmatus*
Short-billed dowitcher	*Limnodromus griseus*
Solitary sandpiper	*Tringa solitaria*
Western sandpiper	*Calidris mauri*
Willet	*Catoptrophorus semipalmatus*
Piscivores	
American white pelican	*Pelecanus erythrorhynchos*
Black-crowned night heron	*Nycticorax nycticorax*
Caspian tern	*Sterna caspia*
Forster's tern	*Sterna forsteri*
Great egret	*Ardea alba*
Greater yellowlegs	*Tringa melanoleuca*
Western gull	*Larus occidentalis*
Common loon	*Gavia immer*
Pelagic cormorant	*Phalacrocorax pelagicus*
Western grebe	*Aechmophorus occidentalis*

Source: Csuti et al. (2001)

[a] Oregon state-listed as vulnerable.

[b] Federally listed as a species of concern.

[c] Listed as a critical species on the ODFW sensitive species list (Oregon Natural Heritage Information Center 2004).

ODFW – Oregon Department of Fish and Wildlife

2.2.3.1.1 Herbivorous Birds

Two common herbivores using the Study Area are Canada goose (*Branta canadensis*) and mallard (*Anas platyrhynchos*). Canada geese are common in the vicinity of the Study Area throughout the year (Puchy and Marshall 1993), some as year-round residents and others as overwintering visitors. Canada geese typically nest on the ground near open water, often in vegetated marshes (Csuti et al. 2001). These geese preferentially feed on the shoots of terrestrial and aquatic plants but will also eat aquatic invertebrates (Ehrlich et al. 1988). Mallards are also very common. Some mallards are present in the summer during breeding season, while others overwinter along the Willamette River (Puchy and Marshall 1993). Mallards are dabbling ducks that forage in open-water areas on aquatic plants and invertebrates (Csuti et al. 2001; Drilling et al. 2002) and nest on the ground near water (Ehrlich et al. 1988). During breeding season, mallards also consume invertebrates to meet metabolic requirements.

2.2.3.1.2 Carnivorous and Omnivorous Birds

Seven diving omnivorous and carnivorous bird species may be resident in the LWR (Table 2-4). The resident omnivores are cinnamon teal (*Anas cyanoptera*), wood duck (*Aix sponsa*), and American coot (*Fulica americana).* A fairly common breeding duck, the cinnamon teal is found throughout Oregon (Puchy and Marshall 1993); they typically overwinter south of Oregon, but some remain in western Oregon throughout the year (Csuti et al. 2001). As a dabbling duck, cinnamon teal forages in vegetated shoreline areas on a mix of aquatic plants and aquatic invertebrates, such as molluscs, midges, and larvae (Ehrlich et al. 1988). They typically nest on the ground in marshes, meadows, or other low-vegetation habitats near open water (Puchy and Marshall 1993).

The wood duck is relatively uncommon in the Willamette Valley, but some are year-round residents (Puchy and Marshall 1993). A perching duck, the wood duck prefers to nest in woodland habitats, often in trees and snags near water (Ehrlich et al. 1988). They feed in shallow water, mainly on seeds and aquatic plants, but are also known to eat aquatic insects (Csuti et al. 2001).

The American coot is locally abundant in the Willamette Valley, usually as a year-round resident (Puchy and Marshall 1993). Their floating nests are usually built under vegetative cover; marshes are a common nesting location (Ehrlich et al. 1988). The American coot is a diving duck and feeds mostly on aquatic plants, occasionally (especially when young) consuming aquatic insects, crustaceans, worms, and other invertebrates (Csuti et al. 2001).

Four species of diving carnivores may use the Study Area: the American dipper (*Cinclus mexicanus*), common merganser (*Mergus merganser*), hooded merganser (*Lophodytes cucullatus*), and pied-billed grebe (*Podilymbus podiceps*). Both merganser species are locally common breeders in the Willamette Valley, with some being year-round residents (Csuti et al. 2001). Mergansers prefer to nest in tree cavities near open water (Ehrlich et al. 1988; Kitchen and Hunt 1969). Common mergansers feed primarily by diving for whatever small fish are abundant, but they will also eat aquatic invertebrates, especially as hatchlings (Csuti et al. 2001). Hooded mergansers are smaller and eat more aquatic invertebrates (e.g., crustaceans, aquatic insects) than do common mergansers (Csuti et al. 2001); they are also known to feed on small fish (Bendell and McNicol 1995).

American dippers are mostly year-round residents (Puchy and Marshall 1993), prefering smaller, fast-flowing streams but occasionally found along larger rivers, ponds, and lakes (Csuti et al. 2001). They usually nest in stream banks or cliffs along flowing water, and feed mostly on aquatic insects and larvae (Ehrlich et al. 1988).

Pied-billed grebes range from uncommon to common breeders in the Willamette Valley, but many individuals overwinter in the area (Csuti et al. 2001; Puchy and Marshall 1993). They forage in open water for aquatic insects, crayfish, small fish, and other aquatic invertebrates and typically build floating nests in quiet waters, usually under the cover of emergent vegetation (Csuti et al. 2001; Ehrlich et al. 1988).

2.2.3.1.3 Sediment-Probing Invertivorous and Omnivorous Birds

Sediment-probing birds consume mostly infaunal and epifaunal invertebrates and may incidentally ingest more sediment than birds in other feeding guilds. Accordingly, exposure of sediment-probing birds to sediment contamination is higher than exposure of other groups, such as herbivorous birds and dabbling ducks. Sediment-probing species that breed in the vicinity of the Study Area include spotted sandpiper (*Actitis macularia*), sora (*Porzana carolina*), killdeer (*Charadrius vociferus*), Virginia rail (*Rallus limicola*), and common snipe (*Gallinago gallinago*). Twenty-eight shorebird beaches have been identified in the LWR (Saban and Andersen 2004).

Spotted sandpipers are locally common breeders in the Willamette Valley, and some are present year-round (Puchy and Marshall 1993). They build ground nests amid herbaceous vegetation and usually feed nearby along shallow gravel shorelines and beaches (Ehrlich et al. 1988). Typically eating insects and benthic invertebrates such as crustaceans, molluscs, and worms (Csuti et al. 2001), some sandpipers are known to ingest relatively large amounts of sediment while feeding (Beyer et al. 1994).

Soras are common breeders along the Willamette but typically do not overwinter (Puchy and Marshall 1993). They build floating nests in emergent vegetation along lakes and streams and are more omnivorous than the other species in this guild (Csuti et al. 2001). They feed on seeds, insects, and aquatic invertebrates (Ehrlich et al. 1988).

Killdeer are locally abundant in the Willamette Valley and most are year-round residents (Puchy and Marshall 1993). They feed mostly on flying insects, such as beetles, dragonflies, and grasshoppers, but may also eat crayfish and other benthic invertebrates (Csuti et al. 2001). Killdeer nest on the ground in a variety of habitats near open water (Ehrlich et al. 1988).

The Virginia rail and the common snipe are also common breeders in the Willamette Valley, and some are year-round residents (Puchy and Marshall 1993). Both species nest on the ground. Virginia rails usually nest in marshes with cover from emergent vegetation, and common snipe make their nests in grassy areas near water (Ehrlich et al. 1988). The diet of Virginia rails consists of insects, aquatic invertebrates, and some seeds (Csuti et al. 2001). Common snipe feed by probing into saturated soils in wetlands and very shallow water, feeding largely on insect larvae and worms.

What Type of Habitat Do Spotted Sandpipers Use?

Because they eat invertebrates that live in the sediment, spotted sandpiper could be more exposed than other types of birds to sediment contaminants. They occupy habitat along intertidal beach areas. Using their specialized beaks, sandpipers probe in beach sediments for invertebrate prey (including epibenthic and infaunal invertebrates). Herbaceous vegetation is required for their ground nests (Ehrlich et al. 1988). Sandpipers incidentally ingest sediment as they probe. No published data are available on the home ranges of spotted sandpipers; however, data for other sandpiper species (i.e., buff breasted, upland, stilt, purple, and western sandpipers) indicate a relatively small home range of approximately 0.5 to 5 mile (Butler et al. 2002; Houston and Bowen 2001; Jehl 1973; Klima and Jehl 1998; Lanctot and Laredo 1994; Pierce 1993; Warnock and Takekawa 1995).

Spotted sandpipers are locally common breeders in the Willamette Valley, and some are present year-round (Puchy and Marshall 1993). Spotted sandpipers have been observed in the Study Area. In a June 2004 shorebird reconnaissance survey, 28 shorebird beaches, representing potential sandpiper habitat, were identified (Saban and Andersen 2004). These beaches were characterized by sandy stretches of intertidal sediment. At some, upland vegetation was present. The longest continuous beaches characterized by gentle slopes were located in the downstream portion of the Study Area (including the Willamette River portion off Sauvie Island and at the mouth of Multnomah Channel). Spotted sandpipers were observed foraging at this downstream portion during the reconnaissance survey. Smaller beaches were observed at interspersed locations throughout the Study Area.

2.2.3.1.4 Piscivorous Birds

Piscivorous birds that either reside year-round, or migrate but breed and rear their young in the vicinity of the Study Area include osprey (*Pandion haliaetus*), double-crested cormorant (*Phalacrocorax auritus*), herons (both the green heron [*Butorides virescens*] and great blue heron [*Ardea herodias*]), American bittern (*Botaurus lentiginosus*), bald eagle (*Haliaeetus leucocephalus*), and belted kingfisher (*Ceryle alcyon)*. Consumption of secondary aquatic consumers, such as invertivorous fish, gives piscivorous birds the highest potential exposure to biomagnifying contaminants. Although few species from this guild feed solely on fish, fish make up the majority of the diet for all eight piscivorous species discussed below.

Osprey tend to feed solely on fish. Osprey nests have been observed in or close to the Study Area, indicating that sensitive developmental life stages of this species are potentially exposed to contaminants in the Study Area. They generally feed on slow-moving prey that swim near the water surface (Csuti et al. 2001). Ospreys are present from March until September, with several breeding pairs nesting in or near the Study Area (Henny et al. 2003). Each fall, osprey migrate to western Mexico and Central America (Martell et al. 2001). Nesting success and population growth throughout the Willamette River system, including the Study Area, increased from 1993 to 2001(Henny et al. 2009). In 1993, one osprey nest was observed between RM 0 and RM 26; in 2001, 10 nests were observed, including several within the Study Area boundaries (Henny et al. 2009).

What is the Status of LWR Osprey Populations?

The osprey is of interest because it is a predatory species whose population is rebounding after years of decline. Osprey occupy a unique ecological niche and have been observed nesting and foraging throughout the Willamette River and its tributaries. Osprey feed almost exclusively on fish and have relatively small home ranges. The nesting success and population growth have been monitored in recent years. Osprey populations in the Willamette River and lower Santiam River (a major tributary entering south of Salem) increased at an average annual rate of 13.7% from 1993 until 2001 (Henny et al. 2009).

Similar trends apply in the LWR. Between RM 0 and RM 26, the number of osprey nests increased from one in 1993 to 10 in 2001. The productivity of osprey in this section of the Willamette River in 2001 (1.75 young per all types of nests [occupied, active, and successful]) is similar to the productivity of osprey that Henny et al. (2009) reported for upstream sections of the Willamette River (average of 1.77 young per active nest in the Upper River and Santiam River sections combined), and well above the 0.7 and 0.8 young per nest reported as the minimum required to maintain stable bald eagle and osprey populations (Henny et al. 2009; Wiemeyer et al. 1984).

These data indicate that the osprey nesting population in the LWR (including the Study Area) has increased in recent years and that the productivity is above that necessary to maintain a stable population.

Belted kingfishers also tend to feed solely on fish. They are common, permanent residents throughout most of Oregon, except where open water is generally absent (Marshall et al. 2003). They usually nest in horizontal burrows dug into sandy stream and river banks (Ehrlich et al. 1988). Kingfishers feed wherever they can find small fish (7.5 to 10 cm); they may also eat crayfish, amphibians, and insects (Csuti et al. 2001).

American bittern are uncommon in the Willamette Valley (Marshall et al. 2003). They have a more varied diet than most other species in this guild, feeding on fish, amphibians, crayfish, and insects (Csuti et al. 2001). American bitterns build ground nests amid emergent vegetation, usually in marshes (Ehrlich et al. 1988).

Double-crested cormorants are common breeding birds along the coast and the lower Columbia River, and it is possible that some breed in the vicinity of the LWR (Csuti et al. 2001; Puchy and Marshall 1993). They are present year-round, and many overwinter in the area (Puchy and Marshall 1993). Double-crested cormorants nest in cliffs, trees, and marshes near open water (Csuti et al. 2001). They feed mostly on fish by diving in relatively deep water; they also occasionally feed on aquatic invertebrates, such as crayfish and molluscs (Ehrlich et al. 1988).

The green heron (also called the green-backed heron) is an uncommon year-round resident in the LWR (Puchy and Marshall 1993). They usually nest in trees in riparian woodlands, often in willows (Csuti et al. 2001). Green herons also have a varied diet consisting mainly of small fish and aquatic invertebrates such as crustaceans and snails. They also consume frogs and terrestrial invertebrates (Ehrlich et al. 1988).

The great blue heron is more common and widespread than the green heron and is a year-round resident in the LWR (Puchy and Marshall 1993). Ross Island is the site of an active rookery containing up to 30 nests. They are colonial nesters and usually build their nests in trees or other structures near water. They can use many different habitats and often travel great distances to forage for food (Csuti et al. 2001). Great blue heron feed mainly on fish, but can also consume crustaceans, amphibians, and some upland vertebrates (Ehrlich et al. 1988).

Bald eagles are known to nest throughout the Willamette River (Isaacs and Anthony 2001). The closest known nest to the Study Area is on Ross Island at RM 15. Two old nests are located on Sauvie Island at RM 0 to RM 3 (Isaacs and Anthony 2001). Eagles are year-round residents in western Oregon. In addition, some eagles from farther north overwinter in the area (Puchy and Marshall 1993). Bald eagles nest in treetops or cliffs near large bodies of water (Csuti et al. 2001). Bald eagles feed mainly on fish but, as opportunists, will scavenge on mammals and birds (Ehrlich et al. 1988). The bald eagle is listed as threatened by the State of Oregon and is protected by the federal Bald and Golden Eagle Protection Act (16 USC 668). The bald eagle is also protected under the Migratory Bird Treaty Act (16 USC 703-712).

2.2.3.2 Mammals

Aquatic and semi-aquatic mammals that potentially use the LWR are listed in Table 2-6. These species include beaver (*Castor canadensis*), muskrat (*Ondatra zibethicus*), raccoon (*Procyon lotor*), river otter (*Lutra canadensis*), mink (*Mustela vison*), nutria (*Myocastor coypus*), and California sea lion (*Zalophus californianus*). Nutria were introduced to the area and are considered a nuisance species.

Table 2-6. Mammals Potentially Using the Lower Willamette River

Species	Scientific Name
Beaver	*Castor canadensis*
California sea lion	*Zalophus californianus*
Mink	*Mustela vision*
Muskrat	*Ondatra zibethicus*
Nutria	*Myocastor coypus*
Raccoon	*Procyon lotor*
River otter	*Lutra canadensis*

Source: Csuti et al. (2001)

Potential foraging areas for beaver, muskrat, raccoon, river otter, mink, and nutria are present at many of the habitat sites identified as part of the Adolfson et al. (2000) natural resource inventory. Beaver and nutria are herbivores, although nutria may occasionally eat molluscs. Muskrats are aquatic mammals that dig burrows in banks and feed on vegetation, but may also consume crayfish, fish, turtles, snails, and salamanders (Csuti et al. 2001). Mink and river otter feed on fish, frogs, crayfish, molluscs, small mammals, and small birds (Csuti et al. 2001). Raccoons are omnivores that ingest significant amounts of vegetation (fruits, berries, nuts, and seeds) along with a broad range of other food items (small mammals, fish, amphibians, birds, aquatic invertebrates) and may obtain a significant portion of their food from sources other than the LWR (Csuti et al. 2001).

California sea lions may use the Study Area, primarily from March to mid-May, to forage on runs of spring Chinook and summer and winter steelhead (Foster and Boatner 2002). California sea lions are protected under the Marine Mammals Act; however, they are considered a nuisance in the LWR because they prey on salmonids. They are known to congregate at the Willamette Falls fish ladder and may migrate through Portland Harbor en route to their preferred feeding areas upstream.

2.2.4 Amphibians and Reptiles

There is a paucity of scientific information on the occurrence of amphibians and reptiles in the LWR. However, conditions within the LWR provide limited suitable habitat for these species. Table 2-7 lists the amphibians and reptiles that could be present in or near the Study Area. Of the species listed in Table 2-7, one amphibian species (northern red-legged frog) and two reptile species (painted turtle and western pond turtle) have special status.[20] The species observed during the LWG sampling activities are indicated.

Table 2-7. Amphibians and Reptiles Potentially Present Within the Study Area

Common Name	Scientific Name
Bullfrog	*Rana catesbeiana*
Common garter snake	*Thamnophis sirtalis*
Long-toed salamander	*Ambystoma macrodactylum*
Northern red-legged frog[a, b]	*Rana aurora*
Northwestern garter snake	*Thamnophis ordinoides*
Pacific tree frog[b]	*Pseudacris regilla*
Painted turtle[c]	*Chrysemys picta*
Western pond turtle[c]	*Actinemys marmorata*

[a] Federally listed as a species of concern and Oregon state-listed as a vulnerable species on the ODFW sensitive species list (ODFW 2005).
[b] Identified during LWG sampling activities.
[c] Oregon state-listed as a critical species on the ODFW sensitive species list (ODFW 2005).

[20] Special-status species include federal and state proposed and candidate species.

LWG – Lower Willamette Group
ODFW – Oregon Department of Fish and Wildlife

Most of the native amphibians (e.g., long-toed salamander, northern red-legged frog, and Pacific tree frog) that may be found in the Study Area prefer undisturbed areas offering ephemeral wetlands with emergent vegetation and shallow waters. They are preyed upon by the more ubiquitous and aggressive bullfrogs, which are invasive to the Pacific Northwest and have few predators.

In the LWR, painted turtles may be found in sloughs and ponds that provide shallow, quiescent aquatic areas with open banks and abundant plant growth. The most frequently encountered reptiles in the Willamette Valley are the common and northwestern garter snakes. Both species prefer wet vegetated terrestrial habitats, where they may be found lying under rocks, wood, and grasses. Roadside ditches or embankments may also provide suitable habitat for either species.

An amphibian and reptile reconnaissance survey was conducted to confirm the presence of amphibians within the Study Area. Likely amphibian habitat was identified based on known bank conditions from prior field efforts, information from the Willamette River Natural Resource Inventory (City of Portland 2008), and field observations. Multiple sites within the initial Study Area (between RM 3.5 and RM 9.2) were visited over a 3-day period in June 2002; all representative bank habitats were visited at least twice. The survey confirmed the presence of northern red-legged frogs and Pacific tree frogs (Integral et al. 2004a) (Figure 2-10).

Figure 2-10. Red-Legged Frog Identified During the 2002 Amphibian/Reptile Reconnaissance Survey

Results of the survey are presented in Table 2-8 and in Map 2-5. Low-sloping beaches and steeper riprapped or rocky banks were identified as potential amphibian habitat areas in the LWR (Integral et al. 2004a) (Map 2-5). Although terrestrial habitat requirements for reptiles may be available near the Study Area, reptiles were not observed during the 2002 survey.

Table 2-8. Results of the 2002 Amphibian Reconnaissance Survey

Location	Reconnaissance Survey Results
At mouth of Multnomah Channel	Observed two northern red-legged frogs
International Slip	Observed unidentified egg mass
RM 3.5 (west bank)	No amphibians or reptiles observed
Terminal 4/Slip 1	Observed unidentified egg mass
Terminal 4/Slip3	Unidentifiable frog call noted
Upstream from St. John's Bridge (both west and east bank) between RM 6 and RM 8	No amphibians or reptiles observed
Willamette Cove	No amphibians or reptiles observed
Saltzman Creek, at approximately RM 7.7 (west bank)	No amphibians or reptiles observed
RM 8.5 (west bank)	No amphibians or reptiles observed
Swan Island Lagoon	Pacific tree frog call noted

Source:, Integral et al. (2004a)
RM – river mile

2.2.5 Aquatic Plants

Aquatic plant communities are used by ecological receptors for nesting, breeding, and refuge. Aquatic plant communities also provide food for herbivores and play a role in the cycling of nutrients. Contaminants in the ecosystem may affect individual plants, plant communities, and higher-trophic-level fauna that consume, either directly or indirectly, contaminant residues that might be in the plants. High turbidity, riprap, and other bank modifications prevent the widespread development of dense submerged and emergent plant communities along the riverbanks of the Study Area.

To date, no comprehensive or semi-quantitative vegetation surveys have been conducted specifically within the Study Area to quantify and describe the plant communities. However, two qualitative plant community surveys have been conducted in the LWR (Adolfson et al. 2000; Integral et al. 2004a); species observed during those surveys are listed in Table 2-9. Potential aquatic plant habitats were characterized as part of the aquatic plant reconnaissance survey conducted in 2002 (Integral et al. 2004a), which included the identification of submerged and emergent aquatic plant species throughout the Study Area (Map 2-5). Twenty-six plant species were identified at the Study Area during this survey, most of which were obligate and facultative wetland plant species. Half of the plant species identified are exotic to the LWR.

Table 2-9. Plant Species of the LWR

Common Name	Scientific Name	Wetland Indicator Status[a]
Alfalfa[b]	*Medicago falcata* L.	NA
Bird's foot trefoil[b]	*Lotus corniculatus*	FAC
Black cottonwood	*Populus balsamifera* var. *trichocarpa*	FAC
Bradshaw's lomatium[c]	*Lomatium bradshawii*	FACW
Canada thistle[b]	*Cirsium arvense*	FAC
Cattail	*Typha latifolia*	OBL
Common wetland asters[d]	*Aster* spp.	NA
Columbia River willow	*Salix fluviatilis*	OBL
Common groundsel[b]	*Senecio vulgaris* L.	FACU
Common horsetail	*Equisetum arvense*	FAC
Common rush	*Juncus effuses*	FACW
Common velvet grass[b]	*Holcus lanatus* L.	FAC
Douglas spiraea	*Spiraea douglasii*	FACW
Himalayan blackberry[b]	*Rubus discolor*	FACU
Howell's bentgrass[e]	*Agrostis howellii*	NA
Hitchcock's blue-eyed grass[e]	*Sisyrinchium hitchcockii*	NA
Howellia[f]	*Howellia aquatilis*	OBL
Nelson's sidalcea[g]	*Sidalcea nelsonia*	FAC
Oregon ash	*Fraxinus latifolia* L.	FACW
Oxeye daisy[b]	*Leucanthemum vulgare*	NA
Pacific willow	*Salix lucida*	FACW
Peacock larkspur[e]	*Delphinium pavonaceum*	NA
Piper's willow	*Salix piperii*	FACW
Purple loosestrife[b]	*Lythrum salicaria*	FACW
Red osier dogwood	*Cornus sericea*	FACW
Reed canary grass[b]	*Phalaris arundinacea*	FACW
Scots broom[b]	*Cytisus scoparius*	NA
Sedge	*Carex* spp.	Varies
Smartweed	*Polygonum* spp.	Varies
Snowberry	*Symphoricarpos albus*	FACU
St. John's wort[b]	*Hypericum perforatum*	NA
Sweet clover[b]	*Melilotus alba* Mill.	NA
Teasel[b]	*Dipsacus fullonum*	NA
Wapato	*Sagittaria latifolia*	OBL
Water moss	*Fontinalis antipyretica*	NA
Wayside aster[e]	*Aster vialis*	NA
White-topped aster[e]	*Aster curtus*	NA

Table 2-9. Plant Species of the LWR

Common Name	Scientific Name	Wetland Indicator Status[a]
Willamette daisy[c]	*Erigeron decumbens*	NA
Yellow water-flag iris[b]	*Iris pseudacorus*	OBL

Sources: Adolfson et al. (2000), Integral et al. (2004a)

[a] Indicator status refers to a species of fidelity to wetland environments in the Pacific Northwest (Reed 1996; Cooke 1997) and are defined as follows:

OBL – obligate; high probability of occurrence in regional wetlands

FAC – facultative; moderate probability of occurrence in regional wetlands

FACU – facultative upland; low to moderate probability of occurrence in regional wetlands

FACW – facultative wet; moderate to high probability of occurrence in regional wetlands

NA – status not available

Varies – status varies by species

[b] Exotic species.

[c] Listed as endangered (state and federal).

[d] The aster species were garden varieties, not *Aster curtus* or *Aster vialis.*

[e] Federal species of concern.

[f] Listed as threatened (federal).

[g] Listed as threatened (state and federal).

LWR – Lower Willamette River

In the Adolfson et al. (2000) qualitative survey, fish and wildlife habitats along the shoreline of the LWR were inventoried and 10 distinct habitat classes were identified: bottomland forest, foothill savanna, conifer forest, meadow, shrub, emergent wetland, beach, rock outcrop, open water, and unvegetated/disturbed. Although all of these habitats are present in the vicinity of the LWR, the bottomland forest, emergent wetlands, beach, and open-water habitat classes are the most common, occurring along the shoreline within the Study Area. Historically, bottomland forests were an important component of the Willamette River floodplain system (Sedell and Froggatt 1984), but they have been reduced to a portion of their former extent (Adolfson et al. 2000). A few remnant patches of emergent wetlands are found adjacent to the shoreline. Beach habitats throughout the LWR typically consist of narrow shoreline areas with sand substrate dominated by various annual grasses and perennial shrubs. Open-water habitats exist throughout the LWR in tributaries, sloughs, and side channels, often dominated by aquatic species from bottomland forest, emergent wetland, and scrub/shrub plant communities (Adolfson et al. 2000). Figures 2-11, 2-12, and 2-13 are examples of the vegetation present in the LWR.

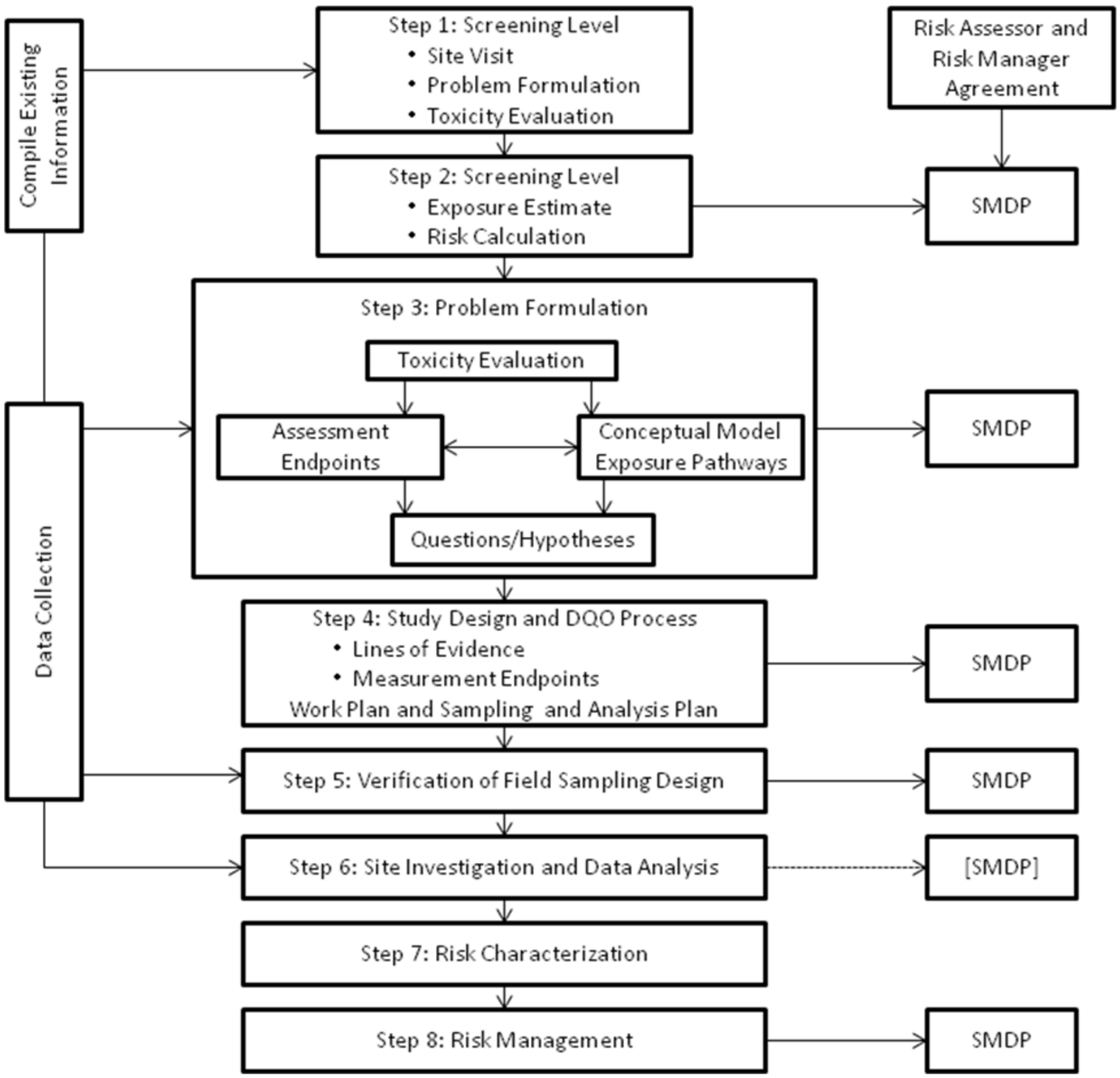
Figure 2-11. Wetland and Upland Vegetation in the LWR

Figure 2-12. Upland Vegetation along the LWR (St. John's Wort, Thistle, Bird's Foot Trefoil)

Figure 2-13. Backwater Marsh Vegetation

3.0 BERA PROBLEM FORMULATION METHODS

This section presents the problem formulation for this BERA. Per EPA guidance for conducting BERAs under the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) (EPA 1997), the problem formulation is developed in Step 3 of the eight-step risk assessment process and identifies specific factors to be addressed in the ERA. Figure 3-1 presents the eight-step process for ERA.



SMDP – scientific/management decision point
Source: EPA (1997)

Figure 3-1. BERA 8-Step Process for Superfund

The problem formulation step of the ERA process includes the following components:

- Refinement of preliminary contaminants of ecological concern (i.e., COPCs)
- Further characterization of the ecological effects of COPCs at the site
- Review and refinement of information on fate and transport, complete exposure pathways, and ecosystems potentially at risk
- Selection of assessment and measurement endpoints
- Development of a CSM and risk questions
- Development of an analysis plan

Elements of the problem formulation were provided as part of Appendix B of the programmatic work plan (Integral et al. 2004a), the draft Ecological PRE (Windward 2005a), and Appendix G of the Comprehensive Round 2 Report (Integral et al. 2007). Discussions between the LWG and EPA led to EPA's development of a problem formulation document. Its intent was to provide a framework for completing the BERA that both addressed implementation of the six steps above and accounted for data and information collected to date. Detailed information on EPA's Problem Formulation for the BERA is presented in Attachment 2.[21] Selected elements are summarized below:

- SLERA and refined screening process for identifying COPCs (Section 3.1)
- Refined CSM (Section 3.2)
- Refined assessment and measurement endpoints (Section 3.3)
- Analysis plan outlining the methods for conducting the BERA (Section 3.4)

3.1 IDENTIFICATION OF ECOLOGICAL COPCS

As part of the SLERA and refined screen conducted according to the procedures outlined in EPA's Problem Formulation (Attachment 2), the complete BERA dataset (i.e., Rounds 1, 2, and 3 data) was screened to compile the BERA COPCs. The BERA dataset is summarized in Section 4. The SLERA, refined screen, and ecological COPCs identified for each ecological receptor group are summarized in Section 5 and detailed in Attachment 5.

[21] EPA's Problem Formulation document is provided as submitted to the LWG. However, footnotes have been provided to indicate where additional agreements between the LWG and EPA further modified the Problem Formulation or where clarification was needed as to how the Problem Formulation was implemented in the BERA.

3.2 REFINED CONCEPTUAL SITE MODEL

The CSM for the BERA is one of the four primary products of Step 3 of the eight-step ERA process (EPA 1997); the others are assessment endpoints, exposure pathways, and risk questions. A CSM describes the relationship between environmental conditions (including those resulting from human activities) and ecological receptors at a site, to the degree that it is known. The BERA CSM describes relationships between contaminants and the resources potentially affected by releases of contaminants from the Study Area. By describing relationships between contaminant sources, transport and exposure pathways, and the ecological receptors in the Study Area, the CSM provides a framework for postulating potential effects of site contaminants on ecological receptors, which, when made specific, become the risk questions and testable hypotheses for the BERA.

Consistent with EPA Superfund guidance (EPA 1997), the ecological receptors selected for assessment in the Portland Harbor BERA were identified from among the organisms using the site by considering the following criteria:

- Societal and cultural significance (i.e., species valued by society or that have special regulatory status – threatened or endangered)
- Ecological significance (i.e., species that serve a unique ecological function)
- Potential level of exposure to likely COPCs at the site (i.e., site usage)
- Relative ability to bioaccumulate likely COPCs at the site
- Sensitivity to likely COPCs at the site
- Availability of sufficient data to assess risks to specific organisms

Based on these criteria, as presented in the BERA CSM, the following ecological receptors were selected for assessment:

- **Benthic invertebrate community**[22] **–** benthic macroinvertebrate community, bivalves (clams), and decapods (e.g., crayfish)
- **Omnivorous fish populations –** largescale sucker, carp, and pre-breeding white sturgeon
- **Invertivorous fish populations –** sculpin, peamouth, and juvenile Chinook salmon[23]
- **Piscivorous fish populations –** smallmouth bass and northern pikeminnow
- **Detritivorous fish individuals –** Pacific lamprey ammocoetes

[22] Clams and crayfish are members of the benthic macroinvertebrate community, but were evaluated separately to provide a population level assessment.

[23] Juvenile Chinook salmon were evaluated at the organism level; all other invertivorous fish receptor species selected were evaluated at the population level.

- **Sediment-probing invertivorous bird populations –** spotted sandpiper
- **Omnivorous bird populations**[24] **–** hooded merganser
- **Piscivorous bird populations –** osprey, bald eagle
- **Aquatic-dependent carnivorous mammal populations –** mink, river otter
- **Amphibian and reptile populations –** amphibians (e.g., including frog and salamander species)
- **Aquatic plant community –** aquatic plant community (e.g., including phytoplankton, periphyton, macrophyte species)

The assessment endpoints for all receptors are based on the protection and maintenance of their populations and the communities in which they live, except that the health of threatened or endangered species is to be protected at the level of the individual organism. Based on EPA's Problem Formulation (Attachment 2), the assessment endpoints were expressed as the survival, growth, and reproduction of organisms in each receptor group.

[24] Belted kingfisher was evaluated in the uncertainty assessment, as previously agreed to by EPA and the LWG. The belted kingfisher ingests a considerable amount of fish, is present year-round, and consumes a variety of prey. Belted kingfisher was included in the uncertainty evaluation to confirm that the evaluations performed on bald eagle, osprey, and merganser are protective of the belted kingfisher.

Uncertainty in Extrapolating from Organism-Level to Population-Level Effects

One of the fundamental challenges in ERA involves establishing the link between effects to an organism and effects to a population or community. Organism-level effects are expressed as an individual's survival, growth, or reproduction, whereas population-level effects influence the population as a whole, for example in numbers, density, or rate of growth (or decline). EPA (1999a) guidance states that assessment endpoints and measures should be associated with sustaining the ecological structure and function of populations and communities rather than individual organisms, unless individuals warrant additional protection in specific cases. Despite the goal of protecting higher-level attributes such as populations, Superfund ERAs typically do not address effects at these higher levels, but instead gather data on individuals in order to predict effects on populations or communities, an approach justified by experience, policy, and judicial decisions (EPA 2004b). According to EPA guidance (EPA 1999a), concentrations expected to protect populations can be extrapolated from those that protect individual organisms against adverse effects using a lines-of-evidence approach including site-specific toxicity tests, bioaccumulation models, and species diversity studies.

The survival, growth, and reproductive benchmark concentrations used in ERAs are typically derived from controlled laboratory experiments in which the effect level is based on a concentration at which the number of test organisms experiencing an adverse effect is significantly greater statistically than the number of control organisms experiencing the same effect. A high degree of uncertainty is associated with the extrapolation from these organism-level attributes to larger-scale influences on the population.

For example, a toxicity benchmark may represent reduced growth in a statistically significant number of fish in a laboratory population, but this same growth effect on that fish species in the field does not necessarily lead to reduced viability of the fish population. Population dynamics are complex, involving multiple feedback loops and compensatory or depensatory mechanisms. In addition, toxicants can alter physiologic, behavioral, and density dynamics by a variety of mechanisms, and it is difficult to predict how the interactions of these responses will result in effects at a population level (Emlen and Springman 2007). A promiscuous or highly polygynous species might easily withstand a stressor that kills of 90% of its population, whereas an immunocompromised population might be adversely affected in the presence of an otherwise harmless pathogen (Emlen and Springman 2007). In a review of 41 toxicity studies that observed both individual traits (i.e., survival, growth, or fecundity) and population growth rates, Forbes and Calow (2002) found that in 81.5% of the toxicity results, the percentage change in population growth rate was less than the percentage change in the most sensitive of the individual traits. This review (2002) indicates that it may be overly conservative to assume that any level of increased mortality or decreased fecundity or growth of organisms will lead to adverse effects on a population or community. In summary, a chemical concentration resulting in organism-level effects might or might not affect a population, and vice versa, and any translation from individual to population must be undertaken with caution (Emlen and Springman 2007).

In light of current standard ERA practice, the uncertainties discussed above, and the prevalence of data related to organism-level attributes, a combination of numerical estimates and best professional judgment should be used to interpret data on ecological relevance. EPA (1997) recommends that additional information be supplied in risk assessments to provide context for the numerical risk estimates; this information may include spatial extent, magnitude of organism-level threshold exceedance, and quality and relevance of the organism-level effect threshold as a predictor of a population- or community-level effect.

Risks to ecological receptors were assessed using the following LOEs:

- The surface water LOE, wherein surface water chemistry data were compared with water TRVs
- The TZW LOE, wherein shallow (< 38 cm) TZW chemistry data were compared with water TRVs
- The site-specific sediment toxicity LOE, wherein site-specific sediment toxicity was measured in laboratory toxicity tests
- The predicted sediment toxicity LOE, wherein sediment chemistry were compared with to site-specific or generic SQGs

- The tissue-residue LOE, wherein site-specific tissue chemistry (both measured and predicted) were compared with a tissue-residue TRV
- The dietary-dose LOE, wherein estimated site-specific dietary doses were compared with a dietary TRV
- The fish condition LOE wherein site-specific fish condition data were compared with literature data on fish condition from locations with elevated concentrations of polycyclic aromatic hydrocarbons (PAHs)

The surface water and TZW LOEs were used to characterize risks to aquatic plants and amphibians. Risks to benthic invertebrates were assessed using several LOEs: site-specific and predicted sediment toxicity, surface water, TZW, and tissue residues (both measured and predicted). Risks to fishes were assessed using four LOEs: surface water, tissue residue (both measured and predicted), dietary, and fish condition. In general, only the dietary LOE, was used to characterize risks to birds and mammals. Risks to osprey and bald eagle, however, were also assessed using the tissue-residue LOE.

The ecological CSM illustrates the pathways that contaminants may follow from primary sources to the ecological receptors through potential exposure pathways. The exposure pathways were classified as one of four categories for each receptor:

- Complete and significant – Exposure pathway is complete and expected to provide the greatest potential for exposure.
- Complete and significance unknown – Exposure pathway is complete but the proportion of a receptor's contaminant dose relative to doses of the same contaminant via other pathways is unknown; the receptor could receive a significant proportion of the contaminant dose when combined with other pathways or other contaminants.
- Complete and insignificant – Exposure pathway is complete but not likely to significantly contribute to a receptor's exposure.
- Incomplete – Receptors cannot be exposed via the pathway.

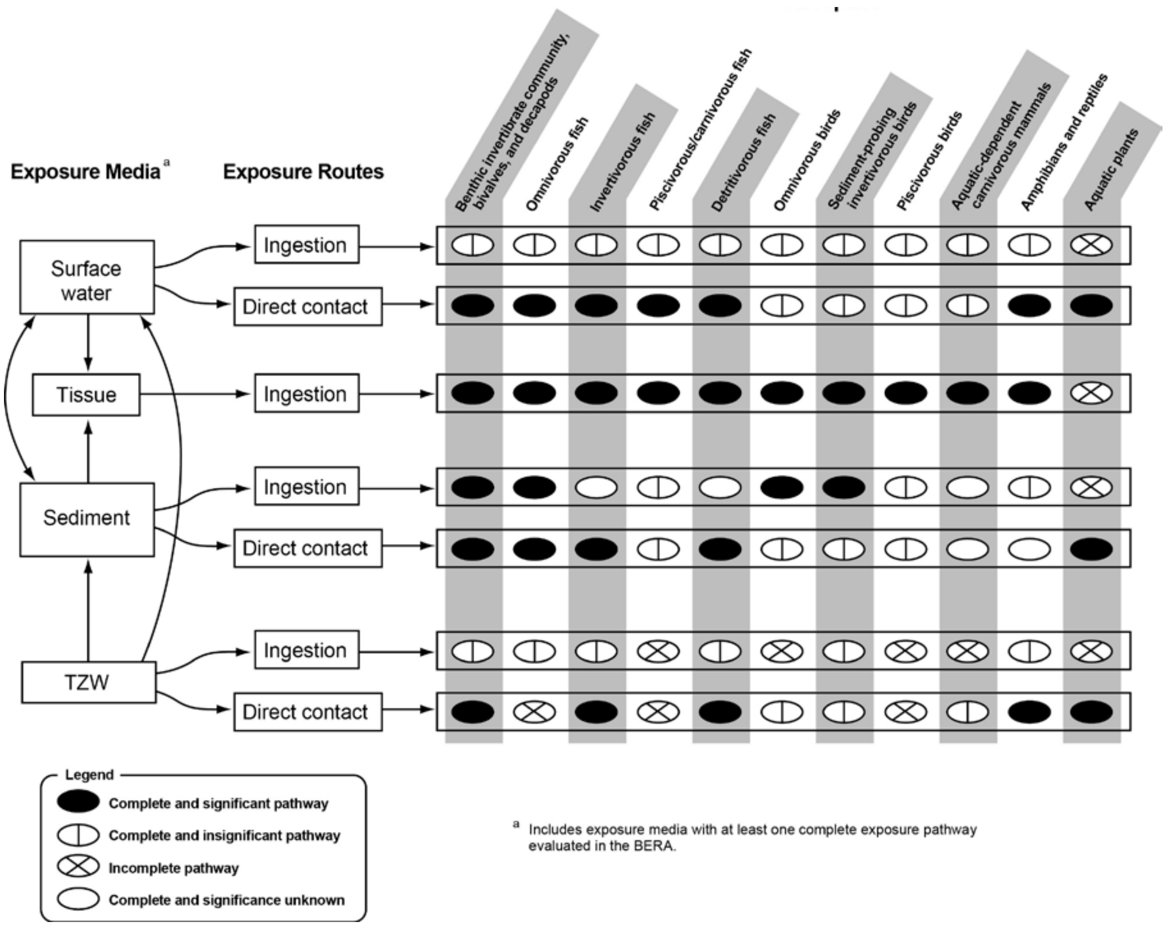
Complete and significant pathways were quantitatively assessed in this BERA. Pathways that were complete and significance unknown were qualitatively assessed to a level of certainty dependent on available toxicological studies and exposure data. Insignificant and incomplete pathways were not addressed further.

The refined ecological CSM (EPA 2008j), which was derived partly from previous ecological CSMs for the Study Area (Windward 2005a; Integral et al. 2004b; Integral et al. 2007), is presented in Attachment 2. The ecological CSM consists of three components:

- A diagram describing contaminant sources within the Study Area

- Illustrations describing the transport of contaminants from their sources to environmental media to ecological receptors
- Descriptions of the mechanisms by which ecological receptor groups are exposed to contaminants in site media

A simplified version of EPA's refined ecological CSM is presented in Figure 3-2. The only exposure media shown are those evaluated in the BERA, for the assessment endpoints and measurement endpoints identified by EPA (see Section 3.3).[25] The simplified CSM also presents tissue as a separate exposure medium; in the refined ecological CSM (Attachment 2), exposure to tissue is identified as a dietary exposure route under each abiotic exposure medium.



Note: The complete and detailed ecological CSM per EPA is provided in Attachment 2.

Figure 3-2. Simplified Ecological CSM

[25] The simplified CSM does not include receptors that were not evaluated in the BERA such as zooplankton or terrestrial plants in the riparian zone; however, they are shown the expanded CSM in Attachment 2.

3.3 ASSESSMENT ENDPOINTS AND MEASUREMENT ENDPOINTS

Table 3-1 presents the selected assessment endpoints, measurement endpoints, and LOEs that were evaluated for each ecological receptor . Assessment endpoints are characteristics of selected environmental receptors that are to be protected (EPA 1997). Questions and hypotheses to test suspected interactions between receptors and contaminants lead to the selection of measurement endpoints that quantify exposure to contaminants via pathways identified in the CSM and/or effects from that exposure. Each measurement endpoint is evaluated with one or more LOEs. An LOE is represented by the results of specific decision criteria or evaluations being applied to a set of exposure or effects data. An LOE is used alone or in combination with other LOEs to help address risk questions. Details on the assessment endpoints and measurement endpoints are presented in EPA's Problem Formulation (Attachment 2) and reproduced in Table 3-1.

Table 3-1. Assessment and Measurement Endpoints and Lines of Evidence for the Portland Harbor BERA

Assessment Endpoint[a]	Target Ecological Receptors	Measures of Effect and Exposure (Measurement Endpoints)	Lines of Evidence in Support of Measurement Endpoints
Benthic Invertebrates			
Survival, growth, and reproduction of benthic macroinvertebrates	Benthic macroinvertebrates (e.g., amphipods, isopods, bivalves, gastropods, oligochaetes, insects, decapods)	Survival and growth of laboratory-exposed invertebrates	Survival and biomass of *Chironomus dilutus* and *Hyalella azteca* exposed to site sediments compared with reference area sediments
		Bulk surface sediment contaminant concentrations	Concentrations in site sediment compared with site-specific sediment quality values (SQVs) derived from models predicting reduced survival or growth based on Portland Harbor surface sediment concentrations and toxicity reported for both *Hyalella* and *Chironomus* endpoints.
			Concentrations in site sediment compared with national consensus-based SQGs (PECs and related quotients), and effects-based SQGs (PELs and related quotients)
		Surface water contaminant concentrations	Concentrations in surface water compared with state WQS, national AWQC, or effects-based values derived from the literature that are protective of benthic macroinvertebrate survival, growth, and reproduction
		TZW contaminant concentrations	Concentrations in shallow TZW compared with state WQS, national AWQC, or effects-based values derived from the literature that are protective of benthic macroinvertebrate survival, growth, and reproduction
		Benthic macroinvertebrate tissue concentrations (modeled or measured in field-collected organisms or estimated in laboratory-exposed organisms)	Empirical (field-collected) whole-body concentrations of epibenthic organisms compared with tissue TRVs
			Steady-state estimates of laboratory-exposed whole-body concentrations in *Lumbriculus* compared with tissue TRVs
			Predicted (BSAF[b]) whole-body concentrations of *Lumbriculus* compared with tissue TRVs

Table 3-1. Assessment and Measurement Endpoints and Lines of Evidence for the Portland Harbor BERA

Assessment Endpoint[a]	Target Ecological Receptors	Measures of Effect and Exposure (Measurement Endpoints)	Lines of Evidence in Support of Measurement Endpoints
Survival, growth, and reproduction of bivalves	Clams, mussels	Bivalve tissue concentrations (modeled or measured in field-collected organisms or estimated in laboratory-exposed organisms)	Empirical (field-collected) whole-body concentrations in *Corbicula fluminea* and freshwater mussels compared with tissue TRVs Steady-state estimates of laboratory-exposed whole-body concentrations in *Corbicula fluminea* compared with tissue TRVs Predicted (BSAF[b]) whole-body concentrations in *Corbicula fluminea* compared with tissue TRVs
		Survival and growth of clams used in bioaccumulation testing	*Corbicula fluminea* survival and growth compared with control data from bioaccumulation tests
		Survival and growth of laboratory-exposed invertebrates[c]	Survival and biomass of *Chironomus dilutus* and *Hyalella azteca* exposed to site sediments, compared with reference sediments
		Surface water contaminant concentrations[c]	Concentrations in surface water compared with state WQS, national AWQC, or effects-based values derived from the literature that are protective of benthic macroinvertebrate survival, growth, and reproduction
		TZW contaminant concentrations[c]	Concentrations in shallow TZW compared with state WQS, national AWQC, or effects-based values derived from the literature that are protective of benthic macroinvertebrate survival, growth, and reproduction
		Bulk sediment contaminant concentrations[c]	Concentrations in site sediment compared with site-specific SQVs and national consensus-based SQGs (PECs and related quotients) and effects-based SQGs (PELs and related quotients)

Table 3-1. Assessment and Measurement Endpoints and Lines of Evidence for the Portland Harbor BERA

Assessment Endpoint[a]	Target Ecological Receptors	Measures of Effect and Exposure (Measurement Endpoints)	Lines of Evidence in Support of Measurement Endpoints
Survival, growth, and reproduction of decapods	Crayfish	Decapod tissue contaminant data (modeled or field-collected)	Empirical whole-body concentrations in crayfish compared with tissue TRVs
			Predicted (BSAF or FWM) whole-body concentrations in crayfish compared with tissue TRVs
		Bulk sediment contaminant concentrations[c]	Concentrations in site sediment compared with site-specific SQVs and national consensus-based SQGs (PECs and related quotients) and effects-based SQGs (PELs and related quotients)
		Surface water contaminant concentrations[c]	Concentrations in surface water compared with state WQS, national AWQC, or effects-based values derived from the literature that are protective of benthic macroinvertebrate survival, growth, and reproduction
		TZW contaminant concentrations[c]	Concentrations in shallow TZW compared with state WQS, national AWQC, or effects-based values derived from the literature that are protective of benthic macroinvertebrate survival, growth, and reproduction
Fish			
Survival, growth, and reproduction of omnivorous fish	Carp,[d, e] white sturgeon, largescale sucker[d, f, g]	Surface water contaminant concentrations[h]	Concentrations in surface water compared with state WQS, national AWQC, or effects-based values derived from the literature that are protective of fish survival, growth, and reproduction
		Fish tissue contaminant concentrations[i] (field-collected) from species-specific exposure areas	Empirical whole-body concentrations compared with tissue TRVs
		Species- or feeding-guild-specific dietary dose of contaminants based on prey and incidentally ingested sediment from species-specific exposure areas	Dietary dose (including incidental sediment ingestion) compared with dietary TRVs

Table 3-1. Assessment and Measurement Endpoints and Lines of Evidence for the Portland Harbor BERA

Assessment Endpoint[a]	Target Ecological Receptors	Measures of Effect and Exposure (Measurement Endpoints)	Lines of Evidence in Support of Measurement Endpoints
		Fish condition or prevalence of lesions (primarily for PAHs)[j]	Correlation of lesion prevalence with areas of contamination and/or comparison to lesion-based TRVs (if relevant to receptor species)
Survival, growth, and reproduction[j] of invertivorous fish	Chinook salmon[f, k] peamouth, sculpin[f]	Surface water contaminant concentrations[h]	Concentrations in surface water compared with state WQS, national AWQC or effects-based TRVs reported in the literature
		TZW contaminant concentrations[h]	Concentrations in shallow TZW compared with state WQS, national AWQC or effects-based TRVs reported in the literature[l] (sculpin only)
		Fish tissue contaminant concentrations (modeled or field-collected) from species-specific exposure areas	Empirical whole-body concentrations compared with tissue TRVs Predicted (BSAF or FWM) whole-body concentration compared with tissue TRVs (sculpin only)
		Species- or feeding-guild-specific dietary dose[m] of contaminants in prey and incidentally ingested sediment from species-specific exposure areas	Dietary dose (including incidental sediment ingestion) compared with dietary TRVs
Survival, growth, and reproduction of piscivorous fish	Northern pikeminnow, smallmouth bass	Surface water exposure contaminant concentrations[h]	Concentrations in surface water compared with reported state WQS, national AWQC, or effects-based TRVs reported in the literature
		Field-collected fish tissue contaminant concentrations from species-specific exposure areas	Empirical whole-body concentrations compared with tissue TRVs Predicted (BSAF or FWM) whole-body concentrations compared with tissue TRVs (smallmouth bass only)
		Species- or feeding-guild-specific dietary dose of contaminants in prey and incidentally ingested sediment from species-specific exposure areas	Dietary dose (including incidental sediment ingestion) compared with dietary TRVs
Survival and growth of detritivorous fish	Pacific lamprey ammocoete[j]	Surface water contaminant concentrations[h]	Concentrations in surface water compared with state WQS, national AWQC, or literature-based values that are protective of early life stages.
		TZW contaminant concentrations[h]	Concentration in shallow TZW compared with state WQS, national AWQC, or effects-based values reported in the literature that are protective of early life stages[l]

Table 3-1. Assessment and Measurement Endpoints and Lines of Evidence for the Portland Harbor BERA

Assessment Endpoint[a]	Target Ecological Receptors	Measures of Effect and Exposure (Measurement Endpoints)	Lines of Evidence in Support of Measurement Endpoints
		Fish tissue contaminant concentrations	Empirical whole-body concentration compared with tissue TRV
Birds			
Survival, growth, and reproduction of invertivorous birds	Spotted sandpiper[d]	Species-specific dietary dose of contaminants in prey and incidentally ingested sediment from shorebird assessment areas	Dietary dose (including incidental sediment ingestion) compared with dietary TRV
Survival, growth, and reproduction of omnivorous birds	Hooded merganser	Species-specific dietary dose of contaminants in prey and incidentally ingested sediment from species-specific assessment areas	Dietary dose (including incidental sediment ingestion) compared with dietary TRV
Survival, growth, and reproduction of piscivorous birds	Osprey, bald eagle	Species-specific dietary dose of contaminants in prey and incidentally ingested sediment from species-specific assessment areas	Dietary-based approach incorporating food chain transfer of contaminants from appropriate fish species (assuming all exposure comes from prey fish) and incidental sediment ingestion[n]
		Egg contaminant concentrations	Measured concentrations in osprey eggs compared with egg- or embryo-based TRVs for DDT and metabolites, PCBs, and dioxin-like compounds
Mammals			
Survival, growth, and reproduction of aquatic-dependent mammals	Mink, river otter	Species-specific dietary dose of contaminants in prey and incidentally ingested sediment from species-specific assessment areas	Dietary dose compared with dietary TRVs

Table 3-1. Assessment and Measurement Endpoints and Lines of Evidence for the Portland Harbor BERA

Assessment Endpoint[a]	Target Ecological Receptors	Measures of Effect and Exposure (Measurement Endpoints)	Lines of Evidence in Support of Measurement Endpoints
Amphibians			
Survival, growth, and reproduction of amphibians	Frogs, salamanders	Surface water contaminant concentrations from amphibian assessment areas	Concentrations in surface water compared with state WQS, national AWQC, or effects-based values reported in the literature that are protective of sensitive life stages
		TZW contaminant concentrations	Concentrations in shallow TZW compared with state WQS, national AWQC, or effects-based values reported in the literature that are protective of sensitive life stages
Aquatic Plants			
Survival, growth, and reproduction of aquatic plants	Phytoplankton, periphyton, macrophytes	Surface water contaminant concentrations from aquatic plant assessment areas	Concentrations in surface water compared with state WQS, national AWQC, or effects-based values derived from the literature that are protective of sensitive life stages (e.g., germination, emergence, early life stage growth)
		TZW contaminant concentrations	Concentrations in shallow TZW compared with state WQS, national AWQC, or effects-based values derived from the literature that are protective of sensitive life stages (e.g., germination, emergence, early life stage growth)

[a] The assessment endpoints for all receptors are based on protection and maintenance of their populations and the communities in which they live, except that the health of threatened or endangered species is to be protected at the individual organism level. Per the SOW, EPA's Problem Formulation (Attachment 2) and as stated in the Programmatic Work Plan (Integral et al. 2004b), the assessment endpoints were expressed as the survival, growth, and reproduction of each receptor group.

[b] For TBT, a BSAF may be derived from site-specific data if sufficient data are available and a relationship between sediment and tissue can be found. If not, then a screening level based on a sediment concentration of 6,000 ng/g OC (based on 2 % OC), which represents a dry-weight concentration of 120 ng/g is used (Meador et al. 2002a).

[c] Although these measures of exposure and effect are components of the benthic invertebrate community, the bivalve population and decapod population assessment endpoints are presented separately in this table. Evaluation of sediment toxicity to *Chironomus* and *Hyalella* and comparison of surface water and shallow TZW concentrations to TRVs were each conducted and presented only once as part of the benthic invertebrate community assessment. Similarly, comparison of sediment concentrations to published SQGs also occurred and was presented only once as part of the benthic community assessment.

[d] Considered representative of receptor that incidentally ingests a significant amount of sediment.

[e] Carp is not a receptor of concern for the ERA; whole-body fish tissue (i.e., carp) was analyzed for dioxin-like contaminants, including PCB congener analysis, and, for these contaminants, is a surrogate for other fish species.

[f] Used to assess fish exposure to PAHs through the diet. Analysis included an evaluation of whether these compounds are found in the diet of the fish receptors, as well as in tissue.

[g] Represents a resident broadcast spawner. Therefore, exposure to sensitive early life stages and eggs is assessed for all contaminants, including PAHs and dioxin-like compounds.

[h] Comparison of water concentrations to AWQC or water-based TRVs was conducted for all trophic guilds collectively (and presented once in the BERA) because the water criteria and water effects data incorporate multiple fish species and life stages in their derivation.

[i] Tissue-based TRV approach for dioxin-like contaminants using literature values and incorporating TEQs based on the World Health Organization TEFs. Risk from other compounds assessed in the uncertainty analysis (contaminant-specific, carp only).

[j] There is no documented linkage between fish condition or prevalence of lesions and fish survival, growth, or reproduction, nor is there is any documented linkage between fish condition or prevalence of lesions and risk to fish populations.

[k] Juvenile Chinook salmon and Pacific lamprey ammocoetes were evaluated at the organism level because they have special status (juvenile Chinook is federally threatened and Pacific lamprey is an Oregon state sensitive species of special concern to Tribes); effect thresholds based on reproduction used as a surrogate for growth in juvenile Chinook salmon and Pacific lamprey ammocoetes.

[l] TZW exposure pathway for fish receptors only considered complete and significant for sculpin and lamprey ammocoetes. The ecological CSM also shows complete TZW exposure pathway for suckers, carp, and sturgeon but categorizes the pathway as insignificant.

[m] Dietary dose compared with dietary TRVs also includes stomach content data or other approaches refined specifically for PAH (Chinook salmon only).

[n] Dioxin-like contaminants assessed using a TEQ approach based on appropriate surrogate fish tissue data. TRVs are based on the most sensitive life stages.

AWQC – ambient water quality criteria
BERA – baseline ecological risk assessment
BSAF – biota-sediment accumulation factor
CSM – conceptual site model
DDT – dichlorodiphenyltrichloroethane
EPA – US Environmental Protection Agency
ERA – ecological risk assessment
FWM – food web model
OC – organic carbon
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
PEC – probable effects concentration
PEL – probable effects level
SOW – scope of work
SQG – sediment quality guideline
SQV—site-specific sediment quality value
TBT – tributyltin
TEF – toxic equivalency factor
TEQ – toxic equivalent
TRV – toxicity reference value
TZW – transition zone water
WQS – water quality standards

What Are Assessment and Measurement Endpoints?

An assessment endpoint is an explicit expression of the ecological value that is to be protected (EPA 1992). Ecological values include those receptors, roles, and processes that are vital to ecosystem function; that provide critical resources such as habitat and fisheries; and that are perceived as having high value by humans (e.g., important to Tribal, commercial, and sport fisheries, or intrinsic value as judged by the general public). An assessment endpoint defines both the valued entity (e.g., health of a particular receptor group) and the attribute of the entity to be protected. Assessment endpoints provide direction for the risk assessment and are the basis for the analyses.

Hypotheses (expressed as risk questions) to test suspected interactions between receptors and contaminants lead to the selection of measurement endpoints that quantify exposure to contaminants via pathways identified in the CSM, effects from that exposure, or both. EPA ERA guidelines (1998) define measures of exposure and effect 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 assessment endpoint, measure of exposure, and measure of effect constitutes an LOE to evaluate risk. Each measurement endpoint may be evaluated with one or more LOEs. The results of an LOE are used alone or in combination with other LOEs to help address risk questions.

Assessment endpoints and hypotheses (expressed as risk questions) are summarized in the following bullets. These endpoints and risk questions are based on those provided in EPA's Problem Formulation (Attachment 2). In some cases, language has been modified to clarify the framework used for conducting the BERA.

- **Assessment endpoint: Survival, growth, and reproduction of benthic invertebrates –** The benthic invertebrate risk assessment was designed to answer the following questions:
  - Are the survival and biomass of benthic invertebrates, as indicated by *Hyalella azteca* and *Chironomus dilutus*, exposed to bulk sediments from Portland Harbor below biological effect thresholds that represent unacceptable effects?
  - Do contaminant concentrations in bulk surface sediment from Portland Harbor exceed SQGs derived from site-specific models that reliably predict effects based on survival and growth of benthic macroinvertebrates exposed to Portland Harbor sediment?
  - Do contaminant concentrations in bulk sediments from Portland Harbor exceed sediment quality thresholds that reliably predict reductions in the survival, reproduction, or growth of benthic macroinvertebrates, bivalves, or decapods?
  - Are the survival and growth of bivalves, as indicated by the survival and growth of the bivalve *Corbicula fluminea,* exposed to whole sediments from Portland Harbor, below biological effect thresholds that represent unacceptable effects?

- Are contaminant concentrations in Willamette River surface water or shallow TZW from Portland Harbor greater than the toxicity thresholds that are protective of the survival, growth, or reproduction of benthic macroinvertebrates (including bivalves and decapods)?
- Are contaminant concentrations in whole-body tissues of laboratory-exposed or field-collected benthic macroinvertebrates, bivalves, or decapods higher than tissue-residue thresholds for survival, reproduction, or growth of benthic invertebrates?

- **Assessment endpoint: Survival, growth, and reproduction of fish –** The fish risk assessment was designed to answer the following questions:
  - Are contaminant concentrations measured in field-collected or predicted for whole-body tissues of invertivorous, omnivorous, piscivorous, or detritivorous fish in Portland Harbor higher than tissue-residue thresholds for survival, reproduction, or growth?
  - Do tissue concentrations in prey or other potentially ingested media (i.e., sediment or water) from Portland Harbor exceed the acceptable concentrations for the survival, reproduction, or growth of invertivorous, omnivorous, or piscivorous fish consuming those media?
  - Are contaminant concentrations in surface water from Portland Harbor greater than the toxicity thresholds for survival, growth, or reproduction of invertivorous, omnivorous, piscivorous, and detritivorous[26] fish?
  - Are detritivorous fish more or less sensitive to waterborne contaminants than the species used to develop existing water quality criteria and TRVs?
  - Are contaminant concentrations in shallow TZW greater than the toxicity thresholds for survival, growth, or reproduction of invertivorous or detritivorous fish?
- **Assessment endpoint: Survival, growth, and reproduction of birds and mammals –** The risk assessment for aquatic-dependent birds and mammals was designed to answer the following questions:
  - Do tissue concentrations in fish and/or benthic invertebrate prey and other potentially ingested media (i.e., water or sediment) from Portland Harbor exceed the acceptable concentrations for the survival, reproduction, or growth of piscivorous, omnivorous, or invertivorous birds or aquatic mammals consuming those media?

[26] Detritivorous fish (i.e., lamprey ammocoetes) were evaluated for growth and survival only because they do not reproduce in the Study Area.

   - Do contaminant concentrations in eggs of piscivorous birds exceed egg-based toxicity thresholds for hatchability and survival of chicks?

- **Assessment endpoint: Survival, growth, and reproduction of amphibians –** The amphibian risk assessment was designed to answer the following question:
   - Do contaminant concentrations in surface waters of the Willamette River or shallow TZW exceed TRVs for the survival, reproduction, or growth of amphibians?
- **Assessment endpoint: Survival, growth, and reproduction of aquatic plants –** The risk assessment for aquatic plants was designed to answer the following question:
   - Do contaminant concentrations in surface waters of the Willamette River or shallow TZW exceed TRVs for the survival, reproduction, or growth of aquatic plants?

3.4 ANALYSIS PLAN

The analysis plan describes the specific approaches and methods for conducting the risk calculations used to evaluate the LOEs for risk questions and assessment endpoints (EPA 1997, 2004a). The general BERA analysis plan was developed by EPA and is presented as part of EPA's Problem Formulation (Attachment 2). Details on the analytical approaches used to evaluate risks to each ecological receptor group are provided in the risk assessment methods for each LOE in Sections 6.0 through 10.0.

A summary of the major components described in the BERA analysis plan is provided below:

- **Exposure assessment –** All exposure pathways classified as complete and significant in the ecological CSM (Figure 3-2) were evaluated quantitatively. Exposure concentrations in sediment, water, and tissue are based on concentrations in environmental samples at an ecological scale relevant to the receptor being evaluated or as required by EPA.
- **Effects assessment –** The effects assessment consisted of two general approaches. For most ecological receptors, the effects assessment used contaminant- and media-specific TRVs or SQGs for COPCs at the site. The TRVs provide estimates of contaminant concentrations that, if not exceeded, should protect ecological receptors from unacceptable adverse effects on survival, growth, or reproduction (i.e., the assessment endpoint for most receptor groups). An additional approach directly evaluated the effects of Portland Harbor COPCs on the survival, growth, and reproduction of the benthic community by directly measuring the toxicity of sediments to which benthic invertebrates are exposed .

- **Risk characterization –** As part of the risk characterization, information on contaminant exposure and effects was integrated to estimate risks to the receptors. Different risk estimation methods were used for the various measurement endpoints and LOEs evaluated in this BERA.

  A description of risks was developed to provide information needed to interpret the risk estimates, including identification of thresholds for adverse effects on assessment endpoints. Uncertainties associated with the risk estimates were also described. In addition to presenting quantitative descriptions of ecological risks and threshold concentrations for adverse ecological effects, the risk characterization also presents information on the significance of the identified risks, including the location and spatial extent of site contamination exceeding adverse effect thresholds, and the degree to which adverse effect thresholds are exceeded.

- **Weight-of-evidence (WOE) framework –** When multiple LOEs are available for assessing risk to an ecological receptor, a framework is needed to reconcile any inconsistencies as well as to determine the reliability of all available LOEs for a given receptor-COPCpair. EPA's Problem Formulation (Attachment 2) provided one such WOE framework. However, EPA recognized that the WOE framework proposed in the problem formulation would not account for differences in relative strength of different LOEs (e.g., the proposed WOE framework does not allow differences in the quality of TRVs across chemicals to lead to different weights on the TRV LOE for different COPCs). Because of this limitation, the proposed WOE framework was not used in the BERA. EPA (2009c) has acknowledged the limitations of the WOE framework and, given the absence of a workable framework, stated that WOE issues will be addressed through BERA review and FS scoping. A qualitative WOE was applied in the BERA to arrive at risk conclusions.

- **Uncertainty analysis –** The BERA formulation stipulated that the general methods for conducting the uncertainty analysis should include the following three features:

  - Incorporate various scenarios for exposure and effects in the risk estimation process to capture the range of uncertainties in assumptions
  - Express numeric risk calculations as point estimates with statistical measures of uncertainty (e.g., confidence limits, percentiles)
  - Conduct sensitivity analyses in which risk parameter values are iteratively varied to examine the effect of variability in the parameter on the risk estimate (e.g., probabilistic risk analysis).

These methods were applied in the uncertainty analysis of this BERA. The uncertainties associated with each LOE for each receptor are discussed in Sections 6.0 though 10.0.

4.0 BERA DATA

Numerous data were collected by the LWG in support of the Portland Harbor RI/FS during three major rounds of sampling from RM 0 to approximately RM 28 of the Willamette River. Additional data were compiled from non-LWG sampling events within this stretch of the Willamette River. The LWG and non-LWG sampling events included in the site characterization and risk assessment (SCRA) dataset are described in detail in Appendix A of the draft final RI (Integral et al. 2011). Details on all SCRA data collected are presented in Sections 2.0 and 5.0 of the draft final RI. This section is intended to provide a general overview of the data used in the BERA.

This BERA used a subset of the data that make up the SCRA dataset (hereafter referred to as the BERA dataset). The BERA focused on the data collected between RM 1.9 and RM 11.8, defined as the Study Area. The following data are included in the BERA dataset:

- Chemistry data for 1,469 surface sediment samples
- Chemistry data for 315 whole-body fish and invertebrate tissue samples and stomach contents
- Chemistry data for 313 surface water samples
- Chemistry data for 192 TZW samples collected adjacent to nine sites[27]
- Chemistry data for five osprey egg tissue samples
- Bioassay data from *Chironomus dilutus* and *Hyalella azteca* tests conducted with 269 surface sediment samples
- Sediment toxicity data from bioaccumulation testing of *Corbicula fluminea* and *Lumbriculus* sp. using sediments collected from 33 sampling locations

Qualitative data collected from the Study Area provide additional information with which to charaterize the ecological setting and risks to ecological receptors:

- Field observations of fish health
- Descriptions of benthic macroinvertebrate (infaunal and epifaunal) community structure
- Data from a reconnaissance survey of aquatic plant habitat and amphibian and reptile habitat
- Data from a reconnaissance survey of shorebird beach habitat

[27] Two areas are adjacent to the Arkema site.

Surface sediment and fish tissue data collected from the downstream reach (RM 0 to RM 1.9), Multnomah Channel, and the downtown reach (RM 11.8 to RM 15.3) were also evaluated in the BERA. These non-Study Area sediment and tissue data were compared with Study Area data, and the risk to ecological receptors from these samples was also evaluated. The following data from the downstream reach, Multnomah Channel, and downtown reach were evaluated:

- Chemistry data from 45 surface sediment samples
- Chemistry data from 11 whole-body fish and invertebrate tissue samples

Data from the upriver reach of the LWR (from RM 15.3 to RM 28.4) were also used in the BERA. Upriver sediment data were used to define reference conditions to evaluate individual toxicity samples for inclusion in the benthic toxicity model (Section 6.0). Upriver fish tissue data were used for comparison purposes in evaluating tissue residues in Study Area fish (Section 7.0). The following upriver reach data were used in the risk characterizations:

- Chemistry data from 22 surface sediment samples
- Chemistry data from 19 whole-body fish tissue samples and stomach contents
- Bioassay data from *Chironomus dilutus* and *Hyalella azteca* tests conducted with 22 surface sediment samples

Surface sediment and surface water data collected from the upriver reach were also compiled to establish background concentrations for the Study Area. These data and methods used to define background concentrations are presented in Section 7.0 of the draft final RI (Integral et al. 2011).

Bird egg tissue data were compiled for samples collected from a considerable distance upriver (n = 5 egg samples, Willamette River RM 69 to RM 77). Upriver bird egg tissue data were used for comparison purposes in evaluating tissue residues in the five Study Area bird eggs (Section 8.0).

More detailed descriptions of the BERA dataset are presented in Section 4.1. Non-Study Area data from the upstream reach, Multnomah Channel, and downtown reach are presented in Section 4.2. Upriver sediment and tissue data are summarized in Section 4.3. Attachment 3 presents the data management rules (e.g., summation, organic carbon [OC]-normalization, treatment of field replicates) that were applied to the BERA dataset. Attachment 4 presents all of the chemistry data used in the BERA for COPCs identified in Section 5.0. Specifically, electronic data files containing chemistry data for the Study Area, chemistry data for the non-Study Area, predicted tissue concentration data, and compiled EPCs are included in Attachment 4, Parts B, C, D and E, respectively.

4.1 STUDY AREA DATA

The BERA dataset includes only those matrices relevant for ecological exposure pathways: surface sediment (0 to 30.5 cm), benthic invertebrate and fish tissue, bird egg tissue, surface water, and shallow TZW (0 to 38 cm). The BERA dataset is summarized, by medium, in Table 4-1 and described in more detail in the following subsections. Qualitative reconnaissance-level and habitat surveys were not used to directly develop risk estimates (and are not included in Table 4-1); however, these data were used in developing the CSM and to provide context for the characterization of risks. These data are briefly summarized at the end of this section.

Table 4-1. Overall Summary of BERA Dataset

Medium	Data Type	Number of Samples
Study Area Data (RM 1.9 – RM 11.8)		
Surface sediment	Contaminant concentrations in all surface sediment collected within the 0-to-30.5-cm depth horizon in the Study Area	1,469
Sediment toxicity tests	Toxicity response endpoints for surface sediment samples tested with *Chironomus dilutus* and *Hyalella azteca*	269
Invertebrate, whole-body fish tissue, fish stomach contents	Contaminant concentrations in field -collected clams, multiplate-collected invertebrates, crayfish, mussels, black crappie, brown bullhead, carp, juvenile Chinook salmon, largescale sucker, northern pikeminnow, peamouth, sculpin, smallmouth bass, juvenile white sturgeon, Pacific lamprey ammocoetes, juvenile Chinook salmon stomach contents, juvenile white sturgeon stomach contents, and laboratory-exposed worm and clam tissue[a]	315
Bird egg tissue	Chemical concentrations in osprey egg tissue	5
Surface water	Chemical concentrations in surface water collected by the LWG using a peristaltic pump and XAD-2 Infiltrex™ 300 system (column and filter) and in surface water collected during non-LWG sampling events	313
Shallow TZW	Chemical concentrations in shallow (0 to 38 cm) TZW sampled using a peeper (0 to 38 cm), Trident® probe, or Geoprobe	192
Downstream Reach Data (RM 0 – RM 1.9)		
Surface sediment	Chemical concentrations in surface sediment collected within the 0-to-30.5-cm depth horizon in the downstream reach of the LWR	21
Invertebrate, whole-body fish tissue	Chemical concentrations in field-collected clams, crayfish, and sculpin	5
Multnomah Channel Data (Multnomah Channel)		
Surface sediment	Chemical concentrations in surface sediment collected within the 0-to-30.5-cm depth horizon from below the mouth of Multnomah Channel	7

Table 4-1. Overall Summary of BERA Dataset

Medium	Data Type	Number of Samples
Downtown Reach Data (RM 11.8 to 15.3)		
Surface sediment	Chemical concentrations in surface sediment collected within the 0-to-30.5-cm depth horizon in the downtown reach of the LWR	17
Sediment toxicity tests	Chemical concentrations in surface sediment samples tested with *Chironomus dilutus* and *Hyalella aztec*a	2
Invertebrate, whole-body fish tissue	Chemical concentrations in field-collected clams, crayfish, and sculpin	6
Upriver Data (RM 15.3 – RM 28.4)		
Surface sediment[b]	Chemical concentrations in surface sediment collected within the 0-to-30.5-cm depth horizon in the upriver reach of the LWR	22[c]
Sediment toxicity tests	Chemical concentrations in surface sediment samples tested with *Chironomus dilutus* and *Hyalella aztec*a	22
Whole-body fish tissue	Chemical concentrations in brown bullhead, carp, juvenile Chinook salmon, smallmouth bass, Pacific lamprey ammocoetes, and juvenile Chinook salmon stomach contents	18
Bird egg tissue	Chemical concentrations in osprey egg tissue	5

[a] Survival and growth data relative to control for laboratory-exposed clams (*Corbicula*) were also collected.

[b] Additional upriver surface sediment chemistry data were used to establish background sediment concentrations. See Section 7.0 of the draft final RI (Integral et al. 2011) for a presentation of background data.

[c] Twenty-three sediment chemistry results were available from 22 sampling locations where toxicity testing was also conducted.

BERA – baseline ecological risk assessment
LWG – Lower Willamette Group
LWR – Lower Willamette River
RI – remedial investigation
RM – river mile
TZW – transition zone water

4.1.1 Surface Sediment

Surface sediment chemistry from the Study Area included in the BERA dataset includes LWG-collected data (from various sampling events in Rounds 1, 2, and 3) and non-LWG-collected data that were of sufficient quality to support the BERA. Table 4-2 presents a summary of the surface sediment samples included in the BERA dataset from the Study Area. Map 4-1 presents all the Study Area surface sediment sampling locations included in the BERA dataset, including those sediment samples that were submitted for toxicity testing. Map 4-2 presents the locations of the beach sediment transect samples that were also collected from the Study Area and included in the BERA dataset.

Table 4-2. Summary of Study Area Surface Sediment Data Evaluated in the BERA

Sampling Event	Sampling Period	No. of Samples[a]	Contaminants Analyzed in Sediment
LWG-Collected Data			
Round 1 co-located surface sediment	October to November 2002	44	PCB Aroclors, butyltins, dioxins and furans, herbicides, metals, PAHs, PCB congeners, organochlorine pesticides, phenols, phthalates, SVOCs, VOCs
Round 1 HHRA beach sediment[b]	October 2002	22	PCB Aroclors, herbicides, metals, PAHs, organochlorine pesticides, phenols, phthalates, SVOCs
Round 2A shorebird and HHRA beach sediment[b]	July to November 2004	26	PCB Aroclors, dioxins and furans, metals, PAHs, PCB congeners, organochlorine pesticides, phenols, phthalates, SVOCs
Round 2A benthic sediment	December 2005	35	PCB Aroclors, butyltins, dioxins and furans, herbicides, metals, PAHs, PCB congeners, organochlorine pesticides, petroleum, phenols, phthalates, SVOCs, VOCs
Round 2A groundwater pathway assessment co-located sediment grabs	November and December 2005	37	Butyltins, dioxins and furans, herbicides, metals, PAHs, organochlorine pesticides, petroleum, phenols, phthalates, SVOCs, VOCs
Round 2A sediment cores[c]	September to November 2004	46	PCB Aroclors, butyltins, dioxins and furans, metals, PAHs, organochlorine pesticides, petroleum, phenols, phthalates, SVOCs
Round 2B sediment cores[c]	October 2005	35	PCB Aroclors, butyltins, dioxins and furans, herbicides, metals, PAHs, organochlorine pesticides, petroleum, phenols, phthalates, SVOCs, VOCs
Round 2A sediment grabs[d]	July to November 2004	525[d]	PCB Aroclors, butyltins, dioxins and furans, herbicides, metals, PAHs, PCB congeners, organochlorine pesticides, petroleum, phenols, phthalates, SVOCs, VOCs
Round 3 sediment from upstream and downstream[e]	January 2007	9	PCB Aroclors, dioxins and furans, metals, PAHs, organochlorine pesticides, petroleum, phenols, phthalates, SVOCs, VOCs
Willamette Cove sampling and analysis	September 2007	1	PCB Aroclors, butyltins, dioxins and furans, herbicides, metals, PAHs, PCB congeners, organochlorine pesticides, petroleum, phenols, phthalates, SVOCs, VOCs
Round 3B co-located sediment grabs	October 2007	21	PCB Aroclors, butyltins, dioxins and furans, metals, PAHs, PCB congeners, organochlorine pesticides, phenols, phthalates, SVOCs
Round 3B sediment grabs[d]	November 2007	163[d]	PCB Aroclors, dioxins and furans, metals, PAHs, PCB congeners, organochlorine pesticides, petroleum, phenols, phthalates, SVOCs

Table 4-2. Summary of Study Area Surface Sediment Data Evaluated in the BERA

Sampling Event	Sampling Period	No. of Samples[a]	Contaminants Analyzed in Sediment
Non-LWG-Collected Data			
2005 O&M dredge sediment characterization[f]	May 2005	85	PCB Aroclors, butyltins, dioxins and furans, herbicides, metals, PAHs, organochlorine pesticides, petroleum, phenols, phthalates, SVOCs, VOCs
City outfall pilot project	August 2002	18	PCB Aroclors, herbicides, metals, PAHs, organochlorine pesticides, petroleum, phenols, phthalates, SVOCs
City outfall sediment investigation	October 2002	85	Herbicides, metals, PAHs, PCB congeners, organochlorine pesticides, petroleum, phenols, phthalates, SVOCs
Gasco source control evaluation	April 2001	10[g]	Metals, PAHs, VOCs
McCormick & Baxter RI Phase 3	October 1999	12	Dioxins and furans, metals, PAHs, phenols
PAH in surface sediments	June 1997	32	PAHs, SVOCs
Portland Harbor sediment investigation	September 1997	140	PCB Aroclors, butyltins, dioxins and furans, herbicides, metals, PAHs, organochlorine pesticides, phenols, phthalates, SVOCs
Portland Shipyard environmental audit	November 1997 to December 1998	8	PCB Aroclors, butyltins, metals, PAHs, phthalates, SVOCs, VOCs
Portland Shipyard sediment investigation	March to April 1998	58	PCB Aroclors, butyltins, metals, PAHs, organochlorine pesticides, phenols, phthalates, SVOCs, VOCs
RM11E sediment[h]	May to June 2009	59	PCB Aroclors, butyltins, dioxins and furans, metals, PAHs, organochlorine pesticides, petroleum, phenols, phthalates, SVOCs, VOCs
Terminal 4 Abatement Phase 1 – Construction Phase 1 – Dredging and Capping[h]	December 2008	18	PCB Aroclors, metals, PAHs, organochlorine pesticides, petroleum, phthalates, SVOCs
Terminal 4 EE/CA	March to May 2004	43	PCB Aroclors, metals, PAHs, organochlorine pesticides, petroleum, phthalates, SVOCs
US Moorings sediment investigation 2002	September 2002	2	PCB Aroclors, butyltins, metals, PAHs, organochlorine pesticides, petroleum, phenols, phthalates, SVOCs

Table 4-2. Summary of Study Area Surface Sediment Data Evaluated in the BERA

Sampling Event	Sampling Period	No. of Samples[a]	Contaminants Analyzed in Sediment
Willamette River 1998 data	January 1998	12	PCB Aroclors, butyltins, metals, PAHs, phthalates

[a] Includes field replicates.

[b] HHRA and shorebird beach samples were collected as transect composite samples of surface sediment along beach areas.

[c] Surface sediment samples from cores collected within the 0-to-30-cm depth included.

[d] Benthic toxicity testing was conducted for a total of 269 co-located samples from these sampling events. This total includes 215 bioassay samples from the Round 2A sediment dataset and 54 bioassay samples from the Round 3B sediment dataset. These data were used in the benthic risk assessment.

[e] The Round 3 sediment from upstream and downstream sampling event was named prior to defining the RI/FS Upstream and Downstream Reaches of the LWR. Samples including the Study Area dataset from this sampling event were located within the boundaries of the Study Area (RM 1.9 to 11.8).

[f] Only sample data associated with sediments that were not ultimately dredged are included in the BERA dataset

[g] Surface sediment from sediment cores were collected from both the 0-to-10-cm and 10-to-20-cm depth horizons at five locations.

[h] Sediment sample contaminant results were only used for the benthic community evaluation in order to be consistent with data lockdown agreements between the LWG and EPA.

BERA – baseline ecological risk assessment
EE/CA – engineering evaluation/cost analysis
FS – feasibility study
HHRA – human health risk assessment
LWG – Lower Willamette Group
O&M – operation and maintenance
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
RI – remedial investigation
RM – river mile
SVOC – semivolatile organic compound
VOC – volatile organic compound

All non-LWG data included in the SCRA database (see Appendix A of the draft final RI (Integral et al. 2011)) were of acceptable data quality for risk evaluation (Category 1/QA2), as agreed to among LWG and EPA in the programmatic work plan (Integral et al. 2004b). All surface sediment data included in the BERA dataset were collected from within the top 30.5 cm (sample depths varied) of the sediment horizon and located within the Study Area. The definition of surface sediment as the top 30.5 cm was based on three bathymetric surveys that provided trends in the magnitude, direction (i.e., shallowing versus deepening), and spatial distribution of riverbed elevation changes (Integral 2004b). Early RI investigations (e.g., sediment trend analysis, SPI, time-series bathymetry studies) suggested the potential for Study Area-wide, small-scale (≤ 30 cm in depth) surface sediment disturbance or movement during winter (rainy season) flow regimes. The measured maximum net bathymetric change over the 25-month period between the January 2002 and February 2004 surveys was less than 30 cm (1 ft) over 90% of the ISA. The apparent redox potential discontinuity depth, which can be used as an estimate of the depth of bioturbation, in the Study Area ranged between < 1.5 cm and > 6 cm (SEA 2002). Surface sediment samples collected from areas that have since been dredged or capped were not included in the SCRA and BERA datasets because these samples no longer represent the current condition of the Study Area.

Surface sediment data were used to estimate exposure concentrations for relevant ecological receptors based on direct contact (i.e., benthic invertebrates, fish, and aquatic plants) and dietary exposure (i.e., fish and wildlife). The chemistry and bioassay results of surface sediment samples are presented in Attachment 4 and Attachment 6, respectively.

4.1.2 Benthic Invertebrate and Fish Tissue

Benthic invertebrate and fish tissue chemistry data from the Study Area included in the BERA dataset were selected from various LWG sampling events, as summarized in Table 4-3.[28] Benthic tissue in the BERA dataset included field-collected tissue samples for crayfish,[29] clams (*Corbicula fluminea*), freshwater mussels (Western pearlshell [*Margaritifera falcata*] and winged floater [*Anodonta nuttalliana*]), and epibenthic invertebrates and zooplankton collected with multiplate samplers.[30]

[28] Although the Study Area boundary is defined as RM 1.9 to RM 11.8, several carp composite samples collected during Round 3 included individual organisms from beyond the site boundary (i.e., composites represented fish collected between RM 0 and RM 3 and between RM 9 and RM 12).

[29] Crayfish were not identified to species; however, only one crayfish species, the western freshwater crayfish (*Pacifastacus leniusculus*), is known to occur in the Study Area.

[30] Multiplate tissue data were used in the evaluation of risks to the benthic inverterbate community tissue-residue assessment (Section 6.4) and as a dietary component for specific pelagic fish and wildlife receptors (Sections 7.2 and 8.1, respectively). Due to the design of the multiplate sampler, the multiplate tissue data represent invertebrate exposure primarily from overlying water rather than directly from sediments. The uncertainty associated with use of these data in the evaluation of benthic invertebrates is discussed in Section 6.4.

Table 4-3. Summary of Study Area Tissue Data Evaluated in the BERA

Sampling Event and Period	Species (Target Length)	No. of Composite Samples[a]	Contaminants Analyzed in Tissue
Round 1A tissue sampling, June 2002	Juvenile Chinook salmon (90 mm)	6	PCB Aroclors, metals, PAHs, organochlorine pesticides, phenols, phthalates, SVOCs
Round 1 tissue sampling, July to November 2002	Black crappie[b] (225 to 300 mm)	4	PCB Aroclors, dioxins and furans, metals, PCB congeners, organochlorine pesticides, SVOCs
	Brown bullhead[b] (225 to 300 mm)	6	PCB Aroclors, dioxins and furans, metals, PAHs, PCB congeners, organochlorine pesticides, phenols, phthalates, SVOCs
	Carp[c] (508 to 677 mm)	6	PCB Aroclors, dioxins and furans, metals, PAHs, PCB congeners, organochlorine pesticides, phenols, phthalates, SVOCs
	Clam	3	PCB Aroclors, butyltins, metals, PAHs, organochlorine pesticides, phenols, phthalates, SVOCs
	Crayfish	27	PCB Aroclors, dioxins and furans, metals, PAHs, PCB congeners, organochlorine pesticides, phenols, phthalates, SVOCs
	Largescale sucker (300 mm)	6	PCB Aroclors, metals, PAHs, organochlorine pesticides, phenols, phthalates, SVOCs
	Northern pikeminnow (250 mm)	6	PCB Aroclors, metals, organochlorine pesticides, SVOCs
	Peamouth (200 mm)	4	PCB Aroclors, metals, organochlorine pesticides, SVOCs
	Sculpin (90 mm)	27	PCB Aroclors, dioxins and furans, metals, PAHs, PCB congeners, organochlorine pesticides, phenols, phthalates, SVOCs
	Smallmouth bass (200 mm)	14	PCB Aroclors, dioxins and furans, metals, PAHs, PCB congeners, organochlorine pesticides, phenols, phthalates, SVOCs

Table 4-3. Summary of Study Area Tissue Data Evaluated in the BERA

Sampling Event and Period	Species (Target Length)	No. of Composite Samples[a]	Contaminants Analyzed in Tissue
Round 2A benthic tissue sampling, November to December 2005	Clam	33	Butyltins, dioxins and furans, metals, PAHs, PCB congeners, organochlorine pesticides, phenols, phthalates, SVOCs
	Laboratory-exposed clam	35	Butyltins, dioxins and furans, metals, PAHs, PCB congeners, organochlorine pesticides, phenols, phthalates, SVOCs
	Laboratory-exposed worm	35	Butyltins, dioxins and furans, metals, PAHs, PCB congeners, organochlorine pesticides, phenols, phthalates, SVOCs
Round 2A juvenile Chinook salmon tissue sampling, May 2005	Juvenile Chinook salmon (stomach contents)	5	PAHs, PCB congeners, organochlorine pesticides, SVOCs
	Juvenile Chinook salmon (50 to 80 mm)	9	Butyltins, dioxins and furans, metals, PAHs, PCB congeners, organochlorine pesticides, phenols, phthalates, SVOCs
Round 2A multiplate tissue sampling, September 2005	Invertebrates	7	Dioxins and furans, metals, PCB congeners, organochlorine pesticides, SVOCs
Round 2 mussel tissue sampling, November to December 2005	Mussels	7	Butyltins, dioxins and furans, metals, PAHs, PCB congeners, organochlorine pesticides, phenols, phthalates, SVOCs
Round 2B lamprey tissue sampling, November 2005	Pacific lamprey ammocoete	1	Dioxins and furans, PCB congeners, organochlorine pesticides, SVOCs
Round 3A lamprey tissue sampling, September 2006	Pacific lamprey ammocoete and macropthalmia[d]	5	Butyltins, dioxins and furans, metals, PAHs, PCB congeners, organochlorine pesticides, phenols, phthalates, SVOCs
Round 3A juvenile sturgeon sampling, February 2007	Juvenile white sturgeon (stomach contents)	3	Metals, PAHs, PCB congeners, organochlorine pesticides, SVOCs
	Juvenile white sturgeon (42 to 60 in.)	15	Butyltins, dioxins and furans, metals, PAHs, PCB congeners, organochlorine pesticides, phenols, phthalates, SVOCs

Table 4-3. Summary of Study Area Tissue Data Evaluated in the BERA

Sampling Event and Period	Species (Target Length)	No. of Composite Samples[a]	Contaminants Analyzed in Tissue
Round 3B biota sampling, August to November 2007	Clam	7	Butyltins, dioxins and furans, metals, PAHs, PCB congeners, organochlorine pesticides, phenols, phthalates, SVOCs
	Crayfish	5	Butyltins, dioxins and furans, metals, PAHs, PCB congeners, organochlorine pesticides, phenols, phthalates, SVOCs
	Carp[e] (508 to 607 mm)	9	Butyltins, dioxins and furans, metals, PAHs, PCB congeners, organochlorine pesticides, phenols, phthalates, SVOCs
	Sculpin (90 mm)	12	Butyltins, dioxins and furans, metals, PAHs, PCB congeners, organochlorine pesticides, phenols, phthalates, SVOCs
	Smallmouth bass[e] (225 to 355 mm)	18	Butyltins, dioxins and furans, metals, PAHs, PCB congeners, organochlorine pesticides, phenols, phthalates, SVOCs
USGS and USFWS bird egg tissue sampling; between May 2008 and September 2009	Osprey egg	5[f]	Dioxins and furans, mercury, PCB Aroclors, PCB congeners, organochlorine pesticides

a Whole-body tissue composite samples only, except where noted; sample count includes field replicates. All whole-body tissue composite samples in cases where gender could be determined include both genders.

b Whole-body tissue was collected for black crappie and brown bullhead in support of the BHHRA and was used to estimate dietary exposure for wildlife receptors.

c Whole-body tissue was collected for carp in support of the BHHRA and was used to estimate dietary exposure for fish (i.e., northern pikeminnow) and wildlife receptors. Carp was used as a surrogate ecological receptor for the tissue-residue approach for dioxins and furans and for dioxin-like PCB congeners in whole-body tissue.

d Both lamprey ammocoete and macropthalmia tissue were collected and analyzed. In the remainder of this document, the term “Pacific lamprey ammocoete tissue” refers to both ammocoete and macropthalmia tissues. Both life stages are representative of the early life stages when lamprey are present in the Study Area.

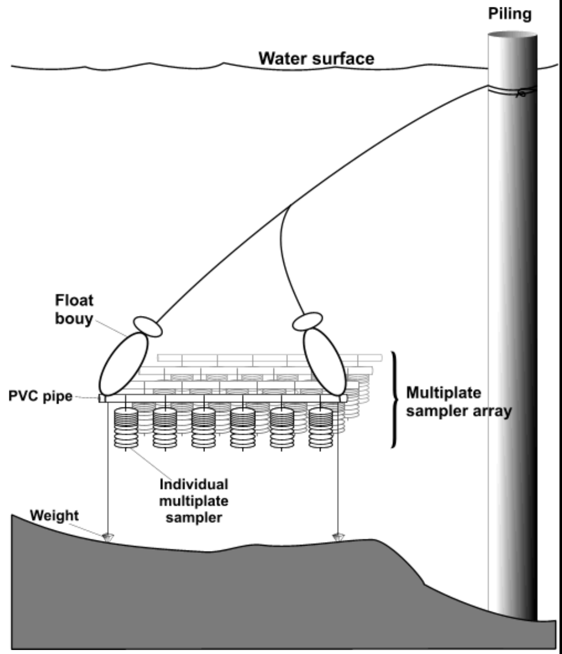
e Whole-body tissue composite concentrations for carp and smallmouth bass were estimated using fillet and remaining body tissue concentrations.

f Samples were of individual eggs, not composites.

BERA – baseline ecological risk assessment
BHHRA – baseline human health risk assessment
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
SVOC – semivolatile organic compound
USFWS – US Fish and Wildlife Service
USGS – US Geological Survey

What is a Multiplate Sampler?

Multiplate samplers were used as an artificial substrate for passive collection of invertebrates. They were EPA-style, round Hester-Dendy samplers consisting of 14 3-in.-diameter plates made of tempered hardboard separated by spacers. The space between the individual plates varied from 3 to 12 mm, which allows for a range of species to colonize the open substrate. Each complete sampling device consisted of 24 multiplate samplers for a total surface area of 2.78 m^2. Anchored with a double-buoy system, the multiplate sampler arrays kept the samplers at a constant distance of approximately 1.5 to 2 m above the river bottom to prevent excessive siltation. The multiplate samplers were retrieved from a tag line either tied to a nearby piling or attached to a second anchor, which was placed near the shoreline. A small unmarked plastic foam float was attached to the second anchor. The multiplate samplers were deployed for 6 weeks. The organisms on 21 of the multiplate samplers at each location were composited into an invertebrate tissue sample, and the organisms on the remaining 3 multiplate samplers were composited into a sample for taxonomic analysis.



Tissue samples of clams (*Corbicula fluminea*) and worms (*Lumbriculus variegatus*) exposed to surface sediments in a laboratory were also included in the BERA dataset. Maps 4-3 though 4-13 present all the invertebrate and fish tissue composite sampling locations from the Study Area included in the BERA dataset.

All fish tissue samples were collected in the field. Data represent whole-body tissue composite samples of mixed gender. Composite samples of tissue were analyzed for 11 fish species: largescale sucker (*Catostomus macrocheilus*), carp (*Cyprinus carpio*), juvenile Chinook salmon (*Oncorhynchus tshawytscha*), sculpin (*Cottus* spp.), peamouth (*Mylocheilus caurinus*), smallmouth bass (*Micropterus dolomieui*), northern pikeminnow (*Ptychocheilus oregonensis*), black crappie (*Pomoxis nigromaculatus*), brown bullhead (*Ameiurus nebulosus*), juvenile (pre-breeding) white sturgeon (*Acipenser transmontanu*s), and Pacific lamprey ammocoetes (*Entosphenus tridentata*, formerly *Lampetra tridentata*). Whole-body tissue composite concentrations for carp and smallmouth bass collected during the Round 3B sampling event were estimated by combining fillet concentrations measured for the baseline human health risk assessment (BHHRA) and remaining body tissue concentrations. Tissue component concentrations were adjusted by the relative weight of the fillet and the remaining body tissue concentration. Data from the contaminant analysis of juvenile Chinook salmon and juvenile sturgeon stomach content samples from within the Study Area were also included in the BERA dataset.

Fish composite samples included fish collected over various stretches of the river within the Study Area, as detailed below.

- Smallmouth bass tissue composite samples included individual fish collected over 1- and 0.5-mile stretches in Rounds 1 and 3, respectively. Based on Round 1 sampling results, the target total length range for smallmouth bass collected in Round 3 was modified from the original 200-mm minimum to a range of 225 to 355 mm; however, fish larger than 335 mm were still retained during sampling and included for composite samples.
- Largescale sucker, peamouth, and northern pikeminnow tissue composite samples included individual fish collected over 2-mile stretches in Round 1. The targeted lengths of individuals whose tissue was retained for compositing were 300 mm for largescale sucker, 200 mm for peamouth, and 250 mm for northern pikeminnow.
- Carp tissue composites included individual fish collected over 3- and 4-mile stretches in Round 1 and Round 3, respectively. The target size for carp retained for compositing in both Round 1 and Round 3 was 508 to 677 mm.
- Black crappie and brown bullhead tissue composites included individual fish collected over 3-mile stretches in Round 1. For tissue compositing, both species had the same target length of 225 to 300 mm.
- Juvenile (pre-breeding) white sturgeon tissue composite samples included individual fish collected over 3-mile stretches in Round 3. The target length for juvenile (pre-breeding) white sturgeon retained for compositing was 1,067 to 1,524 mm (42 to 60 in.).

For each target species, fish were collected in the size range consumed by humans. As stated in the Round 1 field sampling plan (FSP), the minimum of the target size range is no less 75% of the maximum of the target size range, in accordance with EPA guidance (SEA et al. 2002).

Composite samples consisted of equal masses of tissue from 5 to 10 fish of the same species of similar body size from the same sampling location (SEA et al. 2002). The number of fish per composite was chosen to be consistent with previous fish tissue studies completed in the LWR.

Fish and invertebrate tissue data were used to estimate exposure concentrations for relevant pathways (e.g., dietary exposure) and ecological receptors (i.e., benthic invertebrates, fish, and wildlife). The tissue chemistry results from the Study Area samples are presented in Attachment 4.

4.1.3 Bird Egg Tissue

Non-LWG-collected bird egg tissue chemistry data from the Study Area were included in the BERA to evaluate risks to piscivorous birds. Samples of individual osprey egg tissues were collected from five locations in the Study Area between May 2008 and September 2009. Data were collected by the US Geological Survey (USGS) and US Fish and Wildlife Service (USFWS). Egg samples were analyzed for the following contaminant groups: PCB Aroclors, dioxins and furans, mercury, PCB congeners, and organochlorine pesticides.[31] Map 4-14 presents the osprey egg tissue sampling locations from the Study Area included in the BERA dataset.

4.1.4 Surface Water

Surface water data collected from the Study Area used in the BERA dataset include all Round 2 and Round 3 LWG-collected data, as well as non-LWG data, as summarized in Table 4-4. Study Area data from all three sampling events conducted during Round 2 and all four sampling events conducted during Round 3 are included in the BERA dataset. Map 4-15 presents all the Study Area surface water sampling locations included in the BERA dataset

During Round 2, surface water sampling was performed in three separate events (November 2004, March 2005, and July 2005) to assess the seasonality of water flow levels, with the following sampling rationale:

- **Round 2, Event 1 (November 2004) –** conducted during mid- to late fall to capture potentially elevated contaminant concentrations in the river from rainfall runoff
- **Round 2, Event 2 (March 2005) –** selected by EPA to coincide with the early exposure period of amphibian egg masses
- **Round 2, Event 3 (July 2005) –** timed to coincide with low-flow conditions, when any effects of groundwater discharge to the water column would be most pronounced

Twenty-three surface water locations were sampled within the Study Area during each Round 2 sampling event; surface water samples collected were either near-bottom samples (i.e., collected within 1 to 3 ft of the river bottom) or integrated water column (IWC) samples. Round 2 surface water samples were collected from 20 single-point locations (three of which were vertically integrated) and from 3 cross-sectional river transect locations at RM 4.0, RM 6.3, and RM 11.0 (vertically and horizontally integrated). The Round 2 sampling events occurred during discharge conditions that were lower than the historical average; the Study Area discharge during Round 2 sampling ranged from 5,700 cubic feet per second (cfs) to 24,000 cfs.

[31] Bird egg tissue was also analyzed for polybrominated diphenyl ethers. However, because these data were assigned a QA/QC level of Category 2, they are not included as part of the SCRA database. Only non-LWG data assigned a QA/QC level of Category1/QA2 were considered acceptable for risk evaluation, as agreed to among LWG and EPA in the programmatic work plan (Integral et al. 2004b).

Table 4-4. Summary of Study Area Surface Water Data Evaluated in the BERA

Sampling Event	Sampling Period	Sampling Method	Water Depth	No. of Samples[a]	Contaminants Analyzed
LWG-Collected Data					
Round 2A, surface water event 1	November to December 2004	Peristaltic pump	Near bottom or vertically and horizontally integrated	25	PCB Aroclors, butyltins, herbicides, metals, PAHs, organochlorine pesticides, phenols, phthalates, SVOCs
		XAD	Near bottom or vertically and horizontally integrated	8[b]	Dioxins and furans, PAHs, PCB congeners, organochlorine pesticides, phthalates, SVOCs
Round 2A, surface water event 2	March 2005	Peristaltic pump	Near bottom or vertically and horizontally integrated	27	PCB Aroclors, butyltins, herbicides, metals, PAHs, organochlorine pesticides, phenols, phthalates, SVOCs
		XAD	Near bottom or vertically and horizontally integrated	8[b]	Dioxins and furans, PAHs, PCB congeners, organochlorine pesticides, phthalates, SVOCs
Round 2A, surface water event 3	July 2005	Peristaltic pump	Near bottom or vertically and horizontally integrated	25	PCB Aroclors, butyltins, herbicides, metals, PAHs, organochlorine pesticides, phenols, phthalates, SVOCs
		XAD	Near bottom or vertically and horizontally integrated	8[b]	Dioxins and furans, PAHs, PCB congeners, organochlorine pesticides, phthalates, SVOCs
Round 3, surface water event 1	January 2006	Peristaltic pump	Mid-channel, 1 ft from the bottom and 3 ft below the surface	3	Butyltins, herbicides, metals, PAHs, phenols, phthalates, SVOCs
		XAD	Mid-channel, 1 ft from the bottom and 3 ft below the surface	2[b]	Dioxins and furans, PAHs, PCB congeners, organochlorine pesticides, SVOCs
Round 3, surface water event 2	September 2006	Peristaltic pump	Near bottom, near surface, or vertically integrated	12	Butyltins, herbicides, metals, PAHs, phenols, phthalates, SVOCs
		XAD	Near bottom, near surface, or vertically integrated	12[b]	Dioxins and furans, PAHs, PCB congeners, organochlorine pesticides, SVOCs
Round 3, surface water event 3	November 2006	Peristaltic pump	Near bottom, near surface, or vertically integrated	40	Butyltins, herbicides, metals, PAHs, phenols, phthalates, SVOCs
		XAD	Near bottom, near surface, or vertically integrated	38[b]	Dioxins and furans, PAHs, PCB congeners, organochlorine pesticides, SVOCs

Table 4-4. Summary of Study Area Surface Water Data Evaluated in the BERA

Sampling Event	Sampling Period	Sampling Method	Water Depth	No. of Samples[a]	Contaminants Analyzed
Round 3, surface water event 4	January to March 2007	Peristaltic pump	Near bottom or near surface	44	Butyltins, herbicides, metals, PAHs, phenols, phthalates, SVOCs
		XAD	Near bottom or near surface	38[b]	Dioxins and furans, PAHs, PCB congeners, organochlorine pesticides, SVOCs
Non-LWG-Collected Data					
Siltronic supplemental in-river transition zone	May 2005	Peristaltic pump	Near bottom, 0 to1ft above mudline	23	PAHs, SVOCs, VOCs

[a] Sample count includes field replicates.

[b] For XAD samples, the filter and water column samples were combined at the same location to derive a sample count.

BERA – baseline ecological risk assessment
LWG – Lower Willamette Group
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
SVOC – semivolatile organic compound
VOC – volatile organic compound
XAD –Infiltrex™ 300 system with an XAD-2 resin column

During Round 3 sampling, four additional sampling events took place to capture additional seasonal water flows (January 2006, September 2006, and October 2006), and a high-flow event was conducted in January 2007, with the following sampling rationale:

- **Round 3, Event 1 (January 2006) –** completed at the request of EPA during flood conditions; only two transect locations in the Study Area were sampled (at RM 4.0 and RM 11.0).
- **Round 3, Event 2 (September 2006) –** conducted during low-flow conditions when precipitation was minimal and groundwater discharge effects on the water column were anticipated to be high relative to river flow. Five Study Area transect locations (at RM 2.0, RM 4.0, RM 6.3, RM 11.0, and the mouth of Multnomah Channel) were sampled.[32]
- **Round 3, Event 3 (November 2006) –** selected to occur during the early rainy season in order to sample a storm of sufficient duration to result in substantial flow from major outfalls. Five Study Area transect locations (at RM 2.0, RM 4.0, RM 6.3, RM 11.0, and the mouth of Multnomah Channel) and 12 single-point locations were sampled.
- **Round 3, Event 4 (January – March 2007[33]) –** conducted during a high-flow period (i.e., when the discharge was forecasted to exceed 50,000 cfs for a 3-week period, which is the established lower threshold of significant sediment transport in the LWR). Five Study Area transect locations (at RM 2.0, RM 4.0, RM 6.3, RM 11.0, and the mouth of Multnomah Channel) and 12 single-point locations were sampled.

An additional upstream location was sampled at RM 16 during all Round 3 sampling events; this sample was not evaluated as part of the BERA but was used to establish background concentrations for the Study Area (see Section 7.0 of the draft final RI (Integral et al. 2011)). Several types of surface water samples were collected during the Round 2 and Round 3 events, including single-point near-bottom samples, single-point near-surface samples, and cross-sectional river (vertically and horizontally) transect water column samples. Surface water samples were collected using two sampling methods: a peristaltic pump or an Infiltrex™ 300 system with an XAD-2 resin column (XAD). Peristaltic pump surface water samples were collected using a low volume of water (10 to 20 L) over a short period of time (i.e., 30 minutes); samples were analyzed for conventional parameters and for various contaminants (i.e., metals, pesticides,

[32] Two additional cross-sectional river transect locations were added during Round 3 sampling at RM 2.0 and the mouth of the Multomah Channel. In addition, during Round 3 sampling, the transects at stations RM 2.0 and RM 11.0 were subdivided into three lateral segments across the river (i.e., east shoreline to navigation channel, navigation channel, and navigation channel to west shoreline).

[33] The high-flow event (Event 4) of Round 3 was conducted over 3 months in order to sample during the targeted high-flow period.

herbicides, and semivolatile organic compounds [SVOCs]). Surface water samples that were analyzed for hydrophobic organic compounds (i.e., dioxins and furans, PCBs, PAHs, pesticides, and phthalates) were collected using an XAD system for a high volume of water (i.e., approximately 34 to 1,000 L) at a pre-determined rate.

What Sampling Methods Were Used to Collect Surface Water?

Two types of sampling devices were used by LWG to collect surface water samples from the Study Area: peristaltic pump (as shown on the right) and XAD. Two general sample types were collected using these two sampling devices. Single-point samples were collected from discrete single locations either near the bottom (1 to 3 ft above the river bottom) or near the surface (1 to 3 ft below the water surface). Transect samples were composite samples collected from multiple discrete locations over a transect line. Transect samples were either horizontally integrated across the entire width of the river channel or were vertically integrated at various points (water depths) across the width of the river channel.

Samples analyzed for standard chemical and conventional variables were collected using a Standard MasterFlex peristaltic pump attached to a sampling tube. The sampling tube was lowered to the desired depth using a hydraulic or electric winch. Water was then pumped and collected through the sampling tube, and the outflow of the pump was directed through a Tee-splitter into two containers equipped with magnetic stirring devices. Equal volumes were pumped into each of two 20-L containers, one polycarbonate and the other stainless steel or glass. Samples to be analyzed for trace metals, butyltins, and conventional parameters were taken from the polycarbonate container. Samples to be analyzed for organic compounds were taken from the stainless steel or glass container. Both filtered and unfiltered samples were collected using the peristaltic pump. Filtered samples were collected by placing a 0.45-µm Teflon™ filter to the tubing outlet prior to collecting water in a sampling bottle.

XAD sampling enabled more precise analytical results for organic contaminants. The XAD sampling unit is designed to concentrate dissolved contaminants in surface water during the collection process. During sampling, large volumes of river water were pumped through Teflon™-lined polyethylene tubing, passing first through a 140-µm stainless-steel pre-filter and subsequently through a 0.5-µm glass fiber filter cartridge before passing through 250 g of Amberlite XAD-2 resin beads packed inside stainless-steel canisters. This procedure retains particulates on the filters and extracts dissolved organic contaminants onto the resin, eliminating the need to collect, store, and transport large volumes of water. The filter and the column analytical results were combined to determine total concentrations in the water column.

Calibration procedures were not performed for the peristaltic pumps. The calibration procedure for the XAD involved use of a 2-L graduated cylinder to measure the volume of water pumped through the XAD column every 30 minutes. Measured volume was then compared with the XAD flow meter reading to verify the accuracy of the total volume and rate pumped at the sampling location.

The XAD system also incorporated a glass fiber filter to capture the particulate fraction of the water and had an ultra-low analytical detection limit (DL) for hydrophobic organic compounds. The filter and the column analytical results were combined to determine total concentrations in the water column.

Surface water data were used to estimate exposure concentrations in surface water benthic invertebrates, fish, amphibians, and aquatic plants. The chemistry results from the Study Area surface water samples are presented in Attachment 4.

4.1.5 Transition Zone Water

TZW data used in the BERA dataset included all shallow (0 to 38 cm) data collected by LWG in Round 2, as well as non-LWG TZW samples collected using a Geoprobe adjacent to the Siltronic site, as indicated in Table 4-5 and shown in Map 4-16. During Round 2, TZW was sampled by LWG between October 3 and December 2, 2005, to characterize contaminated groundwater discharge to the LWR.

Table 4-5. Summary of Study Area Transition Zone Water Data Evaluated in the BERA

Sampling Site	Sampling Method	No. of Shallow Samples[a]	Contaminants Analyzed in Transition Zone Water
ARCO Terminal 22T	Peeper (unfiltered)	2	Metals, PAHs, petroleum, SVOCs, VOCs
	Trident® probe (filtered)	5	Metals, PAHs, petroleum, SVOCs
	Trident® probe (unfiltered)	5	Metals, PAHs, petroleum, SVOCs, VOCs
Arkema	Peeper (unfiltered)	8	Metals, PAHs, pesticides, SVOCs, VOCs
	Trident® probe (filtered)	10	Metals, pesticides
	Trident® probe (unfiltered)	12	Metals, PAHs, pesticides, SVOCs, VOCs
ExxonMobil Oil Terminal	Trident® probe (filtered)	12	Metals, PAHs, petroleum, SVOCs
	Trident® probe (unfiltered)	11	Metals, PAHs, petroleum, SVOCs, VOCs
Gasco	Peeper (unfiltered)	3	Metals, PAHs, petroleum, SVOCs, VOCs
	Trident® probe (filtered)	4	Metals, PAHs, petroleum, SVOCs
	Trident® probe (unfiltered)	5	Metals, PAHs, petroleum, SVOCs, VOCs
Gunderson	Peeper (unfiltered)	6	Metals, PAHs, SVOCs, VOCs
	Trident® probe (filtered)	2	Metals
	Trident® probe (unfiltered)	3	Metals, PAHs, SVOCs, VOCs
Kinder Morgan Linnton Terminal	Peeper (unfiltered)	5	Metals, PAHs, petroleum, SVOCs, VOCs
	Trident® probe (filtered)	3	Metals, PAHs, petroleum, SVOCs
	Trident® probe (unfiltered)	4	Metals, PAHs, petroleum, SVOCs, VOCs
Rhône-Poulenc	Peeper (unfiltered)	2	Herbicides, metals, PAHs, SVOCs, VOCs
	Trident® probe (filtered)	7	Dioxins and furans, herbicides, metals, pesticides
	Trident® probe (unfiltered)	8	Dioxins and furans, herbicides, metals, PAHs, pesticides, SVOCs, VOCs
Siltronic[b]	Peeper (unfiltered)	7	Metals, PAHs, petroleum, SVOCs, VOCs
	Trident® probe (filtered)	6	Metals, PAHs, petroleum, SVOCs
	Trident® probe (unfiltered)	6	Metals, PAHs, petroleum, SVOCs, VOCs
	Geoprobe (unfiltered)[b]	41	Metals, PAHs, SVOCs, VOCs

Table 4-5. Summary of Study Area Transition Zone Water Data Evaluated in the BERA

Sampling Site	Sampling Method	No. of Shallow Samples[a]	Contaminants Analyzed in Transition Zone Water
Willbridge Bulk Fuels Terminal	Peeper (unfiltered)	3	Metals, PAHs, petroleum, SVOCs, VOCs
	Trident® probe (filtered)	6	Metals, PAHs, petroleum, SVOCs
	Trident® probe (unfiltered)	6	Metals, PAHs, petroleum, SVOCs, VOCs

[a] Sample count includes field replicates.

[b] Non-LWG TZW data were collected from the Siltronic sampling site using a Geoprobe in May and June 2005 and were included in the BERA dataset. All other TZW data were collected by LWG from October to December 2005.

BERA – baseline ecological risk assessment
LWG – Lower Willamette Group
PAH – polycyclic aromatic hydrocarbon
SVOC – semivolatile organic compound
TZW – transition zone water
VOC – volatile organic compound

TZW samples were collected adjacent to the following nine sites: Kinder Morgan Linnton Terminal, ARCO Terminal 22T, ExxonMobil Oil terminal, Gasco, Siltronic, Rhône-Poulenc, former Arkema facility, Willbridge bulk fuels terminal, and Gunderson (see discussion in Section 6.6). TZW samples were collected with a Trident® (30-cm) probe, Geoprobe®, or small-volume peeper. TZW samples were collected at depths up to 150 cm; however, the TZW data in the BERA dataset included only samples from the interval that incorporated the biologically active zone (≤ 10 cm), where exposure of receptors could occur. The Trident probe is a direct-push system equipped with temperature, conductivity, and water sampling probes. With the Trident probe, TZW is collected through a small-diameter, Teflon®-coated, stainless steel probe with a port on the end covered by steel mesh. The Geoprobe® is comparable to the Trident probe. *In situ* porewater samplers (modified Hesslein samplers) were placed vertically in the sediment column by divers and left in place to equilibrate for a 3-week period. After the equilibration period, the peepers were retrieved and samples were collected by inserting a needle through the membrane to extract water (Integral 2006a).

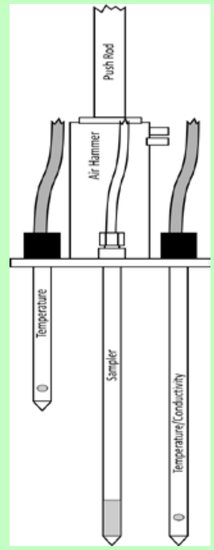
Shallow TZW data were used to estimate TZW exposure concentrations for benthic invertebrates, sculpin, lamprey ammocoetes, aquatic plants, and amphibians. The chemistry results from the Study Area TZW samples are presented in Attachment 4.

How Was Transition Zone Water Collected?

The transition zone is the interstitial area in which groundwater and surface water each make up some percentage of the water that occupies the space between sediment particles. The shallow TZW interval (0 to 38 cm) includes the depth at which exposure of receptors could occur. TZW samples were collected with a Trident® (30-cm) probe, Geoprobe®, or small-volume peeper.

The Trident (as shown on the right) is a direct-push system equipped with temperature, conductivity, and water sampling probes. Oxidation-reduction potential and pH were also measured at most sampling locations. With the Trident probe, TZW is collected through a small-diameter, Teflon®-coated, stainless steel probe that has a port on the end covered by steel mesh. Some samples were collected with a Geoprobe®, which is similar to the Trident probe.

Small-volume peepers were also used to sample TZW. Peepers are constructed of 6- by 18-in. acrylic plates machined to create multiple sample collection spaces or ports covered with a 5-µm membrane. Peeper ports were filled with anoxic distilled water prior to deployment. The peepers were vertically installed within the top 30 cm of sediment by divers. After an equilibration period, the peepers were retrieved, and samples were collected by inserting a needle through the membrane to extract water from the ports.



4.1.6 Qualitative and Reconnaissance-Level Data

Several reconnaissance surveys were conducted in support of the BERA. The results were not used directly to calculate risk estimates but rather to help characterize the ecological setting of the Study Area (Section 2.0) or to provide additional information for characterizing risks to specific receptor groups (in Sections 6.0 to 10.0). The following qualitative data were collected from the Study Area to support the BERA:

- **Field observations of fish health –** Observations of internal and external fish condition were noted during Round 3 sampling of juvenile white sturgeon in February and March 2007. A summary of the field observations is presented as part of the fish risk assessment in Section 7.0.
- **Descriptions of benthic macroinvertebrate (infaunal and epifaunal) community structure –** Benthic community structure data from the Study Area were collected as part of several sampling events. In October 2002, 22 grab samples were collected between RM 2.4 and RM 9.8 for taxonomic analysis of infaunal species. Twelve multiplate samplers were deployed for a 6-week period in the summer of 2002 between RM 3.5 and RM 9.2 and 10 multiplate samples were deployed for 6 weeks in summer 2005 between RM 2 and RM 11. Epifaunal species colonizing multiplate samplers were collected for tissue and taxonomic analyses in the summers of 2002 and 2005. Results from 2002 are reported in Attachment B2 of Appendix B of the Programmatic Work Plan (Integral et al. 2004b) and those from 2005 in a field sampling report[34]

[34] Samplers were deployed primarily for epibenthic and pelagic tissue collection. Taxonomic data were not included in the subsequent Comprehensive Round 2 Report (Integral et al. 2007).

(Windward 2005b). These data were used to help characterize the benthic invertebrate community in the Study Area as presented in Section 2.0.

- **Data from a reconnaissance survey of aquatic plant, amphibian, andreptile habitat –** A reconnaissance survey was conducted in summer 2002 to determine the presence or absence of aquatic plants, amphibians, and reptiles in the Study Area and to identify potential habitat areas for these receptors. Results are detailed in Attachment B2 of Appendix B of the Programmatic Work Plan (Integral et al. 2004b). The habitat areas identified are indicated in the amphibian and aquatic plant risk assessments in Sections 9.0 and 10.0, respectively.
- **Data from a reconnaissance survey of shorebird beach habitat –** During Round 2 sampling in June 2004, a shorebird beach habitat survey was conducted by EPA and LWG to identify potential shorebird beach habitats for sediment sampling and to document shorebird use of the Study Area. The habitat areas identified are included in the wildlife risk assessment in Section 8.0.

4.2 NON-STUDY AREA DATA

Per EPA, data collected from certain locations outside the boundaries of the Study Area were also evaluated in the BERA, as noted in Table 4-6 (sediment) and Table 4-7 (tissue):

- Downstream reach (RM 0 to RM 1.9)
- Multnomah Channel (LWR to Sauvie Island Bridge)
- Downtown reach (RM 11.8 to RM 15.3)

Table 4-6. Surface Summary of Non-Study Area Surface Sediment Data Evaluated in the BERA

LWR Reach	Sampling Event	Sampling Period	No. of Samples
Downstream reach (RM 0 – RM 1.9)	Round 3 sediment from upstream and downstream	January 2007	17
	Round 3B co-located sediments	October to November 2007	4
Multnomah Channel (LWR to Sauvie Island Bridge)	Round 3B sediment grabs	October to November 2007	7
Downtown reach (RM 11.8 – RM 15.3)	Round 3 sediment from upstream and downstream	January to February 2007	6
	Round 3B co-located sediments	October to December 2007	5
	Round 3B sediment cores	January 2008	1
	Round 3B sediment grabs	December 2007	5

BERA – baseline ecological risk assessment
LWR – Lower Willamette River

RM – river mile

Table 4-7. Summary of Non-Study Area Tissue Data Evaluated in the BERA

LWR Reach	Sampling Event and Sampling Period	Species	No. of Samples[a]
Downstream reach (RM 0 – RM 1.9)	Round 3B biota sampling, August to November 2007	Clam	1
	Round 3B biota sampling, August to November 2007	Crayfish	2
	Round 3B biota sampling, August to November 2007	Sculpin	2
Multnomah Channel (Sauvie Island Bridge to mouth of Columbia River)	USGS/USFWS osprey egg tissue sampling, May 2008 to September 2009	Osprey	5[b]
Mid-Willamette River (RM 69 to RM 77)	USGS/USFWS osprey egg tissue sampling, May 2008 to September 2009	Osprey	5[b]
Downtown reach (RM 11.8 – RM 15.3)	Round 3B biota sampling, August to November 2007	Clam	2
	Round 3B biota sampling, August to November 2007	Crayfish	2
	Round 3B biota sampling, August to November 2007	Sculpin	2

[a] Whole-body tissue composites, except where noted; sample count includes field replicates.
[b] Individual bird egg tissue samples.

BERA – baseline ecological risk assessment
LWR – Lower Willamette River
RM – river mile
USFWS – US Fish and Wildlife Service
USGS – US Geological Survey

These non-Study Area sediment and tissue data were compared with Study Area data, and risk to ecological receptors from these samples was also evaluated. No surface water was collected from these non-Study Area reaches. The chemistry results of non-Study Area sediment and tissue data from these reaches are presented in Attachment 4.

4.3 UPRIVER REACH DATA

Surface sediment and fish tissue data collected from the upriver reach of the LWR (from RM 15.3 to RM 28.4) were also evaluated as part of the BERA. Upriver sediment data were evaluated to define sediment bioassay testing samples to represent reference conditions in the benthic toxicity model (Section 6.0). Upriver fish tissue data were used for comparison purposes in evaluating tissue residues in Study Area fish (Section 7.0).

Surface sediment and surface water data collected from the upriver reach were also compiled to establish background concentrations for the Study Area. These data and methods used to define background concentrations are presented in Section 7.0 of the draft final RI (Integral et al. 2011).

4.3.1 Surface Sediment

Sediment toxicity tests and chemistry analyses were conducted on 22 sediment samples collected by LWG in Round 2 and Round 3 from the upriver reach (RM 15.3 to RM 28.4). Eighteen locations were sampled during Round 2 activity in November 2004, and four additional locations were sampled during Round 3 activity in November 2007. Surface sediment data included in the upriver reach dataset were collected from within the top 30.5 cm of the sediment horizon. The chemistry and bioassay results of surface sediment samples from the upriver reach are presented in Section 7.0 of the draft final RI (Integral et al. 2011).

4.3.2 Biota Tissue

Fish tissue chemistry data collected during four LWG sampling events include samples from the upriver reach (Table 4-8). Whole-body tissue composites from upriver locations were collected for four fish species: juvenile Chinook salmon, smallmouth bass, brown bullhead, and Pacific lamprey (as ammocoetes and macropthalmia). A single composite sample of juvenile Chinook salmon stomach contents collected at RM 17 was also included in the upriver dataset. Osprey egg tissue collected by USGS and USFWS in May 2008 to September 2009 were collected from the upriver reach (Table 4-8). The chemistry results from the fish and bird egg tissue samples collected from the upriver reach are presented in Attachment 4.

Table 4-8. Summary of Non-Study Area Tissue Data Evaluated in the BERA

LWR Reach	Sampling Event and Sampling Period	Species	No. of Samples[a]
Upriver reach (RM 15.3 to RM 28.4)	Round 1A tissue sampling, June 2002 (above RM 26)	Juvenile Chinook salmon	1
	Round 1 tissue sampling, October to November 2002 (RM 21 to RM 23.8)	Brown bullhead	3
		Smallmouth bass	6
	Round 2A juvenile Chinook salmon tissue sampling, May 2005 (RM 17)	Juvenile Chinook salmon (stomach contents)	1
		Juvenile Chinook salmon	3
	Round 3, lamprey tissue sampling, September 2006 (RM 17 and RM 19)	Pacific lamprey ammocoete (n = 3) and macropthalmia (n = 1)	4

[a] Whole-body tissue composite samples only, except where noted; sample count includes field replicates.

BERA – baseline ecological risk assessment

LWR – Lower Willamette River

RM – river mile

USFWS – US Fish and Wildlife Service

USGS – US Geological Survey

5.0 IDENTIFICATION OF COPCS

As stated in Section 3.1, ecological COPCs were identified using the complete Study Area dataset identified for the BERA (Section 4.0). COPCs were identified in order to focus the list of contaminants for quantitative evaluation of ecological risks. COPCs were identified by conducting a screening-level analysis in which maximum concentrations in various media are compared with conservative thresholds of toxicity. Contaminants whose maximum concentrations do not exceed conservative thresholds of toxicity and that are not identified as COPCs were excluded from further consideration as a source of unacceptable risk to ecological receptors.

A comprehensive COPC screen was conducted again for this BERA using all data available in the BERA dataset (from Rounds 1, 2, and 3). The screening of COPCs in this BERA was conducted in two tiers as directed by EPA (2008j). The first tier was a SLERA in which maximum concentrations (i.e., maximum detected concentration or maximum DL) in surface sediment, tissue, surface water, TZW, and dietary doses were compared with screening-level thresholds provided by EPA. The second tier was a refined screen of contaminants that passed through the first tier. In the refined screen, maximum detected concentrations were compared with screening-level thresholds and additional factors were evaluated (e.g., frequency of detection, nutritional role) in order to identify ecological COPCs (see Attachment 5 for additional details).

The following COPCs were identified across all LOEs for each receptor group:

- **Invertebrates –** 106 COPCs, consisting of 20 metals, 2 butyltins, 21 individual PAHs or PAH sums, 4 phthalates, 12 SVOCs, 6 phenols, 16 pesticide or pesticide sums, total PCBs, 2,3,7,8-tetrachlorodibenzo-*p*-dioxin (TCDD), 18 volatile organic compounds (VOCs), 3 total petroleum hydrocarbons (TPH), cyanide, and perchlorate
- **Fishes –** 72 COPCs, consisting of 19 metals, 4 butyltins, 17 individual PAHs or PAH sums, bis(2-ethylhexyl) phthalate (BEHP), 3 SVOCs, total PCBs, 7 pesticide and pesticide sums, 18 VOCs, cyanide, and perchlorate
- **Wildlife (birds and mammals) –** 24 COPCs, consisting of 11 metals, 4 individual PAHs or PAH sums, 2 phthalates, total PCBs, dioxin toxic equivalent (TEQ), PCB TEQ, total TEQ, and 3 pesticide or pesticide sums
- **Amphibians and aquatic plants –** 65 COPCs, consisting of 6 metals, monobutyltin, 16 individual PAHs, BEHP, 3 SVOCs, total PCBs, 6 pesticide or pesticide sums, 18 VOCs, gasoline-range hydrocarbons, cyanide, and perchlorate

The complete SLERA and refined screening process, including the thresholds used, are presented in Attachment 5. The following sections summarize the methods and results of the screening process (for both the SLERA and refined screen):

- Section 5.1 summarizes the SLERA and refined screen process used to identify COPCs.
- Sections 5.2, 5.3, 5.4, and 5.5 present the COPCs identified for each ecological receptor group: benthic invertebrates, fish, wildlife (birds and mammals), and amphibians/aquatic plants, respectively.

5.1 SUMMARY OF SLERA AND REFINED SCREEN

The screening of COPCs in this BERA was conducted in two tiers as directed by EPA (2008j). The first tier was a SLERA, and the second tier was a refined screen of contaminants that passed through the first tier.

5.1.1 SLERA

The SLERA was conducted as the first-tier screening step for identifying ecological COPCs. Figure 5-1 presents a flow chart of the SLERA process.

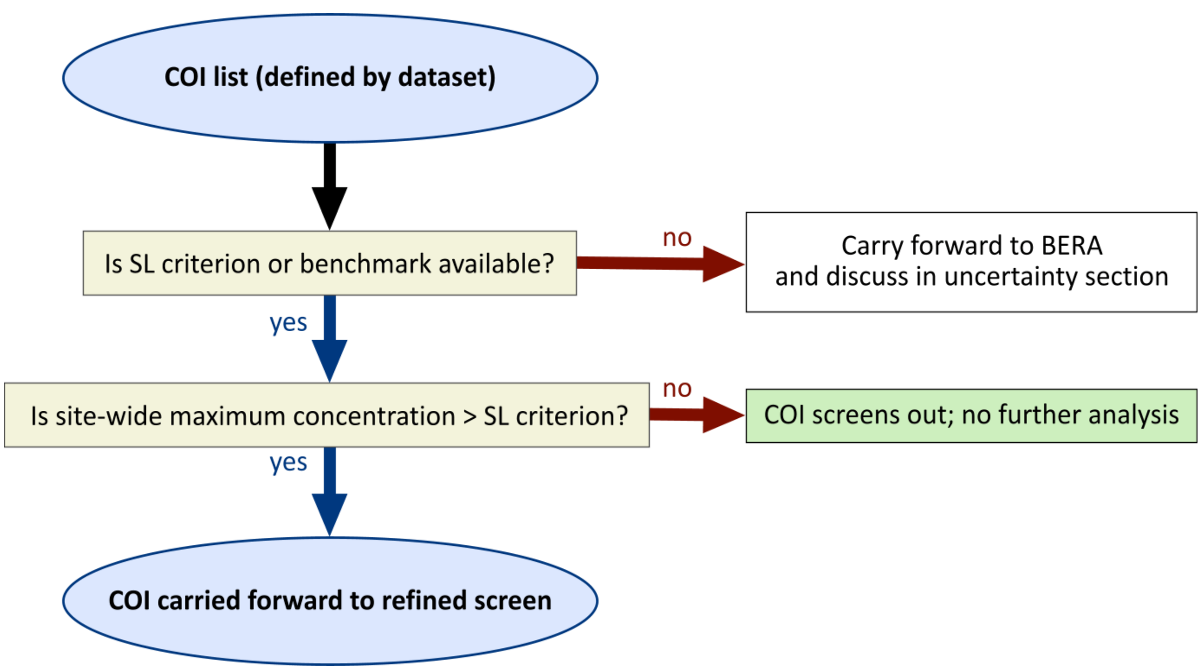


Figure 5-1. The SLERA Process – Step 1 for Identifying COPCs

In the SLERA, all contaminants detected in each medium (i.e., surface sediment, benthic invertebrate tissue, fish tissue,[35] surface water, and shallow TZW) from the Study Area in the BERA dataset were identified as medium-specific COIs. COIs for the wildlife dietary assessment were those contaminants detected in both surface sediment and tissue of fish or invertebrates. Metals and PAHs detected in both surface sediment and tissue samples were identified as COIs for the fish dietary assessment.

COIs were screened in each medium and for receptor-specific dietary scenarios. For benthic invertebrates and fish tissue, COIs were screened on a receptor-specific basis. Maximum COI concentrations (i.e., maximum detected values or maximum DLs) were compared with medium-specific (or dietary) screening-level thresholds. COIs were dropped from further consideration if the maximum value did not exceed the screening-level thresholds. COIs within a given medium that exceeded screening-level thresholds were retained for further evaluation in the refined screen. COIs with no screening-level thresholds were not evaluated but were retained for discussion in the relevant uncertainty section.

5.1.2 Refined Screen

The refined screen was conducted as the second step for identifying ecological COPCs. Figure 5-2 presents a flow chart of the refined screen process.

The COIs carried forward from the SLERA were evaluated in the refined screen using the following steps:

- **Steps 1 and 2 –** For each medium and for the dietary assessment, the maximum detected concentrations for each COI were first compared with the respective screening-level thresholds. If detected concentrations did not exceed screening-level thresholds, the COI was not evaluated further.
- **Step 3 –** For COIs retained in Step 2, if the detection frequency was less than 5% and the maximum DL was less than the screening-level threshold, the COI was evaluated further based on three considerations: medium, magnitude of exceedance, and bioaccumulation potential. If the medium was surface water, shallow TZW, clam tissue, crayfish tissue, sculpin tissue, or smallmouth bass tissue, the COI was retained. The COI was also retained for further evaluation if the maximum detected concentration was at least 5 times as great as the screening threshold or the log K_{OW} of an organic COI was greater than or equal to 4.0.

[35] PAHs were not retained as COIs for benthic invertebrates or fish tissue but were evaluated for more relevant exposure pathways, per agreement with EPA.

- **Step 4 –** Concentrations of tissue or dietary COIs retained in Step 2 or 3 that were also identified as nutritionally essential were compared with available nutritional information. The tissue or dietary COI was eliminated if the maximum detected concentration was less than the "nutritionally essential" concentration.[36]

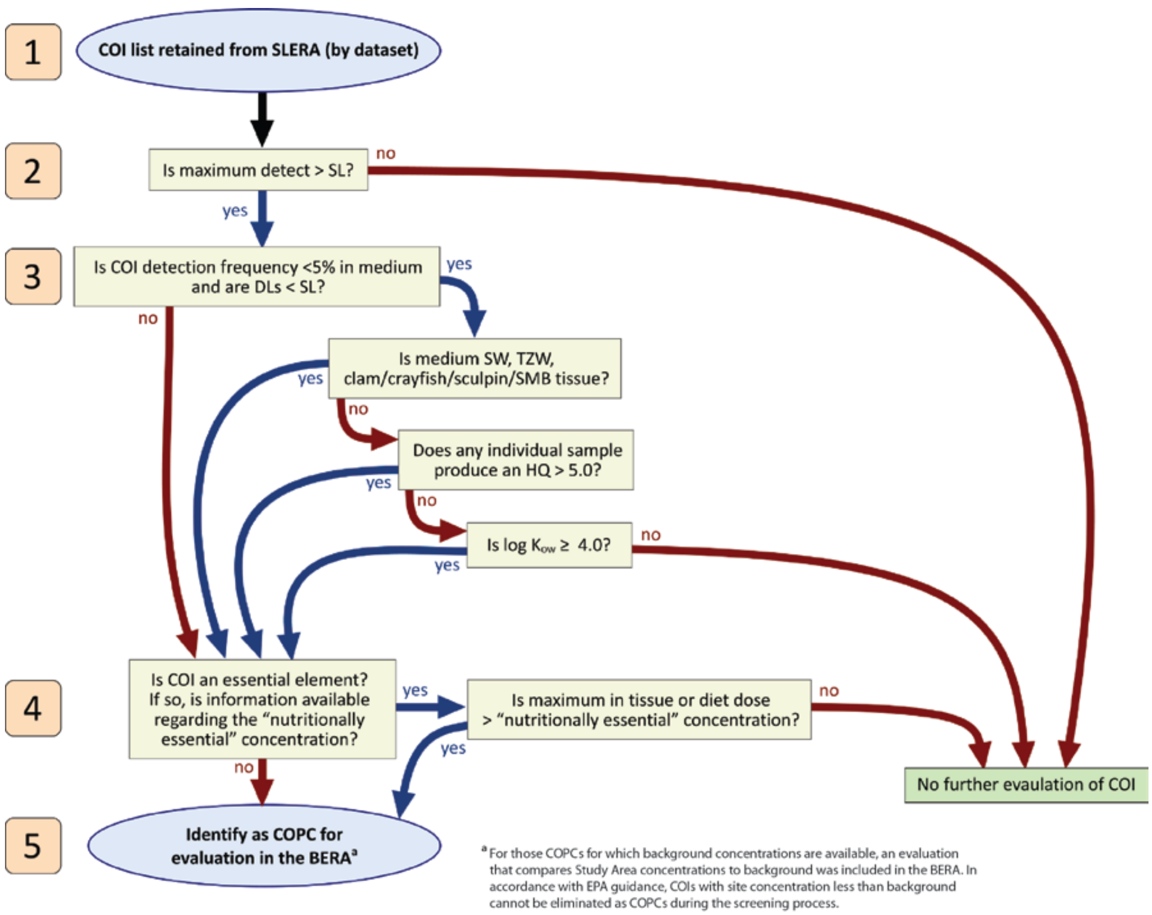


Figure 5-2. The Refined Screening Process – Step 2 for Identifying COPCs

5.1.3 Identification of COPCs

As a final step of the refined screen (Step 5), COPCs were identified for each abiotic medium (i.e., surface sediment, surface water, and shallow TZW), receptor-specific benthic invertebrates, receptor-specific fish tissue, and receptor-specific dietary scenario. COIs that did not screen out during the steps of the refined screen were identified as ecological COPCs for evaluation in the BERA. These COPCs are presented by receptor group in the following sections.

[36] Ultimately, the nutritional role of selected metals was not used to eliminate any COI; as required by EPA, Step 4 was not implemented because definitive information of high certainty was lacking.

5.2 BENTHIC INVERTEBRATE COPCS

COPCs for benthic invertebrates were identified for surface sediment, tissue, surface water, and shallow TZW. These COPCs were further evaluated in the benthic risk assessment, as described in Section 6.0.

The screening thresholds used in the SLERA and refined screen varied for each medium. Screening thresholds are presented in Attachment 5. The following screening thresholds were used to derive benthic invertebrate COPCs:

- **Surface sediment –** The lowest of the SQGs provided by EPA was used to screen surface sediment data. The TPH SQGs were derived by EPA and its partners using the Alaska TPH TRVs (EPA 2008a).
- **Tissue –** Thresholds for screening aquatic tissue residues are either based on a fifth percentile lowest-observed-adverse-effect level (LOAEL) (derived from Appendix B of the Ecological PRE (Windward 2005a) or from Dyer et al. (2000)) or calculated as the product of EPA ambient water quality criteria (AWQC) and a bioconcentration factor (BCF). The approach for developing aquatic tissue-residue screening-level thresholds was developed by EPA and its partners (EPA 2005g) for data evaluation in the Ecological PRE. The tissue residue screening threshold for 2,3,7,8-TCDD is based on Isensee (1978).
- **Surface Water and TZW –** Surface water and TZW screening-level thresholds are represented by chronic water TRVs. Water TRVs were developed using a selection hierarchy, agreed upon by LWG and EPA, of water quality regulatory thresholds and literature-based thresholds, including national and proposed State of Oregon water quality standards (WQS), Tier II values (Suter and Tsao 1996), final chronic values (FCVs) for individual PAH compounds (Table 3-4 of EPA 2003c), Canadian water environmental quality guidelines, Oregon Department of Environmental Quality (ODEQ) guidance values (ODEQ 2006), or literature-derived values. EPA provided TRVs for five of the chemical groups that are blended to form gasoline (EPA 2008a). Average fractions of these components in gasoline were used to convert the total gasoline-range hydrocarbon concentration into gasoline fraction concentrations for comparison with the TRV. Any one gasoline fraction exceeding its TRV was grounds for identifying gasoline as a COPC.

The screening resulted in identification of 104 COPCs for benthic invertebrates across all four media, as listed in Table 5-1: 20 metals, 2 butyltins, 21 individual PAHs or PAH sums, 4 phthalates, 12 SVOCs, 6 phenols, 16 pesticide or pesticide sums, total PCBs, 2,3,7,8-TCDD, 16 VOCs, 3 TPH, cyanide, and perchlorate. Details are presented in Attachment 5.

Table 5-1. Benthic Invertebrate COPCs

	LOE			
COPC	**Sediment**	**Tissue**	**Surface Water**	**TZW**
Metals				
Aluminum		X		
Arsenic	X	X		
Barium				X
Beryllium				X
Cadmium	X	X		X
Chromium	X			
Cobalt				X
Copper	X	X		X
Iron				X
Lead	X			X
Magnesium				X
Manganese	X			X
Mercury	X			
Nickel	X			X
Potassium				X
Selenium	X			
Silver	X			
Sodium				X
Vanadium				X
Zinc	X	X	X	X
Butyltins				
Monobutyltin ion			X	
Tributyltin ion		X		
PAHs				
2-Methylnaphthalene	X			X
Acenaphthene	X			X
Acenaphthylene	X			
Anthracene	X			X
Benzo(a)anthracene	X		X	X
Benzo(a)pyrene	X		X	X
Benzo(b)fluoranthene				X
Benzo(g,h,i)perylene	X			X

Table 5-1. Benthic Invertebrate COPCs

	LOE			
COPC	**Sediment**	**Tissue**	**Surface Water**	**TZW**
Benzo(k)fluoranthene	X			X
Chrysene	X			X
Dibenzo(a,h)anthracene	X			X
Fluoranthene	X			X
Fluorene	X			X
Total HPAHs	X			
Indeno(1,2,3-cd)pyrene	X			X
Total LPAHs	X			
Naphthalene	X		X	X
Total PAHs	X			
Phenanthrene	X			X
Pyrene	X			X
Total benzofluoranthenes	X			
Phthalates				
BEHP	X	X	X	
Butyl benzyl phthalate	X			
Dibutyl phthalate		X		
Di-n-octyl phthalate	X			
SVOCs				
1,2-Dichlorobenzene				X
1,4-Dichlorobenzene				X
1,2,4-Trichlorobenzene	X			
1,2-Dichlorobenzene	X			
1,4-Dichlorobenzene	X			
Benzoic acid	X			
Benzyl alcohol	X			
Carbazole	X			
Dibenzofuran	X			X
Hexachlorobenzene	X			
Hexachlorobutadiene	X			
n-Nitrosodiphenylamine	X			
Phenols				
2,4-Dimethylphenol	X			

Table 5-1. Benthic Invertebrate COPCs

	LOE			
COPC	**Sediment**	**Tissue**	**Surface Water**	**TZW**
2-Methylphenol	X			
4-Methylphenol	X			
4-Nitrophenol				
Pentachlorophenol	X			
Phenol	X			
PCBs				
Total PCBs	X	X	X	
Dioxins and Furans				
2,3,7,8-TCDD	X			
Pesticides				
2,4′-DDD[a]			X	X
2,4′-DDT[a]			X	X
4,4′-DDD[a]	X	X	X	X
4,4′-DDE[a]	X			X
4,4′-DDT			X	X
Sum DDD	X			
Sum DDE	X			
Sum DDT	X			
Total DDx	X	X	X	X
Aldrin	X			
Chlordane (cis & trans)	X			
Total chlordane	X			
Dieldrin	X			
Endrin	X			
Heptachlor epoxide	X			
gamma-HCH	X			
VOCs				
1,1-Dichloroethene				X
cis-1,2-Dichloroethene				X
1,2,4-Trimethylbenzene				X
1,3,5-Trimethylbenzene				X
Benzene				X
Carbon disulfide				X

Table 5-1. Benthic Invertebrate COPCs

	LOE			
COPC	**Sediment**	**Tissue**	**Surface Water**	**TZW**
Chlorobenzene				X
Chloroethane				X
Chloroform				X
Ethylbenzene			X	X
Isopropylbenzene				X
Toluene				X
Trichloroethene			X	X
m,p-Xylene				X
o-Xylene				X
Total xylenes				X
TPH				
Diesel-range hydrocarbons	X			
Gasoline-range hydrocarbons	X			X
Residual-range hydrocarbons	X			
Other Contaminants				
Cyanide				X
Perchlorate				X

[a] These DDT metabolites in surface water and TZW were evaluated as a component of total DDx.

BEHP – bis(2-ethylhexyl) phthalate
COPC – contaminant of potential concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
HCH – hexachlorocyclohexane
HPAH – high-molecular-weight polycyclic aromatic hydrocarbon
LOE – line of evidence
LPAH – low-molecular-weight polycyclic aromatic hydrocarbon
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
SVOC – semivolatile organic compound
TCDD – tetrachlorodibenzo-*p*-dioxin
total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT and 4,4′-DDT)
TPH – total petroleum hydrocarbons
TZW – transition zone water
VOC – volatile organic compound

As shown in Table 5-2, nine COIs were not identified as a COPC for at least one medium despite having been retained from the SLERA (i.e., maximum value exceeded a screening-level threshold). The rationale for excluding these COIs as a COPC for a given medium is that no detected concentration exceeded the corresponding screening-level threshold. The potential risks to benthic invertebrates from these contaminants are unknown. The percentage of samples where the COI was undetected but the DL exceeded the screening-level TRVs is noted in Table 5-2. This percentage is low (< 30%) for most contaminants: diethyl phthalate, dimethyl phthalate, 1,3-dichlorobenzene, and heptachlor (in sediment); diethyl phthalate and beta-hexachlorocyclohexane (HCH) (in tissue); 2,4′-dichlorodiphenyldichloroethylene (DDE) (in surface water); and selenium and styrene (in TZW). The DLs for most crayfish tissue samples (84%) exceeded TRVs for dibutyl phthalate and dimethyl phthalate. A COI that was not retained as a COPC for a particular medium may have been retained as a COPC for another medium or for a subset of the same medium. For example, the COI dibutyl phthalate was retained as a tissue COPC for field-collected clams and laboratory-exposed worms but not for crayfish.

Some COIs could not be screened because no screening-level thresholds were available. These COIs are listed in Table 5-3.

Table 5-2. Benthic Invertebrate COIs Not Retained as COPCs Following the Refined Screen

Chemical	COI Carried Over from SLERA but Not Retained as COPC				Rationale for Exclusion
	Sediment	Tissue	Surface Water	TZW	
Metals					
Selenium				X	Although 26% of non-detected TZW samples had DLs > screening-level TRV, no detected TZW concentration was > screening-level TRV. Selenium was retained as a sediment COPC because maximum detected concentration was > screening-level TRV.
Phthalates					
Dibutyl phthalate		X, for cray-fish only			Although 84% of non-detected crayfish tissue samples had DLs > screening-level TRV, contaminant was never detected in crayfish tissue. Dibutyl phthalate was retained as a tissue COPC for field-collected clams and laboratory-exposed worms because their maximum detected concentrations were > screening-level TRV.
Diethyl phthalate	X	X			Less than 1% of non-detected sediment samples had DLs > SQG; no detected sediment concentration was > SQG. Although 13% of non-detected crayfish tissue samples had DLs > screening-level TRV, contaminant was never detected in crayfish tissue when DL was < screening-level TRV.
Dimethyl phthalate	X	X			Less than 1% of non-detected sediment samples had DLs > SQG; no detected sediment concentration was > SQG. Although 8% of non-detected clam and 84% of non-detected crayfish tissue samples were > screening-level TRV, contaminant was never detected in clam or crayfish tissue when DL was < screening-level TRV.
SVOCs					
1,3-Dichlorobenzene	X				Less than 1% of non-detected sediment samples had DLs > SQG; no detected sediment concentration was > SQG.
VOCs					
Styrene				X	Less than 1% of non-detected TZW samples had DLs > screening-level TRV; but no detected TZW concentration was > screening-level TRV.

Table 5-2. Benthic Invertebrate COIs Not Retained as COPCs Following the Refined Screen

Chemical	COI Carried Over from SLERA but Not Retained as COPC				Rationale for Exclusion
	Sediment	Tissue	Surface Water	TZW	
Pesticides					
2,4′-DDE			X		Less than 1% of non-detected surface water samples had DLs > screening-level TRVs; no detected surface water concentration was > screening-level TRV. Total DDx (which includes 2,4′-DDE) was evaluated as surface water COPC because maximum detected total DDx concentration was > screening-level TRV.
Heptachlor	X				Only 1% of non-detected sediment samples had DLs > SQG; no detected sediment concentration was > SQG.
beta-HCH		X			Only 3% of non-detected clam tissue samples had DLs > screening-level TRV; no detected clam tissue result was > screening-level TRV.

Note: Exclusion as a COPC for a given medium does not preclude retention as a COPC for another medium.

COI – contaminant of interest
COPC – contaminant of potential concern
DL – detection limit
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
HCH – hexachlorocyclohexane
SLERA – screening-level ecological risk assessment
SQG – sediment quality guideline
SVOC – semivolatile organic compound
total DDx – sum of all six DDT isomers (2,4′-DDD; 4,4′-DDD; 2,4′-DDE; 4,4′-DDE; 2,4′-DDT; and 4,4′-DDT)
TRV – toxicity reference value
TZW – transition zone water
VOC – volatile organic compound

Table 5-3. Benthic Invertebrate COIs with No TRVs

	No TRV Available[b]			
COI[a]	**Sediment**	**Tissue**	**Surface Water**	**TZW**
Metals				
Aluminum	X		X	X
Calcium				X
Chromium (hexavalent)	X			
Cobalt	X			
Magnesium	X			
Manganese		X		
Thallium	X			
Tin	X			
Titanium	X			X
Vanadium	X			
Butyltins				
Monobutyltin ion	X	X		
Dibutyltin ion	X	X		
Tetrabutyltin	X	X		
Tributyltin ion	X			
PAHs				
1,6,7-Trimethylnaphthalene	X			
1-Methylnaphthalene	X			
1-Methylphenanthrene	X			
2,6-Dimethylnaphthalene	X			
Benzo(e)pyrene	X			
Dibenzothiophene	X			
Perylene	X			
Dioxins and Furans				
1,2,3,4,6,7,8-Heptachlorodibenzofuran	X	X	X	
1,2,3,4,6,7,8-Heptachlorodibenzo-*p*-dioxin	X	X	X	X
1,2,3,4,7,8,9-Heptachlorodibenzofuran	X	X	X	X
1,2,3,4,7,8-Hexachlorodibenzofuran	X	X	X	X
1,2,3,4,7,8-Hexachlorodibenzo-*p*-dioxin	X	X	X	
1,2,3,6,7,8-Hexachlorodibenzofuran	X	X	X	X
1,2,3,6,7,8-Hexachlorodibenzo-*p*-dioxin	X	X	X	
1,2,3,7,8,9-Hexachlorodibenzofuran	X	X	X	

Table 5-3. Benthic Invertebrate COIs with No TRVs

	No TRV Available[b]			
COI[a]	Sediment	Tissue	Surface Water	TZW
1,2,3,7,8,9-Hexachlorodibenzo-*p*-dioxin	X	X	X	
1,2,3,7,8-Pentachlorodibenzofuran	X	X	X	X
1,2,3,7,8-Pentachlorodibenzo-*p*-dioxin	X	X	X	
2,3,4,6,7,8-Hexachlorodibenzofuran	X	X	X	
2,3,4,7,8-Pentachlorodibenzofuran	X	X	X	X
2,3,7,8-Tetrachlorodibenzofuran	X	X	X	X
VOCs				
1,1,1,2-Tetrachloroethane	X			
1,1,2,2-Tetrachloroethane	X			
1,1-Dichloroethene	X			
1,2,3-Trichloropropane	X			
1,2-Dichloroethane	X			
Acetone	X			
Benzene	X			
Chloroform	X			
cis-1,2-Dichloroethene	X			
Dichlorodifluoromethane	X			
Ethylbenzene	X			
Isopropylbenzene	X			
Methyl isobutyl ketone	X			
Methyl n-butyl ketone	X			
Methyl tert-butyl ether	X			
Methylene chloride	X			
Methylethyl ketone	X			
Styrene	X			
Toluene	X			
trans-1,2-Dichloroethene	X			
Vinyl chloride	X			
m,p-Xylene	X			
o-Xylene	X			
Total xylenes	X			
SVOCs				
2-Chloronaphthalene	X			

Table 5-3. Benthic Invertebrate COIs with No TRVs

	No TRV Available[b]			
COI[a]	**Sediment**	**Tissue**	**Surface Water**	**TZW**
3-Nitroaniline	X			
4-Chloroaniline	X		X	
4-Nitroaniline	X			
Aniline	X		X	
Bis(2-chloro-1-methylethyl) ether	X			
Bis(2-chloroethoxy) methane		X		
Bis(2-chloroisopropyl) ether	X			
Benzoic acid		X		
Benzyl alcohol		X		
Diphenyl	X			
Hexachloroethane	X			
Nitrobenzene		X		
Phenols				
2,3,4,5-Tetrachlorophenol	X			
2,3,4,6-Tetrachlorophenol	X			
2,3,5,6-Tetrachlorophenol	X			
2,4,5-Trichlorophenol	X			
2,4,6-Trichlorophenol	X			
2,4-Dichlorophenol	X			
2-Chlorophenol	X			
4-Chloro-3-methylphenol	X			
4-Nitrophenol		X		
Pesticides				
alpha-Endosulfan	X			
beta-Endosulfan	X			
Total endosulfan	X			
Endosulfan sulfate	X			
Endrin aldehyde	X			
Endrin ketone	X			
alpha-HCH	X			
beta-HCH	X			
delta-HCH	X			
Methoxychlor	X			

Table 5-3. Benthic Invertebrate COIs with No TRVs

	No TRV Available[b]			
COI[a]	Sediment	Tissue	Surface Water	TZW
Mirex	X			
Toxaphene	X			
TPH				
Residual-range hydrocarbons				X
Diesel-range hydrocarbons				X
Total diesel-residual hydrocarbons				X
TPH				X
Herbicides				
2,4,5-T	X			
2,4-D	X			
2,4-DB	X		X	
Dichloroprop	X			
MCPA	X			
MCPP	X		X	
Silvex	X			
Other Contaminants				
Ammonia	X			
Cyanide	X			
Perchlorate	X			

[a] Includes benthic invertebrate COIs based on any medium (i.e., sediment, tissue, surface water, or TZW) for which no screening TRV is available. If a COI is a component of a total for which there is a screening-level (i.e., components of total DDx and total chlordane), the component COIs are not included in this table, because these COIs were evaluated as part of a sum.

[b] Blank cell indicates that contaminant is not a COI for a given medium (i.e., contaminant was either not analyzed or not detected) or TRV was identified.

2,4-D – 2,4-dichlorophenoxyacetic acid
2,4-DB – 4-(2,4-dichlorophenoxy)butyric acid
2,4,5-T – 2,4,5-trichlorophenoxyacetic acid
COI – contaminant of interest
MCPA – 2-methyl-4-chlorophenoxyacetic acid
MCPP – methylchlorophenoxypropionic acid
PAH – polycyclic aromatic hydrocarbon
SVOC – semivolatile organic compound
TPH – total petroleum hydrocarbons
TRV – toxicity reference value
TZW – transition zone water
VOC – volatile organic compound

5.3 FISH COPCS

The SLERA and refined screen identified COPCs for fish in tissue, surface water, and shallow TZW. These COPCs were further evaluated in the fish risk assessment (Section 7.0). Diet COPCs were also identified for fish to be evaluated in the dietary risk LOE.

The screening thresholds used in the SLERA and refined screen varied for each medium. Screening thresholds are presented in Attachment 5. The following screening thresholds were used to derive fish COPCs:

- **Tissue –** Aquatic tissue-residue screening-level thresholds are either based on a fifth percentile LOAEL (derived from Appendix B of the Ecological PRE (Windward 2005a) or from Dyer et al. (2000)) or calculated as the product of EPA AWQC and a BCF. The approach for developing aquatic tissue-residue screening-level thresholds was developed by EPA and its partners (EPA 2005g) for data evaluation in the Ecological PRE. A fifth percentile LOAEL was considered an appropriate threshold for screening.
- **Diet –** Receptor-specific diet-based screening-level thresholds were derived for prey tissue and sediment based on dietary-dose screening-level thresholds (expressed as mg/kg body weight [bw]/day). Provided by EPA (2008f), fish dietary-dose screening-level thresholds are based on no-observed-adverse-effect level (NOAEL) TRVs derived from the toxicological literature. These diet-based screening-level thresholds were used to screen prey tissue and sediment concentrations.
- **Surface Water and TZW –** Surface water and TZW screening-level thresholds are represented by chronic water TRVs. These TRVs were developed using a selection hierarchy, agreed upon by LWG and EPA, of water quality regulatory thresholds and literature-based thresholds, including national and proposed State of Oregon WQS, Tier II values (Suter and Tsao 1996), FCVs for individual PAH compounds (Table 3-4 of EPA 2003c), Canadian water environmental quality guidelines, ODEQ guidance values (ODEQ 2006), or literature-derived values. EPA provided TRVs for five of the chemical groups that are blended to form gasoline (EPA 2008a). Average fractions of these components in gasoline were used to convert the total gasoline-range hydrocarbon concentration into gasoline fraction concentrations for comparison with the TRVs. Any one gasoline fraction exceeding its TRV was grounds for identifying gasoline as a COPC.

As listed in Table 5-4, 72 COPCs were identified for fish based on the SLERA and refined screening steps across all media and for the dietary LOE: 19 metals, 4 butyltins, 17 individual PAHs or PAH sums, BEHP, 3 SVOCs, total PCBs, 7 pesticide and pesticide sums, 18 VOCs, cyanide, and perchlorate. Details are provided in Attachment 5.

Table 5-4. Fish COPCs

	LOE			
COPC	**Tissue**	**Diet**	**Surface Water**	**TZW**
Metals				
Aluminum	X			
Antimony	X			
Barium				X
Beryllium				X
Cadmium	X	X		X
Chromium	X			
Cobalt				X
Copper	X	X		X
Iron				X
Lead	X			X
Magnesium				X
Manganese				X
Mercury	X	X		
Nickel				X
Potassium				X
Selenium				X
Sodium				X
Vanadium				X
Zinc	X		X	X
Butyltins				
Monobutyltin ion		X	X	
Dibutyltin ion		X		
Tetrabutyltin		X		
Tributyltin ion		X		
PAHs				
2-Methylnaphthalene				X
Acenaphthene				X
Anthracene				X
Benzo(a)anthracene			X	X
Benzo(a)pyrene		X	X	X
Benzo(b)fluoranthene				X
Benzo(g,h,i)perylene				X
Benzo(k)fluoranthene				X
Chrysene				X

Table 5-4. Fish COPCs

	LOE			
COPC	**Tissue**	**Diet**	**Surface Water**	**TZW**
Dibenzo(a,h)anthracene				X
Fluoranthene				X
Fluorene				X
Indeno(1,2,3-cd)pyrene				X
Naphthalene			X	X
Total PAHs		X		
Phenanthrene				X
Pyrene				X
Phthalates				
BEHP	X		X	
SVOCs				
1,2-Dichlorobenzene				X
1,4-Dichlorobenzene				X
Dibenzofuran				X
PCBs				
Total PCBs	X		X	
Pesticides				
2,4′-DDD[a]			X	X
2,4′-DDT[a]			X	X
4,4′-DDD[a]	X		X	X
4,4′-DDE[a]				X
4,4′-DDT	X		X	X
Total DDx	X		X	X
beta-HCH	X			
VOCs				
1,1-Dichloroethene				X
1,2,4-Trimethylbenzene				X
1,3,5-Trimethylbenzene				X
Acrolein				X
Benzene				X
Carbon disulfide				X
Chlorobenzene				X
Chloroethane				X
Chloroform				X
cis-1,2-Dichloroethene				X

Table 5-4. Fish COPCs

COPC	LOE			
	Tissue	Diet	Surface Water	TZW
Ethylbenzene			X	X
Isopropylbenzene				X
Styrene				X
Toluene				X
Trichloroethene			X	X
m,p-Xylene				X
o-Xylene				X
Total xylenes				X
TPH				
Gasoline-range hydrocarbons				X
Other Contaminants				
Cyanide				X
Perchlorate				X

[a] These DDT metabolites in surface water and TZW were evaluated as a component of total DDx.

BEHP – bis(2-ethylhexyl) phthalate
COPC – contaminant of potential concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
HCH – hexachlorocyclohexane
LOE – line of evidence
PAH – polycyclic aromatic hydrocarbon
PCB –polychlorinated biphenyl
SVOC – semivolatile organic compound
total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT and 4,4′-DDT)
TPH – total petroleum hydrocarbons
TZW – transition zone water
VOC – volatile organic compound

As shown in Table 5-5, 12 COIs were not identified as COPCs for at least one medium despite having been retained fom the SLERA (i.e., maximum value exceeded a screening-level threshold). The rationale for excluding these COIs as a COPC for a given medium is that no detected concentration exceeded corresponding screening-level thresholds. The potential risks to fish from these contaminants are unknown. The percentage of samples in which the COI was undetected but the DL exceeded screening-level TRVs is noted in Table 5-5. This percentage is low (< 30%) for some contaminants: butyl benzyl phthalate, hexachlorobutadiene, and endrin (in tissue); 2,4,-DDE (in surface water); and selenium and styrene (in TZW). The percentage of samples whose DL exceeded screening-level TRVs is higher for several phthalates and hexachlorobenzenes (i.e., alpha-, beta-, and delta-HCH). A COI that was not retained as a COPC for a particular medium may have been retained as a COPC for another medium or for a subset of the same medium. For example, BEHP was retained as a surface water COPC and as a tissue COPC for largescale sucker, sculpin, and smallmouth bass, but not as a tissue COPC for juvenile Chinook salmon.

Table 5-5. Fish COIs Not Retained as COPCs Following the Refined Screen

Chemical	COI Carried Over from SLERA but Not Retained as COPC				Rationale for COPC Exclusion
	Tissue	Dietary	Surface Water	TZW	
Metals					
Selenium				X	Although 26% of non-detected TZW samples had DLs > screening-level TRV, no detected TZW concentration was > screening-level TRV.
Phthalates					
BEHP	X				Although 36% of non-detected juvenile Chinook salmon tissue samples had DLs > screening-level TRV, contaminant was not detected in juvenile Chinook salmon tissue when DL was < screening-level TRV. BEHP was retained as a surface water COPC and a tissue COPC for largescale sucker, sculpin, and smallmouth bass because maximum detected concentrations were > screening-level TRVs.
Butyl benzyl phthalate	X				Although 57% of non-detected juvenile Chinook salmon tissue samples DLs were > screening-level TRV, no detected concentration was > screening-level TRV.
Dibutyl phthalate	X				Although 50% of non-detected largescale sucker, 67% of non-detected juvenile Chinook salmon, 63% of non-detected sculpin, and 45% of non-detected smallmouth bass tissue samples had DLs > screening-level TRV, no detected concentration was > screening-level TRV. Chemical was not detected in largescale sucker, sculpin, or smallmouth bass tissue even when DL was < screening-level TRV.
Diethyl phthalate	X				Although 17% of non-detected largescale sucker, 27% of non-detected juvenile Chinook salmon, and 100% (n = 1) of lamprey ammocoete tissue samples had DLs > screening-level TRV, contaminant was not detected when DL was < screening-level TRV.
SVOCs					
Hexachloro-butadiene	X				Only 6% of non-detected sculpin samples had DLs > screening-level TRV; no detected concentration was > screening-level TRV.
VOCs					
Styrene				X	Less than 1% of non-detected TZW samples had DLs > screening-level TRV; no detected TZW concentration was > screening-level TRV.

Table 5-5. Fish COIs Not Retained as COPCs Following the Refined Screen

Chemical	COI Carried Over from SLERA but Not Retained as COPC				Rationale for COPC Exclusion
	Tissue	Dietary	Surface Water	TZW	
Pesticides					
2,4′-DDE			X		Less than 1% of non-detected surface water samples had DLs > screening-level TRV, no detected surface water concentration was > screening-level TRV. Total DDx (which includes 2, 4′-DDE) was evaluated as surface water COPC because maximum detected total DDx concentration was > screening-level TRV.
Endrin	X				Although17% of non-detected largescale sucker, 9% of non-detected sculpin, and 12% of non-detected smallmouth bass tissue samples had DLs > screening-level TRV, no detected concentration was > screening-level TRV.
alpha-HCH	X				Although17% of non-detected largescale sucker, 6% of non-detected sculpin, 18% of non-detected smallmouth bass, and 67% of non-detected northern pikeminnow tissue samples had DLs > screening-level TRV, no detected concentration was > screening-level TRV. Chemical was not detected in largescale sucker or northern pikeminnow tissue even when DL was < screening-level TRV.
beta-HCH	X				Although 17% of non-detected largescale sucker, 29% of non-detected smallmouth bass, and 67% of non-detected northern pikeminnow tissue samples had DLs > screening-level TRV, no detected concentration was > screening-level TRV. Contaminant was not detected in largescale sucker or northern pikeminnow tissue even when DL was < screening-level TRV. Beta-HCH was retained as a surface water COPC and a tissue COPC for sculpin because maximum detected concentrations were > screening-level TRV.
delta-HCH	X				Although 17% of non-detected largescale sucker, 6% of non-detected sculpin, 9% of non-detected smallmouth bass, and 67% of non-detected northern pikeminnow tissue samples had DLs > screening-level TRV, no detected concentration was > screening-level TRV. Contaminant was not detected in largescale sucker, smallmouth bass, or northern pikeminnow tissue even when DL was < screening-level TRV.

Note: Exclusion as a COPC for a given medium does not preclude retention as a COPC for another medium.

BEHP – bis(2-ethylhexyl) phthalate
COI – contaminant of interest
COPC – contaminant of potential concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DL – detection limit
DDT – dichlorodiphenyltrichloroethane
HCH – hexachlorocyclohexane
SLERA – screening-level ecological risk assessment
SVOC – semivolatile organic compound
total DDx – sum of all six DDT isomers (2,4′-DDD; 4,4′-DDD; 2,4′-DDE; 4,4′-DDE; 2,4′-DDT; and 4,4′-DDT)
TRV – toxicity reference value
TZW – transition zone water
VOC – volatile organic compound

Certain fish COIs could not be screened because no screening-level thresholds were available. These COIs are listed in Table 5-6.

Table 5-6. Fish COIs with No TRVs

	No TRV Available[b]			
COI[a]	**Tissue**	**Dietary**	**Surface Water**	**TZW**
Metals				
Aluminum			X	X
Antimony		X		
Calcium				X
Chromium		X		
Manganese	X	X		
Nickel		X		
Thallium		X		
Titanium				X
Butyltins				
Monobutyltin ion	X			
Dibutyltin ion	X			
PAHs				
1-Methylnaphthalene		X		
2- Methylnaphthalene		X		
Benzo(e)pyrene		X		
Dibenzothiophene		X		
Perylene		X		
Alkylated PAHs		X		
SVOCs				
4-Chloroaniline			X	
Aniline			X	
Benzoic acid	X			
Benzyl alcohol	X			
Bis(2-chloroethoxy) methane	X			
Phenols				
4-Chloro-3-methylphenol	X			
4-Nitrophenol	X			
TPH				
Residual-range hydrocarbons				X

Table 5-6. Fish COIs with No TRVs

	No TRV Available[b]			
COI[a]	Tissue	Dietary	Surface Water	TZW
Diesel-range hydrocarbons				X
Total diesel-residual hydrocarbons				X
TPH				X
Herbicides				
2,4-DB			X	
MCPP			X	

[a] Includes fish COIs based on any medium (i.e., tissue, dietary, surface water, or TZW) for which no screening TRV is available.

[b] Blank cell indicates that contaminant is not a COI for a given medium (i.e., contaminant was either not analyzed or not detected) or TRV was identified.

2,4-DB – 4-(2,4-dichlorophenoxy)butyric acid
COI – contaminant of interest
MCPP – methylchlorophenoxypropionic acid
PAH – polycyclic aromatic hydrocarbon
SVOC – semivolatile organic compound
TPH – total petroleum hydrocarbons
TRV – toxicity reference value
TZW – transition zone water

5.4 WILDLIFE COPCS

Diet COPCs were identified for birds and mammals to be evaluated in the dietary risk LOE (Section 8.1). Bird egg COPCs were also identified for birds to be evaluated in the bird egg residue LOE (Section 8.2).

Bird- and mammal-specific diet screening thresholds were used in the SLERA. Screening thresholds are presented in Attachment 5. The following screening thresholds were used to derive bird and mammal COPCs:

- **Diet –** Receptor-specific diet-based screening level thresholds were derived for prey tissue and sediment based on dietary-dose screening-level thresholds (expressed as mg/kg bw/day). Provided by EPA (2008f),wildlife dietary-dose screening-level thresholds are based on either EPA's ecological soil screening level (Eco-SSL) documents or NOAEL TRVs derived from the toxicological literature. These diet-based screening-level thresholds were used to screen prey tissue and sediment concentrations.
- **Bird egg tissue –** Pprovided and recommended by EPA (2008f, j) , the bird egg tissue thresholds are based on NOAEL TRVs derived from the toxicological literature. These bird egg tissue screening-level thresholds were used to screen prey tissue concentrations.

Twenty-three COPCs were identified for birds through two LOEs, and twelve COPCs for mammals were identified based on one LOE. These COPCs are presented in

Table 5-73 and include: 11 metals, 3 individual PAHs or PAH sums, 2 phthalates, total PCBs, dioxin TEQ, PCB TEQ, total TEQ, and 3 pesticide or pesticide sums. Details are presented in Attachment 5.

Table 5-7. Wildlife COPCs

	LOE		
COPC	**Bird Diet**	**Bird Egg**	**Mammal Diet**
Metals			
Aluminum	X		X
Antimony			X
Arsenic	X		
Cadmium	X		
Chromium	X		
Copper	X		X
Lead	X		X
Mercury	X		X
Selenium	X		X
Thallium	X		
Zinc	X		
PAHs			
Benzo(a)pyrene	X		
Total HPAHs			X
Total PAHs[a]	X		
Phthalates			
BEHP	X		
Dibutyl phthalate	X		
PCBs and Dioxins			
Total dioxin/furan TEQ	X	X	X
Total PCBs	X	X	X
PCB TEQ	X	X	X
Total TEQ	X	X	X
Pesticides			
Aldrin	X		
4,4'-DDE		X	
Sum DDE	X	X	
Total DDx	X		X

[a] LPAH and HPAH were identified as COPCs for birds in the SLERA (Attachment 5) but (as per EPA direction) were evaluated as total PAHs (LWG 2010).

BEHP – bis(2-ethylhexyl) phthalate
COPC – contaminant of potential concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
HPAH – high-molecular-weight polycyclic aromatic hydrocarbon
LOE – line of evidence
LPAH – low molecular-weight polycyclic aromatic hydrocarbon
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
TEQ – toxic equivalent
total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT and 4,4′-DDT)

Despite having a maximum value that exceeded a screening-level threshold, dibutyl phthalate was not identified as a COPC for osprey (Table 5-8). Although at least one DL exceeded the TRV, concentrations detected in osprey prey tissue did not exceeded the threshold tissue concentration (TTC) and concentrations detected in sediment did not exceed the threshold sediment concentration (TSC). The potential dietary risk to osprey from dibutyl phthalate is unknown. However, risk is negligible because carp was the only osprey prey whose tissue concentrations were greater than the TTC—a determination made on the basis of DLs in carp tissue samples. Furthermore, carp is likely a small component of the osprey diet (see Section 8.1.3.2.2). In all other samples of osprey prey, detected concentrations or DLs were less than the TTC. Dibutyl phthalate was retained as a COPC for spotted sandpiper and hooded merganser because maximum detected prey tissue concentrations (invertebrate tissue) exceeded receptor-specific TTCs.

Table 5-8. Wildlife COIs Not Retained as COPCs Following the Refined Screen

Chemical	COI Carried Over from SLERA but Not Retained as COPC		Rationale for COPC Exclusion
	Dietary	Bird Egg	
Phthalates			
Dibutyl phthalate	X		Although 40% of non-detected carp tissue samples had DLs > osprey TTC, no detected osprey prey tissue concentration was > TTC or sediment concentration was > TSC. All other fish prey and sediment samples had detected values, with DLs < screening-level TRVs. Contaminant was retained as a dietary COPC for spotted sandpiper and hooded merganser because maximum prey tissue concentrations (field-collected clams) was > screening-level TTC.[a]

[a] COI was not carried over from the SLERA for other wildlife receptors (mink and river otter).

COI – contaminant of interest
COPC – contaminant of potential concern
DL – detection limit
SLERA – screening-level ecological risk assessment
TRV – toxicity reference value
TTC – threshold tissue concentration
TSC – threshold sediment concentration

Certain wildlife COIs could not be screened because no screening-level dietary TRVs were available. These COIs are listed in Table 5-9.

Table 5-9. Wildlife COIs with No TRVs

	No TRV Available[b]	
COI[a]	**Dietary**	**Bird Egg**
Metals		
Antimony	X[c]	
Manganese	X	
Silver	X	
PAHs		
1-Methylnaphthalene	X[c]	
2-Methylnaphthalene	X[c]	
Benzo(e)pyrene	X	
Dibenzothiophene	X[c]	
Perylene	X	
Alkylated PAHs	X	
SVOCs		
Benzoic acid	X	
Benzyl alcohol	X[c]	
Carbazole	X	
Dibenzofuran	X	
Hexachloroethane	X[c]	
n-Nitrosodiphenylamine	X	
Phenols		
2-Methylphenol	X	
4-Chloro-3-methylphenol	X	
4-Methylphenol	X[c]	
Phenol	X[c]	

a Includes bird and mammal dietary evaluation COIs for which no screening TRV is available. TRVs were available for all bird egg COIs.

b Blank cell indicates that contaminant is not a COI for a given medium (i.e., contaminant was either not analyzed or not detected).

c No bird dietary screening-level threshold was available; however, a mammal dietary threshold was available.

COI – contaminant of interest
PAH – polycyclic aromatic hydrocarbon
SVOC – semivolatile organic compound
TRV – toxicity reference value

5.5 AMPHIBIAN AND AQUATIC PLANT COPCS

The SLERA and refined screen identified COPCs for amphibians and aquatic plants in surface water and shallow TZW. These COPCs were further evaluated in the amphibian and aquatic plant risk assessments (Sections 9.0 and 10.0, respectively).

Screening thresholds in surface water and TZW (presented in Attachment 5) were used to derive amphibian and aquatic plant COPCs. For both media, screening-level thresholds are represented by chronic water TRVs. Water TRVs were developed according to a selection hierarchy agreed upon by LWG and EPA of water quality regulatory thresholds and literature-based thresholds; these values included national and proposed State of Oregon WQS, Tier II values (Suter and Tsao 1996), FCVs for individual PAH compounds (Table 3-4 of EPA 2003c), Canadian water environmental quality guidelines, ODEQ guidance values (ODEQ 2006), and literature-derived values. EPA provided TRVs for five of the chemical groups that are blended to form gasoline (EPA 2008a). Average fractions of these components in gasoline were used to convert the total gasoline-range hydrocarbon concentration into gasoline fraction concentrations for comparison with the TRVs. Any one gasoline fraction exceeding its TRV was grounds for identifying gasoline as a COPC.

As presented in Table 5-10, 64 COPCs were identified for amphibians and aquatic plants through two LOEs: 15 metals, monobutyltin, 16 individual PAHs, BEHP, 3 SVOCs, total PCBs, 6 pesticide or pesticide sums, 18 VOCs, gasoline-range hydrocarbons, cyanide, and perchlorate. Details are presented in Attachment 5.

Table 5-10. Amphibian and Aquatic Plant COPCs

	LOE	
COPC	**Surface Water**	**TZW**
Metals		
Barium		X
Beryllium		X
Cadmium		X
Cobalt		X
Copper		X
Iron		X
Lead		X
Magnesium		X
Manganese		X
Nickel		X
Potassium		X
Selenium		X
Sodium		X

Table 5-10. Amphibian and Aquatic Plant COPCs

COPC	LOE	
	Surface Water	TZW
Vanadium		X
Zinc	X	X
Butyltins		
Monobutyltin ion	X	
PAHs		X
2-Methylnaphthalene		X
Acenaphthene		X
Anthracene		X
Benzo(a)anthracene	X	X
Benzo(a)pyrene	X	X
Benzo(b)fluoranthene		X
Benzo(g,h,i)perylene		X
Benzo(k)fluoranthene		X
Chrysene		X
Dibenzo(a,h)anthracene		X
Fluoranthene		X
Fluorene		X
Indeno(1,2,3-cd)pyrene		X
Naphthalene	X	X
Phenanthrene		X
Pyrene		X
Phthalates		X
BEHP	X	
SVOCs		
1,2-Dichlorobenzene		X
1,4-Dichlorobenzene		X
Dibenzofuran		X
PCBs		
Total PCBs	X	
Pesticides		
2,4′-DDD[a]	X	X
2,4′-DDT[a]	X	X
4,4′-DDD[a]	X	X
4,4′-DDE[a]		X
4,4′-DDT	X	X

Table 5-10. Amphibian and Aquatic Plant COPCs

	LOE	
COPC	**Surface Water**	**TZW**
Total DDx	X	X
VOCs		
1,1-Dichloroethene		X
1,2,4-Trimethylbenzene		X
1,3,5-Trimethylbenzene		X
Acrolein		X
Benzene		X
Carbon disulfide		X
Chlorobenzene		X
Chloroethane		X
Chloroform		X
cis-1,2-Dichloroethene		X
Ethylbenzene	X	X
Isopropylbenzene		X
Styrene		X
Toluene		X
Trichloroethene	X	X
m,p-Xylene		X
o-Xylene		X
Total xylenes		X
TPH		
Gasoline-range hydrocarbons		X
Other Contaminants		
Cyanide		X
Perchlorate		X

[a] These DDT metabolites in surface water and TZW were evaluated as a component of total DDx.

BEHP – bis(2-ethylhexyl) phthalate
COPC – contaminant of potential concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
LOE – line of evidence
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
SVOC – semivolatile organic compound
total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT and 4,4′-DDT)
TPH – total petroleum hydrocarbons
TZW – transition zone water
VOC – volatile organic compound

As shown in Table 5-11, three COIs were not identified as a COPC for at least one medium despite having been retained fom the SLERA (i.e., maximum value exceeded a screening-level threshold). The rationale for excluding these COIs as a COPC for a given medium is that no detected concentration exceeded corresponding screening-level thresholds. The potential risks to benthic invertebrates from these contaminants are unknown. The percentage of samples in which the COI was undetected but the DLs exceeded screening-level TRVs is noted in Table 5-11. This percentage is low (< 30%) for all three COIs. A COI that was not retained as a COPC for a particular medium may have been retained as a COPC for another medium or for a subset of the same medium.

Table 5-11. Aquatic Plant and Amphibian COIs Not Retained as COPCs Following the Refined Screen

	COI Carried Over from SLERA but Not Retained as COPC		
Contaminant	**Surface Water**	**TZW**	**Rationale for COPC Exclusion**
Metals			
Selenium		X	Although 26% of non-detected TZW samples had DLs > screening-level TRV, no detected TZW concentration > screening-level TRV.
VOCs			
Styrene		X	Less than 1% of non-detected TZW samples had DLs > screening-level TRVs; no detected TZW concentration > screening-level TRV.
Pesticides			
2,4′-DDE	X		Less than 1% of non-detected surface water samples had DLs > screening-level TRVs; no detected surface water concentration > screening-level TRV. Total DDx (which includes 2,4′-DDE) was evaluated as surface water COPC because maximum detected total DDx concentration > screening-level TRV.

Note: Exclusion as a COPC for a given medium does not preclude retention as a COPC for another medium.

COI – contaminant of interest
COPC – contaminant of potential concern
DL – detection limit
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
SLERA – screening-level ecological risk assessment
total DDx – sum of all six DDT isomers (2,4′-DDD; 4,4′-DDD; 2,4′-DDE; 4,4′-DDE; 2,4′-DDT; and 4,4′-DDT)
TRV – toxicity reference value
TZW – transition zone water

Certain COIs could not be screened because no screening-level thresholds were available. These COIs are listed in Table 5-12.

Table 5-12. Aquatic Plant and Amphibian COIs with no TRVs

	No TRV Available[b]	
COI[a]	**Surface Water**	**TZW**
Metals		
Aluminum	X	X
Calcium		X
Titanium		X
Dioxins and Furans		
1,2,3,4,6,7,8-Heptachlorodibenzofuran	X	
1,2,3,4,6,7,8-Heptachlorodibenzo-*p*-dioxin	X	X
1,2,3,4,7,8,9-Heptachlorodibenzofuran	X	X
1,2,3,4,7,8-Hexachlorodibenzofuran	X	X
1,2,3,4,7,8-Hexachlorodibenzo-*p*-dioxin	X	
1,2,3,6,7,8-Hexachlorodibenzofuran	X	X
1,2,3,6,7,8-Hexachlorodibenzo-*p*-dioxin	X	
1,2,3,7,8,9-Hexachlorodibenzofuran	X	
1,2,3,7,8,9-Hexachlorodibenzo-*p*-dioxin	X	
1,2,3,7,8-Pentachlorodibenzofuran	X	X
1,2,3,7,8-Pentachlorodibenzo-*p*-dioxin	X	
2,3,4,6,7,8-Hexachlorodibenzofuran	X	
2,3,4,7,8-Pentachlorodibenzofuran	X	X
2,3,7,8-Tetrachlorodibenzofuran	X	X
SVOCs		
4-Chloroaniline	X	
Aniline	X	
TPH		
Residual-range hydrocarbons		X
Diesel-range hydrocarbons		X
Total diesel-residual hydrocarbons		X
Total petroleum hydrocarbons		X
Herbicides		
2,4-DB	X	
MCPP	X	

[a] Includes aquatic plant or amphibian COIs based on any medium (i.e., surface water or TZW) for which no screening TRV is available.

[b] Blank cell indicates that contaminant is not a COI for a given medium (i.e., contaminant was either not analyzed or not detected).

2,4-DB – 4-(2,4-dichlorophenoxy)butyric acid
COI – contaminant of interest
MCPP – methylchlorophenoxypropionic acid
SVOC – semivolatile organic compound
TRV – toxicity reference value
TPH – total petroleum hydrocarbons
TZW – transition zone water

6.0 BENTHIC INVERTEBRATE RISK ASSESSMENT

This section presents the revised baseline risk assessment for benthic macroinvertebrates in the Study Area. The BERA Problem Formulation (Section 3.0) identifies three receptors to represent aquatic benthic invertebrates: the benthic macroinvertebrate community as a whole, bivalve (clam and mussel) populations, and decapod (crayfish) populations. Four main components were used to characterize risks to benthic macroinvertebrates:

- An assessment of actual or predicted sediment toxicity, a process that included sediment toxicity testing, the development and comparison of Study Area sediment concentrations to site-specific sediment quality values (SQVs), and comparison to generic national freshwater SQGs[37] and associated mean quotients[38] (MQs)
- A tissue-residue assessment in which both empirical and predicted chemical concentrations in Study Area tissue were compared to tissue-residue TRVs
- A water assessment in which chemical concentrations in surface water were compared to water TRVs derived for the protection of aquatic organisms
- An assessment of risk to benthic macroinvertebrates from exposure to contaminated TZW adjacent to nine upland facilities within the Study Area

The different measures of exposure and effect for each receptor are presented in Table 6-1.

Table 6-1. Measures of Exposure and Effect for Benthic Invertebrate Receptors

		Benthic Invertebrates		
Medium	**LOE**	**Benthic Community[a]**	**Bivalves**	**Crayfish**
Bulk sediment	Measured toxicity to representative benthic invertebrate test species following laboratory exposure to field-collected sediment	X	X	Included in benthic community evaluation
	Predicted effects based on a comparison of sediment concentrations to generic national SQGs and site-specific SQVs	X	Included in benthic community evaluation	Included in benthic community evaluation

[37] SQGs are sediment contaminant thresholds that have been published and used for decision-making at other sites throughout the country. SQVs are contaminant thresholds that were developed specifically for the Portland Harbor project based on site-collected data, but have not yet been approved by EPA.

[38] A mean quotient is the average exceedance factor of all chemicals compared to their respective SQG in a given sample.

Table 6-1. Measures of Exposure and Effect for Benthic Invertebrate Receptors

		Benthic Invertebrates		
Medium	**LOE**	**Benthic Community**[a]	**Bivalves**	**Crayfish**
Tissue	Predicted effects based on a comparison of field-collected tissue-residue concentrations to tissue TRVs	X	X	X
	Predicted effects based on a comparison of laboratory-exposed tissue-residue concentrations[b] to tissue TRVs	X	X	NE[c]
	Predicted effects based on a comparison of estimated tissue-residue concentrations (estimated using a mechanistic model or BSARs) to tissue TRVs	X	X	X
Surface water	Predicted effects based on a comparison of water chemical concentrations to TRVs	X	Included in benthic community evaluation	Included in benthic community evaluation
TZW	Predicted effects based on a comparison of shallow TZW chemical concentrations to TRVs	X	Included in benthic community evaluation	NE

[a] Although an LOE not formally selected for the BERA, benthic community structure represented by community successional stage was also evaluated in those areas posing unacceptable risks to the benthic community based on toxicity, tissue residue, and water LOEs.

[b] Laboratory bioaccumulation test organisms were the clam *Corbicula fluminea* and the oligochaete worm *Lumbriculus variegatus.*

[c] No laboratory bioaccumulation tests were conducted using adult crayfish (there are currently no approved experimental protocols); tests based on other invertebrates were used as a surrogate.

[d] No crayfish habitat was identified in the TZW investigation areas.

BERA – baseline ecological risk assessment
BSAR – biota-sediment accumulation regression
LOE – line of evidence
NE – not evaluated
SQV – sediment quality value
TRV – toxicity reference value
TZW – transition zone water

In the sediment toxicity assessment, the toxicity test results were statistically compared to negative control results and numerically compared to characteristic upstream reference thresholds; these thresholds were derived using a reference envelope approach developed for EPA (2008) for application in the BERA. These direct measures of toxicity are considered to be the primary LOE in the assessment of benthic risks.

Although extensively tested, toxicity was not measured at every sediment sampling location within the river. To predict the presence or absence of sediment toxicity throughout the river, site-specific SQVs were derived from the available paired toxicity and chemistry data for surface sediments. Two numerical models were applied to this purpose: the floating percentile model (FPM) (Avocet 2003) and the logistic regression model (LRM) (Field et al. 1999a). Both models were used to identify chemical thresholds that could predict toxicity with a quantified degree of reliability from relationships between the empirical toxicity and surface sediment chemistry data from Portland Harbor. The FPM with the most balanced error rates and the LRM selected by EPA were carried forward to help assess benthic risks. Contaminant concentrations in surface sediment samples that exceeded their respective site-specific SQVs were used as a secondary LOE of the potential for benthic toxicity. Although direct measures of toxicity are assumed to provide more certainty for benthic risk assessment, the comparison of sediment chemical concentrations to site-specific SQVs provides increased spatial resolution in the absence of toxicity test data. The potential magnitude of toxicity was also evaluated using MQs derived from the site-specific SQVs.

At EPA's direction, national freshwater SQGs and mean SQG quotients using a threshold of 0.7 were evaluated for use in predicting risks to the benthic community in Portland Harbor. In the BERA, these guidelines are collectively referred to as generic SQGs. Application of these generic SQGs to sediment from Portland Harbor is affected by numerous uncertainties, including the fact that none of these SQGs were derived using any information from Portland Harbor. The following generic SQGs were evaluated for reliability in predicting toxicity in Portland Harbor sediment:

- Probable effects levels (PELs) (Smith et al. 1996)
- Probable effects concentrations (PECs) (MacDonald et al. 2000)
- Mean quotients of the PECs and PELs

In the LOE based on benthic invertebrate tissue residues, sample-by-sample data were compared to TRVs associated with reduced survival, growth, and reproduction. The TRVs were selected from published studies and approved by EPA for use in the BERA. A tissue concentration greater than the TRV was used to indicate the potential for deleterious effects on benthic invertebrates.

Three types of tissue concentration were used in the comparison to TRVs: those measured in benthic invertebrates collected directly from the field, those measured in laboratory specimens exposed to sediment collected from the field, and those predicted for organisms that may reside in the field where sediment was collected but not assessed by either of the preceding methods. In this last case, a mechanistic model or accumulation factor derived from site-specific tissue and sediment data was used to predict tissue concentrations for contaminants demonstrating a link between sediment and tissue concentrations. The uncertainty associated with the tissue-residue findings is lowest for field-collected benthic invertebrates, except for metals.[39] Uncertainty associated with the laboratory-exposed benthic invertebrates is greater because the duration of the test could not fully replicate field conditions. Although the greatest uncertainty is associated with

[39] The utility of the tissue-residue approach for metals is highly uncertain (EPA 2007e).

use of sediment chemistry to predict tissue residues, the results improved the spatial resolution for assessing benthic risks.

In the surface water assessment, sample results were compared to TRVs selected from state WQS, federal AWQC, Tier II, or other water quality values approved by EPA for use in the BERA. In the TZW assessment, individual porewater sample results were compared to surface water TRVs.

Benthic community structure represented by community successional stage was also evaluated in the Study Area, with a focus on those areas posing unacceptable risk to the benthic community based on toxicity, tissue-residue, and water LOEs. Although an LOE not formally selected for the BERA, benthic community structure information helped to set the context for risk conclusions.

All benthic invertebrate COPCs identified in the SLERA and refined screen for each LOE were evaluated in this assessment. In addition, selected chemicals that were not identified as COPCs because no screening criteria were available were re-evaluated in SQV model development to identify potential site-specific criteria. The methods used to characterize risk for each LOE are described in EPA's Problem Formulation (Attachment 2) and in each of the following subsections. Specific uncertainties associated with each LOE are discussed in the individual LOE sections.

The overall conclusions regarding the benthic community, including a synoptic analysis of uncertainty, are found in Section 6.7, which presents the final determination of COPCs, key uncertainties in the exposure assessment and effects characterization, and interpretation of the BERA findings. Figure 6-1 presents a flowchart of the benthic invertebrate risk assessment section organization.

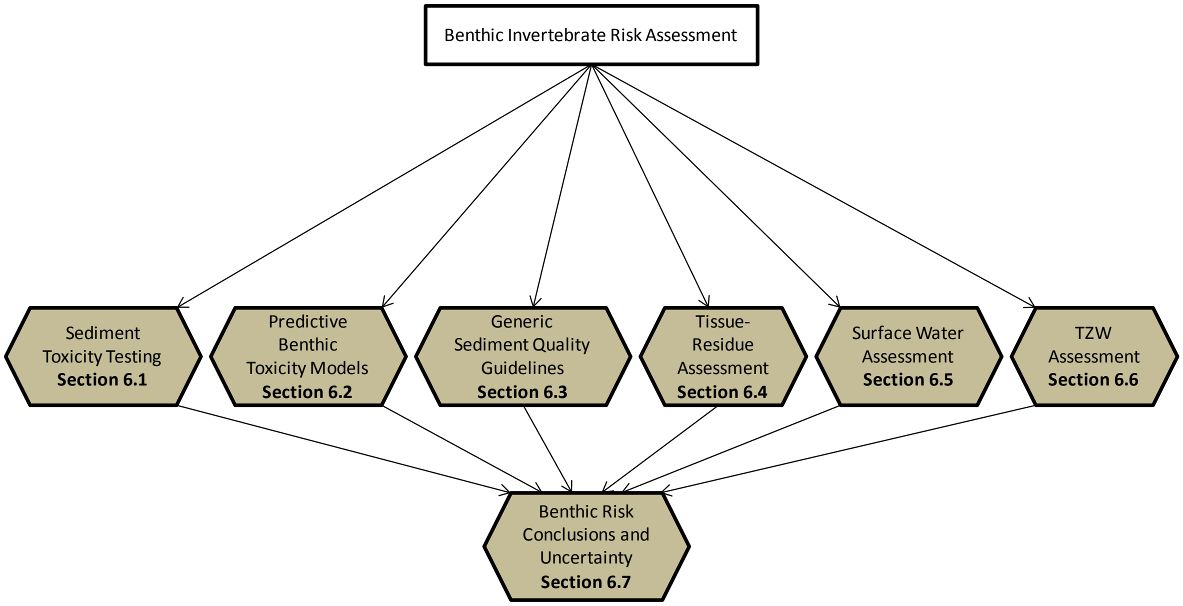


Figure 6-1. Overview of Benthic Invertebrate Risk Assessment Section Organization

6.1 SEDIMENT TOXICITY TESTING

The adverse effects on macroinvertebrates caused by exposure to contaminants in surface sediments were assessed by performing two sediment toxicity tests on 293 surface sediment samples from the LWR: the 10-day survival and biomass test using the midge *Chironomus dilutus* (formerly *C. tentans)* and the 28-day survival and biomass test using the amphipod *Hyalella azteca*. The biomass endpoint was defined as mean biomass (final biomass divided by the initial count of test organisms rather than by the count of survivors per American Society for Testing and Materials [ASTM] (ASTM 2007)) to incorporate the effect of mortality on growth, and hereafter is therefore referred to as the biomass endpoint. An independent endpoint based solely on sublethal effects (i.e., growth) was therefore not evaluated in the BERA.

Toxicity Samples Available for Assessing Risk

Two hundred ninety-three toxicity sample results were available to support the Portland Harbor BERA:

- 269 from the Study Area
- 2 from the downtown reach (at RM 12.2)
- 22 from the upriver reach (RM 15.4 to 25.5)

Since the toxicity tests were conducted, 13 of the tested locations in the Study Area have been dredged, likely changing the surface sediment chemical concentrations. These 13 samples were therefore excluded from the assessment of potential benthic community risks, resulting in 256 toxicity tests for characterization of the Study Area for this LOE. However, the chemical and toxicity test results for these excluded samples were retained for development of the predictive models and the reliability analysis.

Toxicity was determined based on the statistical difference ($p \leq 0.05$) between test and negative control sample responses and on exceedance of reference thresholds for survival and biomass derived from upriver samples. The survival of *Corbicula fluminea* from the 33 bioaccumulation tests conducted in Round 2 was used to assess potential toxicity to bivalves. These results are considered qualitative because the *C. fluminea* test was designed to evaluate bioaccumulation rather than toxicity. Crayfish are addressed in the risk conclusions because receptor-specific toxicity testing was not conducted. Sections 6.1.1 and 6.1.2 present the sediment toxicity assessment for macroinvertebrates and bivalves, respectively.

The details of the macroinvertebrate assessment are presented as follows:

- Section 6.1.1.1 presents the reference envelope approach (MacDonald and Landrum 2008). Reference envelope values (REVs) were developed to be used along with statistical tests to create four toxicity effect levels: Level 0 (L0 [non-toxic]), Level 1 (L1 [low toxicity]), Level 2 (L2 [moderate toxicity]), and Level 3 (L3 [high toxicity]). Additional details on the reference envelope approach are presented in Attachment 6 (Part B).
- Section 6.1.1.2 presents the toxicity assessment based on the sediment toxicity tests. The 256[40] sediment samples from the Study Area were classified using the toxicity categories defined by the REVs.
- Section 6.1.1.3 presents the uncertainty analysis of the sediment toxicity test results.
- Section 6.1.1.4 presents a summary of the sediment toxicity assessment and uncertainty evaluation.

The details of the bivalve assessment (Section 6.1.2) are presented as follows:

- Section 6.1.2.1 presents the toxicity assessment based on the bivalve mortality.
- Section 6.1.2.2 presents the uncertainty analysis of the bivalve sediment toxicity assessment.

Figure 6-2 presents a flowchart of the sediment toxicity testing section organization.

[40] This number excludes the 13 sediment samples from locations that have since been dredged.

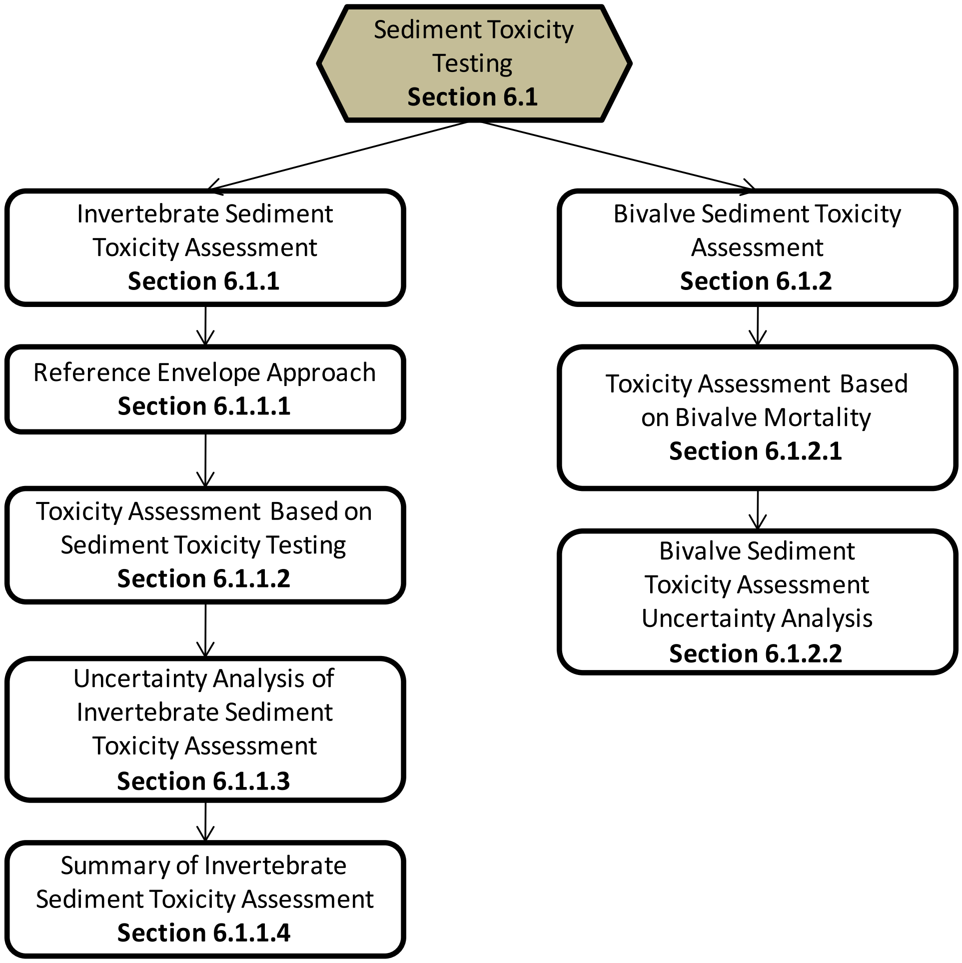


Figure 6-2. Overview of Sediment Toxicity Testing Section Organization

6.1.1 Invertebrate Sediment Toxicity Assessment

Section 6.1.1.1 presents the reference envelope approach for identifying toxic and non-toxic samples. Section 6.1.1.2 presents the evaluation of the sediment toxicity test results from the Study Area. The uncertainties associated with the biological effects levels are discussed in Section 6.1.1.3, followed by a summary of the assessment in Section 6.1.1.4.

6.1.1.1 Reference Envelope Approach

By agreement with EPA, the sediment toxicity tests were evaluated using the reference envelope approach described in MacDonald and Landrum (2008) (Attachment 6 [Part B]). In this approach, both the negative control and an effect threshold representing reference conditions are used to identify sediment samples that are likely to be associated with adverse effects to benthic invertebrates.

The Reference Envelope Approach

A reference envelope provides the range of values associated with toxicity responses or other attributes measured in reference sediments (sediment collected near the site of concern, representing background conditions that result from either global or localized rather than Study Area contaminant inputs, but exhibiting little or no sediment toxicity (ASTM 2007)). A reference envelope incorporates the expected spatial and temporal variability among reference locations and, as such, represents a normal or expected range of values for a given parameter for a given watershed or geographic area. A reference envelope can be used to interpret toxicity test results in that values falling outside of the reference envelope are considered different from reference.

The locations used to represent reference conditions for the Portland Harbor Study Area lie upstream of, or at the upstream end of, the Study Area and conform to data quality requirements recommended in MacDonald and Landrum (2008) (Attachment 6 [Part B]). To derive a reference threshold, control-normalized responses (treatment mean divided by control mean) for each endpoint were fit to a variety of theoretical probability distributions. A model was selected, in consultation with EPA, using visual comparison of plots of the theoretical and empirical distributions, statistical goodness-of-fit criteria, and best professional judgment. The fifth percentile of the selected model for each endpoint was designated as the REV. Two additional thresholds (80% and 90% of the REV) were derived to further classify the magnitude of toxicity.

Empirical treatment and control means from the Study Area that do not differ statistically or whose control-normalized values are higher than the REV are considered non- toxic (L0). Empirical treatment and control means that differ statistically and whose control-normalized values are below the REV were classified as having low, moderate, or high toxicity depending of the magnitude of difference from the REV (within 90% of the REV was low, between 90% and 80% of the REV was moderate, and less than 80% of the REV was high).

Twenty-six toxicity tests performed on sediment samples collected at 22 upriver reach locations were evaluated for inclusion in the reference envelope. Multiple biological and chemical criteria were used to select the reference area dataset based on the recommendations of MacDonald and Landrum (2008), including toxicity test standard performance criteria, control-adjusted response rate criteria based on the National Sediment Inventory (EPA 2004c), and four sets of chemical criteria: the Regional Sediment Evaluation Team (RSET) interim screening levels, PECs, mean PEC quotients (PEC-Q_{dw}), and sum of the equilibrium partitioning sediment benchmark (ESB) toxicity units (TUs) for PAHs ($\sum$ESB-TU_{PAHs}). The criterion based on simultaneously extracted metals and acid volatile sulfide recommended by MacDonald and Landrum (2008) was not incorporated into the chemical evaluation because these two parameters were not measured in the sediment samples.

Toxicity test and sediment chemistry data from 16 samples representing 15[41] upriver reach locations met all the selection criteria, and the associated toxicity test data were included in the reference envelope evaluation. In addition, toxicity test data from two locations in the upstream end of the Study Area (RM 10.6 and RM 11.2) were included in the reference envelope evaluation at the request of EPA. Map 6-1 presents the locations of the 18 samples used to derive the REVs.

[41] A sediment sample collected at one location was tested twice; both toxicity test results were included in the reference dataset.

Control-normalized toxicity test responses (treatment mean response divided by control mean response) from each reference sampling location were fitted to a range of theoretical probability distributions using @Risk software. The statistical distribution that best described each empirical distribution was selected, in consultation with EPA, using statistical and graphical results provided by @Risk and best professional judgment. The lower 5th percentile of each best-fitting survival and biomass distribution was used to represent the normal range of reference responses relative to control responses (i.e., the REV).

Test responses were then classified as having no, low, moderate, or high toxicity based on the magnitude of response relative to the REV and two thresholds selected by EPA (further details are presented in Attachment 6 [Part A]). A threshold based on 90% of the REV defined the boundary between low and moderate toxicity and a value that was 80% of the REV defined the boundary between moderate and high toxicity. Statistical differences between the test and control results were also accounted for in the classification. Table 6-2 presents the REVs and the two reference thresholds used to define toxicity categories in the BERA.

Table 6-2. Biological Effects Levels Based on the REV

Test and Endpoint	REV (%)[a]	90% Threshold (%)	80% Threshold (%)
Chironomus dilutus survival[b]	93.9	84.	75.1
Chironomus dilutus biomass[b]	91.0	81.9	72.8
Hyalella azteca survival[b]	88.1	79.3	70.5
Hyalella azteca biomass[b]	73.6	66.2	58.9

a 5th percentile of negative control-adjusted (test divided by negative control) survival and biomass endpoints for reference sampling locations.

b The test response must also be statistically lower than the negative control response (one-tailed test, $p \leq 0.05$) to consider the sediment as having an adverse effect on benthic invertebrates.

BERA – baseline ecological risk assessment

REV – reference envelope value

The results of the amphipod and chironomid Study Area toxicity tests were compared to batch-specific negative controls to identify sediment samples with significantly lower responses (one-tailed, parametric or non-parametric *t*-test, with $\alpha = 0.05$ comparison-wise). A toxicity test result that was not significantly different ($p > 0.05$) from a paired control response was considered equivalent to a reference response and did not represent an adverse effect. The toxicity test results for samples with significantly less survival or biomass were then classified based on the magnitude of the responses. Those significant results that were within 90% of the reference response were classified as having low toxicity (L1); those between 80 and 90% of the reference value were considered to be moderately toxic (L2); those with responses less than 80% of the reference response were classified as highly toxic (L3). Figure 6-3 illustrates the relationship between the toxicity categories and the REV.

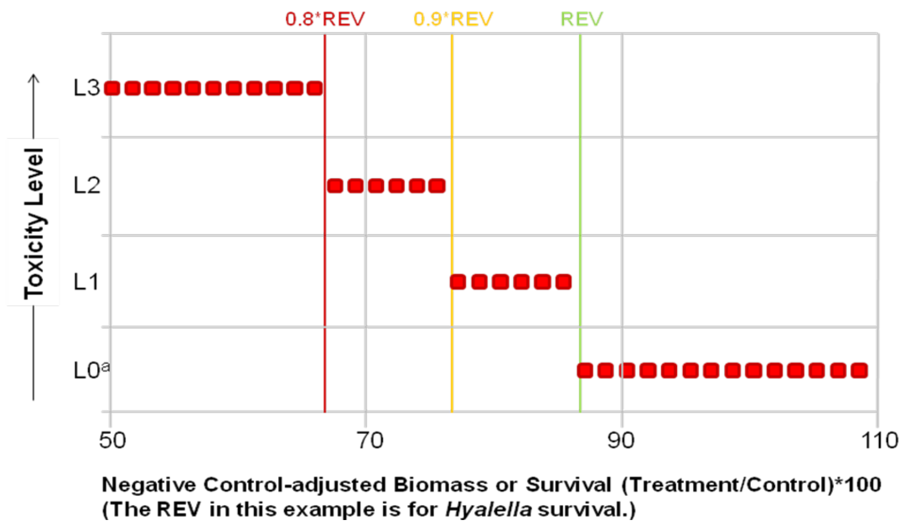


Figure 6-3. Relationship Between the Toxicity Categories and REVs

6.1.1.2 Toxicity Assessment Based on Sediment Toxicity Testing

By agreement between EPA and the LWG (EPA 2010b), the reference envelope approach by MacDonald and Landrum (2008) (Attachment 6 [Part B]) supersedes the approach provided in EPA's Problem Formulation (Attachment 2). Table 6-3 summarizes the toxicity tests results for the 256 toxicity test sampling locations in the Study Area based on a comparison with the negative control and the reference thresholds. A summary of the highest level of toxicity for any endpoint at a given sampling location (i.e., equivalent to a combined endpoint) is also provided.Results varied by endpoint and species. The survival responses were similar between test species, with over 80% of the samples being categorized as non-toxic. *Chironomus* biomass was similar to survival in that more than 74% of the samples were non-toxic. Hyalella biomass had the lowest percentage non-toxic samples (55.9%). Up to 14% (range of 6 to 14%) of the samples were classified as highly toxic, with *Chironomus* biomass accounting for the maximum incidence of toxicity. Intermediate categories of toxicity (low and moderate), combined, accounted for 6.6% (*Hyalella* survival) to 34.8% (*Hyalella* biomass) of the samples, depending on the test organism and endpoint. Based on the maximum toxicity classification for any one endpoint at a sampling location, 41% of the samples are classified as non-toxic, 22.7% as having low toxicity, 18.4% as having moderate toxicity, and 18.0% as highly toxic.Maps 6-2 through 6-5 present the individual endpoint results of the two toxicity tests for the 256 sampling locations in the Study Area. The individual toxicity test results are presented in Attachment 6 (Part A).

Table 6-3. Study Area Toxicity Data Compared to the Negative Control and Reference Thresholds

	Number of Sampling Locations				
	Chironomus		*Hyalella*		Highest Combined Level of Toxicity
Category	Survival	Biomass[a]	Survival	Biomass[a]	
Level 0: Non-toxic	210 (82%)[b]	190 (74.2%)[b]	224 (87.5%)	143 (55.9%)	105 (41.0%)
Level 1: Low toxicity	12 (4.7%)	24 (9.4%)	15 (5.9%)[c]	47 (18.4%)[c]	58 (22.7%)
Level 2: Moderate toxicity	9 (3.5%)	7 (2.7%)	2 (0.8%)	42 (16.4%)	47 (18.4%)
Level 3: High toxicity	25 (9.8%)	35(13.7%)[d]	15 (5.9%)	24 (9.4%)[d]	46 (18.0%)

[a] The biomass endpoint was defined as the average final mass of individuals (initial) in a sample.
[b] The two downtown sampling locations included in the reference envelope fell within these categories.
[c] One sampling location within the downtown reach fell within this category.
[d] Biomass in sampling locations with 100% mortality was not statistically evaluated. However, these locations were included in the group in which biomass exceeded the high threshold.

6.1.1.3 Uncertainty Analysis of Invertebrate Sediment Toxicity Assessment

A quantitative assessment of the likelihood that Portland Harbor bioassays were correctly classified was conducted using a Bayesian likelihood method to calculate the probability that each bioassay sample was assigned to the correct toxicity category. The likelihood calculation quantifies the conditional probability that a particular mean, μ_1, is the true mean response for the sample given the observed replicate data, and accounts for the variance in bioassay replicates as well as the uncertainty in the magnitude of the mean control response (using upper and lower 95% tolerance bounds). The details of this analysis are provided in Attachment 6 (Part C) and summarized here.

The intent of this analysis was to quantify the probability that the true control-normalized sample mean fell between the reference envelope toxicity thresholds indicated by the control-normalized sample mean. The data used in the calculation of likelihoods are the individual test replicates divided by the negative control mean response. When responses are expressed as control-normalized, values ≥ 1 indicate a response that is as good as, if not better than, the control response. The normal likelihood is a simplifying assumption chosen to demonstrate the effects of replicate variance on certainty around point estimates of the mean toxic response. Biomass residuals were strongly normal and resulting probabilities of effect level can be interpreted exactly. Although survival residuals were slightly less normal, with the calculated probabilities for survival responses not as exact, the results described below still provide a good demonstration of the nature of the uncertainty about the mean survival response.

The uncertainty about the true mean bioassay result for a single sample is shown in Figure 6-4 (the likelihood curve). Effect thresholds based on the REV are overlaid on this distribution. In this example, there is approximately a 55% probability that the true mean exceeds the REV (i.e., has greater biomass than the reference response threshold); an 80% probability that the true mean exceeds the L2 threshold; and a 95% probability that the true mean exceeds the L3 threshold. The uncertainty about the true response is high enough, in this example, that it is not possible to confidently predict its toxicity category (although in this case, it could be concluded that the sample is probably L0 [non-toxic] or L1 [low toxicity]). This is a typical example in that the true toxicity classification for the sample is uncertain. The uncertainty is a consequence of defining the toxicity categories too narrowly, given the variability in the empirical bioassay data (the toxicity category interval is sometimes smaller than the variability in the replicate responses).

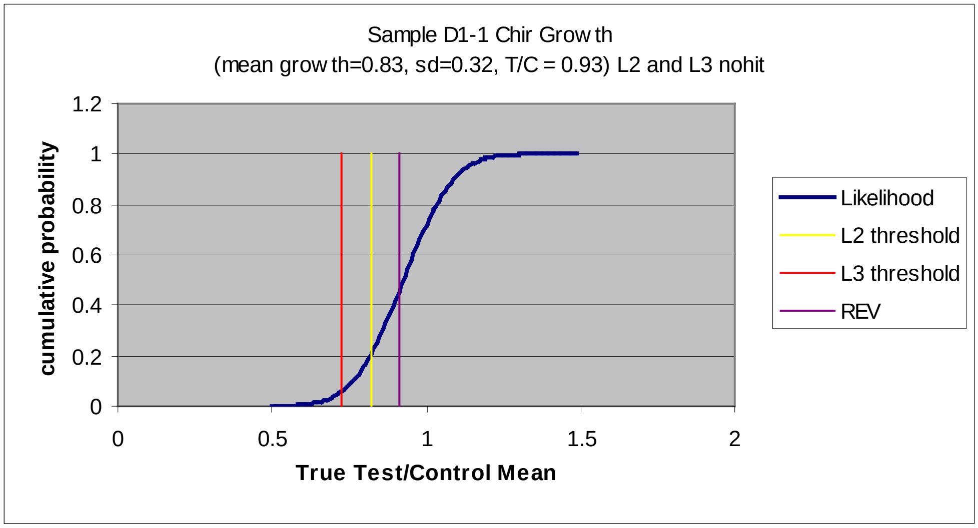
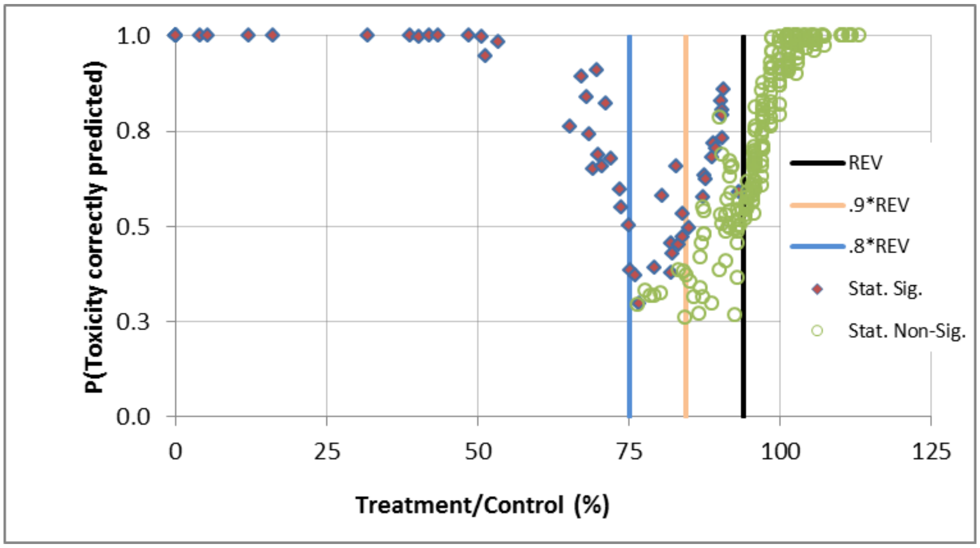


Figure 6-4. Example Distribution of the Probability of a Range of Responses for a Particular Sample from the Study Area

In this analysis, only uncertainty in the test sample mean was considered; however, the control mean also contributes variability and therefore uncertainty to the prediction of toxicity. Additional information on this effect is provided in Attachment 6 (Part C).

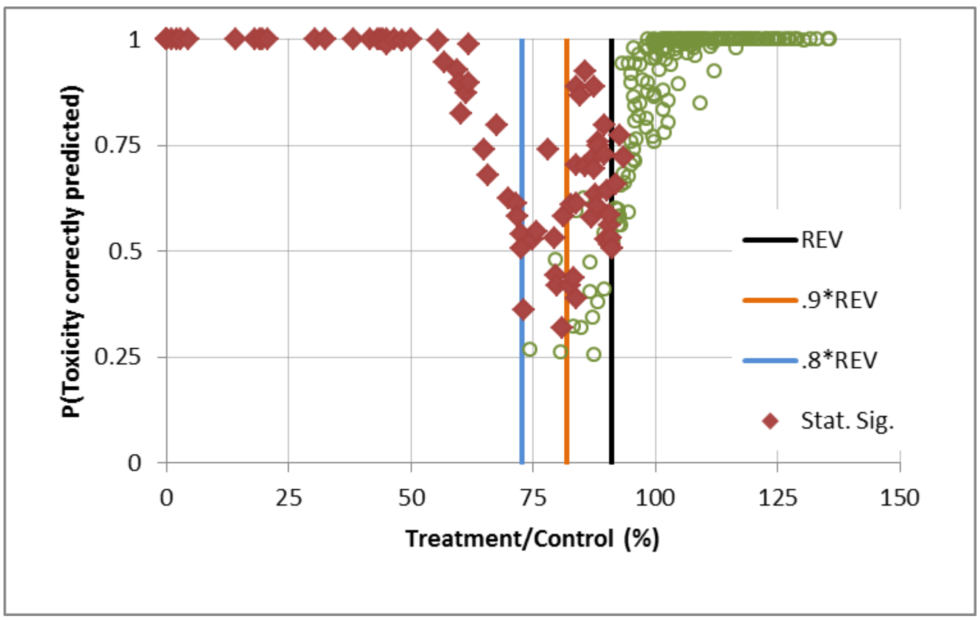
Figures 6-5 through 6-8 present the uncertainty analysis results by bioassay endpoint. The empirical mean control-adjusted bioassay response (i.e., mean treatment response divided by mean control response [T/C]) is on the x-axis and the probability that the “hit” level assigned to that bioassay result is correct is on the y-axis.[42] The thresholds used to define toxicity categories are presented as vertical lines. All points to the right of REV are classified as non-toxic because their observed T/C value exceeded the REV. All open green circles are also classified as non-toxic because they are not significantly different from control. The filled points between REV and 0.9*REV are classified as having low toxicity. The filled points between 0.9*REV and 0.8*REV are classified as moderately toxic, and the filled points to the left of 0.8*REV are classified as highly toxic.

[42] The probability that the bioassay result was correctly assigned is the probability that the true mean exceeds the threshold, given that the observed mean exceeded the threshold. These probabilities are derived from the earlier likelihood calcualtions.



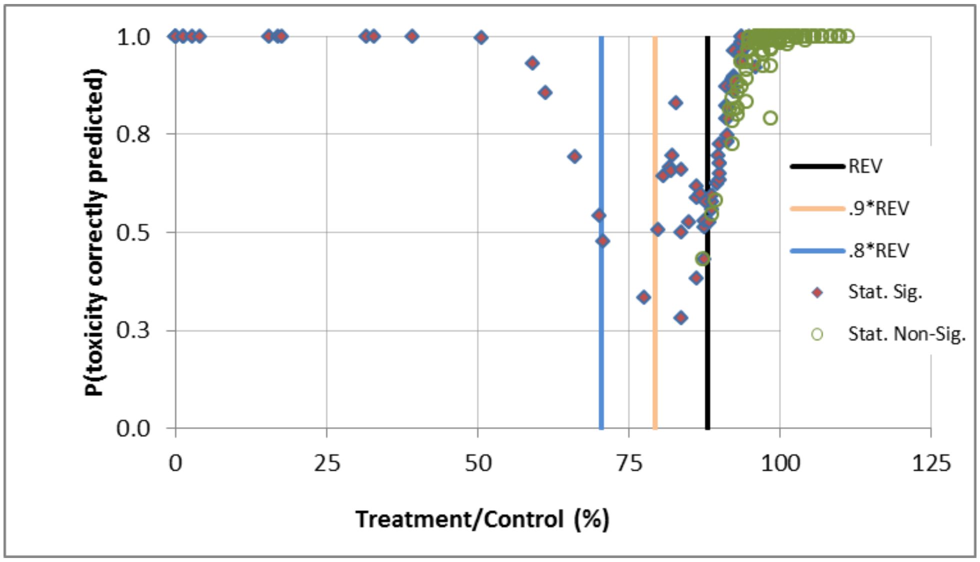
Note: Each dot represents a Portland Harbor bioassay sampling location.

Figure 6-5. Probability of Correctly Predicting Bioassay Hit Classification as a Function of the Empirical Bioassay Response Level – *Chironomus* Survival



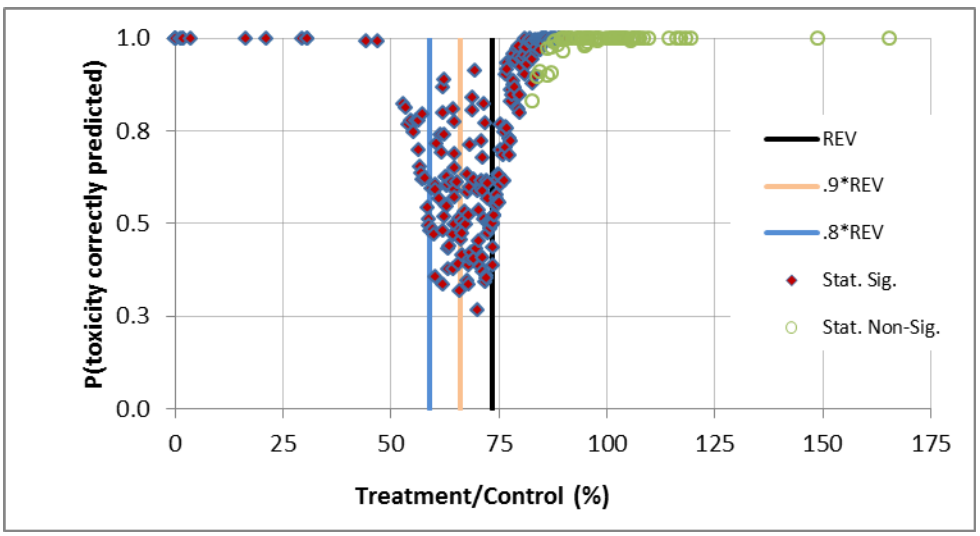
Note: Each dot represents a Portland Harbor bioassay sampling location.

Figure 6-6. Probability of Correctly Predicting Bioassay Hit Classification as a Function of the Empirical Bioassay Response Level – *Chironomus* Biomass



Note: Each dot represents a Portland Harbor bioassay sampling location.

Figure 6-7. Probability of Correctly Predicting Bioassay Hit Classification as a Function of the Empirical Bioassay Response Level – *Hyalella* Survival



Note: Each dot represents a Portland Harbor bioassay sampling location.

Figure 6-8. Probability of Correctly Predicting Bioassay Hit Classification as a Function of the Empirical Bioassay Response Level – *Hyalella* Biomass

For sampling locations that were classified as non-toxic because the mean control-adjusted response exceeded the REV, the probability that the classification was correct was always greater than 50% and usually greater than 80%. Sampling locations classified as non-toxic based on statistical non-significance had mean T/C responses as severe as the L2 category. This latter group contains samples with high uncertainty and very low power for identifying the correct toxicity classification (represented by the green circles to the left of the REV line in Figures 6-5 through 6-8).

Sampling locations classified as highly toxic also had greater than 50% probability of being correctly classified and, in most cases, greater than 80% probability. Sampling locations that were classified as having low toxicity (L1) had a 27 to 93% chance of being correctly classified, although very few samples had probabilities > 80% of being correctly classified. Sampling locations classified moderately toxic (L2) had a 27 to 89% chance of being correctly classified, with few samples having > 80% chance of being correctly classified.

This analysis demonstrates the inherent uncertainty in the toxicity classification scheme used in the BERA. The uncertainty is due to variability in the responses in both the test and the control replicates, which results in low power for the statistical significance test, uncertainty in the severity of the response, or both. Consequently, there is sometimes a large probability that a toxicity response is higher or lower than declared by the BERA classification scheme. This is a source of uncertainty, particularly for benthic toxicity models based on the L2 or moderate toxicity category and the *Hyalella* biomass endpoint.

6.1.1.4 Summary of Invertebrate Sediment Toxicity Assessment

Overall, the majority of the toxicity tests exhibited no or low toxicity (74% to 93%, depending on species and endpoint). When results were combined across species and endpoints, 46 toxicity test sampling locations of the 256 locations in the risk dataset were identified as L3 (high toxicity), based on reduced survival or biomass relative to the reference response in at least one endpoint (see Table 6-3). L2 (moderate toxicity) was the highest effect level among all endpoints at 47 locations; L1 (low toxicity) was the highest response among all endpoints at 58 sampling locations. The *Hyalella* biomass endpoint was primarily responsible for the high number of L1 and L2 hits, and as shown in the uncertainty analysis above, this endpoint has a high degree of uncertainty. No adverse effects (L0) occurred at 105 sampling locations, based on all endpoints.

In cases where L3 was the highest classification at a sampling location, there was usually confirmation from a second endpoint. Reduced chironomid survival and biomass identified the majority of the L3 responses. In the cases where there was only one endpoint defining the L3 response, it was typically a biomass endpoint. Where L2 was the maximum response, it was typically because of the amphipod biomass endpoint, which has a high degree of uncertainty attached to it, as documented in the uncertainty analysis above.

6.1.2 Bivalve Sediment Toxicity Assessment

Toxicity tests based on *Chironomus dilutus* and *Hyalella azteca* are considered the primary indicators of toxic effects on the benthic community, of which bivalves are a part. Results of these tests, presented in Section 6.1.1.2 are considered applicable to the assessment of the potential effects on bivalves. Section 6.1.2.1 presents the qualitative toxicity assessment based on the survival measured during the bioaccumulation tests with *Corbicula fluminea*, as directed in EPA's Problem Formulation (Attachment 2). In these tests a deleterious effect was determined based on a significant difference from mean negative control sample survival. No REV was available for use in the interpretation of this test.

Section 6.1.2.2 presents the toxicity assessment and uncertainty analysis. EPA's Problem Formulation (Attachment 2) also required an assessment of biomass; however, no empirical measures of growth were made as part of the bioaccumulation study (all representations of growth were extrapolations from an initial estimate of average weight), and this endpoint could not be evaluated.

6.1.2.1 Toxicity Assessment Based on Bivalve Mortality

The survival data from the bioaccumulation test with *C. fluminea* performed for the tissue-residue assessment provide a direct measure of the effects on clams in the Study Area. These data are considered qualitative primarily because this bioaccumulation test was not designed to explicitly address sediment toxicity. *C. fluminea* were exposed for 28 days to sediment samples collected at 33 locations in the Study Area (Map 4-3). At the end of the exposure period, adult clam survival in all test sediments ranged from 97.7 to 100.6% of the control survival and no samples were significantly different from their respective negative control.

6.1.2.2 Bivalve Sediment Toxicity Assessment Uncertainty Analysis

The uncertainties associated with this assessment are high because the bioaccumulation tests used in the Portland Harbor BERA were not designed to assess sediment toxicity. To assess the suitability of using toxicity test results for *Chironomus* and *Hyalella* as a surrogate for clams, the midge and amphipod toxicity test results were compared with the bioaccumulation test results for laboratory clams exposed to field-collected sediment. The sediment used in the bioaccumulation testing was collected along tow lines (i.e., areas rather than single grabs), and the locations of these tow lines were superimposed over the *Chironomus* and *Hyalella* toxicity test sediment sampling points to determine where the tow lines overlapped with the sampling locations. Because of the lack of close spatial relationship between the bioaccumulation and toxicity datasets, considerable uncertainty is associated with this analysis.

After superimposing the two sets of sampling locations, toxicity test samples were aggregated according to "nearest neighbor"[43] for comparison to the equivalent bioaccumulation toxicity results (Map 6-6). Where there were multiple toxicity tests associated with the bioaccumulation result, the highest level of toxicity among the bioassays was used to represent the results. Survival results from 21 of the 33 bioaccumulation sediment sampling areas were compared with those from 39 nearby toxicity testing sampling locations. Table 6-4 presents a comparison of the toxicity tests results and bioaccumulation test results.

[43] Nearest neighbor was defined as a bioaccumulation tow that successfully collected benthic tissue and that fell within 133 feet of a bioassay sampling location.

Table 6-4. Comparative Agreement Among Clam and Other Invertebrate Toxicity Tests Based on Survival Endpoint

	Toxicity Test Aggregates (No. of Samples)			
Bioaccumulation Test Endpoint	**Non-Toxic[a]**	**Low Toxicity[b]**	**Moderate Toxicity[c]**	**High Toxicity[d]**
Survival > 80% (n = 21)	13 of 21	5 of 21	0 of 21	3 of 21
Survival ≤ 80% (n = 0)	0 of 21	0 of 21	0 of 21	0 of 21

a Non-toxic was not significantly different from control.
b Low toxicity was defined as significantly different from control, but within 10% of reference threshold.
c Moderate toxicity was defined as significantly different from control and within10 to 20% of reference threshold.
d High toxicity was defined as significantly different from control and exceeding 20% of reference threshold.

Survival results, measured as part of the toxicity tests, agreed with the nearby bioaccumulation toxicity results 76% of the time (19 out of 25 test locations) and indicated no significant toxicity. At the remaining locations, bioassay and bioaccumulation toxicity results did not agree in that no significant mortality occurred in any of the bioaccumulation tests but did occur at varying magnitudes in the bioassays. Overall, there was reasonable concordance between survival measured in the bioaccumulation and toxicity tests.

6.2 PREDICTIVE BENTHIC TOXICITY MODELS

Sediment toxicity within the Study Area where toxicity tests were not conducted was estimated using two models. The models were applied to site-specific data to develop a predictive relationship between surface sediment chemistry and toxicity responses. The FPM developed by RSET (2009) and the LRM developed by the National Oceanic and Atmospheric Administration (NOAA) (Field et al. 1999b) were selected following extensive discussion and review with EPA and other stakeholders. These models allow development of site-specific values (SQVs). The FPM and the LRM use different approaches to first identify the principal chemicals most strongly associated with sediment toxicity and then, for each chemical, develop a site-specific SQV that can be used to predict sediment toxicity to benthic organisms in the LWR with an estimated degree of reliability. It was not possible with the current dataset[44] to validate the performance of either model alone. Both models were therefore used, allowing risk managers to gain insight from and evaluate uncertainties of two quantitative approaches.

The performance of each model was assessed using a suite of reliability measures provided by EPA (2010b). Each measure quantifies an aspect of the relationship between observed toxicity results and those predicted by the model, and can help a risk manager understand the types of errors involved in model output. All of the reliability measures

[44] Low incidence of toxicity at the site reduced the sample sizes available for commonly used cross-validation procedures, and the specificity of site conditions make validation using a different dataset inappropriate.

(EPA 2010b) were calculated from a contingency table using the four primary outcomes of model predictions (see text box):

- True positive (A) and true negative results (D), which are both observed and correctly predicted to be toxic or non-toxic, respectively
- False positive (B) and false negative (C) results, which are observed to be non-toxic but incorrectly predicted to be toxic (B) or observed to be toxic but incorrectly predicted to be non-toxic (C)

Assessing Model Reliability— Primary Outcomes of Model Predictions

A variety of measures of model reliability and error rates can be calculated from the primary four cells of the contingency table below. Four commonly used measures are shown here:

The **false negative rate C/(C+A)** — the number of samples that were toxic based on the toxicity test results but predicted to be non-toxic by the SQVs (false no-hits), divided by the number of samples that were actually toxic (total true hits). This error rate identifies the proportion of toxic samples that was erroneously predicted to be non-toxic.

The **false positive rate B/(B+D)** —the number of samples that were non-toxic based on the toxicity test results but were predicted to be toxic by the SQVs (false hit), divided by the number of samples that were actually non-toxic (total true no-hits). This error rate identifies the proportion of non-toxic samples that was erroneously deemed to be toxic.

The **predicted hit reliability rate A/(A+B)**—the number of correctly predicted toxic results divided by the total number of predicted toxic results. This reliability rate identifies the proportion of samples that were correctly predicted to be toxic.

The **overall reliability rate (A+B)/N** — the number of correctly predicted sampling locations (both true negatives and true positives) divided by the total number of sampling locations. This reliability rate identifies the overall proportion of correctly predicted results.

		Result Predicted by SQV		
		Non-Toxic (≤ SQV)	**Toxic (> SQV)**	
Toxicity Test Result	**Non-Toxic**	True no-hit (negative) D	False hit (positive) B	**Total true no-hits (negative)**
	Toxic	False no-hit (negative) C	True hit (positive) A	**Total true hits (positive)**
		Total predicted no-hits	**Total predicted hits**	**Total sample size (N)**

SQV – sediment quality value

With the procedure defined by EPA (2009a, b), site-specific toxicity thresholds using the reference envelope approach were calculated to define observed toxicity in the models (MacDonald Environmental 2002; Windward 2009a; EPA 2009a, b). The same levels of toxicity used to interpret the empirical toxicity responses (Table 6-2) were used in the models, and both the LRM and FPM were optimized to predict toxicity based on the moderate and high levels of toxicity as defined in Section 6.1.

The details of the benthic predictive models are presented as follows:

- Section 6.2.1 – Summary of the data and process for selecting chemicals to be used in the model

- Section 6.2.2 –FPM process and methods
- Section 6.2.3 – LRM process and methods
- Section 6.2.4 – Derivation of site-specific SQVs using the FPM and LRM
- Section 6.2.5 – Uncertainty associated with toxicity predictions
- Section 6.2.6 – Risk characterization based on predicted toxicity

Figure 6-9 presents a flowchart of the organization of the benthic interpretive models section.

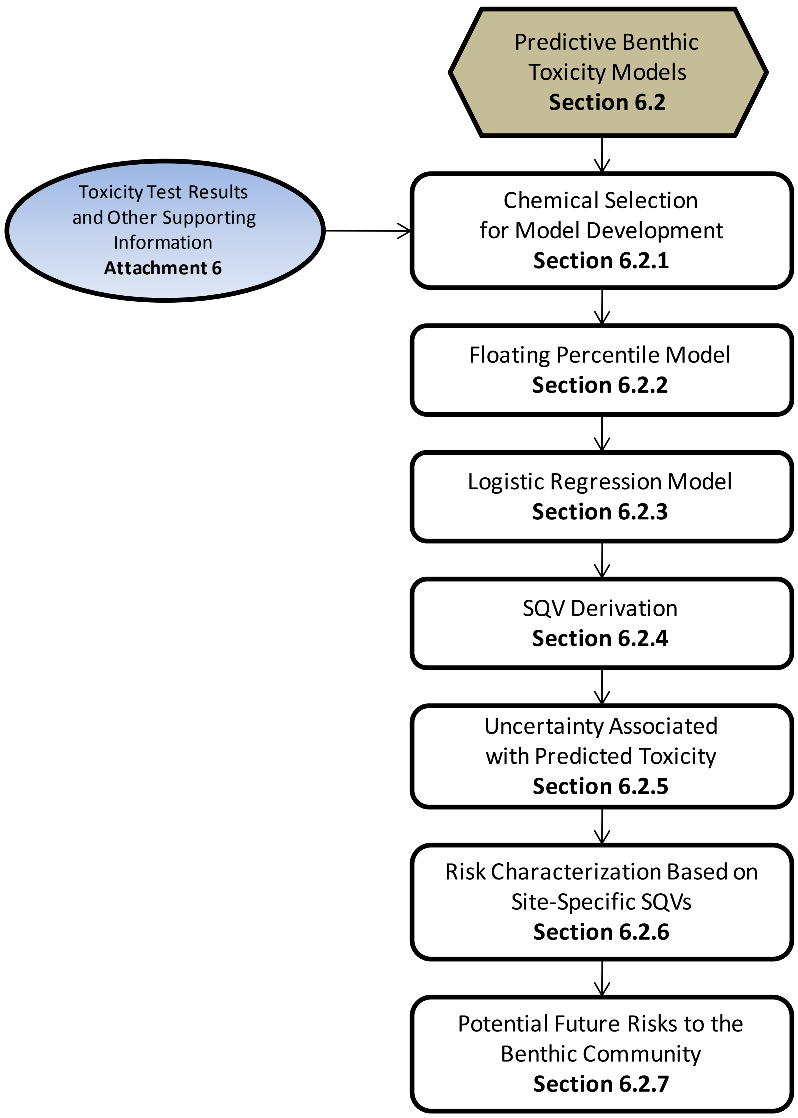


Figure 6-9. Overview of the Predictive Benthic Toxicity Models Section Organization

6.2.1 Chemical Selection for Model Development

The predictive models were developed using the 293 samples with synoptic sediment chemistry and toxicity test data. The toxicity information included results of the 10-day sediment toxicity test measuring survival and biomass in *Chironomus dilutus* and those of the 28-day sediment toxicity test measuring survival and biomass in *Hyalella azteca*. The biological effects levels used in the model were the same as the reference thresholds developed for the interpretation of the empirical toxicity test data (Section 6.1.1.1). In the FPM process, separate models were developed for each of the four endpoints (i.e., *Chironomus* survival, *Chironomus* biomass, *Hyalella* survival, and *Hyalella* biomass), while the LRM developed separate models for pooled species endpoints (survival and biomass). The BERA surface sediment chemistry data used in the models are summarized in Section 4.0.

For individual chemicals in the FPM and LRM models, only detected values were used because undetected chemistry values, which are known only to be "less than" the DL, are not precise enough to develop a predictive relationship between sediment chemistry and sediment toxicity (Avocet 2003). However, when non-detects were part of a chemical group total (e.g., total PCBs), one-half the DL was used in the group totals (summation rules are presented in Attachment 3). Chemical data qualified as N, NJ, and NJT[45] were included in the models, at EPA's direction.

Use of NJ-Qualified Data in the Predictive Models

In the first modeling effort, NJ-qualified surface sediment chemistry data were not included because of uncertainties regarding both the presence and quantity of the chemical. Results of the two predictive models are presented in the *Interpretive Report: Estimating Risks to Benthic Organisms Using Predictive Models Based on Sediment Toxicity Tests* (Windward et al. 2006). EPA (2006a) later questioned the exclusion of NJ-qualified data, stating, "Generally, EPA recommends including the N, NJ, and NJT values for modeling purposes." LWG agreed to revise the surface sediment chemistry dataset by including the NJ-qualified data. Qualified data were used in the predictive modeling effort presented in the *Comprehensive Round 2 Site Characterization Summary and Data Gaps Analysis Report* (Integral et al. 2007), as well as in this BERA.

Eight criteria were used to select chemicals for inclusion in the FPM and LRM models:

1. Only chemicals analyzed in the bioassay dataset could be included.
2. Contaminants specifically excluded or included by EPA direction were respectively excluded or included.
3. Contaminants that had screened in as COPCs during the SLERA process because they exceeded the SLERA SQG were included.

[45] N-qualifier signifies the presumptive evidence of an analyte; for metals, the matrix spike sample recovery was not within control limits, and for organics, the identification was tentative; the analyte exhibited low spectral match parameters but was present. J-qualifier signifies an estimated value. T-qualifier signifies that the value is an average or selected result (following standard project rules).

4. Contaminants without SLERA SQGs were re-evaluated for inclusion in an effort to develop a site-specific SQV for the chemical.
5. Contaminants that were included in a higher level total or covered by another chemical group were excluded to avoid redundancy in the dataset for the FPM. This was not an issue for the LRM because correlations among chemicals do not complicate the process of finding the model with the highest probability of toxicity; therefore both individual chemical constituents and sums were evaluated in the LRM.
6. Contaminants with fewer than 30 detected values were excluded from FPM models because analysts of other datasets from Oregon and Washington have determined that this model requires at least 30 data points to create a usable distribution for the development of SQVs (Avocet 2003). The LRM requires at least 50 detected values for each individual chemical model.
7. Most conventional parameters (specific gravity, liquid limit, individual grain size fraction, percent fines, total organic carbon (TOC), and total solids) were not modeled because they are not considered contaminants. Bulk sediment ammonia and sulfides were retained because they can contribute to toxicity.
8. Finally, for the FPM, contaminants (including conventionals) that met the foregoing seven criteria and whose toxic and non-toxic distributions were significantly different for at least one endpoint using either parametric or non-parametric *t*-tests were included, unless the difference arose for a single L2 endpoint[46] (Attachment 6 [Part D]). This step was not required for the LRM.

Of the analytes (including conventionals) evaluated for inclusion in the model, 43 passed all the selection criteria for the FPM (Nos. 1 through 7 above). These 43 were then tested using parametric and non-parametric *t*-tests (criterion 8 above) for differences between toxic and non-toxic distributions. Ten chemicals were eliminated because there were no significant differences between toxic and non-toxic distributions for any endpoint: mono-, di-, tri-, and tetrabutyltin; alpha- and gamma-hexachlorocyclohexane; methoxychlor; BEHP; butyl benzyl phthalate; and hexachlorobenzene. Arsenic, selenium, and total dioxin and furans were eliminated because toxic and non-toxic distributions differed only for a single L2 endpoint (Table 6-6). Thirty chemicals were accepted for use in the FPM (Table 6-5, Attachment 6 [Part D]).

For the LRM chemical selection criteria (applicable criteria from numbers 1 through 7 above), 63 chemicals, including summed parameters, were included in the individual model development. Individual PCB congeners, dioxin/furans, and most sediment conventionals (except for ammonia and sulfides) were excluded. Perylene, dibenzothiophene, and 1-methylnaphthalene were excluded from model selection because these chemicals were analyzed only in the Phase 3 data collection (60 samples).

[46] Both a parametric t-test using log transformed data and a nonparametric t-test (Mann Whitney U Test) were conducted in order to ensure that the most conservative test was used for each chemical. Tests were one sided and conducted at alpha = 0.05.

Table 6-5. Analytes Included in FPM and LRM

Contaminant	Included in FPM?	Included in LRM?	No. of Detects	No. of Locations Sampled
Metals				
Antimony	No, did not exceed SLERA SQG	Yes	224	293
Arsenic	No, toxic and non-toxic distributions only differed for one L2 endpoint	Yes, exceeds SLERA SQG	293	293
Cadmium	Yes, exceeds SLERA SQG	Yes, exceeds SLERA SQG	291	293
Chromium	Yes, exceeds SLERA SQG	Yes, exceeds SLERA SQG	292	293
Copper	Yes, exceeds SLERA SQG	Yes, exceeds SLERA SQG	293	293
Lead	Yes, exceeds SLERA SQG	Yes, exceeds SLERA SQG	293	293
Mercury	Yes, exceeds SLERA SQG	Yes, exceeds SLERA SQG	289	293
Nickel	Yes, exceeds SLERA SQG	Yes, exceeds SLERA SQG	281	293
Selenium	No, toxic and non-toxic distributions only differed for one L2 endpoint	Yes, exceeds SLERA SQG	120	233
Silver	Yes, exceeds SLERA SQG	Yes, exceeds SLERA SQG	293	293
Zinc	Yes, exceeds SLERA SQG	Yes, exceeds SLERA SQG	293	293
Butyltins				
Tributyltin	No, toxic and non-toxic distributions not significantly different	Yes	73	74
PAHs				
2-Methylnaphthalene	No, PAHs evaluated as intermediate sums	Yes, exceeds SLERA SQG	272	293
Acenaphthene	No, PAHs evaluated as intermediate sums	Yes, exceeds SLERA SQG	274	293
Acenaphthylene	No, PAHs evaluated as intermediate sums	Yes, exceeds SLERA SQG	274	293
Anthracene	No, PAHs evaluated as intermediate sums	Yes, exceeds SLERA SQG	275	293
Benzo(a)anthracene	No, PAHs evaluated as intermediate sums	Yes, exceeds SLERA SQG	281	293

Table 6-5. Analytes Included in FPM and LRM

Contaminant	Included in FPM?	Included in LRM?	No. of Detects	No. of Locations Sampled
Benzo(a)pyrene	No, PAHs evaluated as intermediate sums	Yes, exceeds SLERA SQG	281	293
Benzo(b)fluoranthene	No, PAHs evaluated as intermediate sums	Yes, exceeds SLERA SQG	277	293
Benzo(g,h,i)perylene	No, PAHs evaluated as intermediate sums	Yes, exceeds SLERA SQG	282	293
Benzo(k)fluoranthene	No, PAHs evaluated as intermediate sums	Yes, exceeds SLERA SQG	277	293
Chrysene	No, PAHs evaluated as intermediate sums	Yes, exceeds SLERA SQG	278	293
Dibenzo(a,h)anthracene	No, PAHs evaluated as intermediate sums	Yes, exceeds SLERA SQG	281	293
Fluoranthene	No, PAHs evaluated as intermediate sums	Yes, exceeds SLERA SQG	285	293
Fluorene	No, PAHs evaluated as intermediate sums	Yes, exceeds SLERA SQG	271	293
Indeno(1,2,3-c,d)pyrene	No, PAHs evaluated as intermediate sums	Yes, exceeds SLERA SQG	281	293
Naphthalene	No, PAHs evaluated as intermediate sums	Yes, exceeds SLERA SQG	232	293
Phenanthrene	No, PAHs evaluated as intermediate sums	Yes, exceeds SLERA SQG	282	293
Pyrene	No, PAHs evaluated as intermediate sums	Yes, exceeds SLERA SQG	285	293
Total PAHs	No, PAHs evaluated as intermediate sums	Yes, exceeds SLERA SQG	288	293
Total HPAHs	Yes, exceeds SLERA SQG	Yes, exceeds SLERA SQG	288	293
Total LPAHs	Yes, exceeds SLERA SQG	Yes, exceeds SLERA SQG	283	293
SVOCs				
Benzyl alcohol	Yes, exceeds SLERA SQG	Yes, exceeds SLERA SQG	61	293
Carbazole	Yes, exceeds SLERA SQG	Yes, exceeds SLERA SQG	205	293
Dibenzofuran	Yes, exceeds SLERA SQG	Yes, exceeds SLERA SQG	265	293
Phthalates				
BEHP	No, toxic and non-toxic distributions not significantly different	Yes	209	293
Dibutyl phthalate	No, did not exceed SLERA SQG	Yes	138	293

Table 6-5. Analytes Included in FPM and LRM

Contaminant	Included in FPM?	Included in LRM?	No. of Detects	No. of Locations Sampled
Phenols				
4-Methylphenol	Yes, exceeds SLERA SQG	Yes, exceeds SLERA SQG	123	293
Pentachlorophenol	Yes, exceeds SLERA SQG	No	75	293
Phenol	Yes, exceeds SLERA SQG	Yes, exceeds SLERA SQG	88	293
PCBs				
Total PCBs	Yes, exceeds SLERA SQG	Yes	231	292
Pesticides				
Sum DDD	Yes, exceeds SLERA SQG	Yes	280	292
Sum DDE	Yes, exceeds SLERA SQG	Yes	269	292
Sum DDT	Yes, exceeds SLERA SQG	No	242	292
Total DDx	No, evaluated as part of intermediate sum	Yes	285	292
Aldrin	Yes, exceeds SLERA SQG	Yes	85	272
2,4'-DDD	No, evaluated as part of intermediate sum	Yes	231	292
2,4'-DDE	No, evaluated as part of intermediate sum	Yes	108	292
2,4'-DDT	No, evaluated as part of intermediate sum	Yes	175	292
4,4'-DDD	No, evaluated as part of intermediate sum	Yes	223	283
4,4'-DDE	No, evaluated as part of intermediate sum	Yes	274	292
4,4'-DDT	No, evaluated as part of intermediate sum	Yes	269	292
cis-Chlordane	No, evaluated as part of total chlordane	Yes	154	292
Total chlordane	Yes, exceeds SLERA SQG	Yes	235	292
Dieldrin	Yes, exceeds SLERA SQG	No	48	292
Endrin	Yes, exceeds SLERA SQG (n = 31)	No	31	170
alpha-Endosulfan	No, evaluated as part of total endosulfan	Yes	56	292

Table 6-5. Analytes Included in FPM and LRM

Contaminant	Included in FPM?	Included in LRM?	No. of Detects	No. of Locations Sampled
Total endosulfan	Yes, no SQV but > 30 detects	No	85	292
Endrin ketone	Yes, no SQV but > 30 detects	Yes	70	292
alpha-HCH	No, toxic and non-toxic distributions not significantly different	Yes	74	282
beta-HCH	Yes, no SQV but > 30 detects	Yes	192	292
delta-HCH	Yes, no SQV but > 30 detects	Yes	52	292
cis-Nonachlor	No, evaluated as part of total chlordane	Yes	92	292
trans-Nonachlor	No, evaluated as part of total chlordane	Yes	124	291
Petroleum Hydrocarbons				
Residual range	No, excluded because highly variable mixture with both toxic and non-toxic components. Used PAHs to address toxic components of petroleum	Yes	196	207
Diesel range	No, excluded because highly variable mixture with both toxic and non-toxic components. Used PAHs to address toxic components of petroleum	Yes	202	207
Conventional Contaminants				
Ammonia[a]	Yes, affected model performance	Yes	292	293
Sulfide[a]	Yes, affected model performance	Yes	240	293

BEHP – bis(2-ethylhexyl) phthalate
CERCLA – Comprehensive Environmental Response, Compensation, and Liability Act
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
FPM – floating percentile model
HPAH – high-molecular-weight polycyclic aromatic hydrocarbon
HCH – hexachlorocyclohexane
L2 – Level 2 (moderate toxicity)
LPAH – low-molecular-weight polycyclic aromatic hydrocarbon
LRM – logistic regression model
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
SLERA – screening-level ecological risk assessment
SQG – sediment quality guideline
SVOC – semivolatile organic compound
total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT, and 4,4′-DDT)

Table 6-6. Analytes Not Included in Either the FPM or LRM and Reason for Exclusion

Contaminant	Rationale	No. of Detects	N
Metals			
Aluminum	No, by EPA direction	293	293
Chromium (hexavalent)	No, < 30 detects	3	7
Cobalt	Not analyzed in bioassay dataset	NA	NA
Magnesium	Not analyzed in bioassay dataset	NA	NA
Manganese	Not analyzed in bioassay dataset	NA	NA
Thallium	Not analyzed in bioassay dataset	NA	NA
Tin	Not analyzed in bioassay dataset	NA	NA
Titanium	Not analyzed in bioassay dataset	NA	NA
Vanadium	Not analyzed in bioassay dataset	NA	NA
Butyltins			
Butyltin ion	No SLERA SQG, toxic and non-toxic distributions not significantly different	70	74
Dibutyltin ion	No SLERA SQG, toxic and non-toxic distributions not significantly different	74	74
Tetrabutyltin	No SLERA SQG, toxic and non-toxic distributions not significantly different	34	74
PAHs	**PAHs were evaluated as intermediate sums**	NA	NA
1,6,7-Trimethylnaphthalene	Not analyzed in bioassay dataset	NA	NA
1-Methylnaphthalene	No, distribution covered by LPAHs	58	60
1-Methylphenanthrene	Not analyzed in bioassay dataset	NA	NA
2,6-Dimethylnaphthalene	Not analyzed in bioassay dataset	NA	NA
Benzo(e)pyrene	No, distribution covered by HPAHs	59	60
Total benzofluoranthenes	No, distribution covered by HPAHs	280	293
Perylene	No, distribution covered by HPAHs	60	60
Phthalates			
Butyl benzyl phthalate	Toxic and non-toxic distributions not significantly different	96	293
Diethyl phthalate	No, did not exceed SLERA SQG	44	293
Dimethyl phthalate	No, did not exceed SLERA SQG	19	293
Di-n-octyl phthalate	No, < 30 detects	5	293

Table 6-6. Analytes Not Included in Either the FPM or LRM and Reason for Exclusion

Contaminant	Rationale	No. of Detects	N
SVOCs			
1,2,4-Trichlorobenzene	No, < 30 detects	6	293
1,2-Dichlorobenzene	No, < 30 detects	5	293
1,3-Dichlorobenzene	No, did not exceed SLERA SQG	4	293
1,4-Dichlorobenzene	No, < 30 detects	11	293
2-Chloronaphthalene	No, < 30 detects	0	293
3-Nitroaniline	No, < 30 detects	0	292
4-Chloroaniline	No, < 30 detects	1	292
4-Nitroaniline	No, < 30 detects	1	293
Aniline	No, < 30 detects	16	288
Benzoic acid	No, < 30 detects	26	293
Bis(2-chloroethyl) ether	No, < 30 detects	2	293
Bis(2-chloroisopropyl) ether	Synonym for bis(2-chloro-1-methyethyl)ether	NA	NA
Diphenyl	Not analyzed in bioassay dataset	NA	NA
Hexachlorobenzene	Toxic and non-toxic distributions not significantly different	156	293
Hexachlorobutadiene	No, < 30 detects	24	293
Hexachloroethane	No, < 30 detects	27	293
n-Nitrosodiphenylamine	No, < 30 detects	10	293
Phenols			
2,3,4,5-Tetrachlorophenol	No, < 30 detects	5	293
2,3,5,6-Tetrachlorophenol	No, < 30 detects	3	60
2,4,5-Trichlorophenol	No, < 30 detects	6	293
2,4,6-Trichlorophenol	No, < 30 detects	19	293
2,4-Dichlorophenol	No, < 30 detects	2	293
2-Chlorophenol	No, < 30 detects	1	293
4-Chloro-3-methylphenol	No, < 30 detects	6	293
2,4-Dimethylphenol	No, < 30 detects	1	241
2-Methylphenol	No, < 30 detects	3	293
PCBs			
Aroclor 1016	No, evaluated as part of Total PCBs	0	292
Aroclor 1248	No, evaluated as part of Total PCBs	97	292

Table 6-6. Analytes Not Included in Either the FPM or LRM and Reason for Exclusion

Contaminant	Rationale	No. of Detects	N
Aroclor 1254	No, evaluated as part of Total PCBs	149	292
Aroclor 1260	No, evaluated as part of Total PCBs	218	292
Dioxins/Furans			
2,3,7,8-TCDD	No, < 30 detects	8	70
Total dioxin/furan TEQ	No SLERA SQG, toxic and non-toxic distributions only differed for one L2 endpoint	70	70
Pesticides			
gamma-HCH	Toxic and non-toxic distributions not significantly different	58	292
Heptachlor	No, < 30 detects	20	292
Heptachlor epoxide	No, < 30 detects	6	291
trans-Chlordane	No, evaluated as part of total chlordanes	155	292
beta-Endosulfan	No, evaluated as part of total endosulfans	39	292
Endosulfan sulfate	No, evaluated as part of total endosulfans	3	291
Endrin aldehyde	No, < 30 detects	6	292
Methoxychlor	Toxic and non-toxic distributions not significantly different	53	292
Mirex	No, < 30 detects	7	292
Oxychlordane	No, evaluated as part of total chlordanes	22	291
Toxaphene	No, < 30 detects	0	292
Herbicides			
2,4,5-T	No, < 30 detects	0	48
2,4-D	No, < 30 detects	6	48
2,4-DB	No, < 30 detects	1	48
Dichloroprop	No, < 30 detects	0	48
MCPA	No, < 30 detects	2	48
MCPP	No, < 30 detects	1	48
Silvex	No, < 30 detects	0	48
VOCs			
Various volatile compounds	No SQVs. No VOCs had > 30 detects.	NA	NA
Petroleum Hydrocarbons			
Gasoline range	No, < 30 detects	21	145
Conventional Contaminants			

Table 6-6. Analytes Not Included in Either the FPM or LRM and Reason for Exclusion

Contaminant	Rationale	No. of Detects	N
Cyanide	Not analyzed in bioassay dataset	NA	NA
Perchlorate	Not analyzed in bioassay dataset	NA	NA

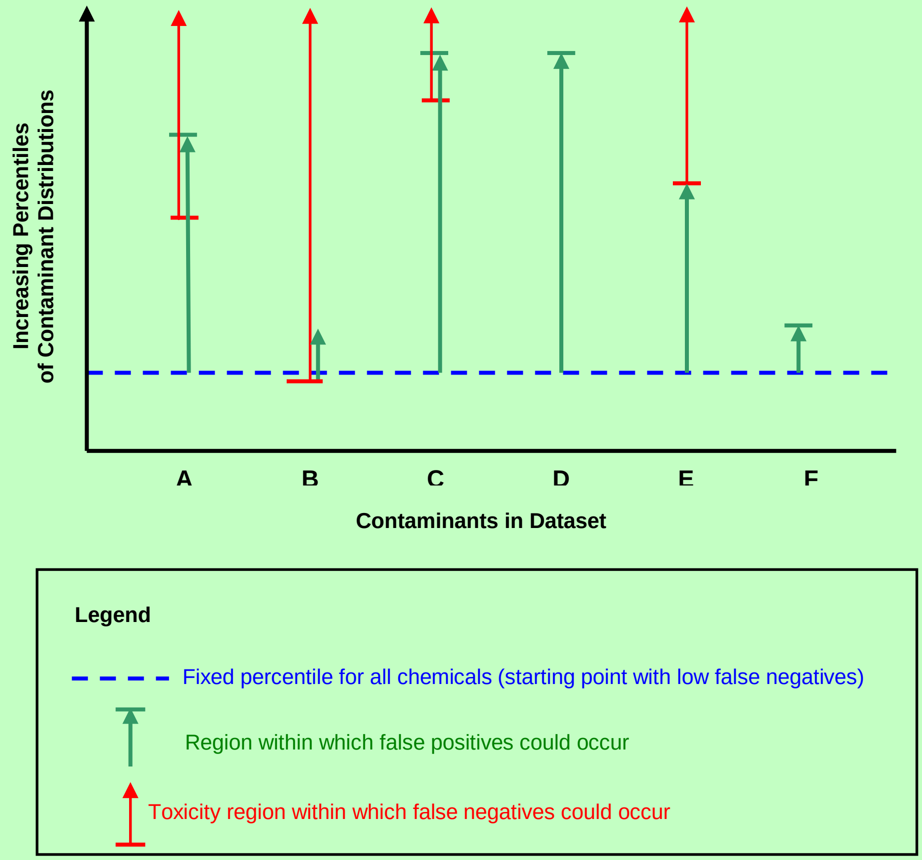
2,4-D – 2,4-dichlorophenoxyacetic acid
2,4-DB – 4-(2,4-dichlorophenoxy)butyric acid
2,4,5-T – 2,4,5-trichlorophenoxyacetic acid
CERCLA – Comprehensive Environmental Response, Compensation, and Liability Act
EPA – US Environmental Protection Agency
FPM – floating percentile model
HCH – hexachlorocyclohexane
HPAH – high-molecular-weight polycyclic aromatic hydrocarbon
LPAH – low-molecular-weight polycyclic aromatic hydrocarbon
LRM – logistic regression model
MCPA – 2-methyl-4-chlorophenoxyacetic acid
MCPP – methylchlorophenoxypropionic acid
N – total number of samples
NA – not applicable
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
SLERA – screening-level ecological risk assessment
SQG – sediment quality guideline
SVOC – semivolatile organic compound
TCDD – tetrachlorodibenzo-*p*-dioxin
TEQ – toxic equivalent
VOC – volatile organic compound

6.2.2 Floating Percentile Model

The FPM uses an iterative search process to identify sets of site-specific SQVs that minimize false-positive prediction error rates across all chemicals for selected cross-chemical false-negative error rates and allows managers to choose the set of final reliability measures and associated SQVs that best meet management objectives.

For each model run, the user selects a maximum acceptable false negative rate (i.e., rate associated with erroneously concluding the sediments are not toxic) and the model then searches the range of chemical concentrations to find the set of thresholds that minimizes the associated false positive rate (i.e., rate associated with erroneously concluding that sediments are toxic). This search process is unlike most other existing SQV sets, which base the SQVs for all chemicals on the same percentile of the toxic (i.e., exceedance of an effects threshold) or non-toxic distribution. After the FPM adjustment process, most chemicals should be at or near a level associated with the onset of toxicity in the dataset, rather than at a level arbitrarily assigned by a fixed percentile (see text box below). In this manner, optimized site-specific SQVs can be developed for a number of different target false negative rates, allowing the trade-offs between false negatives and false positives to be evaluated.

Floating Percentile Model



The basic concept behind the FPM is to select an optimal percentile of individual contaminant concentrations within a multiple contaminant dataset that, collectively, accurately predict toxicity. The selected percentile represents a contaminant-specific threshold that minimizes prediction errors (i.e., false positive and false negative rates), based on paired chemistry-toxicity test samples. This optimized percentile typically occurs within the range in which concentrations associated with no toxicity overlap with those in which toxicity is expressed, for either an individual or pooled toxicity test result.

In the above figure, the y-axis represents the percentile of each contaminant's overall concentration distribution. The green vertical line for each contaminant shows the concentration range over which toxicity did not occur (region where false positives could occur), and the red vertical line shows the range over which toxicity did occur (region where false negatives could occur). The blue dashed line represents an initial minimum threshold percentile associated with correct predictions of no toxicity (low false positive rate) that is selected for all contaminants. The threshold for each individual contaminant is raised until it approaches the concentration associated with the onset of toxicity to minimize incorrect predictions of toxicity (i.e., low false positive rates) and then adjusted further until the rate at which toxicity is correctly predicted over all contaminants is maximized. In the figure, the onset of toxicity within a contaminant distribution varies by contaminant and may even occur at the minimum value (e.g., Contaminant B) or not within the measured range (Contaminant D). Once each contaminant has been individually adjusted upward, the false positive rate will have been significantly reduced while a low false negative rate is retained. Most contaminants should be at or near their actual toxicity range, rather than at a level arbitrarily assigned by a fixed percentile.

In this project, modeling was conducted using the automated FPM Microsoft Excel® spreadsheets provided by RSET (Anderson 2008) (see text box below). For each endpoint and effect level, the FPM was run with five different initial false negative rates in order to observe the sensitivity of the resulting reliability measures and SQVs to the initial false negative rate. All reliability measures requested by EPA (2010b) were calculated for each model run.

Explicit FPM Steps Using the RSET Spreadsheets

1. Create separate FPMCalc.xls file for each endpoint that contains the endpoint toxicity data paired with chemistry data for selected contaminants (enter into Data Table tab).
2. Run FPM for each endpoint and specified range of false negative rates (must run each false negative separately—option to enter a range of false negative rates does not work correctly in the RSET spreadsheets).
   a. Enter false negative rate.
   b. Press "Calculate Floating Percentiles" macro button on "ControlScreen" tab.
   c. For sensitivity analysis, conducted separate runs for 5, 10, 15, 20, 25% false negative rate.

After evaluating the suite of reliability measures that resulted from the set of FPM runs for each endpoint and toxicity level, the model with the most equal false negative and false positive rate for each endpoint (Table 6-7) was selected. Additional reliability measures are presented in Attachment 6 (Part E).

Table 6-7. Selected Error and Reliability Measures for FPM Models with Most Balanced False Positive and False Negative Rates for Each Endpoint

Endpoint by Toxicity Level	Rate[a]	False Negative Rate (%)	False Positive Rate (%)	True Positive Rate (%)[b]	True Negative Rate (%)[c]	Positive Predictive Power (%)[d]	Negative Predictive Power (%)[e]	Overall Reliability (%)
Level 3								
Chironomus biomass	17	16.3	15.6	83.7	84.4	48.0	96.8	84.3
Chironomus survival	18	15.6	12.3	84.4	87.7	45.8	97.9	87.4
Hyalella biomass	25	24.1	22.7	75.9	77.3	26.8	96.7	77.1
Hyalella survival	15	10.5	11.7	89.5	88.3	34.7	99.2	88.4
Level 2								
Chironomus biomass	20	19.2	19.9	80.8	80.1	46.7	95.1	80.2
Chironomus survival	24	22.7	24.9	77.3	75.1	35.4	94.9	75.4
Hyalella biomass	25	24.7	54.5	75.3	45.5	31.4	84.7	52.9

Table 6-7. Selected Error and Reliability Measures for FPM Models with Most Balanced False Positive and False Negative Rates for Each Endpoint

Endpoint by Toxicity Level	Rate[a]	False Negative Rate (%)	False Positive Rate (%)	True Positive Rate (%)[b]	True Negative Rate (%)[c]	Positive Predictive Power (%)[d]	Negative Predictive Power (%)[e]	Overall Reliability (%)
Hyalella survival	15	14.3	11.4	85.7	88.6	36.7	98.8	88.4

[a] Initial maximum false negative rate entered into FPM (maximum false negative rate allowable by FPM for that run).
[b] True positive rate may also be referenced as hit reliability or sensitivity.
[c] True negative rate may also be referenced as no-hit reliability, specificity, or efficiency.
[d] Positive predictive power may also be referenced as predicted hit reliability.
[e] Negative predictive power may also be referenced as predicted no-hit reliability.
FPM – floating percentile model

The FPM identified the maximum concentration within the distribution of the paired toxicity-chemistry dataset as the L3 SQV for 6 of the 30 chemicals entered into the model for all endpoints: lead, nickel, zinc, pentachlorophenol, aldrin, and total chlordane (Table 6-8, column 2). SQVs for these chemicals cannot be used to predict toxicity because these SQVs are minimum values and do not describe the onset of toxicity.

The SQVs for 10 chemicals were set to the maximum non-toxic concentration (apparent effects threshold [AET][47]) for all L3 endpoints (i.e., concentrations higher than these SQVs were always toxic) (Table 6-8, column 3), and SQVs for 10 additional chemicals were set to the AET for two or more L3 endpoints (Table 6-8, column 3). The SQVs for 12 other chemicals were set below the maximum non-toxic concentration for one or more endpoints (typically the biomass endpoints) and were the major chemicals responsible for the model's false positive error rates (Table 6-8, Columns 4 and 5).

Table 6-8. Relationships between FPM L3 SQVs, the Maximum Concentration, and Apparent Effects Thresholds

Contaminant	Number of L3 Endpoints			
	SQV = AET		SQV < AET	
	AET = Max	AET < Max	AET = Max	AET < Max
4-Methylphenol			4 of 4[s]	
Aldrin	4 of 4[b]			
Chromium	3 of 4		1 of 4[c]	

[47] The AET is the maximum chemical concentration associated with a non-toxic sample. This threshold is identified as part of the FPM output and assists in the evaluation of the reliability of the resulting SQVS

Table 6-8. Relationships between FPM L3 SQVs, the Maximum Concentration, and Apparent Effects Thresholds

	Number of L3 Endpoints			
	SQV = AET		SQV < AET	
Contaminant	AET = Max	AET < Max	AET = Max	AET < Max
Lead	4 of 4[b]			
Nickel	4 of 4[b]			
Pentachlorophenol	4 of 4[b]			
Total chlordane	4 of 4[b]			
Total endosulfan	3 of 4[b]		1 of 4[c]	
Zinc	4 of 4[b]			
Benzyl alcohol		4 of 4		
beta-HCH		4 of 4		
Cadmium		4 of 4		
Dieldrin		4 of 4		
Endrin ketone		4 of 4		
Phenol		4 of 4		
Silver		4 of 4		
Sum DDE		4 of 4		
Sum DDT		4 of 4		
Total HPAHs		4 of 4		
Carbazole		3 of 4		1 of 4[c]
Copper	1 of 4	3 of 4		
delta-HCH		3 of 4		1 of 4[c]
Dibenzofuran		2 of 4		2 of 4[c]
Endrin	2 of 4	2 of 4[d]		
Mercury		3 of 4		1 of 4[c]
Sum DDD		2 of 4		2 of 4[c,e]
Total LPAHs		2 of 4		2 of 4[e]
Total PCBs		3 of 4		1 of 4[c]
Ammonia		3 of 4		1 of 4[c]
Sulfide			3 of 4[c,e]	1 of 4[c]

[a] All SQVs (i.e., derived for any endpoint) may contribute to false predictions of toxicity.
[b] SQVs cannot be used to predict toxicity because concentration defining onset of toxicity unknown.
[c] SQV based on biomass endpoint(s) may contribute to false predictions of toxicity.
[d] The L3 SQV for endrin-based *Chironomus* growth is slightly greater (20.8 µg/kg dw) than the AET (20.7 µg/kg dw) but was included in the category of being equivalent to the AET.
[e] SQV based on survival endpoint(s) may contribute to false predictions of toxicity.

AET – apparent effects threshold
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
dw – dry weight
FPM – floating percentile model
HCH – hexachlorocyclohexane
HPAH – high-molecular-weight polycyclic aromatic hydrocarbon
L3 – Level 3 (high toxicity)
LPAH – low-molecular-weight polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
SQV – sediment quality value

6.2.3 Logistic Regression Model

Individual LRMs (Field et al. 1999b; Field et al. 2002; EPA 2005h) were developed for the chemicals shown in Table 6-5. For each chemical, model developers fit 72 unique models from which the best individual model was selected. For each chemical, the 72 individual models were developed using the following:

- Two pooled species endpoints (i.e., *Hyalella* and *Chironomus*)
- Three toxicity levels (i.e., L1, L2, or L3)
- Four sediment chemistry normalizations (i.e., dry weight, OC-normalized, fines-adjusted dry-weight concentrations, and fines-adjusted OC-normalized concentrations)
- Three screening criteria applied independently to each individual chemical model (i.e., exclude toxic samples that were less than or equal to one times the arithmetic mean (1X), two times the arithmetic mean (2X), or two times the geometric mean (2G) of the non-toxic samples; see Attachment 6 (Part F) for more detail).

For each chemical, individual models within the suite of candidate models were scored based on predictive performance within the Portland Harbor dataset and the best model for each chemical was selected. This process did not restrict the set of best individual chemical models to be consistent across chemicals, so that the final suite of models was a mixture of species, toxicity levels, chemistry normalizations, and screening criteria. The set of best individual chemical models was combined into a single multi-chemical model that predicted toxicity for a given sample as the maximum probability of toxicity (pMax) across all chemicals. The final list of individual chemicals in the combined pMax model was reduced to those chemicals that, collectively, provided the best reliability and predictive accuracy. The methods used to develop the individual models and the combined pMax model are briefly summarized below, and described in greater detail in Attachment 6 (Part F).

There were four general steps in the model development process:

1. **Develop set of all possible models (Set 1) –** Develop 72 individual regression models for each chemical (2 species by 3 toxicity levels by 4 sediment chemistry normalizations by 3 screening criteria).
2. **Develop suite of candidate models (Set 2) –** Omit models from Set 1 that do not meet the chi-square goodness-of-fit, gradient, or hit reliability criteria.
3. **Select the best model for each chemical (Set 3) –** Using several reliability criteria, compare among models within Set 2 to subjectively select the single "best" model for each chemical.
4. **Develop the combined multi-chemical pMax model –** Use the models in Set 3 to predict toxicity for the complete dataset (293 bioassay sampling locations) and evaluate reliability metrics. Observed toxicity is represented by pooling across bioassay species and endpoints, at L2 or greater. Chemicals that contribute to overall high false positives or low hit reliability are omitted from the final model.

For the combined set of selected individual chemical models (produced in Step 4 above), predictions of which sampling locations were toxic or not toxic using the pMax were made from the sampling location-specific chemical concentrations. Reliability statistics were calculated where observed toxicity was represented by an overall pooled bioassay response at L2 and L3. The reliability statistics (Attachment 2 of EPA 2010b) for this final combined model are shown in Attachment 6 (Part G) for each pMax threshold from 0.2 to 0.8, in 0.01 increments. Selected reliability metrics for a subset of pMax thresholds are shown in Table 6-9. The characteristics of this dataset, with substantial overlap in chemical concentrations for the two populations of toxic and non-toxic samples, preclude having both error rates at a low level. EPA has selected pMax values of 0.59 (L3) and 0.5 (L2) for use in the benthic risk characterization.

Table 6-9. Selected Error and Reliability Metrics for Selected LRM pMax Model for Pooled Bioassay L2 and L3 Responses

Threshold for Toxicity Predictions	Total Toxic Samples	Total Non-Toxic Samples	False Negative Rate (%)	False Positive Rate (%)	True Positive Rate (%)[a]	True Negative Rate (%)[b]	Positive Predictive Power (%)[c]	Negative Predictive Power (%)[d]	Overall Reliability (%)	Comment
Level 2										
pMax > 0.25	106	187	37.7	37.4	62.3	62.6	48.5	74.5	62.5	Balanced FN/FP
pMax > 0.45	106	187	49.1	18.2	50.9	81.8	61.4	74.6	70.6	FN < 0.5; FP < 0.2
pMax > 0.50	106	187	50.0	9.6	50.0	90.1	74.6	76.1	75.8	FN < 0.5; FP < 0.1
Level 3										
pMax > 0.37	55	238	29.1	29.4	70.9	70.6	35.8	91.3	70.6	Balanced FN/FP
pMax > 0.47	55	238	30.9	18.9	69.1	81.1	45.8	91.9	78.8	FN < 0.5; FP < 0.2
pMax > 0.59	55	238	38.2	10.1	61.8	89.9	58.6	91.1	84.6	FN < 0.5; FP < 0.1
pMax > 0.71	55	238	49.1	7.6	50.9	92.4	60.9	89.1	84.6	FN < 0.5; FP < 0.1

[a] True positive rate may also be referenced as hit reliability or sensitivity.
[b] True negative rate may also be referenced as no-hit reliability, specificity, or efficiency.
[c] Positive predictive power may also be referenced as predicted hit reliability.
[d] Negative predictive power may also be referenced as predicted no-hit reliability.

FN – false negative
FP – false positive
L2 – Level 2 (low toxicity)
L3 – Level 3 (high toxicity)
LRM – logistic regression model
pMax – maximum probability of toxicity

6.2.4 SQV Derivation

Ten sets of SQVs defining two toxicity thresholds (L2 and L3) were derived from the FPM and LRM. The FPM SQVs are based on individual endpoint models, whereas the LRM SQVs are based on combined effects (survival and biomass) endpoints. One or both models include sediment chemical stressors not typically addressed by CERCLA (TPH and sulfides) that appeared to explain some of the observed toxicity. Although SQVs were derived for these chemicals, they were not ultimately used to identify benthic community risk areas. A discussion of the contribution of these chemicals to benthic invertebrate toxicity is provided in the risk characterization and uncertainty sections.

6.2.4.1 FPM SQVs

The final set of SQVs representing the balanced models for each endpoint are presented in Table 6-10.

Table 6-10. L3 and L2 SQVs Derived Using the FPM

Contaminant	Unit	L2 Toxicity CH Biomass	L2 Toxicity CH Survival	L2 Toxicity HY Biomass	L2 Toxicity HY Survival	L3 Toxicity CH Biomass	L3 Toxicity CH Survival	L3 Toxicity HY Biomass	L3 Toxicity HY Survival
Metals									
Cadmium	mg/kg dw	0.714	0.507	3.51	3.51	3.51	3.51	3.51	3.51
Chromium	mg/kg dw	NC	NC	NC	NC	NC	NC	45.9	NC
Copper	mg/kg dw	359	NC	493	562	562	NC	562	562
Mercury	mg/kg dw	0.407	0.722	0.722	0.722	0.624	0.722	0.235	0.722
Silver	mg/kg dw	1.72	1.72	0.285	1.72	1.72	1.72	1.72	1.72
PAHs									
Total HPAHs	µg/kg dw	22,000	610,000	1,300,000	1,300,000	610,000	610,000	1,300,000	1,300,000
Total LPAHs	µg/kg dw	650,000	650,000	1,600	2,000	650,000	2,000	650,000	2,000
SVOCs									
Benzyl alcohol	µg/kg dw	36	36	13	36	36	36	36	36
Carbazole	µg/kg dw	1,100	1,100	30,000	30,000	1,100	2,500	8,500	30,000
Dibenzofuran	µg/kg dw	7,200	170	7,200	7,200	340	7,200	170	7,200
Phenols									
4-Methylphenol	µg/kg dw	96	125	96	260	80	260	260	260
Phenol	µg/kg dw	120	120	22	120	120	120	120	120
PCBs									
Total PCBs	µg/kg dw	500	3,500	3,500	3,500	500	3,500	3,500	3,500

Table 6-10. L3 and L2 SQVs Derived Using the FPM

		L2 Toxicity				L3 Toxicity			
Contaminant	**Unit**	**CH Biomass**	**CH Survival**	**HY Biomass**	**HY Survival**	**CH Biomass**	**CH Survival**	**HY Biomass**	**HY Survival**
Pesticides									
beta-HCH	µg/kg dw	10.8	10.8	10.8	10.8	10.8	10.8	10.8	10.8
delta-HCH	µg/kg dw	2.35	2.35	1.26	2.35	2.35	2.35	1.29	2.35
Dieldrin	µg/kg dw	21.5	21.5	21.5	21.5	21.5	21.5	21.5	21.5
Endrin	µg/kg dw	20.8	20.7	NC	NC	20.8	20.7	NC	NC
Endrin ketone	µg/kg dw	8.5	8.5	8.5	8.5	8.5	8.5	8.5	8.5
Sum DDD	µg/kg dw	114	331	2,460	2,460	114	331	2,460	2,460
Sum DDE	µg/kg dw	906	906	906	906	906	906	906	906
Sum DDT	µg/kg dw	8,110	8,110	17	8,110	8,110	8,110	8,110	8,110
Total endosulfan	µg/kg dw	2.42	NC	1.4	NC	2.42	NC	NC	NC
Conventional Contaminants									
Ammonia	mg/kg dw	276	161	117	334	276	334	168	334
Sulfide	mg/kg dw	38.5	38.5	15.7	336	38.5	38.5	336	336

CH – *Chironomus*
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
dw – dry weight
FPM – floating percentile model
HCH – hexachlorocyclohexane
HPAH – high-molecular-weight polycyclic aromatic hydrocarbon
HY – *Hyalella*
L2 – Level 2 (moderate toxicity)
L3 – Level 3 (high toxicity)
LPAH – low-molecular-weight polycyclic aromatic hydrocarbon
NC – no criterion (unable to derive criterion for this level and endpoint)
PCB – polychlorinated biphenyl
SQV – sediment quality value
SVOC – semivolatile organic compound

6.2.4.2. LRM SQVs

The choice of which pMax threshold to use for predicting risk within the Study Area is a risk management decision, and was made in consideration of project goals, data limitations, and uncertainties. EPA evaluated the reliability results of the LRM pMax model and concluded that pMax thresholds of 0.50 and 0.59 were appropriate for predicting L2 and L3 responses, respectively. These two thresholds have a false positive rate of 10% or less, and overall reliability of 75% or better (Table 6-9). The SQVs based on these pMax values are provided in Table 6-11. Normalizations for each SQV are based on the final model selected by EPA for each chemical. Details regarding these calculations are provided in Attachment 6 (Part F).

Table 6-11. LRM-Derived SQVs

Analyte	Unit	L2 SQV	L3 SQV
Metals			
Chromium	mg/kg OC-fines[a]	2,910	3,260
Copper	mg/kg dw	444	531
Lead	mg/kg dw	196	251
Mercury	mg/kg OC-fines[a]	9.27	11.3
Silver	mg/kg fines	0.456	0.583
Butyltins			
Tributyltin ion	µg/kg dw	3,080	4,260
PAHs			
2-Methylnaphthalene	µg/kg fines	200	270
Acenaphthene	µg/kg fines	5,400	11,000
Acenaphthylene	µg/kg fines	1,600	2,100
Anthracene	µg/kg dw	1,200	1,900
Benzo(a)anthracene	µg/kg fines	12,000	18,000
Benzo(b)fluoranthene	µg/kg fines	14,000	21,000
Benzo(b+k)fluoranthene	µg/kg OC-fines[a]	140,000	190,000
Benzo(g,h,i)perylene	µg/kg fines	11,000	17,000
Benzo(k)fluoranthene	µg/kg OC-fines[a]	140,000	190,000
Chrysene	µg/kg fines	14,000	21,000
Dibenzo(a,h)anthracene	µg/kg fines	1,600	2,300
Fluoranthene	µg/kg fines	29,000	44,000
Fluorene	µg/kg fines	2,300	3,300

Table 6-11. LRM-Derived SQVs

Analyte	Unit	L2 SQV	L3 SQV
Indeno(1,2,3-cd)pyrene	µg/kg fines	11,000	16,000
Phenanthrene	µg/kg fines	27,000	52,000
Pyrene	µg/kg fines	48,000	70,000
Total HPAHs	µg/kg fines	150,000	230,000
Total LPAHs	µg/kg fines	18,000	26,000
Total PAHs	µg/kg fines	43000	59,000
Phthalates			
Dibutyl phthalate	µg/kg OC-fines[a]	11,000	16,000
SVOCs			
Carbazole	µg/kg dw	480	700
Dibenzofuran	µg/kg fines	440	610
Phenols			
Phenol	µg/kg OC-fines[a]	1,400	1,900
PCBs			
Total PCBs	µg/kg fines	1,100	1,600
Pesticides			
2,4'-DDD	µg/kg OC-fines[a]	2,100	2,800
4,4'-DDD	µg/kg dw	350	470
4,4'-DDE	µg/kg OC-fines[a]	5,500	7,500
4,4'-DDT	µg/kg OC-fines[a]	27,000	43,000
cis-Chlordane	µg/kg fines	2.6	3.4
delta-HCH	µg/kg fines	0.77	1.1
Sum DDD	µg/kg fines	220	310
Sum DDE	µg/kg OC	21,000	29,000
Total DDTs	µg/kg dw	1,400	2,000
Petroleum Hydrocarbons			
Diesel-range hydrocarbons	mg/kg OC-fines[a]	10,000	12,000
Conventional Contaminants			
Sulfide	mg/kg dw	102	162

[a] Fines-adjusted OC-normalized concentration.

DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
LPAH – low-molecular-weight polycyclic aromatic hydrocarbon

DDT – dichlorodiphenyltrichloroethane
dw – dry weight
HCH -- hexachlorocyclohexane
HPAH – high-molecular-weight polycyclic aromatic hydrocarbon
L2 – Level 2 (moderate toxicity)
L3 – Level 3 (high toxicity)
LRM – logistic regression model
OC – organic carbon
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
SQV – sediment quality value
SVOC – semivolatile organic compound

6.2.5 Uncertainty Associated with Predicted Toxicity

6.2.5.1 Floating Percentile Model Uncertainties

The FPM is a relatively straightforward model that works iteratively to minimize false positive errors for a selected maximum false negative error rate. No assumptions are required of the data; chemical concentration thresholds for all chemicals are systematically evaluated until a set of thresholds is found that allows no more than the specified percentage of false negative errors and a minimized percentage of false positives. When the model is viewed simply as an iterative sorting and resorting of input data until certain criteria are met, model outcomes have no uncertainties associated with them. Uncertainties about the meaning of model results do exist and are affected by the quality and quantity of the data that are used in the model.

In particular, the size of the dataset available for evaluation, the frequency of toxicity within the dataset, the particular set of chemicals included in the model, the density of each chemical's data, and the strength of the relationship between the dependent toxicity variable and the independent chemistry concentrations all affect model outcomes (i.e., the final set of SQVs and the reliability measures).

The size of the dataset, the frequency of toxicity, and the completeness of chemistry data affect the ability of the user to validate the model using subsets of the dataset. In the case of the Portland Harbor dataset, subdivision of the dataset did not preserve adequate frequency of toxicity and sufficient chemistry data to run the model. This makes it difficult to determine how general the model results are, even across different portions of the Study Area.

For a given dataset of *x* sampling locations and *y* chemicals, the particular set of chemical concentrations modeled and correlations among those concentrations will affect the chemical-specific SQVs that are derived. That is, if two non-identical sets of chemicals are provided to the model, the final SQVs of any chemicals that occur in both sets may not be the same even if the same final false positive and false negative rates are achieved by both sets. This is somewhat, although not exactly, analogous to issues involved with step-wise multiple regression where correlations among variables and the order in which variables are entered into the model can affect the final model. In the FPM, there has been some concern that, for a given set of chemicals, the ordering of chemicals in the model affected model results, but if the model is run correctly this is not the case. Chemicals can be entered in any order,[48] however, the number of chemicals in the model and their cross-correlations do affect the final SQVs that achieve a given set of reliability results.

Different results can be created by correlations among chemical concentrations, density of individual chemical concentrations, frequency of toxicity, and relationship between toxicity and chemistry. Although the effects of correlations among chemicals on final SQVs have been resolved by recent software updates in the FPM model, missing data for certain chemicals and low frequency of toxicity, as in the Portland Harbor dataset, can cause the chemicals that set the false positive or false negative rates in the model to suddenly flip from one subset to a different set due to lack of data for the first subset and available data for the second at particular locations with toxicity.

For each dataset, with its particular data density and frequency of toxicity, several attempts are usually required to identify a set of chemicals that are both independent enough to explain the highest number of toxic pathways and correlated enough to stand in for one another when data are not available for all chemicals at all sampling locations. Some understanding of the toxic mechanisms of the different chemicals and the correlation among the chemicals is needed to feel confident that a final set of chemicals is doing both.

Once that set is determined, the SQVs must be used together to predict the toxicity of the contaminant mixture—they are not independent.[49] Each SQV explains toxicity along with all the other SQVs that were derived from the model except for SQVs that were set equal to the maximum concentration in the dataset (because these SQVs do not define the onset of toxicity).

6.2.5.2 Logistic Regression Model Uncertainties

The LRM is a set of individual stochastic models that allow investigators to quantify the correlative relationship between sediment chemical concentration and incidence of toxicity. As stochastic models, each individual, chemical-specific LRM model contains a random element or random “noise” around the predicted relationship between chemistry and toxicity. This random “noise” is one component of the uncertainty about predictions from the model. Precision in the predicted probabilities of toxicity (*p*) derived from these models is also uncertain and although this uncertainty could be represented by confidence intervals around the “*p*,” these have not been computed. It is

[48] The order in which the chemicals are entered in to the model erroneously appeared to affect the outcome of the model, based on agency review of earlier versions. The effect was tested and found to be an issue in the management and control of the dataset, not the model. Chemicals must be added in alphabetical order in the model for the model to work correctly. The order of the chemicals does not affect model outcomes, as long as they are labeled such that they are added in alphabetical order.

[49] The use of SQVs as a set to determine the potential for toxicity at a particular station requires that all contaminants with SQVs be analyzed at each station. If fewer chemicals are available for evaluation, the toxicity prediction becomes less certain.

important to remember that the predicted "*p*" threshold and the SQV concentrations associated with this "*p*" are expected values from stochastic models, with unquantified levels of random noise and uncertainty. The utility of the LRM approach is not so much in the individual modeled logistic relationships as it is the performance of the set of thresholds measured by the reliability metrics.

Uncertainty exists in the degree to which the underlying correlative relationship fit by the model exists on a chemical-by-chemical basis. The screening step in the modeling process assumes that toxicity of samples with a low concentration for a chemical is caused by higher concentrations of chemicals other than the one under evaluation. Consequently, the screening criteria create a somewhat contrived relationship (of selected toxicity and chemistry data) that is then described by a logistic model.

The reasonableness of this contrived situation is validated somewhat by the acceptance criteria and reliability checks employed for each individual model (see Attachment 6 [Part F]. For example, a set of data that has identical distributions for the chemical concentrations in the toxic and non-toxic sampling locations will tend to generate goodness-of-fit statistics and reliability metrics for the individual model that indicate poor performance, and the individual model for those data would likely be excluded from the final model set. However, the potential for a relationship to be found following the screening step is prevalence-affected; the modeling approach is biased towards accepting individual model endpoints that have a higher prevalence of toxicity. In contrast, some of the reliability metrics used in the model selection process are improved when prevalence is low.

Uncertainty also exists about the generality of the relationships described by the site-specific LRM. Without an independent set of data to validate the model, the use of the LRM SQVs, or any set of SQVs, assumes that the same conditions and relationships present in the modeling dataset exist site wide. The ability to evaluate that uncertainty is complicated by incomplete analyte lists at all sampling locations and chemical mixtures that differ dramatically across the Study Area. Different chemical mixtures generate uncertainty about the extent to which the same chemical antagonisms or synergisms are at work throughout the site. When this set of SQVs is applied to sampling locations with incomplete analyte lists, or in areas that have different or unknown physical or chemical conditions compared to the modeling dataset, the reliability of those toxicity predictions is unknown.

It is important to note that while individual LRM SQVs may be derived, they are best used as a set for all the chemicals listed in Table 6-11 showing individual LRM SQVs. Individual chemical SQVs for a given *p* threshold (e.g., 0.59) will not have the same reliability as the same threshold applied to the pMax for the complete set of 41 chemicals. Each individual chemical LRM was retained because it explained some level of correlation between chemical concentrations and incidence of toxicity. However, the reliability of the LRM SQVs is known only for the full set of models (n = 41 chemicals).

Finally, the specifics of the modeling approach are not fully documented. This model was developed by NOAA for EPA use, and not all of the components of the model are fully described or understood. Uncertainties are associated with several of the decision criteria used to select chemicals for evaluation[50] and to accept or reject individual chemical models. Best professional judgment (undocumented) was utilized at several stages to develop the final set of models. The current understanding of the process is described in Attachment 6 (Part F); but at this point in time the model does not appear to be replicable.

6.2.5.3 Mean Quotient Threshold Uncertainties

The MQ approach was developed to allow evaluation of mixtures of chemicals relative to their SQGs (or SQVs). A number of studies have demonstrated increasing toxicity (both incidence and magnitude) associated with increasing MQs (Long et al. 2006; Fairey et al. 2001; Carr et al. 1996; Long et al. 1998); however, Long et al. (2006) acknowledged that the threshold delineating non-toxic from toxic conditions is often site-specific. Long et al.(2006) further reported that the reliability of the MQ is improved when a relationship between contaminants and effects can demonstrated and when those SQGs that are most predictive of toxicity are used in the calculation of the MQ.

EPA's Problem Formulations states that the MQ threshold of 0.7 represents a 50% probability of significant toxicity if exceeded; however the specific calculation is not provided. This threshold appears to have been based on a study in Indiana Harbor discussed in Long et al. (2006) that used the relationship between the pMax from an LRM and site MQs based on PECs. In that study, an MQ between 2.9 and 5.5 was associated with an incidence of toxicity of 45%. In Portland Harbor, the lack of correlation between contaminant concentrations and presence of toxicity made it difficult to predict toxicity with high reliability.

As an alternative derivation of a site-specific MQ threshold, the range of MQs associated with sediment samples that had no FPM SQV exceedances was examined. For these samples, the maximum MQ was 0.8 and can be considered a no-adverse-effects threshold. A threshold predicting some probability of toxicity would be higher than this value, but the derivation of the MQ associated with any probability of toxicity would require additional investigation (modeling) and discussion of the uncertainty.

6.2.5.4 Uncertainties Related to Low Prevalence of Toxicity and its Effects on Reliability Measures

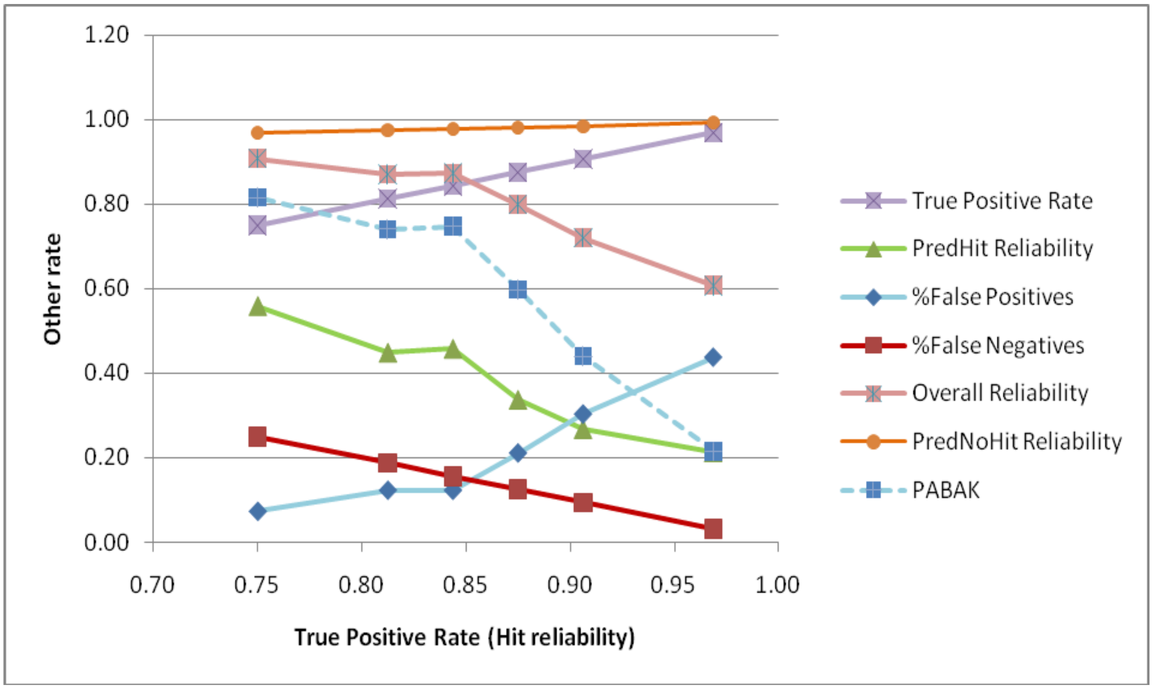
As discussed in previous sections and in depth in EPA's comments on the draft final BERA (2010b), one of the primary obstacles to finding a "best" model to predict toxicity in the Lower Willamette was the low prevalence of toxicity in the Study Area.

[50] As an example, dibutyl phthalate was not identified as a COPC in the SLERA but was included in the LRM.

The low prevalence of toxicity reduced power to detect relationships between toxicity and chemical concentrations (not enough signal to overcome the noise) and to validate any predictive models that were developed. In addition, competing concerns, both within and between involved parties, about the most useful measures of reliability for selecting models, the inter-relatedness of various reliability measures, and conceptual differences between the optimization methods of the FPM and LRM made sorting out the important issues and selecting a “best” model difficult.

All reliability measures used in the BERA are based on relationships among four categories of observed and predicted toxicity: true positives, true negatives, false positives, and false negatives. Each reliability rate has its “mirror” error rate, and all measures reflect a linear combination of these four measures. When the values of any one or two of the measures are constrained to be very low, then other measures will be conversely high.

Observed toxicity rates (prevalence) at the site ranged between 6% and 25% depending on the endpoint and the level of toxicity. Although it was difficult to find concentration thresholds that cleanly differentiated toxic from non-toxic sampling locations, the true positive rate (hit reliability, TP/(TP+FN)) from most models tended to be high (greater than 75%). That is, a large percentage of the truly toxic samples tended to be correctly predicted to be toxic (Figure 6-10). On the other hand, the predicted hit reliability rates (TP/(TP+FP)) tended to be low (less than 50%) – less than half of models’ toxic predictions were correct and predicted hit reliability decreased as the true positive rate increased. For a given model, small changes in the SQVs tended to have a larger effect on the true positive rate (since the fixed number of toxic samples forms the denominator) or the false positive rate by changing the prediction of toxicity for one or two samples. Measures that were relatively insensitive to changes in predictions of toxicity were dominated by predictions of non-toxicity such as predicted no hit reliability and overall reliability.



Note: Prevalence- and bias-adjusted (Cohen's) kappa (PABAK) describes the extent to which a model predicts toxicity at a rate higher than expected by chance (adjusted for prevalence and bias).

Figure 6-10. Example Reliability and Error Rates from Level 3 Chironomid Survival FPM Model

As can be seen in Figure 6-10, most reliability rates are either positively or negatively correlated with one or more other reliabilty or error rate. Although one rate may be more sensitive than another (e.g., PABAK, a measure designed to normalize for prevalence, mimics but is steeper than predicted hit reliability, overall reliabilty, and false negative rate), more than one rate can, and should, be used to assess if a model is performing adequately to meet management needs.

In the end, models with the most balanced probabilities of incorrectly predicting that a toxic sampling location was not toxic (false negative rate) and that a non-toxic sampling location was toxic (false positive rate) were selected (Humphrey 2011).

6.2.6 Risk Characterization Based on Site-Specific SQVs and Mean Quotients

Toxicity to benthic organisms was predicted by comparing surface sediment chemistry to the site-specific SQVs, as well as the MQs derived from the FPM SQVs. Toxicity based on the LRM was predicted by calculating the pMax value for each sample. Samples with pMax $\geq$ 0.59 were considered highly toxic (L3) and samples with 0.50 < pMax < 0.59 were considered moderately toxic (L2).

The FPM SQVs were derived for four different endpoints and two levels of toxicity. MQs were used to integrate the magnitude of SQV exceedances across endpoints for each toxicity level to assist in the interpretation of the spatial distribution of toxicity based on multiple predictions. Individual quotients are calculated by dividing a sample contaminant concentration by its respective SQV, summing the contaminant quotients for an individual sample, and then dividing by the number of quotients. Sample MQs exceeding 0.7 were considered toxic, per EPA's Problem Formulation (Attachment 2). pMax values are already an integration of predictions across chemicals for a given sample, so MQs were not calculated for LRM SQVs.

6.2.6.1 Comparison of Study Area Concentrations to Site-Specific FPM SQVs

SQVs that could reliably predict toxicity were derived for 24 chemicals or chemical sums (five metals, two PAH sums, three SVOCs, two phenolic compounds, total PCBs, and nine pesticides or pesticide sums) and two conventional parameters (ammonia and sulfides) (Table 6-10) of the 30 chemicals evaluated in the FPM. For some of the chemicals, the FPM SQVs for the L2 and L3 effects thresholds were the same value, providing a clear separation between low and high levels of toxicity. For other chemicals, the two FPM SQVs were not equal, creating a range of uncertain predictions of toxic level.

Using site-specific SQVs, toxicity was predicted for up to 1,183 surface sediment sampling locations in the BERA database where only chemistry data were available. A summary of the results of the FPM predictions is provided in Table 6-12.

Table 6-12. Frequency of Exceedance of L2 and L3 FPM SQVs in the Study Area

	L2 – Moderate Toxicity				L3 – High Toxicity			
Contaminant	*Chironomus* Biomass	*Chironomus* Survival	*Hyalella* Biomass	*Hyalella* Survival	*Chironomus* Biomass	*Chironomus* Survival	*Hyalella* Biomass	*Hyalella* Survival
Metals								
Cadmium	74 of 1,126	127 of 1,126	7 of 1,126	7 of 1,126	7 of 1,126	7 of 1,126	7 of 1,126	7 of 1,126
Chromium	NC	NC	NC	NC	NC	NC	63 of 1,122	NC
Copper	15 of 1,122	NC	8 of 1,122	6 of 1,122	6 of 1,122	NC	6 of 1,122	6 of 1,122
Mercury	25 of 1,109	10 of 1,109	10 of 1,109	10 of 1,109	13 of 1,109	10 of 1,109	58 of 1,109	10 of 1,109
Silver	15 of 1,110	15 of 1,110	359 of 1,110	15 of 1,110	15 of 1,110	15 of 1,110	15 of 1,110	15 of 1,110
PAHs								
Total HPAHs	72 of 1,183	8 of 1,183	2 of 1,183	2 of 1,183	8 of 1,183	8 of 1,183	2 of 1,183	2 of 1,183
Total LPAHs	4 of 1,183	4 of 1,183	143 of 1,183	123 of 1,183	4 of 1,183	123 of 1,183	4 of 1,183	123 of 1,183
SVOCs								
Benzyl alcohol	5 of 990	5 of 990	23 of 990	5 of 990	5 of 990	5 of 990	5 of 990	5 of 990
Carbazole	16 of 993	16 of 993	1 of 993	1 of 993	16 of 993	6 of 993	2 of 993	1 of 993
Dibenzofuran	2 of 1,088	46 of 1,088	2 of 1,088	2 of 1,088	27 of 1,088	2 of 1,088	46 of 1,088	2 of 1,088
Phenols								
4-Methylphenol	151 of 1,047	140 of 1,047	151 of 1,047	105 of 1,047	160 of 1,047	105 of 1,047	105 of 1,047	105 of 1,047
Phenol	3 of 1,046	3 of 1,046	38 of 1,046	3 of 1,046	3 of 1,046	3 of 1,046	3 of 1,046	3 of 1,046

Table 6-12. Frequency of Exceedance of L2 and L3 FPM SQVs in the Study Area

	L2 – Moderate Toxicity				L3 – High Toxicity			
Contaminant	***Chironomus* Biomass**	***Chironomus* Survival**	***Hyalella* Biomass**	***Hyalella* Survival**	***Chironomus* Biomass**	***Chironomus* Survival**	***Hyalella* Biomass**	***Hyalella* Survival**
PCBs								
Total PCBs	31 of 908	2 of 908	2 of 908	2 of 908	31 of 908	2 of 908	2 of 908	2 of 908
Pesticides								
beta-HCH	4 of 851	4 of 851	4 of 851	4 of 851	4 of 851	4 of 851	4 of 851	4 of 851
delta-HCH	5 of 848	5 of 848	12 of 848	5 of 848	5 of 848	5 of 848	12 of 848	5 of 848
Dieldrin	2 of 846	2 of 846	2 of 846	2 of 846	2 of 846	2 of 846	2 of 846	2 of 846
Endrin	2 of 700	2 of 700	NC	NC	2 of 700	2 of 700	NC	NC
Endrin ketone	4 of 851	4 of 851	4 of 851	4 of 851	4 of 851	4 of 851	4 of 851	4 of 851
Sum DDD	28 of 900	12 of 900	0 of 900	0 of 900	28 of 900	12 of 900	0 of 900	0 of 900
Sum DDE	2 of 897	2 of 897	2 of 897	2 of 897	2 of 897	2 of 897	2 of 897	2 of 897
Sum DDT	1 of 899	1 of 899	110 of 899	1 of 899	1 of 899	1 of 899	1 of 899	1 of 899
Total endosulfan	28 of 851	28 of 851	NC	NC	40 of 851	NC	NC	NC
Conventional Contaminants[a]								
Ammonia	4 of 200	47 of 200	92 of 200	3 of 200	4 of 200	3 of 200	42 of 200	3 of 200
Sulfide	11 of 198	11 of 198	42 of 198	1 of 198	11 of 198	11 of 198	1 of 198	1 of 198

[a] Conventional parameters are not CERCLA contaminants and were not used to identify areas of benthic risk for the purpose of remediation.

CERCLA – Comprehensive Environmental Response, Compensation, and Liability Act
DDD – dichlorodiphenyldichloroethane
HCH – hexachlorocyclohexane
HPAH – high-molecular-weight polycyclic aromatic hydrocarbon
L2 – Level 2 (moderate toxicity)
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
SQV – sediment quality value

DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
FPM – floating percentile model
L3 – Level 3 (high toxicity)
NC – no criterion
LPAH – low-molecular-weight polycyclic aromatic hydrocarbon
SVOC – semivolatile organic compound

The most frequently exceeded SQV was the 4-methylphenol L3 SQV based on *Chironomus* biomass (160 sampling locations). The L3 SQV for total low-molecular-weight PAHs (LPAHs) was exceeded at 123 sampling locations. Other L3 SQVs that were exceeded at more than 20 sampling locations included chromium (63), mercury (58), dibenzofuran (46), ammonia (42) total PCBs (31), sum dichlorodiphenyldichloroethane (DDD) (28), and total endosulfan (28). Most of these more frequently exceeded SQVs were derived from biomass endpoint models; the total LPAH SQV was the exception in that it was derived from the survival endpoints models. Chromium, 4-methylphenol, and total endosulfan were the three contaminants within the FPM SQV set contributing to false positive classifications (identifying a sample as toxic, when it is not).

The pattern for the contaminants that more frequently exceeded L2 SQVs was similar, with the exceptions that (1) more sampling locations exceeded L2 SQVs (L2 SQVs are typically lower than L3 SQVs), and (2) the SQVs for high-molecular-weight PAHs (HPAHs), sum dichlorodiphenyltrichloroethane (DDT), phenol, benzyl alcohol, cadmium, silver, and sulfides were among the chemicals with more than 20 locations exceeding L2 SQVs.

6.2.6.2 Comparison of Study Area Concentrations to Site-Specific LRM SQVs

Forty-one SQVs were derived using the LRM: five metals, tributyltin (TBT), nineteen PAHs and PAH sums, three SVOCs, phenol, total PCBs, nine pesticides or pesticide sums, diesel-range hydrocarbons, and sulfides (a conventional contaminant) (Table 6-11). All LRM L2 and L3 SQVs had different concentrations representing each level (i.e., L2 and L3 SQVs did not overlap).

The frequency of exceedance of the LRM SQVs within the Study Area is provided in Table 6-13. The greatest number of L3 predictions of toxicity were associated with silver (81 sampling locations), anthracene (42), 2-methylnapthalene (24), total PAHs (23), carbazole (21), diesel-range hydrocarbons (20), and total LPAHs (20). Individual PAHs, PAH sums, diesel-range petroleum hydrocarbons, and carbazole predictions of toxicity tended to be co-located. The incidence of toxicity for all other L3 predictions of toxicity based on the LRM pMax values was less than 20 samples. L2 predictions of toxicity based on pMax values identified similar contaminants as L3 associated with toxicity, in addition to two other metals (lead and mercury), and one other PAH (fluorene) based on more than 20 sampling locations exceeding the L2 thresholds.

Table 6-13. Frequency of Exceedance of L2 and L3 LRM SQVs in the Study Area

Contaminant	LRM Frequency of Exceedance (No. of Samples)	
	L2 – Moderate Toxicity	**L3 – High Toxicity**
Metals		
Chromium	10 of 1,122	9 of 1,122
Copper	10 of 1,122	7 of 1,122
Lead	24 of 1,136	16 of 1,136
Mercury	22 of 1,109	13 of 1,109
Silver	119 of 1,110	81 of 1,110
Butyltins		
Tributyltin ion	5 of 222	4 of 222
PAHs		
2-Methylnaphthalene	25 of 1,113	24 of 1,113
Acenaphthene	16 of 1,183	7 of 1,183
Acenaphthylene	8 of 1,183	6 of 1,183
Anthracene	48 of 1,183	42 of 1,183
Benzo(a)anthracene	9 of 1,183	8 of 1,183
Benzo(b)fluoranthene	9 of 1,067	6 of 1,067
Benzo(b+k)fluoranthene	0 of 1,16	0 of 116
Benzo(g,h,i)perylene	10 of 1,183	8 of 1,183
Benzo(k)fluoranthene	16 of 1,033	11 of 1,033
Chrysene	10 of 1,183	7 of 1,183
Dibenzo(a,h)anthracene	11 of 1,183	6 of 1,183
Fluoranthene	12 of 1,183	7 of 1,183
Fluorene	20 of 1,183	16 of 1,183
Indeno(1,2,3-cd)pyrene	9 of 1,183	8 of 1,183
Phenanthrene	12 of 1,183	8 of 1,183
Pyrene	8 of 1,183	7 of 1,183
Total HPAHs	9 of 1,183	8 of 1,183
Total LPAHs	23 of 1,183	20 of 1,183
Total PAHs	27 of 1,183	23 of 1,183
Phthalates		
Dibutyl phthalate	0 of 1,120	0 of 1,120

Table 6-13. Frequency of Exceedance of L2 and L3 LRM SQVs in the Study Area

Contaminant	LRM Frequency of Exceedance (No. of Samples)	
	L2 – Moderate Toxicity	**L3 – High Toxicity**
SVOCs		
Carbazole	26 of 993	21 of 993
Dibenzofuran	17 of 1,088	14 of 1,088
Phenols		
Phenol	6 of 1,046	5 of 1,046
PCBs		
Total PCBs	1 of 908	1 of 908
Pesticides		
2,4'-DDD	10 of 844	8 of 844
4,4'-DDD	6 of 900	3 of 900
4,4'-DDE	3 of 897	3 of 897
4,4'-DDT	14 of 889	10 of 889
cis-Chlordane	6 of 851	5 of 851
delta-HCH	5 of 848	5 of 848
Sum DDD	10 of 900	5 of 900
Sum DDE	7 of 897	6 of 897
Total DDx	11 of 900	8 of 900
Petroleum Hydrocarbons[a]		
Diesel-range hydrocarbons	25 of 533	20 of 533
Conventional Contaminants[a]		
Sulfide	4 of 198	2 of 198

[a] Total petroleum and conventional parameter SQVs are used to evaluate uncertainty in the predictions of sediment toxicity.

CERCLA – Comprehensive Environmental Response, Compensation, and Liability Act
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
HCH – hexachlorocyclohexane
HPAH – high-molecular-weight polycyclic aromatic hydrocarbon
L2 – Level 2 (moderate toxicity)
L3 – Level 3 (high toxicity)
LPAH – low-molecular-weight polycyclic aromatic hydrocarbon
LRM – logistic regression model
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
SQV – sediment quality value
SVOC – semivolatile organic compound
TBT – tributyltin
total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT, and 4,4′-DDT)

6.2.6.3 Spatial Evaluation of Site-Specific SQV Exceedances

Individual sampling locations that were predicted to be moderately or highly toxic are mapped in Maps 6-7 through 6-10 for the FPM and Map 6-11 for the LRM.

An analysis based on a natural neighbors (NN) spatial interpolation (NN-interpolation)[51] statistical algorithm (de Smith et al. 2008) was used to estimate SQV exceedance areas based on these individual sample points. Where too few detected values were available to allow interpolation, exceedances were displayed as points on the interpolated maps. Because the FPM predictions are made using four different endpoints and two effect levels, the spatial interpolation maps display how the different endpoints overlap. Where more than one endpoint indicates toxicity, predictions are considered relatively certain; single endpoint predictions to toxicity are the least certain. Map 6-12 shows the overlap across FPM endpoint predictions. Sampling locations used in the interpolation are presented on this map to show the effect of data density on interpolations. The interpolations have been bounded using the 30-ft contour or the navigation channel line to reduce the effect of large distances between points on predictions of toxicity.

Benthic toxicity from exposure to metals (specifically cadmium, copper, and lead) was limited in both the spatial extent and magnitude of the L3 exceedances based on both FPM and LRM thresholds. Fewer than 20 locations were predicted to exceed the SQV or pMax threshold for each of these metals. Predicted risks based on chromium and mercury were more widespread, but only based on the *Hyalella* biomass endpoint FPM SQVs. LRM predictions of L3 toxicity from silver were also more widespread. Of the more widespread predictions of metals toxicity, few were represented by contiguous locations and often co-occurred with other contaminants.

FPM SQV exceedances for LPAHs and HPAHs occurred in channel or shoreline sediments between RM 5.0 and RM 6.8, with a few smaller areas along the eastern shoreline at RM 2.8, RM 3.8, RM 4.6, and between RM 9 and RM10. Individual PAH exceedances of LRM pMax thresholds fell within the LPAH and HPAH SQV exceedance areas.

Other SVOCs exceeding their respective SQVs or pMax thresholds included benzyl alcohol, carbazole, dibenzofurans, phenol, and phthalates. Threshold exceedances by these chemicals were limited to only a few samples scattered throughout the Study Area. The samples that exceeded the carbazole and dibenzofuran SQV typically exceeded the total LPAH SQV. Diesel-range petroleum hydrocarbons were co-located

[51] The NN-interpolation algorithm is built into ArcGIS software. It has the advantage over other spatial statistical algorithms of being fully defined, so once the dataset and grid are established, any geographic information system (GIS) analyst applying the algorithm will get the same interpolation result. Other spatial statistical methods require the analyst to use professional judgment to fully define the interpolation algorithm, which introduces subjectivity into the analysis and confounds reproducibility.

with LPAH exceedances in almost all locations (all but three). L3 phenol exceedances were also found within the L3 LPAH exceedance areas.

One chemical, 4-methylphenol, exceeded the L3 SQV at 170 sampling locations. These locations were distributed throughout the river and tended to identify unique locations compared to other SQVs, often exceeding its SQV in otherwise uncontaminated areas. Methylated phenols are readily biodegraded under aerobic conditions, and 4-methylphenol is expected to have a half-life in sediment on the order of days. That 4-methylphenol was found suggests the presence of ongoing sources.

Sediment concentrations of PCBs that were higher than the total PCB SQV occurred at the eastern shoreline between RM 2.0 and RM 2.4, the southwestern portion of International Slip, smaller individual locations between RM 4.0 and RM 4.5 on the eastern shoreline, RM 6.7 along the eastern shoreline, RM 7.7 along the western shoreline, portions of Swan Island Lagoon, the western shoreline between RM 8 and RM 9.2, and a section of the channel and eastern shoreline between RM 11 and RM 11.4.

Few pesticides, with the exception of sum DDD, sum DDT, and total endosulfan, had more than 20 samples exceeding sediment thresholds. Individual sediment samples exceeding the SQVs for beta- and delta-HCH, dieldrin, endrin, and endrin ketone were found in areas of the river between RM 6.8 and RM 7.5 (western shoreline), around RM 11.3 (eastern shoreline), and a few other localized points. These individual sample exceedances typically fell within the areas defined by total DDx exceedances or its constituent groups.

Conventional contaminants, ammonia and sulfides, were only measured at selected locations associated with nearshore areas. Exceedances of their respective SQVs were located throughout the river (Maps 6-13 through 6-16) above RM 4, primarily on the west shoreline. Most exceedances were set by the *Hyalella* biomass SQV.

6.2.6.3 Comparison of Study Area Concentrations to Site-Specific FPM MQs

Map 6-17 portrays the magnitude of the FPM exceedances expressed as the average MQ across all four FPM SQV sets. MQs were exceeded at locations throughout the Study Area; however, exceedances were aggregated at RM 4.6 (east), RM 5.7 (west) between RM 6.2 and 6.8 (west), RM 7.4 (west) and RM 8.6 to RM 9.2. Other exceedances of the MQ threshold tended to be separated by large distances and low concentrations (MQs < 0.7).

6.2.6.4 Comparison of Non-Study Area Concentrations to Site-Specific SQVs

Per EPA's Problem Formulation (Attachment 2), sediment data collected from just outside the boundaries of the Study Area were also evaluated in the BERA. Such data were available from the following areas: Multnomah Channel, the downstream reach (RM 0 to RM 1.9), and the downtown reach (RM 11.8 to RM 15.3). Sediment concentrations of COPCs in these areas were compared to site-specific SQVs, where available. The number of detected concentrations exceeding site-specific FPM SQVs from each of these non-Study Area reaches are presented in Tables 6-14 (L2) and 6-15 (L3). Counts of LRM pMax L2 and L3 exceedances are presented in Table 6-16. Site-specific L3 SQVs or pMax thresholds were each exceeded in one sediment sample collected in the downriver reach for mercury, benzyl alcohol, 4-methyphenol, phenol, total PCBs, and total endosulfan. L3 thresholds were exceeded for mercury, silver, and total LPAHs in the downtown reach. No site-specific thresholds were exceeded in Multnomah Channel.

Table 6-14. FPM SQV L2 Exceedance Summary in Non-Study Areas

	Number of Sampling Locations											
	Downriver				Multnomah Channel				Downtown			
Contaminant	CH Biomass	CH Survival	HY Biomass	HY Survival	CH Biomass	CH Survival	HY Biomass	HY Survival	CH Biomass	CH Survival	HY Biomass	HY Survival
Metals												
Cadmium	4 of 21	8 of 21	0 of 21	0 of 21	0 of 7	0 of 7	0 of 7	0 of 7	1 of 14	1 of 14	0 of 14	0 of 14
Chromium	NC	NC	NC	NC	NC	NC	NC	NC	NC	NC	NC	NC
Copper	0 of 21	NC	0 of 21	0 of 21	0 of 7	NC	0 of 7	0 of 7	0 of 14	NC	0 of 14	0 of 14
Mercury	1 of 21	1 of 21	1 of 21	1 of 21	0 of 7	0 of 7	0 of 7	0 of 7	1 of 14	1 of 14	1 of 14	1 of 14
Silver	0 of 21	0 of 21	0 of 21	0 of 21	0 of 7	0 of 7	0 of 7	0 of 7	1 of 14	1 of 14	3 of 14	1 of 14
PAHs												
Total HPAHs	0 of 21	0 of 21	0 of 21	0 of 21	0 of 7	0 of 7	0 of 7	0 of 7	1 of 14	0 of 14	0 of 14	0 of 14
Total LPAHs	0 of 21	0 of 21	1 of 21	1 of 21	0 of 7	0 of 7	0 of 7	0 of 7	0 of 14	0 of 14	1 of 14	1 of 14
SVOCs												
Benzyl alcohol	1 of 21	1 of 21	3 of 21	1 of 21	0 of 7	0 of 7	0 of 7	0 of 7	0 of 14	0 of 14	0 of 14	0 of 14
Carbazole	0 of 21	0 of 21	0 of 21	0 of 21	0 of 7	0 of 7	0 of 7	0 of 7	0 of 14	0 of 14	0 of 14	0 of 14
Dibenzofuran	0 of 21	0 of 21	0 of 21	0 of 21	0 of 7	0 of 7	0 of 7	0 of 7	0 of 14	0 of 14	0 of 14	0 of 14
Phenols												
4-Methylphenol	0 of 21	0 of 21	0 of 21	0 of 21	0 of 7	0 of 7	0 of 7	0 of 7	0 of 13	0 of 13	0 of 13	0 of 13
Phenol	0 of 21	0 of 21	4 of 21	0 of 21	0 of 7	0 of 7	0 of 7	0 of 7	0 of 13	0 of 13	0 of 13	0 of 13
PCBs												

Table 6-14. FPM SQV L2 Exceedance Summary in Non-Study Areas

	Number of Sampling Locations											
	Downriver				Multnomah Channel				Downtown			
Contaminant	CH Biomass	CH Survival	HY Biomass	HY Survival	CH Biomass	CH Survival	HY Biomass	HY Survival	CH Biomass	CH Survival	HY Biomass	HY Survival
Total PCBs	1 of 21	0 of 21	0 of 21	0 of 21	0 of 7	0 of 7	0 of 7	0 of 7	0 of 18	0 of 18	0 of 18	0 of 18
Pesticides												
beta-HCH	0 of 21	0 of 21	0 of 21	0 of 21	0 of 7	0 of 7	0 of 7	0 of 7	0 of 14	0 of 14	0 of 14	0 of 14
delta-HCH	0 of 21	0 of 21	0 of 21	0 of 21	0 of 7	0 of 7	0 of 7	0 of 7	0 of 14	0 of 14	0 of 14	0 of 14
Dieldrin	0 of 21	0 of 21	NC	NC	0 of 7	0 of 7	NC	NC	0 of 14	0 of 14	NC	NC
Endrin	0 of 21	0 of 21	0 of 21	0 of 21	0 of 7	0 of 7	0 of 7	0 of 7	0 of 14	0 of 14	0 of 14	0 of 14
Endrin ketone	0 of 21	0 of 21	0 of 21	0 of 21	0 of 7	0 of 7	0 of 7	0 of 7	0 of 14	0 of 14	0 of 14	0 of 14
Sum DDD	0 of 21	0 of 21	0 of 21	0 of 21	0 of 7	0 of 7	0 of 7	0 of 7	0 of 14	0 of 14	0 of 14	0 of 14
Sum DDE	0 of 21	0 of 21	1 of 21	0 of 21	0 of 7	0 of 7	0 of 7	0 of 7	0 of 14	0 of 14	1 of 14	0 of 14
Sum DDT	1 of 21	NC	1 of 21	NC	0 of 7	NC	0 of 7	NC	0 of 14	NC	0 of 14	NC
Total endosulfan	1 of 21	NC	1 of 21	NC	0 of 7	NC	0 of 7	NC	0 of 16	NC	0 of 16	NC
Conventionals												
Ammonia	NA	NA	NA	NA	NA	NA	NA	NA	0 of 2	0 of 2	0 of 2	0 of 2
Sulfide	NA	NA	NA	NA	NA	NA	NA	NA	0 of 2	0 of 2	0 of 2	0 of 2

CH – *Chironomus dilutus*
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
FPM – floating percentile model
HCH – hexachlorocyclohexane
HPAH – high-molecular-weight polycyclic aromatic hydrocarbon
HY – *Hyalella azteca*
L2 – Level 2 (moderate toxicity)
LPAH – low-molecular-weight polycyclic aromatic hydrocarbon
NA not analyzed
NC – no criterion
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
SQV – sediment quality value
SVOC – semivolatile organic compound

Table 6-15. FPM SQV L3 Exceedance Summary in Non-Study Areas

Contaminant	Number of Sampling Locations											
	Downriver				Multnomah Channel				Downtown			
	CH Biomass	CH Survival	HY Biomass	HY Survival	CH Biomass	CH Survival	HY Biomass	HY Survival	CH Biomass	CH Survival	HY Biomass	HY Survival
Metals												
Cadmium	0 of 21	0 of 21	0 of 21	0 of 21	0 of 7	0 of 7	0 of 7	0 of 7	0 of 14	0 of 14	0 of 14	0 of 14
Chromium	NC	NC	0 of 21	NC	NC	NC	0 of 7	NC	NC	NC	0 of 14	NC
Copper	0 of 21	NC	0 of 21	0 of 21	0 of 7	NC	0 of 7	0 of 7	0 of 14	NC	0 of 14	0 of 14
Mercury	1 of 21	1 of 21	1 of 21	1 of 21	0 of 7	0 of 7	0 of 7	0 of 7	1 of 14	1 of 14	2 of 14	1 of 14
Silver	0 of 21	0 of 21	0 of 21	0 of 21	0 of 7	0 of 7	0 of 7	0 of 7	1 of 14	1 of 14	1 of 14	1 of 14
PAHs												
Total HPAHs	0 of 21	0 of 21	0 of 21	0 of 21	0 of 7	0 of 7	0 of 7	0 of 7	0 of 14	0 of 14	0 of 14	0 of 14
Total LPAHs	0 of 21	1 of 21	0 of 21	1 of 21	0 of 7	0 of 7	0 of 7	0 of 7	0 of 14	1 of 14	0 of 14	1 of 14
SVOCs												
Benzyl alcohol	1 of 21	1 of 21	1 of 21	1 of 21	0 of 7	0 of 7	0 of 7	0 of 7	0 of 14	0 of 14	0 of 14	0 of 14
Carbazole	0 of 21	0 of 21	0 of 21	0 of 21	0 of 7	0 of 7	0 of 7	0 of 7	0 of 14	0 of 14	0 of 14	0 of 14
Dibenzofuran	0 of 21	0 of 21	0 of 21	0 of 21	0 of 7	0 of 7	0 of 7	0 of 7	0 of 14	0 of 14	0 of 14	0 of 14
Phenols												
4-Methylphenol	1 of 21	0 of 21	0 of 21	0 of 21	0 of 7	0 of 7	0 of 7	0 of 7	0 of 13	0 of 13	0 of 13	0 of 13
Phenol	0 of 21	0 of 21	0 of 21	0 of 21	0 of 7	0 of 7	0 of 7	0 of 7	0 of 13	0 of 13	0 of 13	0 of 13
PCBs												
Total PCBs	1 of 21	0 of 21	0 of 21	0 of 21	0 of 7	0 of 7	0 of 7	0 of 7	0 of 16	0 of 16	0 of 16	0 of 16

Table 6-15. FPM SQV L3 Exceedance Summary in Non-Study Areas

	Number of Sampling Locations											
	Downriver				Multnomah Channel				Downtown			
Contaminant	**CH Biomass**	**CH Survival**	**HY Biomass**	**HY Survival**	**CH Biomass**	**CH Survival**	**HY Biomass**	**HY Survival**	**CH Biomass**	**CH Survival**	**HY Biomass**	**HY Survival**
Pesticides												
beta-HCH	NA	NA	NA	NA	NA	NA	NA	NA	0 of 14	0 of 14	0 of 14	0 of 14
delta-HCH	NA	NA	NA	NA	NA	NA	NA	NA	0 of 14	0 of 14	0 of 14	0 of 14
Dieldrin	0 of 21	0 of 21	0 of 21	0 of 21	0 of 7	0 of 7	0 of 7	0 of 7	0 of 14	0 of 14	0 of 14	0 of 14
Endrin	0 of 21	0 of 21	NC	NC	0 of 7	0 of 7	NC	NC	0 of 14	0 of 14	NC	NC
Endrin ketone	0 of 21	0 of 21	0 of 21	0 of 21	0 of 7	0 of 7	NC	NC	0 of 14	0 of 14	0 of 14	0 of 14
Sum DDD	0 of 21	0 of 21	0 of 21	0 of 21	0 of 7	0 of 7	0 of 7	0 of 7	0 of 14	0 of 14	0 of 14	0 of 14
Sum DDE	0 of 21	0 of 21	0 of 21	0 of 21	0 of 7	0 of 7	0 of 7	0 of 7	0 of 14	0 of 14	0 of 14	0 of 14
Sum DDT	0 of 21	0 of 21	0 of 21	0 of 21	0 of 7	0 of 7	0 of 7	0 of 7	0 of 14	0 of 14	0 of 14	0 of 14
Total endosulfan	1 of 21	NC	NC	NC	0 of 7	NC	NC	NC	0 of 14	NC	NC	NC
Conventionals												
Ammonia	NA	NA	NA	NA	NA	NA	NA	NA	0 of 2	0 of 2	0 of 2	0 of 2
Sulfide	NA	NA	NA	NA	NA	NA	NA	NA	0 of 2	0 of 2	0 of 2	0 of 2

CH – *Chironomus dilutus*
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
FPM – floating percentile model
HCH – hexachlorocyclohexane
HPAH – high-molecular-weight polycyclic aromatic hydrocarbon
HY – *Hyalella azteca*
L3 – Level 3 (high toxicity)
LPAH – low-molecular-weight polycyclic aromatic hydrocarbon
NA – not analyzed
NC – no criterion
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
SQV – sediment quality value
SVOC – semivolatile organic compound

Table 6-16. Frequency of Exceedance of L2 and L3 LRM pMax Thresholds in Non-Study Areas

Contaminant	Frequency of Exceedance					
	Downriver		Multnomah Channel		Downtown	
	LRM L2	LRM L3	LRM L2	LRM L3	LRM L2	LRM L3
Metals						
Chromium	0 of 21	0 of 21	0 of 7	0 of 7	0 of 14	0 of 14
Copper	0 of 21	0 of 21	0 of 7	0 of 7	0 of 14	0 of 14
Lead	0 of 21	0 of 21	0 of 7	0 of 7	0 of 14	0 of 14
Mercury	1 of 21	1 of 21	0 of 7	0 of 7	1 of 14	1 of 14
Silver	0 of 21	0 of 21	0 of 7	0 of 7	1 of 14	1 of 14
Butyltins						
Tributyltin ion	0 of 4	0 of 4	NA	NA	0 of 6	0 of 6
PAHs						
2-Methylnaphthalene	0 of 21	0 of 21	0 of 7	0 of 7	0 of 14	0 of 14
Acenaphthene	0 of 21	0 of 21	0 of 7	0 of 7	0 of 14	0 of 14
Acenaphthylene	0 of 21	0 of 21	0 of 7	0 of 7	0 of 14	0 of 14
Anthracene	0 of 21	0 of 21	0 of 7	0 of 7	0 of 14	0 of 14
Benzo(a)anthracene	0 of 21	0 of 21	0 of 7	0 of 7	0 of 14	0 of 14
Benzo(b)fluoranthene	0 of 21	0 of 21	0 of 7	0 of 7	0 of 14	0 of 14
Benzo(b+k)fluoranthene	NA	NA	NA	NA	NA	NA
Benzo(g,h,i)perylene	0 of 21	0 of 21	0 of 7	0 of 7	0 of 14	0 of 14
Benzo(k)fluoranthene	0 of 21	0 of 21	0 of 7	0 of 7	0 of 14	0 of 14
Chrysene	0 of 21	0 of 21	0 of 7	0 of 7	0 of 14	0 of 14
Dibenzo(a,h)anthracene	0 of 21	0 of 21	0 of 7	0 of 7	0 of 14	0 of 14
Fluoranthene	0 of 21	0 of 21	0 of 7	0 of 7	0 of 14	0 of 14
Fluorene	0 of 21	0 of 21	0 of 7	0 of 7	0 of 14	0 of 14
Indeno(1,2,3-cd)pyrene	0 of 21	0 of 21	0 of 7	0 of 7	0 of 14	0 of 14
Phenanthrene	0 of 21	0 of 21	0 of 7	0 of 7	0 of 14	0 of 14
Pyrene	0 of 21	0 of 21	0 of 7	0 of 7	0 of 14	0 of 14
Total HPAHs	0 of 21	0 of 21	0 of 7	0 of 7	0 of 14	0 of 14
Total LPAHs	0 of 21	0 of 21	0 of 7	0 of 7	0 of 14	0 of 14
Total PAHs	0 of 21	0 of 21	0 of 7	0 of 7	0 of 14	0 of 14

Table 6-16. Frequency of Exceedance of L2 and L3 LRM pMax Thresholds in Non-Study Areas

	Frequency of Exceedance					
	Downriver		Multnomah Channel		Downtown	
Contaminant	LRM L2	LRM L3	LRM L2	LRM L3	LRM L2	LRM L3
Phthalates						
Dibutyl phthalate	0 of 21	0 of 21	0 of 7	0 of 7	0 of 14	0 of 14
SVOCs						
Carbazole	0 of 21	0 of 21	0 of 7	0 of 7	0 of 14	0 of 14
Dibenzofuran	0 of 21	0 of 21	0 of 7	0 of 7	0 of 14	0 of 14
Phenols						
Phenol	1 of 21	1 of 21	0 of 7	0 of 7	0 of 13	0 of 13
PCBs						
Total PCBs	0 of 21	0 of 21	0 of 7	0 of 7	0 of 16	0 of 16
Pesticides						
2,4'-DDD	0 of 21	0 of 21	0 of 7	0 of 7	0 of 14	0 of 14
4,4'-DDD	0 of 21	0 of 21	0 of 7	0 of 7	0 of 14	0 of 14
4,4'-DDE	0 of 21	0 of 21	0 of 7	0 of 7	0 of 14	0 of 14
4,4'-DDT	0 of 21	0 of 21	0 of 7	0 of 7	0 of 14	0 of 14
Sum DDD	0 of 21	0 of 21	0 of 7	0 of 7	0 of 14	0 of 14
Sum DDE	0 of 21	0 of 21	0 of 7	0 of 7	0 of 14	0 of 14
Total DDx	0 of 21	0 of 21	0 of 7	0 of 7	0 of 14	0 of 14
Petroleum Hydrocarbons						
Diesel-range hydrocarbons	0 of 17	0 of 17	0 of 7	0 of 7	0 of 8	0 of 8
Conventionals						
Sulfide	NA	NA	NA	NA	0 of 2	0 of 2

DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
HCH – hexachlorocyclohexane
HPAH – high-molecular-weight polycyclic aromatic hydrocarbon
L2 – Level 2 (moderate toxicity)
L3 – Level 3 (high toxicity)
LPAH – low-molecular-weight polycyclic aromatic hydrocarbon
LRM – logistic regression model
NA – not analyzed
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
SVOC – semivolatile organic compound
TBT – tributyltin
total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT, and 4,4′-DDT)

6.2.7 Potential Future Risks to the Benthic Community

Risk to the benthic community was also assessed both for current conditions in the Study Area and estimated future conditions, per EPA direction. The future condition assessment is based on the maximum bed change scenario presented in the draft RI (Map 3.4-7) and a sample-by-sample evaluation of changes in status of predicted risk in the erosional areas using comparisons to site-specific SQVs. Attachment 18 presents the approach and results of the current and future risk predictions in the erosional areas of the Study Area. For the majority of erosional sediments (approximately 60%), there was no change of status in predicted risk to the benthic community (i.e., the sediment quality was similar at the erosional depth and the surface). This finding is not surprising because the erosional sediments are predicted to be primarily sands. Of the remaining erosional sediments, approximately 24% is predicted to be more contaminated in the future. The last 16% of the erosional area is predicted to be cleaner after the erosional event.

6.3 GENERIC SEDIMENT QUALITY GUIDELINES

Generic, national freshwater SQGs described in the EPA's Problem Formulation (Attachment 2) were evaluated for use in the BERA. These generic SQGs are chemical thresholds identified from either field or laboratory toxicity studies of chemical mixtures or single chemicals conducted under various programs throughout the United States and Canada. Thresholds are typically selected based on a preponderance of evidence or distributional characteristics of the paired effect-concentration data. Per agreement with EPA, generic SQGs were evaluated for use in predicting site-specific toxicity and subsequent risk to the benthic community. This evaluation primarily consisted of comparing the reliability of the generic SQGs in predicting site-specific toxicity with the reliability of the FPM- and LRM-derived site-specific SQVs, summarizing the frequency of exceedances, and presenting the sampling locations exceeding the generic SQGs in maps.

The details of the SQG assessment are presented as follows:

- Section 6.3.1 presents the generic SQGs evaluated in the BERA.
- Section 6.3.2 provides a comparison of the reliability of generic SQGs and site-specific SQVs.
- Section 6.3.3 characterizes risks based on generic SQGs.

Figure 6-11 presents a flowchart of the generic SQGs section organization.

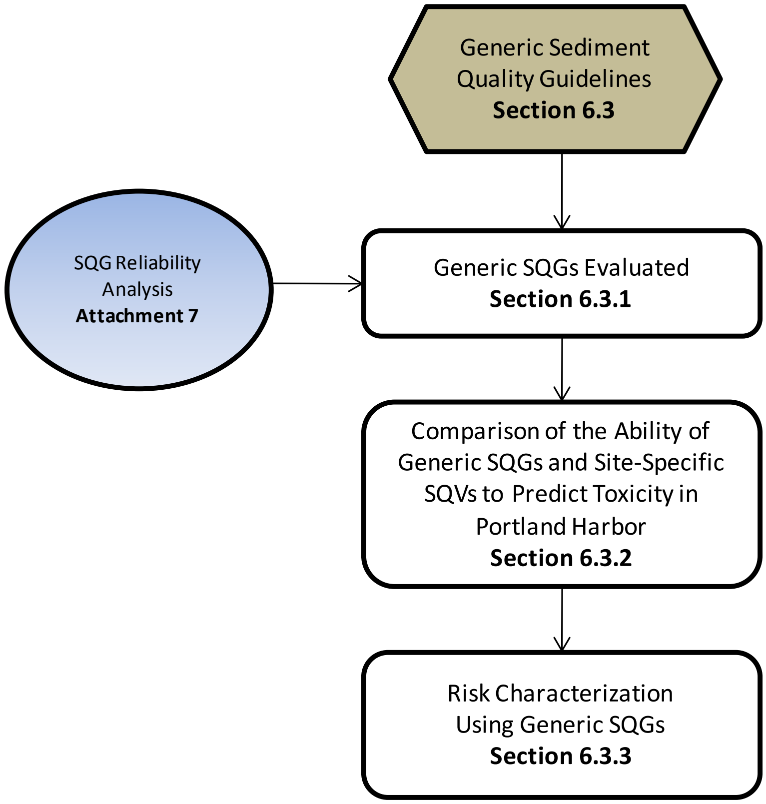


Figure 6-11. Overview of Generic Sediment Quality Guidelines Section Organization

6.3.1 Generic SQGs Evaluated

Of the numerous, published empirical SQGs derived from national datasets, several were included in this assessment at EPA's direction. These SQGs have different purposes or narrative intents. The term "narrative intent" refers to the specific toxicity predictions associated with exposure to sediment contaminant concentrations below, between, or above SQGs. Proper interpretation of these SQGs requires a clearly articulated narrative intent. An understanding of narrative intent is also essential when comparing predictions across LOEs to arrive at overall conclusions about benthic community risk from exposure to chemicals in sediments.

Two sets of SQGs were evaluated for use in assessing risks to benthic communities (Table 6-17).

Table 6-17. Generic SQGs, Their Derivation, and Narrative Intent

SQG	Toxicity Threshold	Narrative Intent	Source
PEC	High	The consensus-based PECs were intended to define the concentration of sediment-associated contaminants above which adverse effects on sediment-dwelling organisms are likely to occur. The PECs were derived as geometric means of SQGs (including ERMs and PELs) in the literature with similar narrative intent. The SQGs are derived from a combination of freshwater and marine toxicity tests.	MacDonald et al. (2000)
PEL	High	The PELs were intended to estimate the concentration of a chemical above which adverse biological effects frequently occurred. The PELs were derived by calculating the geometric mean of the 50th percentile of the effect dataset and the 85th percentile of the no-effect dataset. These SQGs are derived from a national biological effects database that includes freshwater toxicity tests and changes in freshwater benthic community structure.	Smith et al. (1996)

ERM – effects range – median
PEC – probable effects concentration
PEL – probable effects level
SQG – sediment quality guideline

The specific concentration thresholds associated with the generic SQGs are provided in Table 6-18.

Table 6-18. Generic National Freshwater SQGs

Contaminant	PEC	PEL	Unit (dw)
Metals			
Arsenic	33	17	mg/kg
Cadmium	4.98	3.53	mg/kg
Chromium	111	90	mg/kg
Copper	149	197	mg/kg
Lead	128	91.3	mg/kg
Mercury	1.06	0.486	mg/kg
Nickel	48.6	36	mg/kg
Zinc	459	315	mg/kg
PAHs			
2-Methylnaphthalene	No guideline	201	µg/kg
Acenaphthene	No guideline	88.9	µg/kg
Acenaphthylene	No guideline	128	µg/kg
Anthracene	845	245	µg/kg
Benzo(a)anthracene	1,050	385	µg/kg
Benzo(a)pyrene	1,450	782	µg/kg
Chrysene	1,290	862	µg/kg

Table 6-18. Generic National Freshwater SQGs

Contaminant	PEC	PEL	Unit (dw)
Dibenzo(a,h)anthracene	No guideline	135	µg/kg
Fluoranthene	2,230	2,355	µg/kg
Fluorene	536	144	µg/kg
Naphthalene	561	391	µg/kg
Phenanthrene	1,170	515	µg/kg
Pyrene	1,520	875	µg/kg
Total PAHs	22,800		µg/kg
PCBs			
Aroclor 1254	No guideline	340	µg/kg
Total PCBs	676	No guideline	µg/kg
Pesticides			
Chlordane (cis & trans)	No guideline	8.9	µg/kg
Dieldrin	61.8	6.67	µg/kg
Endrin	207	62.4	µg/kg
gamma-HCH	4.99	1.38	µg/kg
Heptachlor epoxide	16	2.74	µg/kg
Sum DDD	28	8.51	µg/kg
Sum DDE	31.3	6.75	µg/kg
Sum DDT	62.9	4.77	µg/kg
Total chlordane	17.6	8.9	µg/kg
Total DDx	572	4,450	µg/kg

DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
HCH – hexachlorocyclohexane
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
PEC – probable effects concentration
PEL – probable effects level
SQG – sediment quality guideline
total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT, and 4,4′-DDT)

SQGs can be used on a chemical-by-chemical basis or across a suite of chemicals. The latter typically involves calculating the “SQG quotient” (sediment concentration divided by SQG) for each chemical in the SQG set for a particular sediment sample and then averaging to get the mean SQG quotient. SQG MQs were also evaluated in the benthic assessment, with the toxicity threshold defined as a quotient of 0.7 per EPA’s Problem Formulation (Attachment 2).

6.3.2 Comparison of the Ability of Generic SQGs and Site-Specific SQVs to Predict Toxicity in Portland Harbor

The ability of the generic SQGs and MQs to predict site-specific toxicity was evaluated by the same series of reliability metrics calculated for the site-specific SQVs. Error rates and other metrics associated with predictions of toxicity were derived for all four endpoints and two effect levels (L2 and L3). Selected reliability measures are provided in Table 6-19, the full suite of reliability measures can be found in Attachment 7.

Overall reliability of the generic SQGs evaluated here ranged from 52.2% to 71.3%. Mean quotient overall reliability ranged from 65.2% to 91.2%, with all but the PEL MQ for *Hyalella* survival (both effect levels), the PEL MQ for L2 *Hyalella* biomass, and PEC MQ for L2 *Hyalella* survival exceeding 80%. The overall reliability of the FPM-derived SQVs ranged from 52.9 to 88.4%, with all but *Hyalella* biomass (at both the L2 and L3 effect levels) and *Chironomus* survival at L2exceeding 80%. The LRM-derived SQVs based on a combined endpoint model for predicting L3 effects had an overall reliability of 85%; L2 LRM SQVs had a lower overall reliability (76%).

False positive rates (rate associated with designating a sample as toxic, when it was not) were greater than 28% for both the PEC and PEL SQGs when predicting toxicity for all endpoints and levels. PEC-MQ and PEL-MQ false positive rates ranged from 4.2 to 20%. FPM-derived SQV false positive rates ranged from 11.4 to 24.9%, except *Hyalella* biomass (54.5% for L2 effects). The LRM SQVs had a false positive rate of 10% for both effect levels.

False negative rates (rate associated with designating a sample as non-toxic, when it was toxic) were higher than 20% for all of the PEC predictions (21.1 to 67.1%) and several of the PEL predictions. PEL predictions of high toxicity (L3) were an exception; in these cases the false negative error rates (5.3 to 20.7%) were comparable to those reported for the FPM-derived SQVs for the L3 effect level (10.5 to 24.1%). PEC-MQ and PEL-MQ false negative rates ranged from 31.6 to 80.8%. The LRM SQVs had a 38% false negative error rate for predicting L3 toxicity levels and 50% for L2 toxicity levels.

The generic SQGs predicted non-toxic samples as reliably as the FPM-derived and LRM-derived SQVs. The PELs had similar reliability as the FPM-derived SQVs for all endpoints at L3 and several endpoints for L2 toxicity predictions. The PECs did not perform as well as the FPM-derived SQVs for any endpoint or effect level.

Table 6-19. Major Reliability Measures for Generic Models

SQV Type	Endpoint	Effect Level	False Negative Rate (%)	False Positive Rate (%)	True Positive Rate (%)[a]	True Negative Rate (%)[b]	Positive Predictive Power (%)[c]	Negative Predictive Power (%)[d]	Overall Reliability (%)
PEC	*Chironomus* survival	L2	40.9	31.3	59.1	68.7	25.0	90.5	67.2
PEC	*Chironomus* biomass	L2	30.8	28.2	69.2	71.8	34.6	91.5	71.3
PEC	*Hyalella* survival	L2	23.8	32.4	76.2	67.6	15.4	97.4	68.3
PEC	*Hyalella* biomass	L2	67.1	36.4	32.9	63.6	23.1	74.1	56.0
PEL	*Chironomus* survival	L2	22.7	49.4	77.3	50.6	21.7	92.6	54.6
PEL	*Chironomus* biomass	L2	15.4	46.9	84.6	53.1	28.0	94.1	58.7
PEL	*Hyalella* survival	L2	9.5	50.7	90.5	49.3	12.1	98.5	52.2
PEL	*Hyalella* biomass	L2	47.9	54.1	52.1	45.9	24.2	74.3	47.4
PEC	*Chironomus* survival	L3	31.3	31.4	68.8	68.6	21.2	94.7	68.6
PEC	*Chironomus* biomass	L3	27.9	29.2	72.1	70.8	29.8	93.7	71.0
PEC	*Hyalella* survival	L3	21.1	32.5	78.9	67.5	14.4	97.9	68.3
PEC	*Hyalella* biomass	L3	41.4	33.0	58.6	67.0	16.3	93.7	66.2
PEL	*Chironomus* survival	L3	15.6	49.8	84.4	50.2	17.2	96.3	53.9
PEL	*Chironomus* biomass	L3	14.0	48.0	86.0	52.0	23.6	95.6	57.0
PEL	*Hyalella* survival	L3	5.3	50.7	94.7	49.3	11.5	99.3	52.2
PEL	*Hyalella* biomass	L3	20.7	50.8	79.3	49.2	14.6	95.6	52.2
PEC-MQ	*Chironomus* survival	L2	63.6	5.6	36.4	94.4	53.3	89.4	85.7
PEC-MQ	*Chironomus* biomass	L2	61.5	4.1	38.5	95.9	66.7	87.8	85.7
PEC-MQ	*Hyalella* survival	L2	33.3	5.9	66.7	94.1	46.7	97.3	92.2

Table 6-19. Major Reliability Measures for Generic Models

SQV Type	Endpoint	Effect Level	False Negative Rate (%)	False Positive Rate (%)	True Positive Rate (%)[a]	True Negative Rate (%)[b]	Positive Predictive Power (%)[c]	Negative Predictive Power (%)[d]	Overall Reliability (%)
PEC-MQ	*Hyalella* biomass	L2	80.8	7.3	19.2	92.7	46.7	77.6	74.4
PEL-MQ	*Chironomus* survival	L2	54.5	15.7	45.5	84.3	33.9	89.7	78.5
PEL-MQ	*Chironomus* biomass	L2	48.1	13.3	51.9	86.7	45.8	89.3	80.5
PEL-MQ	*Hyalella* survival	L2	38.1	16.9	61.9	83.1	22.0	96.6	81.6
PEL-MQ	*Hyalella* biomass	L2	79.5	20.0	20.5	80.0	25.4	75.2	65.2
PEC-MQ	*Chironomus* survival	L3	53.1	5.7	46.9	94.3	50.0	93.5	89.1
PEC-MQ	*Chironomus* biomass	L3	58.1	4.8	41.9	95.2	60.0	90.5	87.4
PEC-MQ	*Hyalella* survival	L3	31.6	6.2	68.4	93.8	43.3	97.7	92.2
PEC-MQ	*Hyalella* biomass	L3	51.7	6.1	48.3	93.9	46.7	94.3	89.4
PEL-MQ	*Chironomus* survival	L3	43.8	15.7	56.3	84.3	30.5	94.0	81.2
PEL-MQ	*Chironomus* biomass	L3	46.5	14.4	53.5	85.6	39.0	91.5	80.9
PEL-MQ	*Hyalella* survival	L3	36.8	17.2	63.2	82.8	20.3	97.0	81.6
PEL-MQ	*Hyalella* biomass	L3	55.2	17.4	44.8	82.6	22.0	93.2	78.8

[a] True positive rate may also be referenced as hit reliability or sensitivity.
[b] True negative rate may also be referenced as no-hit reliability, specificity, or efficiency.
[c] Positive predictive power may also be referenced as predicted hit reliability.
[d] Negative predictive power may also be referenced as predicted no-hit reliability.

L2 – Level 2 (moderate toxicity)
L3 – Level 3 (high toxicity)
PEL – probable effects level
PEC – probable effects concentration
MQ – mean quotient

6.3.3 Risk Characterization Using Generic SQGs

Maps 6-18 through 6-21 present the sediment chemistry sampling locations in the Study Area identified as exceeding the generic SQG thresholds or an MQ threshold of 0.7 (these maps include both bioassay and chemistry-only sampling locations). A summary of SQG exceedances, by chemical, is provided in Table 6-20.

Table 6-20. Exceedance Frequency of PEC and PEL in Study Area Sediment

	Exceedance Frequency (No. of Sampling Locations)	
Contaminant	**PEC**	**PEL**
Metals		
Arsenic	8 of 1,390	17 of 1,390
Cadmium	7 of 1,382	8 of 1,382
Chromium	20 of 1,378	30 of 1,378
Copper	66 of 1,378	49 of 1,378
Lead	49 of 1,392	76 of 1,392
Mercury	3 of 1,365	21 of 1,365
Nickel	30 of 1,376	52 of 1,376
Zinc	38 of 1392	80 of 1,392
PAHs		
2-Methylnaphthalene	No guideline	66 of 1,369
Acenaphthene	No guideline	256 of 1,439
Acenaphthylene	No guideline	111 of 1,439
Anthracene	81 of 1,439	156 of 1,439
Benzo(a)anthracene	131 of 1,439	253 of 1,439
Benzo(a)pyrene	121 of 1,439	198 of 1,439
Chrysene	144 of 1,439	187 of 1,439
Dibenzo(a,h)anthracene	No guideline	167 of 1,439
Fluoranthene	136 of 1,439	135 of 1,439
Fluorene	85 of 1,439	163 of 1,439
Naphthalene	52 of 1,440	66 of 1,440
Phenanthrene	145 of 1,439	256 of 1,439
Pyrene	180 of 1,439	266 of 1,439
Total PAHs	104 of 1,439	No guideline
PCBs		
Aroclor 1254	No guideline	24 of 1,164
Total PCBs	48 of 1,163	96 of 1,163
Pesticides		
Chlordane (cis & trans)	No guideline	1 of 188
Dieldrin	1 of 1,107	10 of 1,107

Table 6-20. Exceedance Frequency of PEC and PEL in Study Area Sediment

Contaminant	Exceedance Frequency (No. of Sampling Locations) PEC	PEL
Endrin	0 of 1,107	0 of 1,107
gamma-HCH	29 of 1,107	92 of 1,107
Heptachlor epoxide	1 of 1,107	11 of 1,107
Sum DDD	119 of 1,155	250 of 1,155
Sum DDE	52 of 1,152	142 of 1,152
Sum DDT	89 of 1,154	301 of 1,154
Total chlordane	43 of 1,106	68 of 1,106
Total DDx	38 of 1,155	7 of 1,155

DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
HCH – hexachlorocyclohexane
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
PEC – probable effects concentration
PEL – probable effects level
total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT, and 4,4′-DDT)

6.4 TISSUE-RESIDUE ASSESSMENT

One LOE for evaluating risks to the benthic invertebrate community and bivalve and crayfish populations is tissue residues. This LOE integrates multiple exposure pathways for invertebrates, including direct contact with sediment and water, ingestion of sediment and water, and ingestion of prey. COPCs were identified in the SLERA and refined screen using screening-level tissue TRVs (Attachment 5). These COPCs were further evaluated by comparing the COPC concentrations in benthic invertebrate tissue with baseline tissue TRVs.

The details of the tissue-residue assessment are presented as follows:

- Section 6.4.1 presents a summary of the assessment methods
- Section 6.4.2 provides a summary of the COPCs identified in the SLERA and refined screen that are evaluated in the tissue-residue LOE. The tissue-residue LOE was used for only a subset of COPCs selected by EPA (Attachment 9).
- A summary of the residue data is presented in Section 6.4.3. Tissue-residue concentrations for each COPC, when detected or predicted, are used as evidence of exposure. All tissue-residue data are presented in Attachment 4.
- Baseline tissue TRVs (Section 6.4.4), developed in cooperation with EPA, represent thresholds that identify the lowest adverse effects levels. Additional details on the development of the baseline tissue TRVs are presented in Attachment 9.

- Section 6.4.5 presents the risk characterization results, uncertainties, and COPCs identified for the tissue-residue LOE. In Section 6.7, these COPCs are compared with the risk characterization for the other LOEs (i.e., sediment toxicity and surface water) to identify contaminants for which the preponderance of evidence indicates unacceptable risk to benthic invertebrates in the Study Area.

Figure 6-12 presents a flowchart of the benthic tissue-residue assessment section organization.

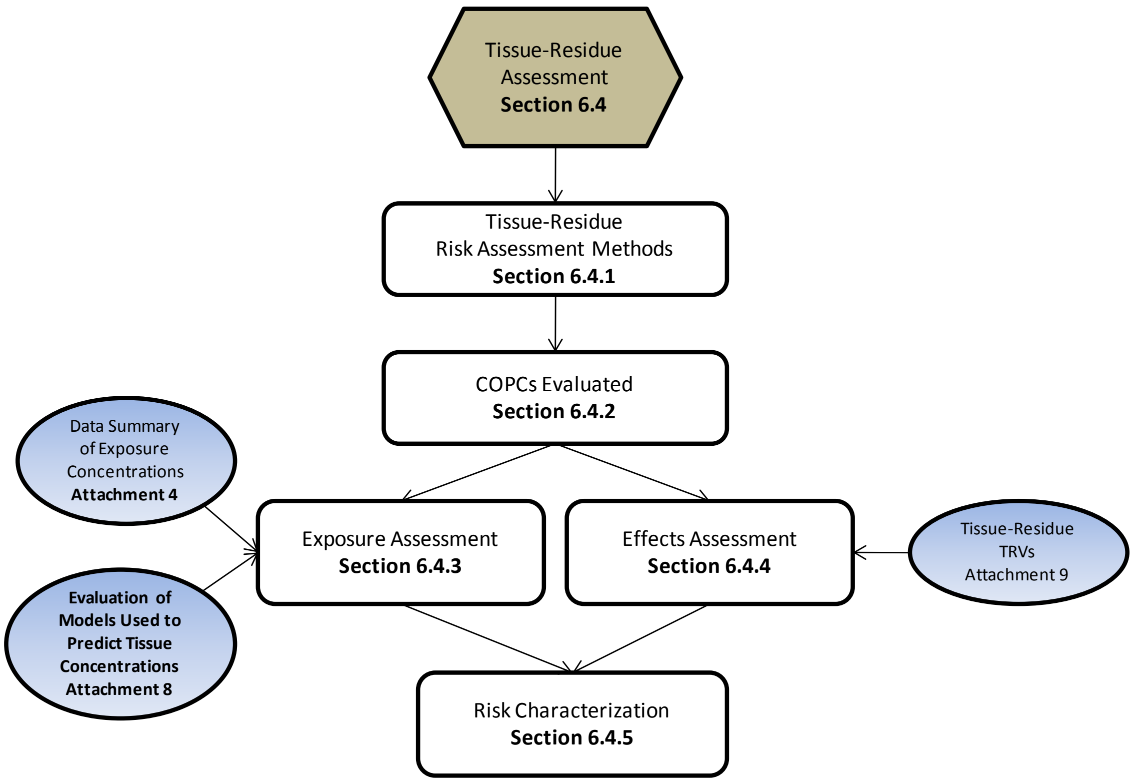


Figure 6-12. Overview of Benthic Tissue-Residue Assessment Section Organization

6.4.1 Tissue-Residue Risk Assessment Methods

The tissue-residue approach for benthic invertebrates is based on an evaluation of COPCs in benthic tissue. Tissue concentrations were measured or predicted in various benthic organisms including clams, mussels, worms, and crayfish. Tissue concentrations in epibenthic organisms collected from artificial samplers were also measured. Tissues with measured concentrations were either collected in the field or exposed to field-collected sediments under laboratory conditions. Field-collected benthic invertebrates included the Asiatic clam (*Corbicula fluminea*), mussels (western pearlshell mussel [*Margaritifera falcata*]) and winged floater [*Anodonta nuttalliana*]), crayfish (the western crayfish [*Pacifastacus leniusculus*]), and epibenthic organisms collected with multiplate samplers. Laboratory exposure of the freshwater oligochaete (*Lumbriculus variegatus*) and the Asiatic clam (*Corbicula fluminea*) was performed to estimate tissue concentrations for other common sediment-exposed benthic invertebrates (Windward and Integral 2005b). Laboratory-exposed tissue concentrations were adjusted to estimate steady-state conditions, since the duration of the tests was shorter than that required to achieve an equivalent of a field-exposure. In areas without co-located tissue-residue data, tissue concentrations were predicted for clams, worms, and crayfish where a relationship between sediment and tissue could be demonstrated.

All the tissue data were used in the risk assessment for the benthic community; only clam and mussel tissue-residue data were used in the risk assessment for bivalve populations, and only crayfish data were used in the risk assessment for crayfish populations (Table 6-1). Potential risks were identified by comparing individual tissue sample residues to toxicologically based TRVs.

HQs were derived for receptor-COPC pairs using the following equation:

$$HQ = \frac{EPC}{TRV}$$ **Equation 6-1**

Where:

HQ = hazard quotient

EPC = exposure point concentration (mg/kg wet weight [ww])

TRV = toxicity reference value (mg/kg ww)

EPA's Problem Formulation (Attachment 2) directed the LWG to identify a chemical as a COPC if tissue-residue concentrations exceeded the TRVs (HQs ≥ 1 in any one benthic tissue sample; accordingly, comparison of tissue concentrations to TRVs was conducted on an individual sample basis. A TRV exceedance in a single or isolated tissue sample should not be interpreted as posing an unacceptable risk to the benthic community, bivalve population, or crayfish population. Rather, the value of point-by-point comparisons to TRVs is that they make it possible to look at the spatial distribution of TRV exceedances. This issue is addressed in the risk conclusions (Section 6.7), which also evaluates the importance of benthic invertebrate COPCs across multiple LOEs.

Assessment Based on Individual Samples

In the tissue-residue assessment, risks to the benthic invertebrate community and to populations of clams and crayfish were determined by evaluating each individual tissue sample. This is a conservative approach for evaluating risks to individual organisms, and it relies on inferences with little scientific basis because population-level processes may compensate for adverse effects on individuals (Pastorok et al. 2001). In other ERAs, risks to invertebrates have been assessed using the 95th UCL concentrations (Windward 2007; MacDonald Environmental 2002) or estimates of the median concentrations (Kaiser-Hill 2006).

Several methods have been used in an attempt to address population-level effects, but no consensus on approach currently exists. In Oregon, acceptable risk to a population is defined as ≤ 10% chance that more than 20% of the total population would be adversely exposed (adverse exposure is defined as greater than the LD50 [dose that is lethal to 50% of an exposed population] or LC50 [concentration that is lethal to 50% of an exposed population] for a hazardous substance based on studies with route and exposure duration that simulated field-exposure conditions of the ecological receptor) (ODEQ 1998). Note that exceedance of such a threshold may or may not result in a change in a true population-level endpoint (e.g., density, growth rate, age structure of the population).

In light of current standard ERA practice and the prevalence of data related to organism-level attributes, a combination of numerical estimates and best professional judgment should be used to interpret data on ecological relevance. EPA (1997) recommends that additional information be supplied in risk assessments to provide context for the numerical risk estimates; this information may include spatial extent, magnitude of organism-level threshold exceedance, and quality and relevance of the organism-level effect threshold as a predictor of a population- or community-level effect.

6.4.2 COPCs Evaluated

Eleven COPCs for benthic invertebrate tissue residues were identified in the SLERA and refined screen (Table 6-21). With the exception of aluminum, these COPCs were evaluated further to assess risks to benthic invertebrates. Per EPA (2008e), aluminum was not evaluated because there was insufficient information in the literature to derive an effect threshold for benthic invertebrates. Aluminum concentrations in the Study Area were at or below regional background levels (Section 7.0 of the draft final RI (Integral et al. 2011)), and thus it was not considered to be a site-related chemical.

Table 6-21. Benthic Invertebrate Tissue COPCs

	Tissue Type					
	Field-Collected				Laboratory-Exposed	
COPC	Clam	Mussels	Crayfish	Epibenthic Invertebrates	Clam	Worm
Metals						
Aluminum	X	X	X	X	X	X
Arsenic						X
Cadmium	X	X				X
Copper	X		X	X	X	X
Zinc	X	X				X
Butyltins						
Tributyltin	X				X	X

Table 6-21. Benthic Invertebrate Tissue COPCs

	Tissue Type					
	Field-Collected				Laboratory-Exposed	
COPC	**Clam**	**Mussels**	**Crayfish**	**Epibenthic Invertebrates**	**Clam**	**Worm**
Phthalates						
BEHP					X	
Dibutyl phthalate	X					X
PCBs						
Total PCBs	X					X
Pesticides						
4,4′-DDD[a]	X				X	X
Total DDx	X				X	X

[a] 4,4′-DDD was identified as a COPC in the SLERA (Attachment 5). Per EPA direction, because a TRV could be derived for this DDT metabolite, 4,4′-DDD was evaluated as an individual metabolite in this assessment.

BEHP – bis(2-ethylhexyl) phthalate
COPC – contaminant of potential concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
EPA – US Environmental Protection Agency
PCB – polychlorinated biphenyl
SLERA – screening-level ecological risk assessment
total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT, and 4,4′-DDT)
TRV – toxicity reference value

Twenty-three tissue COIs were not evaluated in the SLERA and refined screen (Table 6-22) because no toxicological data were available. The risks to benthic invertebrates from these chemicals therefore cannot be assessed by the tissue-residue LOE. Risks to individual dioxin and furans other than 2,3,7,8-TCDD could not be evaluated by the tissue-residue LOE because a tissue TRV is available only for 2,3,7,8-TCDD. For vertebrates, dioxins and furans are evaluated as a toxicity-weighted sum based on the toxicity of each congener relative to 2,3,7,8-TCDD, using toxic equivalency factors (TEFs) based on their common mechanism for toxicity; but TEFs are not available for benthic invertebrates. In addition, risks to butyltins other than TBT could not be evaluated by the tissue-residue LOE because a tissue TRV was only developed for TBT. TBT is more toxic to aquatic organisms, including invertebrates, than any other butyltin. In mussels, for example, TBT was found to be an order of magnitude more toxic than dibutyltin (Widdows and Page 1993). Similar results have been found for other invertebrate species, with monobutylin being less toxic than dibutyltin (California DTSC 2003). Risks due to the three butyltins not evaluated are assumed to be lower than risk associated with the more toxic TBT, which was evaluated.

Table 6-22. Benthic Invertebrate Tissue COIs with No TRVs

COI	
Metals	
Manganese	
Butyltins	
Monobutyltin ion[a]	Tetrabutyltin[a]
Dibutyltin ion[a]	
Dioxins/Furans	
1,2,3,4,6,7,8-Heptachlorodibenzofuran	1,2,3,7,8,9-Hexachlorodibenzofuran
1,2,3,4,6,7,8-Heptachlorodibenzo-*p*-dioxin	1,2,3,7,8,9-Hexachlorodibenzo-*p*-dioxin
1,2,3,4,7,8,9-Heptachlorodibenzofuran	1,2,3,7,8-Pentachlorodibenzofuran
1,2,3,4,7,8-Hexachlorodibenzofuran	1,2,3,7,8-Pentachlorodibenzo-*p*-dioxin
1,2,3,4,7,8-Hexachlorodibenzo-*p*-dioxin	2,3,4,6,7,8-Hexachlorodibenzofuran
1,2,3,6,7,8-Hexachlorodibenzofuran	2,3,4,7,8-Pentachlorodibenzofuran
1,2,3,6,7,8-Hexachlorodibenzo-*p*-dioxin	2,3,7,8-Tetrachlorodibenzofuran
SVOCs	
Benzoic acid	Bis(2-chloroethoxy) methane
Benzyl alcohol	Nitrobenzene
Phenols	
4-Nitrophenol	

[a] Risks due to mono-, di-, and tetrabutyltin are assumed to be lower than that associated with TBT, which was evaluated.

COI – contaminant of interest

SVOC – semivolatile organic compound

TBT – tributyltin

TRV – toxicity reference value

In addition, four tissue COIs were not retained as COPCs because no detected concentrations exceeded TRVs (although at least one DL exceeded a TRV): dibutyl phthalate, diethyl phthalate, dimethyl phthalate, and beta-HCH (see Table 5-2). It is possible that the tissue-residue concentrations of these four COIs exceeded the corresponding TRVs in those tissue samples whose DLs were greater than TRVs.

Uncertainties Associated with the Tissue-Residue Approach for Metals

Five metals were identified as COPCs for benthic invertebrates in the SLERA and refined screen. Ecotoxicologists have a range of opinions about whether it is useful to apply the tissue-residue approach to metals, and this has been a very active area of recent scientific research and discourse (Meyer et al. 2005) The uptake, distribution, and disposition of metals are typically species-specific and governed by highly specific biochemical processes that alter the metal form and involve facilitated or active transport. For example, some organisms take up metals and sequester them into “storage” compartments in chemical forms that have little toxicological potency, whereas other organisms actively excrete excess metals (EPA 2007e). These differences create difficulties when interpreting the toxicological significance of whole-body residues and increase the uncertainty when extrapolating adverse effects across different exposure routes, durations, and species.

6.4.3 Exposure Assessment

This section presents the exposure assessment using tissue-residue data collected in the field as well as tissue residues resulting from laboratory bioaccumulation testing or predicted using BSARs presented in Attachment 8.

6.4.3.1 Empirical Tissue EPCs

Empirical tissue EPCs were derived from both field-collected and laboratory-exposed benthic invertebrates. COPC concentrations in composite tissue samples collected from the Study Area function as EPCs for clams, mussels, crayfish, and epibenthic invertebrates. The field sampling locations are shown on Maps 4-3 through 4-5. Due to the design of the multiplate sampler used to collect epibenthic invertebrates (see Section 4.1.2), the epibenthic invertebrate tissue data represent exposure primarily from overlying water rather than direct sediment exposure. Thus, uncertainty is associated with the use of these data in the assessment of benthic invertebrates.

COPC concentrations were also analyzed in organisms exposed to site sediments under laboratory conditions. Because the exposure duration may have been less than the time required to reach steady-state under laboratory conditions, there was some concern that concentrations for neutral organic compounds could underestimate steady-state tissue residues. COPC concentrations in laboratory-exposed tissue samples were therefore adjusted to estimate theoretical steady-state conditions, according to US Army Corps of Engineers (USACE) procedures (McFarland 1995). The steady-state adjustment method is provided in Attachment 3. Map 4-3 shows the sampling locations for sediments used in the laboratory bioaccumulation tests with the clam *Corbicula fluminea* and the oligochaete worm *Lumbriculus variegatus*.

Summary tables and raw data, including all COPC concentrations for each tissue sample, are presented in Attachment 4. Steady-state adjusted concentrations are also provided in the attachment.

6.4.3.2 Predicted Tissue EPCs

Per EPA's Problem Formulation (Attachment 2), the mechanistic model and BSARs were used to predict EPCs from sediment concentration data at sediment sampling locations where tissue-residue data were not collected. The predictive models used in the BERA were selected to provide methodological consistency between BERA tissue-residue predictions and risk-based PRGs for the FS. The models are presented in the draft bioaccumulation modeling report for the Portland Harbor RI/FS Windward (2009b).

The mechanistic model was available for predicting total PCB, pesticide, and dioxin and furan concentrations.[52] The mechanistic model was not used for other COPCs because it is appropriate only for hydrophobic organic chemicals (Arnot and Gobas 2004). Site-specific

[52] Because dioxins and furans were not classified as tissue COPCs for any benthic invertebrate receptors, the mechanistic model was not used to predict their tissue concentrations.

BSARs were selected for benthic invertebrate tissue COPCs that met appropriate regression analysis assumptions, had a statistically significant positive slope ($p < 0.05$), had an $r^2 > 0.30$, and were not modeled mechanistically. Windward (2009b) presents the details of the BSAR analysis and the mechanistic bioaccumulation model.

The mechanistic model was used to predict total PCB and total DDx concentrations in all benthic invertebrate tissues (i.e., clams, worms, and crayfish). Of the benthic tissue COPCs that were not modeled mechanistically, only TBT in laboratory-exposed clams and laboratory-exposed worms met the criteria noted above (Table 6-23). Modeled data are presented in Attachment 4. BSARs were used to predict TBT concentrations in laboratory-exposed worms. The laboratory-exposed clam TBT BSAR was rejected because it failed to predict correctly the empirical field clam tissue TBT data (see Attachment 8). Therefore, only the laboratory-exposed worm TBT BSAR was used to predict tissue concentrations; those predicted laboratory-exposed worm concentrations are the only information available for estimating TBT bioaccumulation in benthic infauna.

Table 6-23. Summary of Benthic Invertebrate Tissue COPCs and Selected Models Used to Predict Tissue Concentrations

	Field-Collected				Laboratory-Exposed			
	Clam		Crayfish		Clam		Worm	
COPC	Tissue Concentration Predicted?	Selected Model	Tissue Concentration Predicted?	Selected Model	Tissue Concentration Predicted?	Selected Model	Tissue Concentration Predicted?	Selected Model
Metals								
Arsenic	No[a]	NA	No[a]	NA	No[a]	NA	No[a]	NA
Cadmium	No[a]	NA	No[a]	NA	No[a]	NA	No[a]	NA
Copper	No[a]	NA	No[a]	NA	No[a]	NA	No[a]	NA
Zinc	No[a]	NA	No[a]	NA	No[a]	NA	No[a]	NA
Butyltins								
Tributyltin	No[a]	NA	No[a]	NA	No[b]	NA	Yes	BSAR
Phthalates								
BEHP	No[c]	NA	No[c]	NA	No[a]	NA	No[a]	NA
Dibutyl phthalate	No[c]	NA	No[c]	NA	No[c]	NA	No[c]	NA
PCBs								
Total PCBs	Yes	Mechanistic	Yes	Mechanistic	Yes	Mechanistic	Yes	Mechanistic
Pesticides								
Total DDx	Yes	Mechanistic	Yes	Mechanistic	Yes	Mechanistic	Yes	Mechanistic

a Site-specific BSARs were not selected for these COPCs because they did not meet the appropriate BSAR analysis assumptions (Windward 2009b), did not have a statistically significant positive slope ($p < 0.05$), or had an $r^2 < 0.30$.

b The laboratory-exposed clam TBT BSAR was rejected because it fails at predicting the empirical field clam tissue TBT data. A BSAR unable to predict empirical, field-collected data was judged to be an inappropriate model for predicting tissue concentrations elsewhere.

c No appropriate BSAR model could be developed because too few sediment and tissue detected concentration data pairs were available (n = 5).

BEHP – bis(2-ethylhexyl) phthalate
BSAR – biota-sediment accumulation regression
COPC – contaminant of potential concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
NA – not applicable
PCB – polychlorinated biphenyl
TBT – tributyltin
total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT, and 4,4′-DDT)

Distinguishing Field and Laboratory Exposure Regimes

A significant relationship between sediment and tissue TBT concentrations was observed for laboratory-exposed clams but not for field-exposed clams. One possible explanation is that the laboratory exposures took place in a static renewal system, where suspended particulate, dissolved organic, and aqueous phase TBT concentrations are more likely (than in the field) to be correlated with the sampled sediment. In the field, the clams feed from the water column, and dispersal by currents would reduce the correlation between co-located sediment and tissue samples.

For the other benthic tissue COPCs (arsenic, cadmium, copper, zinc, BEHP and dibutyl phthalate), no relationship was apparent between co-located sediment and benthic tissue concentrations (Table 6-23). This lack of relationship suggests that the organisms are bioregulating their tissue residues (e.g., for the essential metals copper and zinc), that the exposure source is not limited to local sediments, or both. In the absence of either an empirical relationship between co-located sediment and tissue concentrations, or a mechanistic basis for relating the two, it is not possible to develop a BSAR. Therefore, benthic tissue EPCs for arsenic, cadmium, copper, zinc, BEHP and dibutyl phthalate could not be predicted from co-located sediment concentrations.

The predicted benthic tissue EPCs for total PCBs and total DDx in worms, clams, and crayfish, and for TBT in laboratory-exposed worms are presented in Attachment 4.

6.4.3 Effects Assessment

This section presents the tissue LOAEL TRVs developed for COPCs in cooperation with EPA; the approach is presented in Attachment 9.[53] Acceptable TRV studies included aquatic invertebrate studies with adverse effects on survival, growth, and reproduction. As outlined in EPA's TRV memorandum (Attachment 9), both marine and freshwater studies were included in the TRV development. Per EPA, an acute-to-chronic ratio (ACR) was applied to convert an acute mortality LOAEL into a LOAEL for effects on reproduction and growth (i.e., a chronic LOAEL) because concentrations required to elicit acute mortality are generally higher than those that reduce growth or reproduction. The ACR was applied to all mortality LOAELs if the exposure duration was $\leq$ 21 days. The chemical-specific ACRs are presented in Table 6-24. If fewer than five studies were available, the lowest tissue concentration associated with adverse effects was selected as the benthic tissue-residue LOAEL. If five or more studies were available, a species sensitivity distribution (SSD) was calculated, and the 10th percentile of the SSD was selected as the LOAEL. An SSD displays available data as a plot of the toxicity data for each species on the x-axis and the cumulative probability on the y-axis. Table 6-24 presents the benthic tissue-residue LOAELs for the 10 COPCs.

[53] The decisions and compromises made during the negotiation process are captured in the e-mails presented in Attachment 1.

Table 6-24. Benthic Tissue-Residue LOAEL TRVs

COPC	TRV (mg/kg ww)	Derivation	ACR	Key Uncertainties
Metals				
Arsenic	2.00	LOAEL based on 10th percentile SSD	3.803	Benthic invertebrates may actively regulate metals tissue concentrations; limited number (five) of toxicological studies were available.
Cadmium	0.35	LOAEL based on 10th percentile SSD	8.3	Benthic invertebrates may actively regulate metals tissue concentrations.
Copper	7.67	LOAEL based on 10th percentile SSD	3.23	Benthic invertebrates may actively regulate metals tissue concentrations.
Zinc	24.07	LOAEL based on 10th percentile SSD	2	Benthic invertebrates may actively regulate metals tissue concentrations.
Butyltins				
Tributyltin	0.15	LOAEL based on 10th percentile SSD	12.69	Low uncertainty in derivation of the TRV. However, the TBT TRV (0.15 mg/kg ww) is one-fourth of the sublethal effect threshold (3 mg/kg dw or 0.6 mg/kg assuming 20% moisture) (Meador et al. 2002a) proposed for protection of juvenile salmonid prey, which is based on reduced growth in multiple species. In addition, the SSD includes gastropod imposex as an endpoint, which only affects a subclass of gastropods. Use of this TRV likely overpredicts toxicity to the benthic community.
Phthalates				
BEHP	3.12	LOAEL derived from Sanders et al. (1973)	not used	Only two acceptable toxicity studies were identified.
Dibutyl phthalate	3,855	LOAEL derived from Hudson et al. (1981)	8.3	Only one acceptable toxicity study was identified.
PCBs				
Total PCBs	1.32	LOAEL based on 10th percentile SSD	8.6	Low uncertainty: 15 acceptable toxicity studies were identified. The test organisms included annelids, amphipods, daphnids, decapods, and insects. The PCBs included Aroclors 1016, 1242, 1254, and 1268 and PCB congeners 153 and 101.
Pesticides				
4,4′-DDD	1.81	LOAEL derived from Lotufo et al. (2000)	8.3	Only two acceptable toxicological studies were identified.
Total DDx	0.97	LOAEL based on 10th percentile SSD	8.3	A limited number (8) of toxicological studies based on crustaceans and annelids was available.

ACR – acute-to-chronic ratio
BEHP – bis(2-ethylhexyl) phthalate
COPC – contaminant of potential concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
LOAEL – lowest-observed-adverse-effect level
PCB – polychlorinated biphenyl
SSD – species sensitivity distribution
TBT – tributyltin
total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT, and 4,4′-DDT)
TRV – toxicity reference value
ww – wet weight

Tissue-Residue TRV Uncertainties

Species sensitivity distributions could be developed for 7 of the 10 COPCs, given the number of studies reporting effects. For the remaining three COPCs (BEHP, dibutyl phthalate, and 4,4′-DDD), only one or two toxicological studies were available in the literature, resulting in greater uncertainties regarding the tissue-residue evaluations.

The application of ACRs to mortality studies is a conservative approach because the ACR is calculated as a ratio, with the numerator as the concentration that is lethal to or causes a non-lethal effect in 50% of an exposed population (i.e., the LC50 or the EC50 [concentration that causes a non-lethal effect in 50% of an exposed population]) and the denominator as the chronic NOAEL or the maximum acceptable toxicant concentration (MATC), the MATC being the geometric mean of the NOAEL and the LOAEL determined from the growth, reproduction, or survival endpoints (Raimondo et al. 2007). Hence, the application of an ACR may adjust the tissue-residue concentration to a level lower than a concentration that caused adverse effects, which, when included in the SSD, may produce a more conservative TRV.

6.4.5 Risk Characterization

This section presents the risk estimates for benthic invertebrates based on the tissue-residue LOE.

6.4.5.1 Risk Characterization Process

The risk characterization based on the tissue-residue LOE for benthic invertebrates was conducted by evaluating individual benthic invertebrate tissue samples. HQs were calculated on a sample-by-sample basis for all tissue samples within the Study Area, in accordance with the EPA's Problem Formulation (Attachment 2).

Contaminants with HQs ≥ 1 for any individual benthic invertebrate tissue sample were identified as COPCs. For all COPCs, the spatial distribution and magnitude of HQs, and the associated exposure and effects assumptions were evaluated to provide a more detailed understanding of impacts on benthic invertebrates. The evaluation of COPCs and associated uncertainties were further examined to arrive at risk conclusions for benthic invertebrates (Section 6.7).

6.4.5.2 Risk Characterization Results and Uncertainty Evaluation

Table 6-25 presents the frequency of COPC HQs ≥ 1 for all benthic invertebrate tissue samples that were field-collected or laboratory-exposed.[54] The COPCs were identified for each individual receptor (i.e., field-collected clams, mussels, crayfish, and epibenthic invertebrates, and laboratory-exposed clams and worms) in the SLERA (Attachment 5). Eight of the 10 COPCs had at least one HQ ≥ 1.

[54] Tissue residues for neutral organic compounds in laboratory-exposed organisms are based on steady-state adjusted concentrations.

Table 6-25. Number of Individual Benthic Invertebrate Empirical Samples with LOAEL HQs ≥ 1

	Number of Samples with HQs ≥ 1 (Maximum HQ)					
	Field-Collected Tissue				Laboratory-Exposed Tissue	
COPC	Clam	Mussel	Crayfish	Epibenthic Invertebrates	Lab Clam	Lab Worm
Metals						
Arsenic	NA	NA	NA	NA	NA	2 of 35 (1.5)
Cadmium	0 of 38	0 of 7	NA	NA	NA	0 of 35
Copper	32 of 38 (1.8)	NA	32 of 32 (2.6)	0 of 2	0 of 35	1 of 35 (2.6)
Zinc	34 of 38 (2.2)	5 of 7 (1.7)	NA	NA	NA	27 of 35 (1.3)
Butyltins						
Tributyltin	1 of 34 (3.5)	NA	NA	NA	1 of 35 (4.5)	1 of 35 (11)
Phthalates						
BEHP	NA	NA	NA	NA	1 of 35 (2.8)[a]	NA
Dibutyl phthalate	0 of 38	NA	NA	NA	NA	0 of 35[a]
PCBs						
Total PCBs	1 of 41 (2.0)	NA	0 of 32	NA	NA	8 of 35 (7.5)[a]
Pesticides						
4,4′-DDD	0 of 41	NA	NA	NA	0 of 35[a]	1 of 35 (1.2)[a]
Total DDx	0 of 41	NA	NA	NA	1 of 35 (2.2)[a]	2 of 35 (3.2)[a]

[a] Values for neutral organic compounds are based on concentrations that have been adjusted to represent steady-state conditions (see Attachment 3). All other values are based on empirical laboratory concentrations.

BEHP – bis(2-ethylhexyl) phthalate
COPC – contaminant of potential concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
NA – not applicable (chemical was not a COPC for the receptor)
PCB – polychlorinated biphenyl
total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT, and 4,4′-DDT)

Table 6-26 presents the frequency of HQs ≥ 1 for all benthic invertebrate tissue concentrations that were predicted for the three COPCs with a significant sediment-tissue relationship.

Table 6-26. Number of Individual Sediment Samples Predicted to Have LOAEL HQs ≥ 1

	Number of Samples Predicted to Have HQs ≥ 1 (Maximum HQ)		
COPC	Clam[a]	Crayfish	Worm
Butyltins			
Tributyltin	Not predicted[b]	Not predicted[b]	27 of 272 (149)
PCBs			
Total PCBs	5 of 1,100 (12)	20 of 1,100 (20)	15 of 1,100 (19)
Pesticides			
Total DDx	12[c] of 1,128 (6.7)	13[d] of 1,128 (9.1)	15[e] of 1,128 (10)

a The mechanistic model derived tissue concentrations for filter feeders (i.e., clams), hence tissue concentrations were not derived for field and lab clams

b Not predicted; no significant relationship between sediment and tissue

c Six of the 12 HQs ≥ 1 are based on N-qualified data.

d Six of the 13 HQs ≥ 1 are based on N-qualified data.

e Six of the 15 HQs ≥ 1 are based on N-qualified data.

COPC – contaminant of potential concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
PCB – polychlorinated biphenyl
total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT, and 4,4′-DDT)

From the empirical and predicted tissue-residue data, the following tissue-residue COPCs for the benthic community were identified (i.e., HQ ≥ 1) based on the following receptors:

- **Worms –** arsenic, copper, zinc, TBT, total PCBs, 4,4′-DDD, and total DDx
- **Bivalves –** copper, zinc, TBT, BEHP, total PCBs, and total DDx
- **Crayfish –** copper, total PCBs, and total DDx

Maps 6-22 through 6-26 show the HQs by location for both empirical data and predicted tissue-residue data for all COPCs except arsenic and BEHP. The arsenic HQs ≥ 1 were located at RM 3.7 (east bank) and RM 7.4 (east bank). The BEHP HQ ≥ 1 was located at the RM 8.8 (west bank). For total PCBs and total DDx, the maps present the NN-interpolated areas with predicted tissue concentrations resulting in HQs ≥ 1.

Three metals (arsenic, copper, and zinc) were identified as benthic community COPCs using the tissue-residue LOE. Two key principles (EPA 2007e) should be considered before concluding that metals pose an unacceptable risk to the benthic community:

- The environmental chemistry of metals strongly influences their fate and effects on ecological receptors.

- The toxicokinetics and toxicodynamics of metals depend on the metal, the form of the metal or metal compound, and the organism's ability to regulate and store the metal.

Ecotoxicological conclusions based on whole-body tissue residues of metals are highly uncertain, especially when the toxicity threshold is based on interspecies extrapolation. Species-specific biochemical processes alter the uptake, distribution, and disposition of metals in the organism. These differences confound the interpretation of ecotoxicological significance of whole-body residues and increase the uncertainty when extrapolating adverse effects across different exposure routes, durations, and species. Hence, the uncertainty associated with the tissue TRVs for these four metals is high. Moreover, as discussed in Section 6.4.3.2, no predictive relationship could be derived for arsenic, copper, and zinc and only a weak relationship between sediment and field-collected clam tissue was found for cadmium. Both bioregulation and environmental chemistry contribute uncertainty, with the consequence that the relationship between sediment and whole-body tissue residue is weak or non-existent in the Study Area. Taking into account the uncertainties about metals bioavailability, bioaccumulation, and bioregulation, the cumulative uncertainty in metals risk estimates based on the tissue-residue LOE is very high.

Risk to benthic invertebrates from arsenic in the Study Area is limited spatially. Arsenic was greater than its tissue TRV at only two locations (east banks at RM 3.7 and RM 7.4), based on laboratory-exposed worm tissues. In contrast, copper and zinc HQs ≥ 1 based on field-collected and laboratory-exposed tissues were found throughout the Study Area. However, both copper and zinc are essential invertebrate nutrients that are metabolically regulated. Table 6-27 lists tissue concentrations for copper and zinc in the benthic invertebrates. For both metals, the maximum concentration is within a factor of 4 of the nutritional threshold,[55] except for copper in laboratory-exposed worms (factor of 9.2). These copper and zinc concentrations are within the range that organisms are able to regulate, whether by active regulation of uptake and elimination or by storage in detoxified forms.

Table 6-27. Copper and Zinc Concentrations in Tissue Compared with Nutritional Values

Tissue Type by Contaminant	Concentration (mg/kg ww)			Ratio of Detected Concentration to Nutritional Value	
	Mean	Range	Nutritional Value	Mean	Maximum
Copper					
Clam	9.2	5.99 – 13.5	5	1.8	2.7
Crayfish	15	10.4 – 20.2	5	3.0	4.0
Laboratory-exposed clam	3.8	2.64 – 5.94	5	0.8	1.2

[55] Nutritional thresholds were provided by EPA (2008d) and are discussed in Attachment 9.

Table 6-27. Copper and Zinc Concentrations in Tissue Compared with Nutritional Values

Tissue Type by Contaminant	Concentration (mg/kg ww) Mean	Range	Nutritional Value	Ratio of Detected Concentration to Nutritional Value Mean	Maximum
Laboratory-exposed worm	2.9	1.83 – 20.2	2.2	1.3	9.2
Multiplate invertebrates	4.5	3.01 – 6	2.2	2.0	2.7
Mussel	1.4	1.01 – 1.82	5	0.3	0.4
Zinc					
Clam	34	19.6 – 54	20	1.7	2.7
Crayfish	17	13.7 – 20.3	20	0.9	1.0
Laboratory-exposed clam	14	10.8 – 16.8	20	0.7	0.8
Laboratory-exposed worm	26	18.2 – 31.5	20	1.3	1.6
Multiplate invertebrates	18.7	12.6 – 24.8	20	0.9	1.2
Mussel	27.0	15.7 – 41.5	20	1.4	2.1

ww – wet weight

TBT was identified as a COPC at the mouth of and in Swan Island Lagoon, and at five other single sample locations (east bank at RM 3.7, east and west banks near RM 5.7, west bank at approximately RM 6.2, and west bank at RM 7.4), based primarily on predicted tissue-residues. At a number of these locations, however, field-collected tissue data indicated no unacceptable risk to benthic invertebrates (i.e., HQ < 1). Thus, the predicted tissue concentrations based on laboratory studies (the strength of the relationship between worm tissue and sediment was considered only moderate [see Attachment 8] and could not be validated with field data) probably overestimate actual exposure levels.

The risk to the benthic invertebrate community from BEHP in tissue is also spatially limited, with only one HQ ≥ 1 in the empirical tissue dataset (RM 8.8, west; HQ = 2.8). Only two toxicological studies were available from which to derive the TRV, making the TRV uncertain.

Total PCB HQs based on predicted tissue-residue concentrations range from 1 to 20 throughout the Study Area. HQs in nine actual tissue samples range from 1.1 to 7.5. Taken together, the predicted and empirical tissue concentrations suggest that total PCBs might pose unacceptable risk to benthic invertebrates in areas around RM 2.3 (east), RM 6.9 (west), RM 8.1 (west), RM 8.8 (west), RM 9.6 (west), and RM 11.4 (east), and in the International Slip, Willamette Cove, and Swan Island Lagoon.

Concentrations of 4,4′-DDD and total DDx in benthic invertebrate tissues do not indicate risk to the benthic invertebrate community, except possibly on the west side of the river between RM 6.9 and RM 7.6. Laboratory-exposed worm tissue concentrations exceeded the 4,4′-DDD TRV at one location (RM 6.9 west, HQ = 1.2). Only two toxicological studies were available from which to derive the TRV, making the TRV uncertain. Total

DDx HQs ≥ 1 based on predicted tissue concentrations occurred along the west bank approximately between RM 7.2 and RM 7.6. Those based on laboratory-exposed clams and worms occurred at RM 6.9 west and RM 7.3 west (HQs = 2.1 and 3.2 for worm, 2.2 for clam).

6.4.5.3 Evaluation of Non-Study Area Concentrations

Per EPA (2008c), benthic invertebrate tissue data collected from just outside the boundaries of the Study Area were also evaluated in the BERA. One clam and two crayfish composite tissue samples[56] were available from the downstream reach (RM 0 to RM 1.9) and two clam and two crayfish composite tissue samples[57] were available from the downtown reach (RM 11.8 to RM 15.3) (Maps 4-3 and 4-4). The tissue data from these non-Study Area reaches are presented in Attachment 4.

A cumulative frequency distribution (CFD) approach was used to evaluate the relationships between non-study and Study Area data. CFDs of COPC concentrations in field-collected clam and crayfish tissue within the Study Area, the downstream reach, and the downtown reach were plotted for the receptor-specific COPCs with tissue TRVs (i.e., clam and copper, zinc, TBT, and total PCBs; crayfish and copper; Figures 6-13 through 6-17). One clam sample collected from the downstream reach had a copper concentration slightly greater than the tissue LOAEL; neither sample from the downtown reach exceeded the copper LOAEL (Figure 6-13). Concentrations of copper in crayfish (Figure 6-14) and of zinc in clams (Figure 6-15) from the downstream and downtown reaches were greater than the LOAEL. This was not the case for TBT (Figure 6-16) and total PCBs (Figure 6-17), whose concentrations in clams were less than the LOAELs.

[56] The clam sample was collected from the east bank at RM 1.6. The crayfish samples were collected from the west and east banks at approximately RM 1.0 and RM 1.5. These samples provide limited spatial coverage of the downstream reach (spanning 0.6 mile).

[57] Samples were collected from the west and east banks at approximately RM 12.0 and RM 12.2, which provides limited spatial coverage of the downtown reach (spanning 0.2 mile).

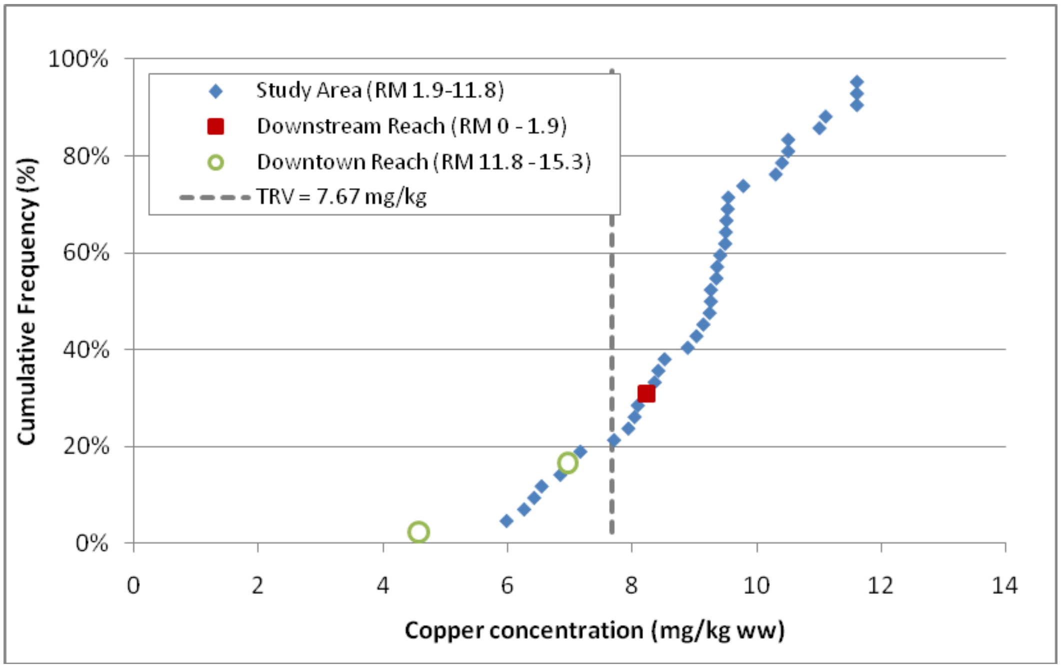


Figure 6-13. CFD of Copper Concentrations in Field-Collected Clam Tissues

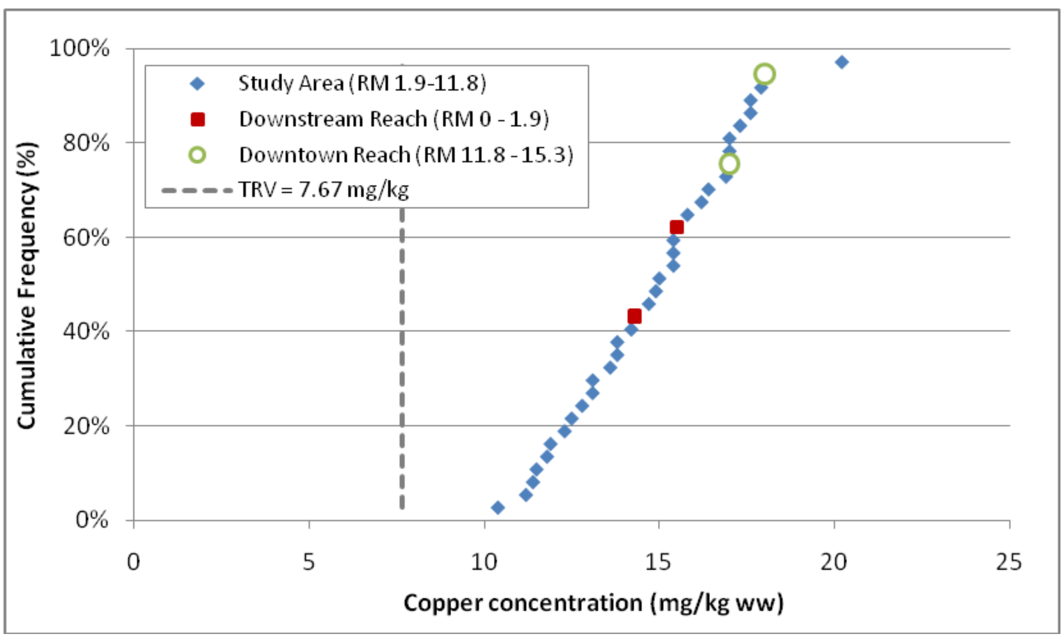


Figure 6-14. CFD of Copper Concentrations in Field-Collected Crayfish Tissues

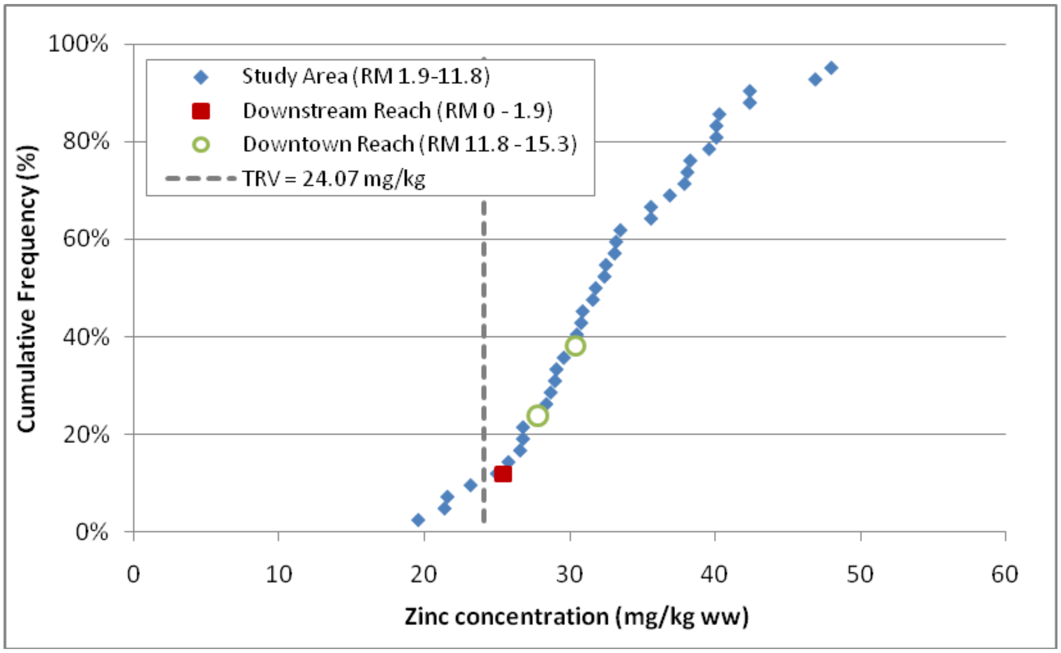
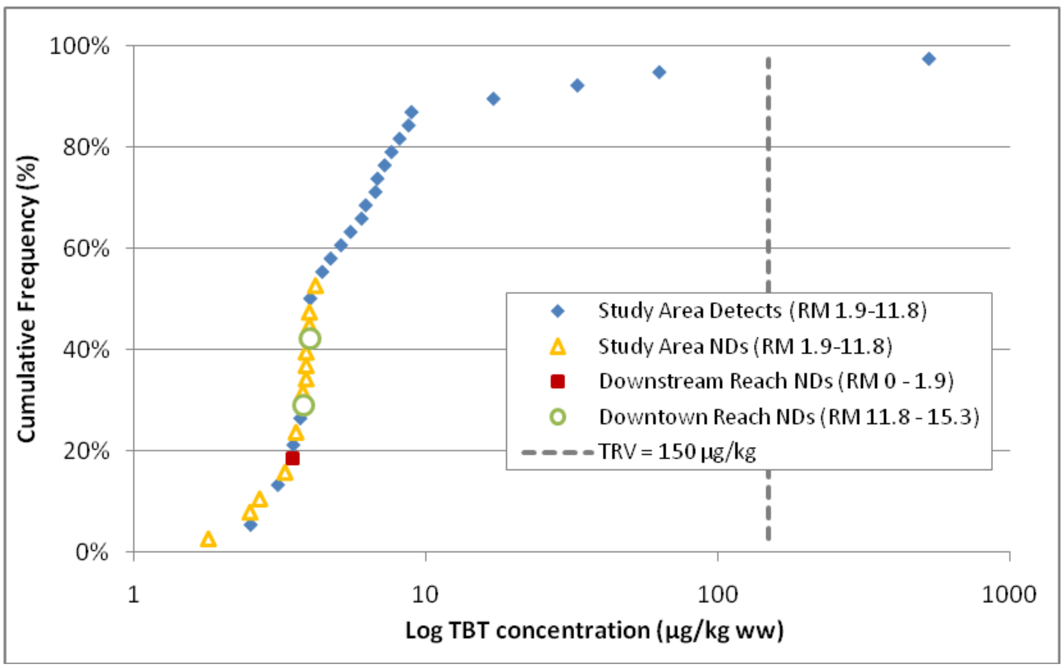
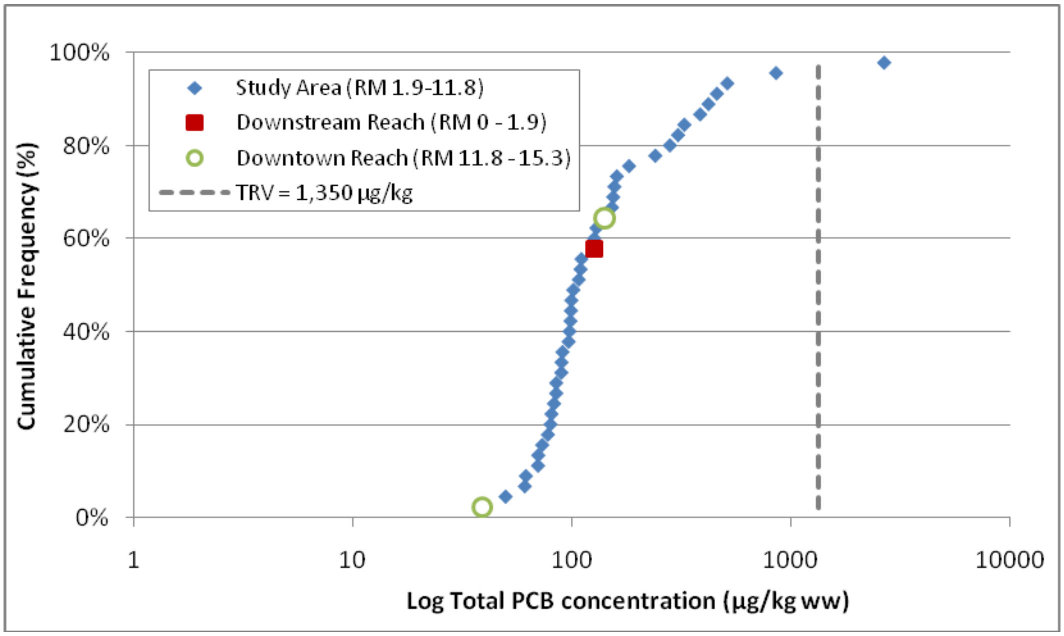


Figure 6-15. CFD of Zinc Concentrations in Field-Collected Clam Tissues



Note: ND = non-detects

Figure 6-16. CFD of TBT Concentrations in Field-Collected Clam Tissues



Note: ND = non-detects

Figure 6-17. CFD of Total PCBs Concentrations in Field-Collected Clam Tissues

6.4.5.4 COIs for Which Risks Cannot Be Quantified

COIs for which risks to benthic invertebrates cannot be quantified based on tissue data are listed in Table 6-28. These COIs represent chemicals for which no TRV is available or for which the maximum DL exceeds a TRV, but detected values do not.

Table 6-28. Benthic Invertebrate Tissue COIs with No Available TRV or with DLs Exceeding SL TRVs

COI	Rationale for Why Risks Cannot Be Quantitatively Evaluated
Metals	
Manganese	Risk to benthic invertebrates based on tissue data unknown; no tissue TRV available
Butyltins	
Monobutyltin ion Dibutyltin ion Tetrabutyltin	Risk to benthic invertebrates based on tissue data unknown; no tissue TRV available; however, TBT is the most toxic butyltin and a TRV is available; risk from other butyltins is expected to be lower than risk from TBT, which was assessed
Dioxin/Furans	
Individual congeners other than 2,3,7,8-TCDD[a]	Risk to benthic invertebrates based on tissue data unknown; no tissue TRV available
SVOCs	
Benzoic acid	Risk to benthic invertebrates based on tissue data unknown; no tissue TRV available
Benzyl alcohol	Risk to benthic invertebrates based on tissue data unknown; no tissue TRV available

Table 6-28. Benthic Invertebrate Tissue COIs with No Available TRV or with DLs Exceeding SL TRVs

COI	Rationale for Why Risks Cannot Be Quantitatively Evaluated
Bis(2-chloroethoxy) methane	Risk to benthic invertebrates based on tissue data unknown; no tissue TRV available
Nitrobenzene	Risk to benthic invertebrates based on tissue data unknown; no tissue TRV available
Phthalates	
Dibutyl phthalate	Risk to crayfish based on tissue data unknown; 84% of non-detected crayfish tissue samples had DLs > SL TRV, but chemical was never detected in crayfish tissue; no risk to dibutyl phthalate was determined for clams and laboratory-exposed worms
Diethyl phthalate	Risk to crayfish based on tissue data unknown; 13% of non-detected crayfish tissue samples had DLs > SL TRV, but chemical was never detected in crayfish tissue
Dimethyl phthalate	Risk to crayfish and clams based on tissue data unknown; 8% of non-detected clam and 84% of non-detected crayfish tissue samples exceed SL TRV, but chemical was never detected in crayfish or clam tissue
Phenols	
4-Nitrophenol	Risk to benthic invertebrates based on tissue data unknown; no tissue TRV available
Pesticides	
Beta-HCH	Risk to clams based on tissue data are unknown; 3% of non-detected clam tissue samples exceed SL TRV, but no detected concentration > SL TRV

[a] Includes the following dioxin and furan congeners: 1,2,3,4,6,7,8-heptachlorodibenzofuran; 1,2,3,7,8,9-hexachlorodibenzofuran; 1,2,3,4,6,7,8-heptachlorodibenzo-p-dioxin; 1,2,3,7,8,9-hexachlorodibenzo-p-dioxin; 1,2,3,4,7,8,9-heptachlorodibenzofuran; 1,2,3,7,8-pentachlorodibenzofuran; 1,2,3,4,7,8-hexachlorodibenzofuran; 1,2,3,7,8-pentachlorodibenzo-p-dioxin; 1,2,3,4,7,8-hexachlorodibenzo-p-dioxin; 2,3,4,6,7,8-hexachlorodibenzofuran; 1,2,3,6,7,8-hexachlorodibenzofuran; 2,3,4,7,8-pentachlorodibenzofuran; 1,2,3,6,7,8-hexachlorodibenzo-p-dioxin; and 2,3,7,8-tetrachlorodibenzofuran

COI – contaminant of interest
DL – detection limit
HCH – hexachlorocyclohexane
SL – screening level
SVOC – semivolatile organic compound
TRV – toxicity reference value
TCDD – tetrachlorodibenzo-*p*-dioxin

6.4.5.5 Summary of Benthic Invertebrate Tissue COPCs

Seven chemicals (arsenic, copper, zinc, TBT, total PCBs, 4,4′-DDD, and total DDx) were identified as tissue-residue COPCs for the lab worms, six chemicals (copper, zinc, TBT, BEHP, total PCBs, and total DDx) were identified as tissue-residue COPCs for bivalves, and three chemicals (copper, total PCBs, and total DDx) were identified as tissue-residue COPCs for crayfish. Table 6-29 presents the HQs, uncertainties, and risk conclusions for each receptor-COPC pair.

In Section 6.7, results of the benthic invertebrate tissue LOE are integrated with those from the other LOEs to determine risk conclusions for benthic invertebrates.

Table 6-29. Summary of Benthic Invertebrate Tissue-Residue COPCs

COPC by Receptor	Exposure Area	HQ[a]	Frequency of HQs ≥ 1 (%)	Exposure Uncertainty	Effects Uncertainty
Lab Worms[b]					
Arsenic	RM 3.7, east; RM 7.4, east	1.1 – 1.5	5.7	Worm tissue data are from the laboratory bioaccumulation test. No relationship was found between tissue and sediment.	Invertebrate tissue-residue LOEs for metals are very uncertain. Limited number of toxicological studies (5) was available for derivation of a TRV. Limited spatial extent of TRV exceedances.
Copper	Mouth of Swan Island Lagoon	2.6	2.9	Worm tissue data are from the laboratory bioaccumulation test. No relationship was found between tissue and sediment.	Invertebrate tissue-residue LOEs for metals are very uncertain, particularly for essential metals. Limited spatial extent of TRV exceedances (1 sample).
Zinc	Site-wide	1.0 – 1.3	77	Worm tissue data are from the laboratory bioaccumulation test. No relationship was found between tissue and sediment.	Invertebrate tissue-residue LOEs for metals are very uncertain, particularly for essential metals.
TBT	Mouth of Swan Island Lagoon	11	2.9	Worm tissue data are from the laboratory bioaccumulation test, which may not replicate field conditions and may alter the bioavailability of TBT. Concentrations in field samples nearby do not reflect the levels predicted by uptake equations based on lab exposures (see Map 6-24)	Limited spatial extent of TRV exceedances (1 sample).

Table 6-29. Summary of Benthic Invertebrate Tissue-Residue COPCs

COPC by Receptor	Exposure Area	HQ[a]	Frequency of HQs ≥ 1 (%)	Exposure Uncertainty	Effects Uncertainty
TBT (predicted)	RM 3.7, east; RM 5.7, east; RM 5.7, west; RM 6.2, east; RM 7.4, east; mouth of Swan Island Lagoon (RM 7.9 to RM 8.7)	1.0 – 149	9.9	Predicted tissue concentrations are based on laboratory-exposed worms, which may not replicate field conditions and may alter the bioavailability of TBT.	Low uncertainty in derivation of the TRV. However, the TBT TRV (0.15 mg/kg ww) is one-fourth of the sublethal effect threshold (3 mg/kg dw) proposed for protection of juvenile salmonid prey (Meador et al. 2002a), which was based on reduced growth in multiple species. Use of this TRV likely overpredicts toxicity.
Total PCBs	RM 2.3, east; RM 3.7, east; Willamette Cove, RM 6.9, west; Swan Island Lagoon; RM 8.8, west; RM 9.7, west	1.1 – 7.5	26	Worm tissue data are from the laboratory bioaccumulation test.	Low uncertainty: 15 acceptable toxicity studies were identified for derivation of TRV. The test organisms included annelids, amphipods, daphnids, decapods, and insects. The PCBs included Aroclors 1016, 1242, 1254, and 1268, and PCB congeners 153 and 101.
Total PCBs (predicted)	RM 2.2, east; RM 3.7, east; Swan Island Lagoon (RM 8.4 to 8.5); RM 11.3, east	1.1-19	1.4	Predicted tissue concentrations were derived using the mechanistic model	Low uncertainty: 15 acceptable toxicity studies were identified for derivation of TRV. The test organisms included annelids, amphipods, daphnids, decapods, and insects. The PCBs included Aroclors 1016, 1242, 1254, and 1268, and PCB congeners 153 and 101.
4,4′-DDD	RM 6.9, west	1.2	2.9	Worm tissue data are from the laboratory bioaccumulation test.	Limited spatial extent of TRV exceedances (1 sample). The LOAEL is based on 2 toxicological studies.
Total DDx	RM 6.9 to RM 7.2, west	2.1 – 3.2	5.7	Worm tissue data are from the laboratory bioaccumulation test.	Limited spatial extent of TRV exceedances (2 samples). Limited number (8) of toxicological studies was available, based on crustaceans and annelids.
Total DDx (predicted)	RM 7.2 to RM 7.6, west	1.1 – 10	1.3	Predicted tissue concentrations were derived using the mechanistic model.	Limited number (8) of toxicological studies was available, based on crustaceans and annelids.

Table 6-29. Summary of Benthic Invertebrate Tissue-Residue COPCs

COPC by Receptor	Exposure Area	HQ[a]	Frequency of HQs ≥ 1 (%)	Exposure Uncertainty	Effects Uncertainty
Bivalves[c]					
Copper (field-collected)	Site-wide	1.0 – 1.8	84	Clam tissue data are based on composites created from multiple transects over broad area in order to collect sufficient tissue mass. No relationship was found between tissue and sediment.	Invertebrate tissue-residue LOEs for metals are very uncertain, particularly for essential metals. The LOAEL was derived from an SSD based on numerous studies using a range of benthic invertebrates as test organisms; however, bivalves are not among the most sensitive species in the SSD. The maximum HQ would be 1.2 if the predicted tissue-residue concentrations were compared to a LOAEL of 11.09 mg/kg ww derived from 8 available bivalve studies.
Zinc (field-collected)	Site-wide	1.1 – 2.2	89	Clam tissue data are based on composites created from multiple transects over broad area in order to collect sufficient tissue mass. No relationship was found between tissue and sediment.	Invertebrate tissue-residue LOEs for metals are very uncertain, particularly for essential metals. The LOAEL was derived from an SSD based on numerous studies using a range of benthic invertebrates as test organisms; however, bivalves are not among the most sensitive species in the SSD. The maximum HQ would be 1.2 if the predicted tissue-residue concentrations were compared to a LOAEL of 42.89 mg/kg ww derived from 8 available bivalve studies.
Zinc (field-collected mussel)	Site-wide	1.1 – 1.7	71	Mussel tissue data were collected from a limited (7) number of locations. No relationship was found between tissue and sediment.	Invertebrate tissue-residue LOEs for metals are very uncertain, particularly for essential metals. All tissue residues were below the nutritional requirement for mollusks. The LOAEL was derived from an SSD based on numerous studies using a range of benthic invertebrates as test organisms; however, bivalves are not among the most sensitive species in the SSD. All HQs would be < 1 if the predicted tissue-residue concentrations were compared to a LOAEL of 42.89 mg/kg ww derived from 8 available bivalve studies.

Table 6-29. Summary of Benthic Invertebrate Tissue-Residue COPCs

COPC by Receptor	Exposure Area	HQ[a]	Frequency of HQs ≥ 1 (%)	Exposure Uncertainty	Effects Uncertainty
TBT (field-collected)	Mouth of Swan Island Lagoon	3.5	2.9	Clam tissue data are based on composites created from multiple transects over broad area in order to collect sufficient tissue mass.	Limited spatial extent of TRV exceedances (1 sampling area in vicinity of shipyard).
TBT (lab-exposed)	Mouth of Swan Island Lagoon	4.5	2.9	Clam tissue data are from the laboratory bioaccumulation test, which may not replicate field conditions and may alter the bioavailability of TBT.	Limited spatial extent of TRV exceedances (1 sample).
BEHP (lab-exposed)	RM 8.8, east	2.8	2.9	Clam tissue data are from the laboratory bioaccumulation test.	The LOAEL is based on 2 toxicological studies.
Total PCBs (field-collected)	Willamette Cove	2.0	2.4	Clam tissue data are based on composites created from multiple transects over broad area in order to collect sufficient tissue mass.	Limited spatial extent of TRV exceedances (1 sampling area).
Total PCBs (predicted)	RM 2.2, east; RM 3.7, east; Swan Island Lagoon (RM 8.4 to 8.5); RM 11.3, east	1.2 – 12	0.5	Predicted tissue concentrations were derived using the mechanistic model	Low uncertainty: fifteen acceptable toxicity studies were identified for derivation of TRV. The test organisms included annelids, amphipods, daphnids, decapods, and insects. The PCBs included Aroclors 1016, 1242, 1254, and 1268, and PCB congeners 153 and 101.
Total DDx (lab-exposed)	RM 7.2, west	2.2	2.9	Clam tissue data are from the laboratory bioaccumulation test.	Limited spatial extent of TRV exceedances (1 sample). Limited number (8) of toxicological studies was available, based on crustaceans and annelids.
Total DDx (predicted)	RM 7.2 to RM 7.6, west	1.1 – 6.7	1.1	Predicted tissue concentrations were derived using the mechanistic model.	Limited number (8) of toxicological studies was available, based on crustaceans and annelids.

Table 6-29. Summary of Benthic Invertebrate Tissue-Residue COPCs

COPC by Receptor	Exposure Area	HQ[a]	Frequency of HQs ≥ 1 (%)	Exposure Uncertainty	Effects Uncertainty
Crayfish					
Copper (field-collected)	Site-wide	1.4 – 2.6	100	Crayfish tissue data were collected in the field. No relationship was found between tissue and sediment.	Invertebrate tissue-residue LOEs for metals are very uncertain, particularly for essential metals. The LOAEL is based on numerous studies using a range of benthic invertebrates as test organisms; no crayfish-specific data are available.
Total PCBs (predicted)	RM 2.2, east; RM 3.7, east; Swan Island Lagoon (RM 8.4 to 8.5); RM 11.3, east	1.1 – 20	1.8	Predicted tissue concentrations were derived using the mechanistic model.	Low uncertainty: 15 acceptable toxicity studies were identified for derivation of TRV. The test organisms included annelids, amphipods, daphnids, decapods, and insects. The PCBs included Aroclors 1016, 1242, 1254, and 1268, and PCB congeners 153 and 101.
Total DDx (predicted)	RM 7.2 to RM 7.6, west	1.2 – 9.1	1.2	Predicted tissue concentrations were derived using the mechanistic model.	Limited number (8) of toxicological studies was available, based on crustaceans and annelids.

[a] Only HQs ≥ 1 are presented. HQs in all other tissue samples were < 1.
[b] All values are empirical unless otherwise noted.
[c] All bivalve data are for clam unless otherwise noted.

BEHP – bis(2-ethylhexyl) phthalate
COPC – contaminant of potential concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
LOE – line of evidence
PCB – polychlorinated biphenyl
RM – river mile
SSD – species sensitivity distribution
TBT – tributyltin
total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT, and 4,4′-DDT)
TRV – toxicity reference value

6.5 SURFACE WATER ASSESSMENT

The surface water assessment is one of several LOEs by which to evaluate risks to the benthic invertebrate community, bivalve population, and crayfish population. Invertebrate surface water COPCs were identified in the SLERA and refined screen using water TRVs based on AWQCs or other TRVs available in the literature (Attachment 5).

The details of this assessment are presented as follows:

- Section 6.5.1 presents a description of the asessment methods
- Section 6.5.2 presents a summary of the COPCs evaluated in the surface water LOE.
- Section 6.5.3 presents an overview of how exposure concentrations were derived. Exposure concentrations in this assessment are represented by detected surface water concentrations from all individual surface water samples. All surface water chemical concentrations are presented in Attachment 4.
- Section 6.5.4 presents a summary of the effects data. Effects data (i.e., water TRVs) in this assessment are the same as those developed for the SLERA and refined screen. Details on the development of the water TRVs are presented in Attachment 10.
- Section 6.5.5 presents the risk characterization results, receptor-COPC pairs, and associated uncertainties. These COPCs are further assessed in the benthic invertebrate risk conclusions (Section 6.7).

Figure 6-18 presents a flowchart showing organization of the benthic invertebrate surface water assessment.

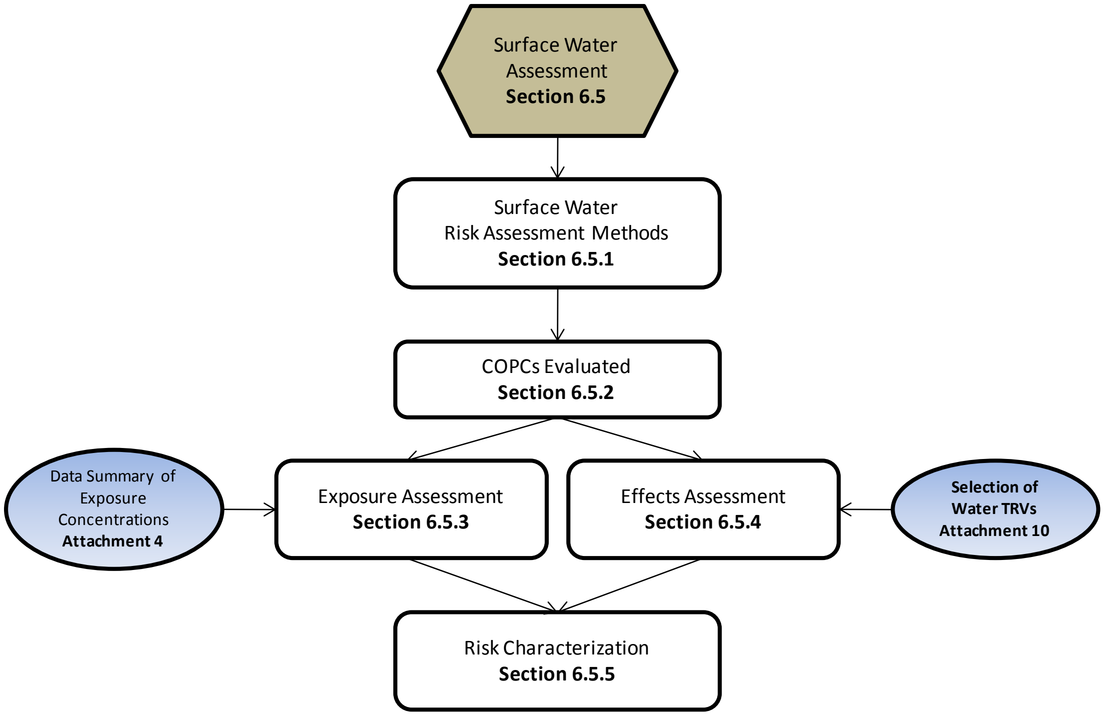


Figure 6-18. Overview of Benthic Invertebrate Surface Water Section Organization

6.5.1 Surface Water Risk Assessment Methods

For this assessment, the same water TRVs as used in the SLERA and refined screen were used to evaluate baseline risks to all targeted benthic invertebrate populations.

The comparison of surface water concentrations to water TRVs was conducted on an individual sample basis per EPA's Problem Formulation (Attachment 2). A sample-by-sample analysis is appropriate for surface water because individual samples of (flowing) surface water capture conditions integrated over a wide area and represent an appropriate spatial scale for a population- and community-level analysis.

HQs were derived for all COPCs using Equation 6-1. Contaminants with HQs $\geq$ 1 for any individual surface water sample were identified as COPCs. For all COPCs, the spatial distribution and magnitude of HQs, the seasonal patterns of HQs, and the associated exposure and effects assumptions were evaluated to provide a more detailed assessment of impacts on the benthic community, bivalve population, and crayfish population. The evaluation of COPCs and associated uncertainties were further examined to arrive at risk conclusions for the benthic invertebrate assessment endpoints (Section 6.7).

6.5.2 COPCs Evaluated

Eleven of the 14 surface water COPCs identified in the SLERA and refined screen (Attachment 5) were evaluated in the BERA. Three individual DDT metabolites identified in the SLERA (2,4′-DDD, 2,4′-DDT, and 4,4′-DDD) were evaluated as part of total DDx and were not evaluated individually; 4,4′-DDT was evaluated both individually and as total DDx because the TRV for DDx is based on 4,4′-DDT. All other COPCs were evaluated in this assessment (Table 6-30).

Table 6-30. Surface Water COPCs Evaluated in the BERA

COPCs	
Metals	
Zinc[a]	
Butyltins	
Monobutyltin	
PAHs	
Benzo(a)anthracene	Naphthalene
Benzo(a)pyrene	
Phthalates	
BEHP	
PCBs	
Total PCBs	
Pesticides	
4,4′-DDT	Total DDx
VOCs	
Ethylbenzene	Trichloroethene

[a] TRV based on dissolved concentration.

BEHP – bis(2-ethylhexyl) phthalate
BERA – baseline ecological risk assessment
COPC – contaminant of potential concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT, and 4,4′-DDT)
TRV – toxicity reference value
VOC – volatile organic compound

Nineteen surface water COIs were not evaluated in the SLERA and refined screen because no toxicological data were available (Table 6-31). Four of these COIs (4-chloroaniline, aniline, 4-(2,4-dichlorophenoxy)butyric acid (2,4-DB), and methylchlorophenoxypropionic acid [MCPP]) were detected infrequently, in isolated areas, and at different times. The risks to benthic invertebrates from these chemicals in surface water are unknown because of the absence of toxicological data. Surface water thresholds are unavailable for individual dioxins and furans other than 2,3,7,8-TCDD. For vertebrates, dioxins and furans are evaluated as a toxicity-weighted sum based on the toxicity of each congener relative to 2,3,7,8-TCDD, using TEFs based on their common mechanism for toxicity; but TEFs are not available for benthic invertebrates.

Table 6-31. Surface Water COIs With No Chronic TRVs

Metals	
Aluminum	
SVOCs	
4-Chloroaniline	Aniline
Herbicides	
2,4-DB	MCPP
Dioxins/Furans	
1,2,3,4,6,7,8-Heptachlorodibenzofuran	1,2,3,7,8,9-Hexachlorodibenzofuran
1,2,3,4,6,7,8-Heptachlorodibenzo-*p*-dioxin	1,2,3,7,8,9-Hexachlorodibenzo-*p*-dioxin
1,2,3,4,7,8,9-Heptachlorodibenzofuran	1,2,3,7,8-Pentachlorodibenzofuran
1,2,3,4,7,8-Hexachlorodibenzofuran	1,2,3,7,8-Pentachlorodibenzo-*p*-dioxin
1,2,3,4,7,8-Hexachlorodibenzo-*p*-dioxin	2,3,4,6,7,8-Hexachlorodibenzofuran
1,2,3,6,7,8-Hexachlorodibenzofuran	2,3,4,7,8-Pentachlorodibenzofuran
1,2,3,6,7,8-Hexachlorodibenzo-*p*-dioxin	2,3,7,8-Tetrachlorodibenzofuran

2,4-DB – 4-(2,4-dichlorophenoxy)butyric acid
COI – contaminant of interest
MCPP – methylchlorophenoxypropionic acid
SLERA – screening-level ecological risk assessment
SVOC – semivolatile organic compound
TRV – toxicity reference value

Aluminum was not identified as a water COPC as per agreement with EPA because the AWQC were developed using toxicity data from acidic waters and are not applicable to the Study Area. Aluminum concentrations in background surface water and sediment were evaluated to identify local sources of aluminum contamination within the Study Area, if any (Section 6.5.5.3). Like aluminum, zinc is naturally occurring in the environment, and background zinc concentrations were also evaluated.

In addition, one COI (2,4′-DDE) was not retained as a COPC in the refined screen because no measured concentration exceeded the TRV (although at least one DL exceeded a TRV). However, 2,4′-DDE was evaluated as a component of total DDx.

6.5.3 Exposure Assessment

This section presents the exposure assessment of the surface water COPCs. An overview of the Study Area surface water sampling program is presented in Section 6.5.2.1, and surface water EPCs for benthic invertebrates are defined in Section 6.5.2.2.

6.5.3.1 Overview of Surface Water Data Collected from the Study Area

The surface water sampling program was designed to characterize the chemical concentrations in the river under low-flow (< 50,000 cfs) and high-flow (> 50,000 cfs) regimes (Section 5.0 of the draft final RI (Integral et al. 2011)). In addition, the surface water study was designed to characterize surface water during stormwater-influenced low flow (i.e., during active runoff into the Study Area). This section provides an overview of exposure concentrations in the Study Area surface water across several seasons and provides an estimate of likely temporal variability in exposure.

6.5.3.1.1 Sampling Events

In Round 2 and Round 3, surface water samples were collected at 38 locations (including single-point near-bottom and near-surface samples[58] and vertically or horizontally integrated transect samples[59]) during seven surface water sampling events from November 2004 to January 2007 (Map 4-15). Four sampling events occurred during low-flow conditions (November 2004, March 2005, July 2005, and September 2006), two sampling events occurred during high-flow conditions (January 2006 and January 2007), and one sampling effort occurred during a stormwater event (November 2006).

The Round 2 surface water data were collected to capture the seasonal differences in water flow conditions in the LWR (Integral 2004a). Round 3 surface water data were collected to provide additional seasonal data as well as event-specific (i.e., storm) data for the LWR (Integral 2006b). Additional non-LWG surface water data were collected in May 2005 at one location in the Study Area (RM 6.4, west) and were included in the BERA dataset. Additional details on the surface water sampling methods and events are described in Section 4.1.4.

6.5.3.1.2 Sampling Types

Single-point surface water samples were collected using a peristaltic pump at most locations. These samples were analyzed for metals, PCB Aroclors, organochlorine pesticides, and SVOCs. In addition, a subset of samples was also collected using an XAD. This XAD method was used to collect samples for the analysis of low-level hydrophobic organic compounds (i.e., dioxins and furans, PCB congeners, PAHs, organochlorine pesticides, and phthalate esters). These hydrophobic chemicals are frequently undetected in surface water when standard analytical methods are used. For certain of these chemicals, very low DLs are needed to allow screening against relevant water quality criteria established for aquatic biota and for the protection of human health via the ingestion of fish (Integral 2006b). The XAD system uses a 0.5-µm glass fiber filter to capture the particulate fraction from a high volume of water, in addition to a resin

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[58] Surface water was collected from two points in the water column. The near-bottom sample was collected at a depth of 1 ft off the river bottom. The near-surface sample was collected 3 ft below the surface.

[59] At transect sampling locations, vertically or horizontally integrated samples consisted of composites of water samples collected from multiple lateral or vertical locations at one cross-section of the river and were designed to estimate an integrated water concentration for that section of the river at one point in time.

column to capture remaining concentrations in the "dissolved" fraction. The analytical results from the filter and the column were combined to estimate total analyte concentrations in the water column. Sampling locations where both peristaltic pump and the XAD collection methods were employed for a given analyte, the XAD results were preferentially used because of the greater sensitivity and lower DLs. A comprehensive summary of the surface water data is presented in Attachment 4.

All water samples collected using the peristaltic pump were analyzed using the whole water sample (i.e., unfiltered). In addition, a portion of each sample was filtered and analyzed for metals. Thus, for zinc (the only metal COPC), data are available for both dissolved and total fractions.

Uncertainty Associated with Surface Water Sampling Methods

Uncertainties are associated with the use of multiple sampling types and methods in the evaluation of ecological exposure to surface water (e.g., duration of sampling time for a single-point grab versus an integrated transect sample, water volume sampled by an XAD versus peristaltic method). Surface water was collected both as single-point samples and as a spatially integrated (vertical or horizontal transect) samples using two types of sampling methods (i.e., the XAD method and the peristaltic method). Samples also were collected over seven sampling events; however, not all surface water locations were sampled at each event. Surface water transect samples provide a measurement over a longer temporal scale, although horizontal transects were sampled at only five locations within the Study Area (at RM 2.0, RM 4.0, RM 6.3, RM 11, and at the mouth of Multnomah Channel).

The multiple temporal and spatial scales over which samples were collected allows for a reasonable characterization of surface water data in the Study Area; however, the relevance of ecological exposure to surface water data collected from the various sampling types is highly uncertain.

6.5.3.1.3 Surface Water COPC Concentrations

All surface water data are presented in Attachment 4. General trends in surface water COPC concentrations are described below. A detailed evaluation of the distributions of surface water chemical concentrations is presented in Section 5.0 of the draft final RI (Integral et al. 2011).

- **Metals and Butyltins –** Zinc and monobutyltin were analyzed only in peristaltic pump samples. Zinc (dissolved) was detected in about half the samples at concentrations ranging from 0.9 to 41.9 µg/L. Butlytin was infrequently detected at concentrations ranging from 2 to 85 µg/L.
- **SVOCs –** SVOCs (PAHs and BEHP) were analyzed in both peristaltic pump and XAD samples, with lower DLs achieved in the XAD samples. PAHs were detected in less than 15% of the peristaltic samples, and at concentrations ranging from 2.4 to 60,500 ng/L. PAHs were detected more often in the XAD samples (detection frequencies of 22% to 90%), with concentrations ranging from 0.018 to 34.5 ng/L. BEHP detection frequencies in the XAD samples were 4 times those in the peristaltic samples. BEHP concentrations in the peristaltic samples ranged from 700 to 68,000 ng/L; those in the XAD samples ranged from 7.75 to 33 ng/L.

- **PCBs –** Total PCBs were analyzed in both peristaltic pump and XAD samples, with lower s in the XAD samples. Total PCB Aroclors were analyzed in peristaltic samples only and had concentrations that ranged from 5.9 to 17 ng/L, with a less than 15% detection frequency. Total PCB congeners were analyzed in XAD samples only and were detected in all samples, at concentrations that ranged from 0.0457 to 12 ng/L.
- **DDx –** Total DDx were analyzed in both peristaltic pump samples and XAD samples, with lower detection limits in the XAD samples. Total DDx analyzed in peristaltic samples were detected at a frequency of 35%, at concentrations that ranged from 1.2 to 20 ng/L. Total DDx analyzed in XAD samples were was detected more frequently—in all samples—and at lower concentrations (0.0372 to 9.76 ng/L). In addition, the maximum concentrations for 4,4′-DDT, and total DDx in the peristaltic samples were based on NJ- or NJT-qualified data. NJ-qualification indicates "the presence of an analyte that has been 'tentatively identified,' and the associated numerical value represents its approximate concentration" (EPA 1999b). The qualification indicates that the validator believed the result was due to analytical interference from a chemical other than the target analyte.

6.5.3.2 Surface Water EPCs

Surface water EPCs in the assessment were represented by detected concentrations in individual surface water samples. Surface water concentrations were compared to water TRVs to characterize risks to benthic invertebrates via exposure to surface water. Surface water COPC concentration data for all individual samples are presented in Attachment 4. Near-bottom surface water samples are called out separately (Section 6.5.5.2.1), as they might be more representative of exposure concentrations for benthic invertebrates.

Invertebrate Exposure to Surface Water Uncertainty

Many benthic organisms live or feed at the interface between surface water and sediment, in a zone known as the benthic boundary layer. The thickness of this layer depends primarily on the velocity of the water flowing over the bottom and the roughness of the bottom surface. Faster flows and minimal roughness tend to result in very shallow boundary layers—in larger rivers, the boundary layer is millimeters to centimeters thick. This intersection between the sediment and the water sets up conditions under which particles may be aggregated or "captured," in turn attenuating the physical forces to which the organisms are exposed, creating turbulent flow, and altering the exchange of dissolved chemicals (including oxygen and waste products) and food within the sediment, the boundary layer, and the overlying water column. Water column and vertically integrated transect samples, not having been collected from the benthic boundary layer where the epibenthic invertebrates reside, may not be representative of benthic community exposure. Any risk to the benthic community based on COPC concentrations in such samples is therefore associated with uncertainty.

6.5.4 Effects Assessment

Surface water chemical concentrations were compared to the effects thresholds as part of the risk characterization process. At the direction of EPA (2008f), chronic water TRVs were developed for all surface water COPCs based on the hierarchy detailed in

Attachment 10. These TRVs, which are listed in Table 6-32 for all surface water COPCs, are based on national AWQC, state WQS, or national Tier II criteria. Developed by considering the sensitivities of fish and invertebrate species, each TRV is intended to be protective of all aquatic receptors, including benthic invertebrates. As indicated in Table 6-32, the chosen level is often significantly lower than the lowest effects concentration for invertebrates.

Table 6-32. Water TRVs for Surface Water COPCs

COPC	Unit	Water TRV	Source	Basis for Water TRV
Metals				
Zinc	µg/L	36.5[a]	AWQC	Derived from a number of acute and chronic studies with both invertebrates and fish; the lowest chronic effect concentrations are 46.7 µg/L for invertebrates and 36.4 µg/L for fish.
Butyltins				
Monobutyltin	ng/L	72[b]	AWQC	Based on TBT AWQC surrogate; derived from acute studies with seven invertebrate species, two chronic studies with *Daphnia*, acute studies with five fish species (including rainbow trout), and one chronic study with fathead minnow; the lowest chronic concentrations are 0.14 µg/L for invertebrates and 0.26 µg/L for fish.
PAHs				
Benzo(a)-anthracene	ng/L	27	Tier II	Derived from one acute study with *Daphnia*, resulting in an LC50 concentration of 10 µg/L.
Benzo(a)pyrene	ng/L	14	Tier II	Derived from one acute study with *Daphnia*, resulting in an LC50 concentration of 5 µg/L.
Naphthalene	ng/L	12,000	Tier II	Derived from studies with *Daphnia* (two acute), rainbow trout (one acute), and fathead minnow (two acute and one chronic); lowest effect concentration is 619 µg/L (chronic) for fish and 2,194 µg/ L (acute) for invertebrates (*Daphnia*). Value was derived by dividing the lowest GMAV (1,600 µg/L, the LC50 for rainbow trout) by an acute adjustment factor.
Phthalates				
BEHP	ng/L	3,000	Tier II	Derived from five studies (four acute and one chronic) with aquatic invertebrates, seven acute studies with fish (including rainbow trout), and one acute study with an amphibian; lowest effect concentrations are 133 µg/L for invertebrates (*Daphnia*) and 160 µg/L for fish.
VOCs				
Ethylbenzene	µg/L	7.3	Tier II	Derived from two acute studies with fish (fathead minnow and guppy); the lowest effect concentration is 8,450 µg/L.

Table 6-32. Water TRVs for Surface Water COPCs

COPC	Unit	Water TRV	Source	Basis for Water TRV
Trichloroethene	µg/L	47	Tier II	Derived from four acute and one chronic study with fish (fathead minnow and flagfish); the lowest effect concentration is 11,057 µg/L (chronic).
PCBs				
Total PCBs	µg/L	0.19	Alternative TRV	An alternative TRV was derived because the default TRV (0.014 µg/L) protects mink from dietary exposure and is not appropriate for evaluation of aquatic receptors.
Pesticides				
4,4′-DDT	µg/L	0.011	Alternative TRV	An alternative TRV was derived because the AWQC-based chronic TRV (0.001 µg/L) protects birds from dietary exposure (specifically eggshell thinning) and is not appropriate for the evaluation of aquatic receptors.
Total DDx	µg/L	0.011	Alternative TRV	

[a] Chronic TRV is based on dissolved criteria; chronic TRV was compared to dissolved concentration measured in Study Area. Attachment 10 presents the method for hardness-adjustment of the TRV

[b] Chronic TRV is based on criterion for TBT.

AWQC – ambient water quality criteria
BEHP – bis(2-ethylhexyl) phthalate
COPC – contaminant of potential concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
GMAV – genus mean acute value
LC50 – concentration that is lethal to 50% of an exposed population
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
TBT – tributyltin
total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT, and 4,4′-DDT)
TRV – toxicity reference value
VOC – volatile organic compound

Because the selected AWQC for total PCBs and 4,4′-DDT are based on protection of mammals and birds, respectively, risk estimates for aquatic receptors based on these TRVs are associated with substantial uncertainty. Therefore, alternative TRVs protective of fish and invertebrates were developed in this BERA using methods consistent with those used for AWQC derivation.

The toxicity data presented in the PCB AWQC document (EPA 1980c), were insufficient to derive a final acute value (FAV) or FCV. A PCB FAV was calculated using the AWQC data plus additional toxicity data published since 1979 and reported in EPA's ECOTOX online database (ECOTOX 2009). This dataset was sufficient to derive an FAV in accordance with AWQC methods. Data and calculations are presented in Attachment 10. The resultant FAV is 1.6 µg/L. The FAV was divided by the geometric mean of the ACRs presented in the AWQC document for PCBs (8.39), to yield an FCV of 0.19 µg/L.

This concentration (0.19 µg/L) was evaluated as an alternative water TRV for total PCBs. For evaluating direct exposure of organisms to water, this alternative water TRV is considered more appropriate than the total PCB criterion (0.014 µg/L), which is based on protection of mink via ingestion of contaminated prey. Both the selected PCB water TRV (0.014 µg/L) and the alternative water TRV (0.19 µg/L) were used to calculate water HQs; however, because the alternative TRV is more appropriate for assessing risk to aquatic organisms (including benthic invertebrates), the alternative TRV was used to determine risk conclusions.

The AWQC document for DDT and metabolites (i.e., DDx) reports sufficient data to calculate criteria for protection of aquatic biota directly exposed to DDx in water (EPA 1980a). The AWQC chronic value, however, was not based on the direct calculations because lower thresholds were necessary to protect aquatic-dependent birds from toxicity via bioaccumulation of DDx up the food chain. The protection of birds is not pertinent to the assessment of risk to aquatic organisms. The direct exposure toxicity data reported in the AWQC document for DDx were used to establish an alternative chronic TRV for protection of aquatic life following the guidelines for development of AWQC (Stephan et al. 1985).

The AWQC document for DDx reports a calculated FAV of 1.1 µg/L for 4,4′-DDT, but insufficient toxicity data were available to derive an FCV or an ACR. Only a single ACR (65) was identified in the AWQC document for DDx. Suter and Tsao (1996) recommend an ACR of 17.9 when fewer than three ACRs are available; however, Raimondo et al. (2007) reported ACRs ranging from 3 to 5 (median 3.6) in four studies of chemicals with a DDT-like mode of action. Consistent with use of the Raimondo et al. (2007) ACRs to derive tissue-residue TRVs for use in this BERA (see Attachment 9), the Raimondo et al. median ACR for chemicals with a DDT-type mode of action was used to calculate the FCV. Dividing the FAV (1.1 µg/L) by the median reported ACR of 3.6 results in an FCV of 0.31 µg/L.

In accordance with the guidelines for AWQC development (Stephan et al. 1985), a final tissue-residue value is appropriate when establishing the alternative TRV, provided that it is likely to be lower than an FCV or final plant value. A tissue-residue-derived water TRV of 0.011 µg/L was calculated by dividing the DDx 10^{th} percentile fish tissue-residue LOAEL (1.6 mg/kg ww) by a bioaccumulation factor (BAF) of 142,960[60] (derived from the DDT AWQC document). Because this concentration of 0.011 µg/L is lower and thus more conservative than the FCV (0.31µg/L) and the lowest plant value (lowest algae value is 0.3 µg/L) (EPA 1980a), it was selected as the alternative water TRV for DDx compounds in water. For evaluating direct exposure of aquatic organisms (i.e., benthic invertebrates) to water, this alternative TRV is considered more appropriate than the AWQC (0.0010 µg/L), which is based on the protection of brown pelican via ingestion of contaminated prey. Both the selected DDT water TRV (0.001µg/L) and the alternative

[60] A BAF of 1,429.60 was based on the lipid-normalized BAF (17,870) and anchovy lipid percent (8%) as presented in the DDT AWQC document.

water TRV (0.011 µg/L) were used to calculate water HQs. Because the alternative TRV is more appropriate for assessing risk to fish, it was used to determine risk conclusions.

Surface Water TRV Uncertainties

TRVs are based on the most sensitive aquatic organism and may overestimate effects to benthic invertebrates that are less sensitive than the species on which the TRV is based. The TRVs established by the AWQC and Tier II sources included toxicity data on a planktonic invertebrate (*Daphnia* sp.) with multiple endpoints for assessing risks to benthic invertebrates (LC50, EC50 based on mortality, growth, and reproduction) for the following six COPCs: zinc, monobutyltin, benzo(a)anthracene, benzo(a)pyrene, naphthalene, and BEHP. These TRVs are considered adequately protective for evaluating risks to the benthic community, despite the fact that they were derived for a planktonic invertebrate.

The relevance of the water TRVs for total PCBs, ethylbenzene, trichloroethene, and DDx compounds in assessing risks to benthic invertebrates is more uncertain. The TRVs for total PCBs, ethylbenzene, and trichloroethene are based on the effects data for fish species and may over- or underestimate risks to the benthic community. The TRV for total DDx is based on the 4,4′-DDT AWQC and is derived from effects data for only one bird species (brown pelican) via ingestion of contaminated prey. Because birds are known to be sensitive to DDx compounds,[61] the water TRV that is protective of birds may overestimate risk to the benthic community. Similarly, the total PCB AWQC is based on the protection of mink via ingestion of contaminated prey. Alternative water TRVs were therefore developed in this BERA for total PCBs and total DDx using toxicity data specific to aquatic organisms and following the methods used to develop AWQC. The alternative TRVs are considered more appropriate for evaluating risks to aquatic organisms directly exposed to surface water. Although both the selected and alterative water TRVs were used to derive water HQs, only the alternative TRV was used to determine risk conclusions.

6.5.5 Risk Characterization

This section presents the risk estimates for invertebrates based on the surface water LOE. An HQ calculation was used to quantify estimated risk. The EPC and TRV are represented by surface water chemical concentrations. Section 6.5.5.1 presents the overall approach used to characterize risks via surface water to benthic invertebrate receptors. Section 6.5.5.2 presents the risk characterization results, uncertainty evaluation, and surface water COPCs. Section 6.5.5.3 presents an evaluation of background concentrations. Section 6.5.5.4 presents a summary of surface water COPCs.

6.5.5.1 Risk Characterization Process

The surface water exposure risk characterization for benthic assessment endpoints was conducted by evaluating individual surface water samples. HQs were calculated on a sample-by-sample basis for all surface water samples within the Study Area, in

[61] The best documented response is eggshell thinning in birds, which can result in embryo mortality and decreased hatchling survival (e.g., Heath et al. 1969; Lincer 1975). Overall avian sensitivity is highly variable. Raptors, waterfowl, passerines, and non-passerine ground birds have been documented to be more susceptible to eggshell thinning than are domestic fowl and other gallinaceous birds, and DDE appears to have been a more potent inducer of eggshell thinning than DDT (EPA 2007b). The leading hypothesis for DDE-induced thinning involves an inhibition by p,p′-DDE (but not by o,p′-DDE, -DDD, or -DDT) of prostaglandin synthesis in the shell gland mucosa (EPA 2007b).

accordance with the EPA's Problem Formulation (Attachment 2). A sample-by-sample exposure analysis is more appropriate for surface water than, say, for sediment because surface water is a well-mixed, flowing medium whose samples represent exposure areas that are larger than the sampling location.

A contaminant with an HQ ≥ 1 for any individual surface water sample was identified as COPC. For all COPCs, the spatial distribution and magnitude of HQs, the seasonal patterns of HQs, and the associated exposure and effects assumptions were evaluated to provide a more detailed assessment of impacts on benthic invertebrates. COPCs and associated uncertainties were further evaluated to arrive at risk conclusions for benthic invertebrates (Section 6.7).

6.5.5.2 Risk Characterization Results and Uncertainty Evaluation

Risks are characterized in the following section based on the frequency and magnitude of TRV exceedances in samples representing the entire water column and those more closely associated with the habitat where the benthic community members reside.

6.5.5.2.1 Risk Characterization

Individual HQs calculated across all surface water samples for all COPCs are shown in Table 6-33. By definition (because the BERA and SLERA TRVs and EPCs were unchanged), all of the COPCs had at least one sample with an HQ ≥ 1.[62] For six COPCs, HQs were ≥ 1 in less than 2% of samples: zinc, monobutyltin, benzo(a)anthracene, benzo(a)pyrene, BEHP, and total PCBs. For two COPCs—ethylbenzene and trichloroethene—only one sample had an HQ ≥ 1. Frequency of exceedance was higher for naphthalene (3.7%), 4,4′-DDT (11%), and total DDx (21%); when alternative TRVs were applied to 4,4′-DDT and total DDx, only one sample exceeded the TRV.

Table 6-33. Number of Surface Water Samples with HQs ≥ 1

COPC	Number of Samples with HQs ≥ 1 (Maximum HQ)	Percentage of Samples with HQs ≥ 1
Metals		
Zinc (dissolved)	1 of 167 (1.1)	< 1%
Butyltins		
Monobutyltin	1 of 167 (1.2)	< 1%
PAHs		
Benzo(a)anthracene	2 of 245 (10)	< 1%
Benzo(a)pyrene	3 of 245 (14)	1.2%

[62] All of the COPCs had at least one sample with an HQ ≥ 1 only when AWQC-based TRVs for Total PCBs and Total DDx are considered.

Table 6-33. Number of Surface Water Samples with HQs ≥ 1

COPC	Number of Samples with HQs ≥ 1 (Maximum HQ)	Percentage of Samples with HQs ≥ 1
Naphthalene	10 of 268 (50)	3.7%
Phthalates		
BEHP	2[a] of 190 (2.3)	1.1%
PCBs		
Total PCBs[b, c]	0 of 160 (0.1)	0%
Pesticides		
4,4′-DDT[d, e]	0 of 170 (0.40)	0%
Total DDx[d, e]	1 of 170 (1.8)	0.60%
VOCs		
Ethylbenzene	1 of 23 (1.6)	3.7%
Trichloroethene	1 of 23 (4.1)	3.7%

[a] An additional two samples had DLs> TRV. The maximum HQ based on a DL is 1.4 for BEHP.

[b] The summary statistics for total PCB concentrations are based on the alternative total PCB TRV of 0.19 µg/L, which is specific to aquatic organisms.

[c] 2 of 160 samples had total PCB concentrations greater than the AWQC-based TRV of 0.014 µg/L, which is specific to protection of mink via consumption of contaminated prey (maximum HQ was 1.2).

[d] The summary statistics for 4,4′-DDT and total DDx concentrations are based on the alternative 4,4′-DDT TRV of 0.011 µg/L, which is specific to aquatic organisms.

[e] 19 of 170 and 35 of 170 samples had 4,4′-DDT and total DDx concentrations, respectively, greater than the TRV of 0.001 µg/L, which is based on protection of birds (maximum HQs were 4.7 and 20, respectively). An additional four samples had DLs that were greater than the AWQC TRV. The maximum HQ based on a DL is 1.6 for both 4,4′-DDT and total DDx.

AWQC – ambient water quality criteria
BEHP – bis(2-ethylhexyl) phthalate
COPC – contaminant of potential concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
DL – detection limit
HQ – hazard quotient
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT, and 4,4′-DDT)
TRV – toxicity reference value
VOC – volatile organic compound

Individual HQs calculated for near-bottom surface water samples for all COPCs are shown in Table 6-34. The percent of samples with HQs ≥ 1 are similar to those in Table 6-33; naphthalene, 4,4′-DDT, and total DDx had the greatest number of samples with HQs ≥ 1.[63]

[63] 4,4′-DDT, and total DDx had a relatively high number of samples with HQ ≥ 1 only when the AWQC-based TRVs is considered.

Table 6-34. Number of Near-Bottom Surface Water Samples with HQs ≥ 1

COPC	Number of Samples with HQs ≥ 1 (Maximum HQ)	Percentage of Samples with HQs ≥ 1
Metals		
Zinc (dissolved)	1 of 91 (1.1)	1.1%
Butyltins		
Monobutyltin	0 of 91	0%
PAHs		
Benzo(a)anthracene	2 of 122 (10)	1.6%
Benzo(a)pyrene	3 of 122 (14)	2.5%
Naphthalene	10 of 145(50)	6.9%
VOCs		
Ethylbenzene	1 of 23 (1.6)	4.3%
Trichloroethene	1 of 23 (4.1)	4.3%
Phthalates		
BEHP	2[a] of 105 (2.3)	1.9%
PCBs		
Total PCBs	0[b] of 86	0%
Pesticides		
4,4′-DDT	0 of 93 (0.4)[c, d]	0%
Total DDx	1 of 93 (1.8) [c, d]	1%

[a] An additional sample had a DL that was greater than the TRV. The maximum HQ based on a DL is 1.4.

[b] 1 of 86 samples (1.1%) had a total PCB concentration greater than the total PCB TRV of 0.014 µg/L.

[c] An additional three samples had DLs greater than the TRV. The maximum HQ based on a DL is 1.6 for both 4,4′-DDT and total DDx.

[d] 11 of 93 and 21 of 93 had 4,4’-DDT and total DDx concentrations, respectively, greater than the TRV of 0.001 µg/L/L (HQs were 4.7 and 20, respectively).

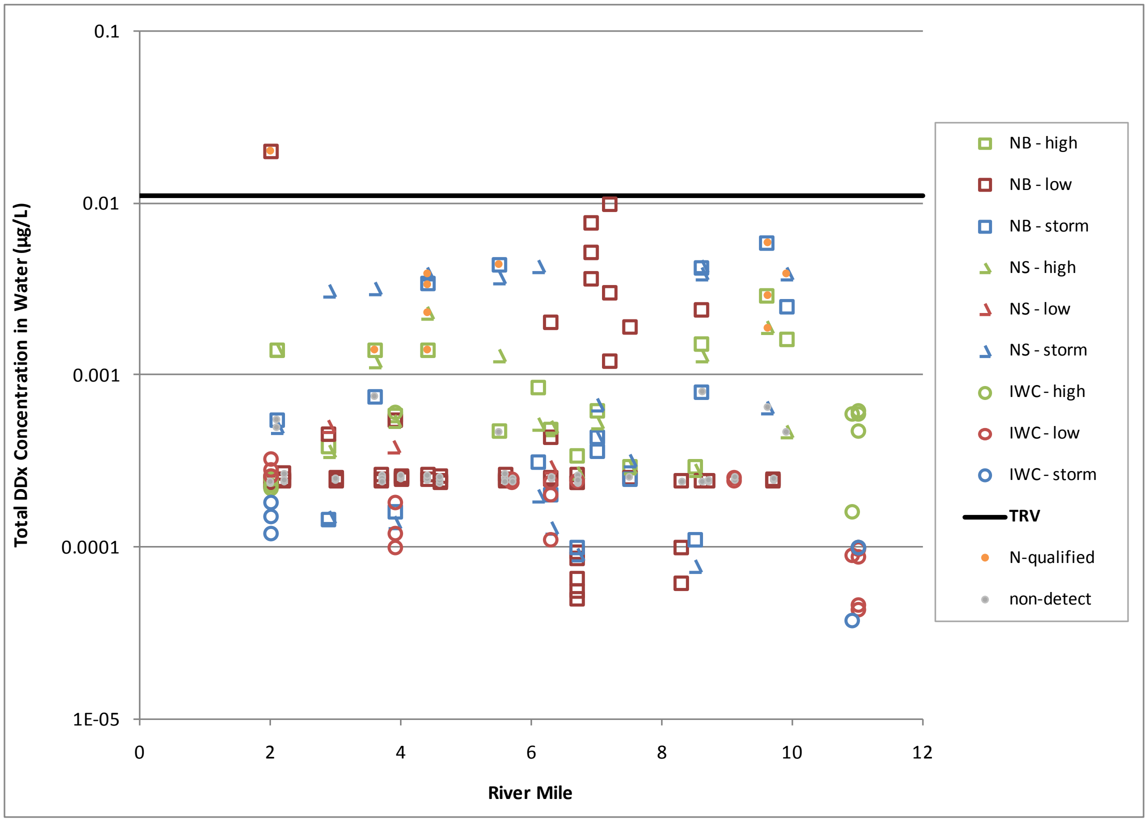
BEHP – bis(2-ethylhexyl) phthalate
COPC – contaminant of potential concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
DL – detection limit
HQ – hazard quotient
LOEC – lowest-observed-effect concentration
NOEC – no-observed-effect concentration
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT, and 4,4′-DDT)
TRV – toxicity reference value
VOC – volatile organic compound

All COPCs had HQs ≥ 1 in at least one surface water sample[64]. Map 6-27 presents the sampling locations with HQs ≥ 1. A discussion of these COPCs and an evaluation of the key uncertainties, including the frequency, location, water depth, and effect of flow condition, and their effects on HQs are presented below.

- **Metals, butyltins, and VOCs –** For four of the COPCs (i.e., zinc, monobutyltin, ethylbenzene, and trichloroethene), calculated HQs are ≥ 1 in only one sample. The monobutyltin evaluation is uncertain because the TRV is based on the AWQC for the more toxic butyltin TBT. Uncertainty is associated with the evaluation of ethylbenzene and trichloroethene because data were spatially limited and because the water TRVs are protective of fish, which may over- or underestimate risks to benthic invertebrates.
- **PAHs, BEHP, and total PCBs –** Four COPCs (i.e., benzo(a)anthracene, benzo(a)pyrene, BEHP, total PCBs) have HQs ≥ 1 in fewer than five samples. The two benzo(a)anthracene samples and three benzo(a)pyrene samples with HQs ≥ 1 were collected near the bottom at RM 6.1 or RM 6.3, during both high- and low-flow conditions. BEHP and total PCBs each have HQs ≥ 1 in two samples at different locations in the river. Only one near-bottom sample has an HQ ≥ 1 for BEHP (HQ = 2.3), and only one near-bottom sample has an HQ ≥ 1 for total PCBs (HQ = 1.2). Uncertainty associated with the PAH evaluation is due, in part, to the use of extrapolated LC50s as the basis of the TRVs. A high degree of uncertainty is associated with the total PCB evaluation because the TRV is protective of mink, and as such may over- or underestimate risks to benthic invertebrates. When the alternative total PCB TRV of 0.19 µg/L that is specific to aquatic organisms is used, no samples exceed the TRV.
- **Naphthalene –** Naphthalene does not have any HQs ≥ 1 for the peristaltic or XAD samples collected during the LWG surface water sampling events. However, naphthalene HQs are ≥ 1 in 5% of the peristaltic samples collected from RM 6.4 on the west bank of the Study Area during a non-LWG sampling event. Naphthalene HQs are ≥ 1 at only this one localized area.
- **DDx –** Total DDx in only one sample (W001, RM 2.0) exceeds the TRV. However, this result is N-qualified, indicating that the elevated concentration was likely due to analytical interference from a different chemical. Six percent of the samples (n = 11) were N-qualified data, indicating a high bias in the results because of potential interference from another analyte. These N-qualified data, including the sample with the highest total DDx concentration (0.0198 µg/L), are considered highly uncertain. Total DDx HQs based on non-N-qualified data (n = 159) range from 0.003 to 0.9.

[64] All COPCs had HQs ≥1 in at least one surface water sample only when AWQC criteria were used for total PCBs and 4,4′-DDT.

The total DDx HQ used in this assessment is based on the alternative TRV of 0.011 µg/L. A high degree of uncertainty is associated with the total DDx AWQC-based TRV (0.001 µg/L), which was derived from bird effects data and cannot be meaningfully applied to benthic invertebrates.



high – high-flow conditions	low – low-flow conditions	NS – near surface
IWC – integrated water column	NB – near bottom	storm – storm flow conditions

Figure 6-19. Total DDx Concentrations in All Surface Water Samples

Additional uncertainty for risks associated with DDx compounds concerns the toxicological basis for the selected TRV. With the lowest acute value for invertebrates of 0.18 µg/L and no corresponding chronic values available, a TRV based on empirical invertebrate data could not be developed. Because the AWQC for 4,4′-DDT is based on protection of birds via ingestion of contaminated prey, an alternative TRV of 0.011 µg/L was selected as the appropriate metric for judging risk to aquatic organisms and as the primary line of evidence to determine risk conclusions.

6.5.5.2.2 Uncertainty Analysis of Surface Water Sampling Methods

The BERA surface water dataset consists of data collected using both peristaltic and XAD samplers. When samples were collected from the same location using both methods, exceedances occurred only in the peristaltic samples because the XAD method results in lower DLs and more accurate analytical results for low-level organic chemicals. A comparison of the surface water samples collected at the same location using both methods was conducted to evaluate if excluding the less accurate peristaltic data affected risks estimates. At least one PAH (i.e., benzo(a)anthracene, benzo(a)pyrene) or pesticide COPC (e.g., 4,4′-DDT and total DDx) was detected at 10 sampling locations. In no case did PAHs detected in peristaltic samples exceed chronic criteria.

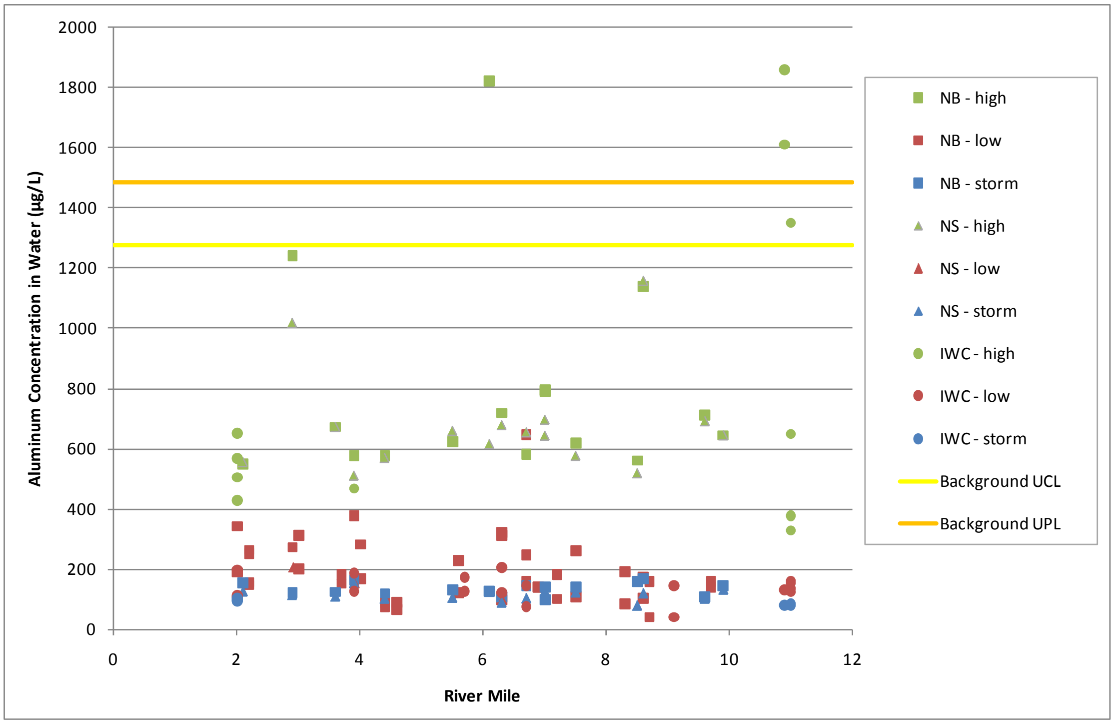
For pesticides, however, peristaltic samples collected during the Round 3 storm event at two locations (W027 at RM 2.9 and W031 at RM 6.1) exceeded the corresponding chronic TRV; the accompanying XAD samples did not. The 4,4′-DDT concentrations in the peristaltic storm event samples from W027 and W031 were 0.0019 µg/L and 0.0029 µg/L, respectively; the total DDx concentrations were 0.0031 µg/L and 0.0043 µg/L, respectively. In comparison, concentrations of 4,4′-DDT in the XAD samples from the same locations were 2 orders of magnitude lower and those of total DDx concentrations from the same sampling event were 1 order of magnitude lower.

During the same sampling event, peristaltic samples exceeded the 4,4′-DDT TRV at two additional locations (W030 at RM 5.5 and W036 RM 8.6). As noted in Section 6.5.3.1.2, only XAD results were used where both XAD and peristaltic samples were available. The addition of these two peristaltic samples would slightly increase the number of locations where the total DDx HQ is ≥ 1.

The peristaltic samples with HQs ≥ 1 were collected during a storm event near the surface of the water over a short period of time (e.g., 20 minutes) and represent a whole water sample. In contrast, the XAD samples were collected over a longer time interval using a column and a filter that produced results more accurately reflecting the total DDx in the sample.

6.5.5.3 Evaluation of Background Concentrations

Aluminum was not identified as a COPC because no acceptable TRV was identified for the circumneutral waters associated with the Study Area. Background concentrations in surface water and sediment were established as part of the RI (Section 7.0 of the draft final RI (Integral et al. 2011)). A comparison of Study Area to background concentrations in sediment and surface water is presented in Attachment 11. The Study Area UCL water concentration of aluminum (460 µg/L) is approximately one-third as great as the background UCL and upper prediction limit (UPL) concentrations (1,278 and 1,485 µg/L, respectively). The Study Area UCL sediment aluminum concentration (24,375 mg/kg dw) is similar to the background sediment UCL and UPL (24,877 and 33,842 mg/kg dw, respectively). Aluminum concentrations for the Study Area were generally below the background UCL and UPL (Figure 6-20).



high – high-flow conditions
IWC – integrated water column
low – low-flow conditions

NB – near bottom
NS – near surface
storm – storm flow conditions

Figure 6-20. Aluminum Surface Water Concentrations Compared to Background Concentrations

From these comparisons, it was concluded that potentially unacceptable risk to benthic invertebrates in the Study Area from aluminum cannot be attributed to sources from within the Study Area. Aluminum and other trace elements are major constituents of the mineral fraction of sediment and contribute to the analytical chemical findings as a result of the acid extraction step during analysis. Because aluminum is not biologically available to benthic invertebrates and not toxic at naturally occurring concentrations generally found in surface water, aluminum is not expected to pose unacceptable risk to benthic invertebrates.

Zinc also occurs naturally as a crustal element in the environment. A background water concentration could not be established because the number of data points was too limited (see Attachment 11). The Study Area UCL concentration of zinc in water (2.5 µg/L) is greater than highest zinc concentration detected in background[65] (range of 1.4 to 2.2 µg/L). The Study Area UCL concentration of zinc in sediment (164 mg/kg dw) is greater than the background sediment UCL and UPL (79 and 110 mg/kg dw, respectively). These data indicate that zinc concentrations are elevated above background

[65] Zinc was detected in only 3 of 22 surface water samples included in the background dataset (see Section 7.0 of the RI).

and that zinc concentrations in the Study Area cannot be solely attributed to background. This is as would be expected in a river within an urbanized basin.

6.5.5.4 COIs for Which Risks Cannot Be Quantified

COIs for which risks to benthic invertebrates cannot be quantified based on surface water data are listed in Table 6-35. No TRV is available for these COIs.

Table 6-35. Benthic Invertebrate Surface Water COIs with no Available TRV

COI	Rationale for Why Risks Cannot Be Quantitatively Evaluated
Metals	
Aluminum	Risk to benthic invertebrates based on surface water data unknown; AWQC not applicable to circumneutral waters of the Study Area
SVOCs	
4-Chloroaniline	Risk to benthic invertebrates based on surface water data unknown; no water threshold available
Aniline	Risk to benthic invertebrates based on surface water data unknown; no water threshold available
Herbicides	
2,4-DB	Risk to benthic invertebrates based on surface water data unknown; no water threshold available
MCPP	Risk to benthic invertebrates based on surface water data unknown; no water threshold available
Dioxins/Furans	
Individual congeners other than 2,3,7,8-TCDD[a]	Risk to benthic invertebrates based on tissue data unknown; no tissue TRVs available

[a] Includes the following dioxin and furan congeners: 1,2,3,4,6,7,8-heptachlorodibenzofuran; 1,2,3,7,8,9-hexachlorodibenzofuran; 1,2,3,4,6,7,8-heptachlorodibenzo-p-dioxin; 1,2,3,7,8,9-hexachlorodibenzo-p-dioxin; 1,2,3,4,7,8,9-heptachlorodibenzofuran; 1,2,3,7,8-pentachlorodibenzofuran; 1,2,3,4,7,8-hexachlorodibenzofuran; 1,2,3,7,8-pentachlorodibenzo-p-dioxin; 1,2,3,4,7,8-hexachlorodibenzo-p-dioxin; 2,3,4,6,7,8-hexachlorodibenzofuran; 1,2,3,6,7,8-hexachlorodibenzofuran; 2,3,4,7,8-pentachlorodibenzofuran; 1,2,3,6,7,8-hexachlorodibenzo-p-dioxin; and 2,3,7,8-tetrachlorodibenzofuran

2,4-DB – 4-(2,4-dichlorophenoxy)butyric acid
COI – contaminant of interest
MCPP – methylchlorophenoxypropionic acid
SVOC – semivolatile organic compound
TCDD – tetrachlorodibenzo-*p*-dioxin
TRV – ecological screening level

6.5.5.5 Summary of Surface Water COPCs

The following surface water COPCs were identified: zinc, monobutyltin, benzo(a)anthracene and benzo(a)pyrene, naphthalene, ethylbenzene, trichloroethene, BEHP, and total DDx. Table 6-36 summarizes the HQs and exposure and effects uncertainties for each surface water COPC. In Section 6.7, results of the surface water LOE are integrated with those from the other LOEs to determine risk conclusions for benthic invertebrates.

Table 6-36. Summary of Benthic Invertebrate Surface Water COPCs

COPC	Risk Exposure Area	HQ[a]	Percentage of All Samples with HQs ≥ 1	Sampling Event with HQs ≥ 1	Key Uncertainties
Metals					
Zinc	Individual sample with HQ ≥ 1 at RM 9.7 west	1.1	0.6%	November 2004 low-flow event	The HQ was 1.1 in one sample only; TRV is based on toxicity to fish and invertebrates; invertebrates may be less sensitive to zinc than are fish; zinc concentrations are elevated in Study Area as compared to background.
Butyltins					
Monobutyltin	Individual sample with HQ ≥ 1 at RM 11 west	1.2	0.6%	Winter 2007 high-flow event	TRV is based on a surrogate TBT TRV; HQ was 1.2 in one sample only; HQs are < 1 for all near-bottom samples; TRV is based on TBT effects data for invertebrates and fish.
PAHs					
Benzo(a)anthracene	Individual samples with HQ ≥ 1 at RM 6.1 and RM 6.3 west	4.1 – 10	0.8%	July 2005 low-flow event and winter 2007 high-flow event	HQs ≥ 1 based on peristaltic samples only at W012 and W031; TRV is based on extrapolated *Daphnia* acute LC50. Findings are consistent with other LOEs.
Benzo(a)pyrene	Individual samples with HQ ≥ 1 at RM 6.1 and RM 6.3 west	1.4 – 14	1%	November 2004 and July 2005 low-flow events, and winter 2007 high-flow event	HQs ≥ 1 based on peristaltic samples only at W012 and W031; TRV is based on extrapolated *Daphnia* acute LC50. Findings are consistent with other LOEs.
Naphthalene	RM 6.4 west	2.9 – 50	4%	May 2005 (non-LWG sampling event)	HQs ≥ 1 based on 10 different peristaltic samples only along the west bank of RM 6.4; TRV is based on risk to fish and invertebrates. Findings are consistent with other LOEs.

Table 6-36. Summary of Benthic Invertebrate Surface Water COPCs

COPC	Risk Exposure Area	HQ[a]	Percentage of All Samples with HQs ≥ 1	Sampling Event with HQs ≥ 1	Key Uncertainties
Phthalates					
BEHP	Individual samples with HQ ≥ 1 at RM 3.9 (transect location) and RM 6.7, west (Willamette Cove)	1.2 and 2.3	1%	November 2006 storm runoff event and winter 2007 high-flow event	HQs ≥ 1 (n = 4 samples) in peristaltic samples only at W005, W010, W017, and W032 (near-bottom sample); two exceedances are based on DLs; TRV is based on risk to fish and invertebrates.
PCBs					
Total PCBs	Individual samples with HQ ≥ 1 at RM 3.7 (International Slip) and RM 6.7, east (Willamette Cove)	1.1 – 1.2	1%	November 2004 low-flow event and March 2005 low-flow event	HQs ≥ 1 in peristaltic samples only at W014 and W004; TRV is based on risks to mink. No samples exceed the alternative water TRV that is based on direct exposure of aquatic organisms to surface water; therefore, there is no indication of an unacceptable benthic community risk.
Pesticides					
Total DDx	Site-wide; the highest HQs that were based on non-N-qualified data were located at RM 7.2 and RM 6.9	1.2 – 20	21%	November 2004, March 2005, and July 2005 low-flow events; November 2006 storm runoff event; and winter 2007 high-flow event	Thirty-one percent of samples with HQs ≥ 1.(n = 11 samples) are based on N-qualified data, in which HQs ranged from 1.4 to 20; HQs based on non-N-qualified data ranged from 1.1 to 9.8; TRV is based on risk to birds. One sample exceeds alternative water TRV that is protective of direct exposure of aquatic organisms to surface water (HQ = 1.8); however, this sample (W001 at RM 2.0) is N-qualified. The indication of analytical interference in the only sample that exceeded a threshold intended to be protective of organisms directly exposed to surface water suggests that no unacceptable risks to the benthic community from surface water are expected.

Table 6-36. Summary of Benthic Invertebrate Surface Water COPCs

COPC	Risk Exposure Area	HQ[a]	Percentage of All Samples with HQs ≥ 1	Sampling Event with HQs ≥ 1	Key Uncertainties
VOCs					
Ethylbenzene	Individual sample with HQ ≥ 1 at RM 6.4, west	1.6	4%	May 2005 (non-LWG sampling event)	Available VOC data are limited to a single sample with an HQ ≥ 1 at RM 6.4 on the west bank. TRV is based on risk to fish; invertebrates may be more sensitive to ethylbenzene.
Trichloroethene	Individual sample with HQ ≥ 1 at RM 6.4, west	4.1	4%	May 2005 (non-LWG sampling event)	Available VOC data are limited to a single sample with an HQ ≥ 1 at RM 6.4 on the west bank. TRV is based on risk to fish; invertebrates may be more sensitive to trichloroethene.

[a] Only HQs ≥ 1 are presented. HQs in all other water samples were < 1.

BEHP – bis(2-ethylhexyl) phthalate
COPC – contaminant of potential concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
HQ – hazard quotient
LC50 – concentration that is lethal to 50% of an exposed population
LOE – line of evidence
LWG – Lower Willamette Group
N – presumptive evidence of a compound
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
RM – river mile
TBT – tributyltin
total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT, and 4,4′-DDT)
TRV – toxicity reference value
VOC – volatile organic compound

6.6 TZW ASSESSMENT

The TZW assessment is an additional LOE that was used to estimate risks to benthic invertebrate populations and communities. This evaluation is also applicable to other ecological receptors, including benthic fish (i.e., sculpin and lamprey ammocoetes), aquatic plants, and amphibians. For the purpose of the BERA, TZW is the porewater associated with the sediment matrix within the top 38 cm[66] of the sediment column. TZW is composed of some percentage of both groundwater and surface water.

The TZW samples evaluated in this assessment were collected primarily during a 2005 sampling effort that focused on the areas offshore of nine upland sites with known or likely pathways for discharge of upland contaminated groundwater to the Study Area. Sampling locations were selected at each of the nine study sites based on results of the groundwater discharge mapping field effort. The RI Appendix C2 presents the process used to select these sites per agreement with EPA, ODEQ, and LWG. The findings of the discharge mapping effort were considered in conjunction with relevant site data (e.g., hydrogeology, surface sediment texture delineation, distribution of COIs in upland groundwater and sediments) to identify zones of possible contaminated groundwater discharge. Additional sampling locations were specified to provide comparative data for TZW quality outside of the potential discharge zones(Integral et al. 2011). Because the primary objective of the TZW sampling effort was to evaluate whether transport pathways from upland contaminated groundwater plumes to the river were complete, TZW target analyte lists varied from site to site and were derived primarily based on the COIs in the upland contaminated groundwater plumes. Therefore, not all COIs in sediments were analyzed in TZW samples. As described in Sections 4.4.3.1 and 6.1.5.2 of the draft final RI (Integral et al. 2011), there may be other groundwater plumes in the Study Area that may be discharging into river sediments where samples have not been collected.

The details of this assessment are presented as follows:

- Section 6.6.1 presents the general approach used to assess risks to benthic inveretbrates from TZW.
- Section 6.6.2 presents a summary of the TZW COPCs evaluated in the BERA. (Some COPCs were not evaluated because no toxicity thresholds were available.)
- Section 6.6.3 presents an overview of the TZW data that were used to represent exposure concentrations and uncertainties associated with TZW exposure data.

[66] This depth represents the maximum depth of a TZW sample used in the BERA evaluation. TZW samples collected by push probe were from the top 30 cm of the sediment column; samples collected via peepers were from the top 38 cm of the sediment column. The Siltronic data represent TZW in the top 31 cm of the sediment column.

Exposure concentrations in this assessment are represented by detected concentrations from all individual TZW samples. All TZW chemical concentrations are presented in Attachment 4.

- Section 6.6.4 presents a summary of the effects data. Effects thresholds (i.e., water TRVs) in this assessment are the same as the screening levels developed for the SLERA and refined screen. Details on the development of the water TRVs are presented in Attachment 10.
- Section 6.6.5 summarizes the risk characterization results.

Figure 6-21 presents a flowchart showing organization of the TZW evaluation.

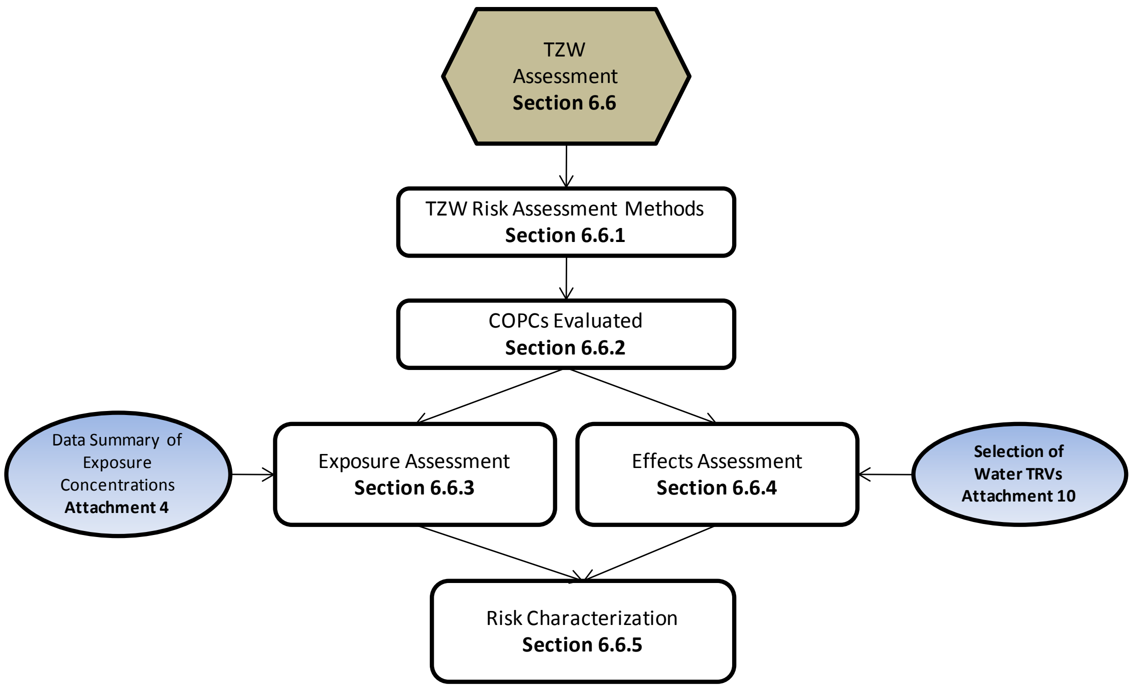


Figure 6-21. Overview of TZW Section Organization

6.6.1 TZW Risk Assessment Methods

TZW HQs were calculated by comparing COPC concentrations in individual TZW samples to chronic water TRVs developed based on a hierarchy of water quality criteria and literature-based TRVs in accordance with the EPA's Problem Formulation (Attachment 2) for surface water. Potentially unacceptable risks were identified based on those COPCs that resulted in HQs ≥ 1. Exposure data, effects data, and the quantitative risk results (i.e., magnitude, spatial distribution, and frequency of HQs) are presented in the following sections. Results of the TZW LOE are integrated with those from the other LOEs in the for benthic invertebrate risk conclusions (Section 6.7).

6.6.2 COPCs Evaluated

Fifty-four of the 58 TZW COPCs identified in the SLERA and refined screen (Attachment 5) are evaluated in the BERA. Four individual DDT metabolites identified in the SLERA (2,4′-DDD, 2,4′-DDT, 4,4′-DDD, and 4,4′-DDE) were evaluated as part of total DDx and were not evaluated individually; 4,4′-DDT was evaluated both individually and as total DDx because the TRV for DDx is based on 4,4′-DDT. Table 6-37 presents the detected COPCs by area.

Table 6-37. COPCs in TZW by Area

	Area									
		Arkema								
COPC	**ARCO**	**Acid Plant Area**	**Chlorate Plant Area**	**Mobil Oil**	**Gasco**	**Gunderson**	**Kinder Morgan**	**Rhône-Poulenc**	**Siltronic**	**Willbridge**
Metals										
Barium (total) [a]	X	X	X	X	X	X	X	X	X	X
Beryllium (total) [a]				X				X		X
Cadmium (dissolved)[b]	X		X	X			X	X		X
Cobalt (total) [a]									X	
Copper (dissolved) [b]								X		
Iron (total) [a]	X	X	X	X	X	X	X	X	X	X
Lead (dissolved) [b]				X	X			X		
Magnesium (total) [a]		X	X					X		
Manganese (total) [a]	X	X	X	X	X	X	X	X	X	X
Nickel (dissolved) [b]			X		X			X		
Potassium (total) [a]			X							
Sodium (total) [a]		X	X							
Vanadium (total) [a]									X	
Zinc (dissolved) [b]	X									
PAHs										
2-Methylnaphthalene	X				X				X	
Acenaphthene	X				X				X	
Anthracene	X				X				X	

Table 6-37. COPCs in TZW by Area

	Area									
		Arkema								
COPC	ARCO	Acid Plant Area	Chlorate Plant Area	Mobil Oil	Gasco	Gunderson	Kinder Morgan	Rhône-Poulenc	Siltronic	Willbridge
Benzo(a)anthracene	X			X	X		X		X	X
Benzo(a)pyrene	X			X	X		X		X	X
Benzo(b)fluoranthene	X				X				X	
Benzo(g,h,i)perylene	X			X	X				X	
Benzo(k)fluoranthene	X				X				X	
Chrysene	X				X				X	
Dibenzo(a,h)anthracene	X				X				X	
Fluoranthene	X				X				X	
Fluorene	X			X	X				X	
Indeno(1,2,3-cd)pyrene	X			X	X				X	
Naphthalene	X	X			X				X	
Phenanthrene	X			X	X				X	
Pyrene	X				X				X	
SVOCs										
1,2-Dichlorobenzene								X		
1,4-Dichlorobenzene								X		
Dibenzofuran					X				X	

Table 6-37. COPCs in TZW by Area

	Area									
		Arkema								
COPC	**ARCO**	**Acid Plant Area**	**Chlorate Plant Area**	**Mobil Oil**	**Gasco**	**Gunderson**	**Kinder Morgan**	**Rhône-Poulenc**	**Siltronic**	**Willbridge**
Pesticides										
4,4′-DDT		X						X		
Total DDx		X						X		
VOCs										
1,1-Dichloroethene									X	
1,2,4-Trimethylbenzene									X	
1,3,5-Trimethylbenzene									X	
Benzene					X				X	
Carbon disulfide		X							X	
Chlorobenzene		X						X		
Chloroethane						X				
Chloroform		X	X							
cis-1,2-Dichloroethene									X	
Ethylbenzene					X				X	
Isopropylbenzene					X				X	
m,p-Xylene									X	
o-Xylene					X				X	
Toluene					X				X	

Table 6-37. COPCs in TZW by Area

COPC	Area									
		Arkema								
	ARCO	Acid Plant Area	Chlorate Plant Area	Mobil Oil	Gasco	Gunderson	Kinder Morgan	Rhône-Poulenc	Siltronic	Willbridge
Total xylenes					X				X	
Trichloroethene									X	
Petroleum Hydrocarbons										
Gasoline-range hydrocarbons [c]	X			X	X		X		X	X
Other Contaminants										
Cyanide					X				X	
Perchlorate		X	X							

[a] Criteria are based on total concentration.

[b] Criteria are based on dissolved concentration.

[c] Gasoline-range hydrocarbons were evaluated as five components (aliphatic hydrocarbons C4-C6, aliphatic hydrocarbons C6-C8, aliphatic hydrocarbons C8-C10, aliphatic hydrocarbons C10-C12, and aromatic hydrocarbons C8-C10). Gasoline-range hydrocarbons was identified as an exceedance if any one of the five gasoline components exceeded its TRV.

COPC – contaminant of potential concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
PAH – polycyclic aromatic hydrocarbon
SVOC – semivolatile organic compound
total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT, and 4,4′-DDT)
TZW – transition zone water
VOC – volatile organic compound

TZW COIs that could not be evaluated because no toxicological data were available to allow development of water TRVs are listed in Table 6-38. The risks to benthic receptors associated with exposure to these contaminants in TZW are therefore unknown. TRVs were unavailable for individual dioxins and furans other than 2,3,7,8-TCDD. For vertebrates, dioxins and furans are evaluated as a toxicity-weighted sum based on the toxicity of each congener relative to 2,3,7,8-TCDD, using TEFs based on their common mechanism for toxicity. Because TEFs are not available for benthic invertebrates, no individual dioxin or furan (other than 2,3,7,8-TCDD) or dioxin group total could be evaluated.

Table 6-38. TZW COIs Without Screening-Level Benchmarks

Metals	
Aluminum	Titanium
Calcium	
Dioxins/Furans	
1,2,3,4,6,7,8-Heptachlorodibenzo-*p*-dioxin	1,2,3,7,8-Pentachlorodibenzofuran
1,2,3,4,7,8,9-Heptachlorodibenzofuran	2,3,4,7,8-Pentachlorodibenzofuran
1,2,3,4,7,8-Hexachlorodibenzofuran	2,3,7,8-Tetrachlorodibenzofuran
1,2,3,6,7,8-Hexachlorodibenzofuran	
Petroleum Hydrocarbons	
Residual-range hydrocarbons	Total diesel-residual hydrocarbons
Diesel-range hydrocarbons	Total petroleum hydrocarbons

COI – contaminant of interest
SLERA – screening-level ecological risk assessment
TZW – transition zone water

By agreement with EPA, aluminum was not identified as a COPC because its AWQC was developed using toxicity data from acidic waters and is not applicable to the circumneutral waters of the Study Area. Aluminum concentrations in background surface water and sediment were evaluated to determine whether a local source of aluminum is present within the Study Area (Section 6.5.5.3.).

In addition, two TZW COIs were not retained as COPCs because no detected concentrations exceeded TRVs (although at least one DL exceeded a TRV): selenium and styrene (see Table 5-2).

6.6.3 Exposure Assessment

This section presents the exposure assessment of the TZW COPCs. An overview of the Study Area TZW sampling program is presented in Section 6.6.3.1, TZW EPCs for benthic invertebrates are defined in Section 6.6.3.2, and uncertainties associated with the exposure data are discussed in Section 6.6.3.3.

6.6.3.1 Overview of TZW Collected from the Study Area

RI Appendix C2 presents the process used by EPA, ODEQ, and the LWG to select the nine TZW sampling areas. TZW sampling locations used in this assessment are presented on Map 4-16.

TZW samples evaluated for ecological exposure were limited to those collected in "shallow" sediment (≤ 38 cm below the mudline), which includes the biologically active zone (typically 10 to 20 cm deep based on sediment profiling imaging data collected in 2001 (SEA 2002)). LWG TZW samples were collected using a Trident® push probe, a Geoprobe®, and small-volume diffusion sampler ("peeper") between October 3 and December 2, 2005. This sampling period corresponded with low river stage and was anticipated to represent higher groundwater discharge to the river. Trident® samples were processed to represent both whole water and dissolved concentrations; Geoprobe® samples represented only whole water. Peeper samples were collected through a 5-µm membrane and are similar to dissolved samples but may contain colloids or very fine particulates. Additional offshore groundwater samples were collected during a non-LWG sampling event in May and June 2005 using the Geoprobe® sampling method. Sampling locations (Map 4-16) were selected along each of the properties (see Table 6-37) based on results of the groundwater discharge mapping field effort (Integral 2006a).

Uncertainty Associated with TZW Sampling Methods

TZW was sampled adjacent to nine upland sites with known or likely pathways for discharge of upland contaminated groundwater to the Study Area (Integral 2006a) and are meant to characterize those areas of likely contaminated groundwater impacts. The TZW data are not representative of, nor should they be used to infer large-area or site-wide risks to benthic fish, benthic invertebrates, aquatic plants, and amphibians from COPCs in TZW (Integral et al. 2011),. TZW target analyte lists varied from site to site and were derived primarily based on the COIs in the upland contaminated groundwater plumes. Therefore, not all COIs in sediments were analyzed in TZW samples. As described in Sections 4.4.3.1 and 6.1.5.2 of the draft final RI (Integral et al. 2011), there may be other groundwater plumes in the Study Area that may be discharging into river sediments where TZW samples have not been collected.

Risks associated with exceedance of TRVs based on comparison with whole-water sample results are also uncertain. Many organic compounds have a very high affinity to organic particulate material. Whole-water samples contain colloidal and larger particulates that may bind organic chemicals. The comparison of dissolved and total concentrations of PAHs, other SVOCs, and DDx compounds suggest that concentrations of many organics are not measurably present (i.e., are below DLs) or do not exceed their associated TRVs when evaluated on a dissolved basis.

6.6.3.2 TZW EPCs

TZW EPCs in this assessment were represented by concentrations in all individual TZW samples collected in the Study Area regardless of sampling method or depth.[67] TZW concentrations were compared to water TRVs. A summary of the chemicals detected in shallow TZW and the range of concentrations is provided in Table 6-39. All TZW data, by site, are presented in Attachment 4.

[67] All TZW samples evaluated in this BERA were within the 0- to 38-cm depth; however, the depth of the different sampling equipment used to collect TZW (i.e., using peeper, Trident® probe, and Geoprobe) varied.

Table 6-39. Summary of TZW Exposure Data

Analyte	Unit	Detection Frequency	Percent Detected	Detected Concentration			DL Range
				Minimum	Maximum	Mean	
Metals							
Barium (total)	µg/L	93 of 93	100%	4.06	4,390	350	NA
Beryllium (total)	µg/L	52 of 93	56%	0.012 JT	1.34	0.19	0.006 – 0.047
Cadmium (dissolved)	µg/L	36 of 55	65%	0.006 J	0.52	0.092	0.002 – 0.158
Cobalt (total)	µg/L	8 of 13	62%	6.1	82	26	5 – 5
Copper (dissolved)	µg/L	9 of 45	20%	0.36 T	3.63	1.2	0.08 – 0.77
Iron (total)	µg/L	106 of 106	100%	173	252,000	50,000	NA
Lead (dissolved)	µg/L	20 of 55	36%	0.01 J	1.61	0.34	0.01 – 0.195
Magnesium (total)	µg/L	106 of 106	100%	1,020	578,000	45,000	NA
Manganese (total)	µg/L	106 of 106	100%	111	66,200 T	5,000	NA
Nickel (dissolved)	µg/L	53 of 55	96%	0.3 J	25.5	4.9	1.5 – 20
Potassium (total)	µg/L	85 of 93	91%	199 J	197,000 JT	9,300	1,000 – 4,410
Sodium (total)	µg/L	91 of 93	98%	3,110 T	37,490,000	1,700,000	1,390 – 2,400
Vanadium (total)	µg/L	9 of 13	69%	11.6	379	91.1	10 – 10
Zinc (dissolved)	µg/L	30 of 55	55%	0.95 T	526	22	0.75 – 6.07
PAHs							
2-Methylnaphthalene	µg/L	27 of 89	30%	0.082	84	13	0.0034 – 0.065
Acenaphthene	µg/L	96 of 102	94%	0.0031 J	399	28	0.0024 – 0.069
Anthracene	µg/L	75 of 102	74%	0.0027 J	63.8	3.4	0.0013 – 0.25

Table 6-39. Summary of TZW Exposure Data

Analyte	Unit	Detection Frequency	Percent Detected	Detected Concentration Minimum	Detected Concentration Maximum	Detected Concentration Mean	DL Range
Benzo(a)anthracene	µg/L	44 of 102	43%	0.0046 J	32.3	2.3	0.0024 – 0.2
Benzo(a)pyrene	µg/L	38 of 102	37%	0.0025 J	37.8	3	0.0018 – 0.062
Benzo(b)fluoranthene	µg/L	31 of 102	30%	0.0042 J	33.3	3.2	0.0022 – 0.064
Benzo(g,h,i)perylene	µg/L	40 of 102	39%	0.0069 J	28.8	2.1	0.0041 – 0.0483
Benzo(k)fluoranthene	µg/L	27 of 102	26%	0.004 J	9	1.2	0.0015 – 0.059
Chrysene	µg/L	46 of 102	45%	0.0033 J	34.5	2.3	0.0014 – 0.3
Dibenzo(a,h)anthracene	µg/L	29 of 102	28%	0.0024 J	3.71	0.38	0.0018 – 0.0483
Fluoranthene	µg/L	67 of 102	66%	0.013 J	106	6.9	0.011 – 1.7
Fluorene	µg/L	82 of 102	80%	0.0075 J	108	11	0.0031 – 0.72
Indeno(1,2,3-cd)pyrene	µg/L	39 of 102	38%	0.0046 J	16.9	1.5	0.0023 – 0.0483
Naphthalene	µg/L	74 of 169	44%	0.048	13,700	1,300	0.0063 – 15
Phenanthrene	µg/L	70 of 102	69%	0.044	362	26	0.004 – 1.1
Pyrene	µg/L	72 of 102	71%	0.017 J	148	8.5	0.0099 – 4.3
SVOCs							
1,2-Dichlorobenzene	µg/L	15 of 136	11%	0.14 J	640	65	0.12 – 5.6
1,4-Dichlorobenzene	µg/L	13 of 128	10%	0.14 J	240	27	0.12 – 0.3
Dibenzofuran	µg/L	51 of 89	57%	0.013 J	8	0.92	0.0079 – 0.042
Pesticides							
2,4'-DDD	µg/L	10 of 14	71%	0.011 J	1.1 J	0.22	0.0033 – 0.004
2,4'-DDT	µg/L	3 of 14	21%	0.0078 NJ	0.093 J	0.037	0.00089 – 0.15

Table 6-39. Summary of TZW Exposure Data

Analyte	Unit	Detection Frequency	Percent Detected	Detected Concentration			DL Range
				Minimum	Maximum	Mean	
4,4'-DDD	µg/L	6 of 14	43%	0.015 J	1.3	0.45	0.0047 – 0.15
4,4'-DDE	µg/L	3 of 14	21%	0.015 J	0.12 J	0.059	0.0039 – 0.93
4,4'-DDT	µg/L	3 of 14	21%	0.84	1.8	1.2	0.005 – 0.15
Total DDx	µg/L	10 of 14	71%	0.049 JT	3.1 JT	0.98	0.0077 – 0.016
VOCs							
1,1-Dichloroethene	µg/L	12 of 136	9%	0.18 J	40.5	8.1	0.13 – 6.1
1,2,4-Trimethylbenzene	µg/L	17 of 41	41%	1.05	69.9	22	1 – 1
1,3,5-Trimethylbenzene	µg/L	16 of 41	39%	0.33	21.6	6.4	0.3 – 0.3
Benzene	µg/L	59 of 136	43%	0.19 J	3,840	150	0.14 – 0.46
Carbon disulfide	µg/L	8 of 136	6%	0.16 J	800	110	0.16 – 8
Chlorobenzene	µg/L	35 of 136	26%	0.15 J	12,000	360	0.14 – 0.27
Chloroethane	µg/L	8 of 136	6%	0.23 J	160	23	0.2 – 12
Chloroform	µg/L	9 of 136	7%	0.14 J	580	120	0.14 – 130
cis-1,2-Dichloroethene	µg/L	44 of 136	32%	0.12 J	67,000	2,100	0.12 – 0.24
Ethylbenzene	µg/L	41 of 136	30%	0.13 J	416	44	0.13 – 6.5
Isopropylbenzene	µg/L	33 of 136	24%	0.11 J	14.5	3.7	0.11 – 5.3
Toluene	µg/L	66 of 136	49%	0.23 J	178	8.6	0.11 – 5.4
Trichloroethene	µg/L	20 of 136	15%	0.14 J	88,500	4,400	0.14 – 0.67
m,p-Xylene	µg/L	45 of 136	33%	0.22 J	293	18	0.22 – 11
o-Xylene	µg/L	53 of 136	39%	0.11 J	150	12	0.11 – 5.1

Table 6-39. Summary of TZW Exposure Data

Analyte	Unit	Detection Frequency	Percent Detected	Detected Concentration Minimum	Detected Concentration Maximum	Detected Concentration Mean	DL Range
Total xylenes	µg/L	58 of 136	43%	0.22 JT	440 T	25	0.22 – 11
Petroleum Hydrocarbons							
Gasoline-range hydrocarbons[a]	mg/L	29 of 57	51%	0.013 JT	4.0 J	0.53	0.013 – 0.17
Conventionals							
Cyanide	µg/L	32 of 34	94%	0.006 J	23.1 J	1.5	0.01 – 1.4
Perchlorate	µg/L	11 of 21	52%	105 T	177,000	47,600	0.4 – 2,0000

[a] Gasoline-range hydrocarbons were evaluated as five components (aliphatic hydrocarbons C4-C6, aliphatic hydrocarbons C6-C8, aliphatic hydrocarbons C8-C10, aliphatic hydrocarbons C10-C12, and aromatic hydrocarbons C8-C10). Gasoline-range hydrocarbons was identified as an exceedance if any one of the five gasoline components exceeded its TRV. Gasoline-range hydrocarbons were not included in the final count of COPCs but are discussed in the evaluation of uncertainties associated with TZW risk.

DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
DL – detection limit
J – estimated concentration
N – presumptive evidence of a compound
NA – not applicable
PAH – polycyclic aromatic hydrocarbon
SVOC – semivolatile organic compound
T – value calculated or selected from multiple results
total DDx – sum of all six DDT isomers (2,4′-DDD; 4,4′-DDD; 2,4′-DDE; 4,4′-DDE; 2,4′-DDT; and 4,4′-DDT)
TZW – transition zone water
VOC – volatile organic compound

6.6.3.3 Uncertainty Associated with Ecological Exposure to TZW

The degree to which the collected TZW samples characterize exposure of organisms living in or on the sediment to TZW is a key uncertainty associated with the ecological evaluation of TZW. Samples were collected from nearshore areas of the river with known or likely pathways for discharge of upland contaminated groundwater to the Study Area. Because these areas include potential habitat for benthic invertebrates, benthic fish, amphibians, and aquatic plants, TZW is considered a complete and significant pathway for these receptors in the CSM for Portland Harbor. Although these organisms reside in the sediment column or are in contact with the sediment surface, the water column rather than the sediment matrix is thought to provide more exposure to contaminants (Hare et al. 2001). The proportion each matrix contributes to the exposure is influenced by how (and if) an organism irrigates its tube for respiration and waste removal, where in the sediment column it lives, its diet, and the configuration and construction of the burrow or tube wall. Lumbriculid and tubificid worms feed head down and take up oxygen by extending their tails into the overlying water column, with no need to irrigate their blind-ended burrows. These species experience greater exposure to porewater than most other benthic organisms, whose exposure is minimized because their tubes or burrows are flushed for purposes of respiration, feeding and waste elimination.

The TZW samples evaluated represent a sediment layer that is deeper than that typically used by benthic organisms. TZW was collected in the top 30 to 38 cm (depending on sampler type), well beyond the biologically active zone of 0 to 20 cm below the mudline. SPI data in the vicinity of the TZW sampling locations suggest that the local biologically active zone is shallower. A maximum depth of 5 cm was measured for the apparent redox potential discontinuity (aRPD) layer (an estimate of the oxygenated layer where the majority of the benthic organisms reside) and a maximum invertebrate feeding void depth of 11.8 cm was observed in SPI sampling locations in the vicinity of the TZW samples.

TZW below the thin (several mm to several cm) oxygenated zone at the sediment-water interface is essentially uninhabitable because it lacks oxygen and typically has low food content (Arnot and Gobas 2004). In addition, porewater below the oxygenated zone is often toxic to burrowing organisms because the decomposing organic material on the bottom releases products that result in the formation of hydrogen sulfide and ammonia (Forbes et al. 1998). Ammonia and sulfides were identified as potentially contributing to benthic invertebrate toxicity observed in Portland Harbor in the development of site-specific SQVs (see Section 6.2).

Burrowing organisms that live below the oxic zone have adaptations that introduce oxygenated overlying water into their tubes or burrows for both respiration and feeding, essentially extending the sediment-water interface into the sediment column (Lee and Swartz 1980). These behaviors result in reduced exposure to porewater. Mechanisms to modify the sediment environment and behaviors that increase the interchange with overlying water vary by species. Burrowing and tube-dwelling organisms actively pump overlying water into their burrows or tubes through the rhythmic beating of pleopods (e.g., crayfish) or cilia, or though body undulations or peristaltic contractions (e.g., some soft-bodied worms) (Riisgard and Larsen 2005). Some burrowing organisms construct a U-shaped burrow or tube with one opening at a slightly higher elevation than the other; this slight difference in height creates a passive flow-through system that minimizes the metabolic energy required to flush their tubes or burrows (Vogel 1994). The entrainment of overlying water into tubes and burrows oxygenates not only the tube or burrow but the sediment surrounding the tube or burrow (Satoh et al. 2007). The presence of oxygen fosters the growth of bacteria, fungi, algae, and protozoa on the walls of the tube and within the adjacent sediment. These biofilms decrease the infiltration of the surrounding porewater into the tube or burrow by decreasing or filling the interstitial spaces. Many organisms living on or in the sediment also secrete mucus to protect soft body parts from abrasion by sediment particles. The mucus may also reduce the transport of contaminants in the porewater across the body wall. This effect is visible in many sediment profile images that show oxygenated sediments outlining burrows extending below the aRPD zone.

Feeding strategy also affects exposure to porewater contaminants. Filter-feeding organisms depend on the flow of water to gather food, extending specialized appendages or structures into the water column to trap particles. For example, a filter-feeding bivalve extends its siphon above the sediment surface and pumps overlying water across the gills, through the mantle cavity, and out the siphon. This action limits exposure to porewater, while supporting both feeding and respiration. Pumping rates can vary by orders of magnitude (2 mL to 6.5 L/individual/hour) depending on the species (Lee and Swartz 1980). Clams account for some of the higher pumping rates, which result in very high dilution of porewater. *Macoma* clams are estimated to ventilate about 10% porewater, even when their siphons are retracted inside their burrows (Winsor et al. 1990). Organisms that feed on organic material below the sediment surface tend to increase the porosity of the sediments, which increases the exchange of porewater with overlying water (Winsor et al. 1990; Krantzberg 1985, as cited in Rasmussen et al. 2000), and which, in turn, dilutes the porewater.

The biological activity of benthic invertebrates can also enhance the exchange of porewater with overlying water by increasing the roughness of the sediment surface. For example, tube openings, a pile of material pushed out of a burrow, and fecal castings make small changes to the surface sediment profile. As water flows over the sediment surface, these topographic features create changes in velocity (and associated pressure fields) at the benthic boundary layer (interface between flowing water and bottom surface) that cause surface water to be entrained into the sediment (Huettel and Rusch 2000; Hoffman 2005; Precht and Huettel 2003).

As a result of these strategies, burrowing organisms have relatively low exposure to porewater compared to surface water. Organisms that live on (versus in) the sediment surface (or are closely tied to the surface) experience even less exposure to porewater.

For the foregoing reasons, the representativeness of the COPC concentrations in porewater within the top 38 cm layer of sediment for purpose of estimating exposure and subsequent risks to benthic organisms is highly questionable.

6.6.4 Effects Assessment

TZW chemical concentrations were compared to the effects thresholds as part of the risk characterization process. At the direction of EPA (2008f), chronic water TRVs were developed for all TZW COPCs through a review of WQS, criteria, published benchmarks, and toxicity data. The hierarchy detailed in Attachment 10 was used when selecting TRVs, all of which were approved by EPA for use in the BERA. Criteria for metals COPCs were hardness-adjusted when appropriate. For individual metals, if the published criteria were based on dissolved concentrations, then the dissolved sample result was compared to the dissolved criterion; otherwise the total concentration for both the sample and criterion were used. Table 6-40 presents the TRVs for all TZW COPCs and their sources. These values were developed based on the sensitivities of fish and invertebrate species and are considered protective of all aquatic receptors, including benthic invertebrates.

Table 6-40. TRVs for TZW COPCs

COPC	TRV (µg/L)	Source
Metals		
Barium	4[a]	Tier II
Beryllium	0.66[a]	Tier II
Cadmium	0.09[b]	AWQC
Cobalt	23[a]	Tier II
Copper	2.74[b]	AWQC
Iron	1,000[a]	AWQC
Lead	0.54[b]	AWQC
Magnesium	82,000[a]	AWQC
Manganese	120[a]	Tier II
Nickel	16.1[b]	AWQC
Potassium	53,000[a]	Tier II
Sodium	680,000[a]	Tier II
Vanadium	20[a]	Tier II
Zinc	36.5[b]	AWQC
PAHs		
2-Methylnaphthalene	2.1	Tier II
Acenaphthene	23	Tier II
Anthracene	0.73	Tier II

Table 6-40. TRVs for TZW COPCs

COPC	TRV (µg/L)	Source
Benzo(a)anthracene	0.027	Tier II
Benzo(a)pyrene	0.014	Tier II
Benzo(b)fluoranthene	0.6774	EPA (2003c)[c]
Benzo(g,h,i)perylene	0.4391	EPA (2003c)[c]
Benzo(k)fluoranthene	0.6415	EPA (2003c)[c]
Chrysene	2.042	EPA (2003c)[c]
Dibenzo(a,h)anthracene	0.2825	EPA (2003c)[c]
Fluoranthene	6.16	Tier II
Fluorene	3.9	Tier II
Indeno(1,2,3-cd)pyrene	0.275	EPA (2003c)[c]
Naphthalene	12	Tier II
Phenanthrene	6.3	Tier II
Pyrene	10.11	EPA (2003c)[c]
VOCs		
1,1-Dichloroethene	25	Tier II
1,2,4-Trimethylbenzene	7.3[d]	Tier II
1,3,5-Trimethylbenzene	7.3[d]	Tier II
Benzene	130	Tier II
Carbon disulfide	0.92	Tier II
Chlorobenzene	64	ODEQ
Chloroethane	47[f]	Tier II
Chloroform	28	Tier II
cis-1,2-Dichloroethene	590	Tier II
Ethylbenzene	7.3	Tier II
Isopropylbenzene	7.3[d]	Tier II
m,p-Xylene	66.67	EPA (2006c)
o-Xylene	13[g]	Tier II
Toluene	9.8	Tier II
Total xylenes	13[g]	Tier II
Trichloroethene	47	Tier II
SVOCs		
1,2-Dichlorobenzene	14	Tier II
1,4-Dichlorobenzene	15	Tier II
Dibenzofuran	3.7	Tier II

Table 6-40. TRVs for TZW COPCs

COPC	TRV (µg/L)	Source
Pesticides[h]		
4,4′-DDT	0.011 (0.001)[i]	Alternative TRV[i] (EPA (2006d))
Total DDx	0.011 (0.001)[i]	Alternative TRV[i] (EPA (2006d))
Petroleum Hydrocarbons		
Gasoline-range hydrocarbons[j]	NV	EPA (2008f)
Aliphatic hydrocarbons C4-C6[k]	128[j]	EPA (2008f)
Aliphatic hydrocarbons C6-C8[k]	54[j]	EPA (2008f)
Aliphatic hydrocarbons C8-C10[k]	9.5[j]	EPA (2008f)
Aliphatic hydrocarbons C10-C12[k]	2.6[j]	EPA (2008f)
Aromatic hydrocarbons C8-C10[k]	212[j]	EPA (2008f)
Other Contaminants		
Perchlorate	9,300[e]	Dean et al. (2004) per EPA (2008f)
Cyanide	5.2	AWQC

a TRV is based on total criterion; TRV was compared to total concentration detected in Study Area.

b TRV is based on dissolved criterion; TRV was compared to dissolved concentration detected in Study Area.

c TRV is based on PAH mixtures.

d TRV is based on criteria for ethylbenzene.

e An ACR of 8.3 was used to calculate a chronic screening value from an acute screening value when no chronic data were available, per agreement with EPA (2008c).

f TRV is based on criteria for 1,1-dichloroethane.

g TRV is based on criteria for xylene.

h TRVs for total DDx and DDT metabolites are based on criteria for 4,4′-DDT.

i Two TRVs were evaluated for DDx and DDT metabolites. The selected TRV was derived based only on aquatic organisms (see text above for details on derivation of the selected TRVs). HQs were also derived using EPA's TRV based on AWQC; however, this AWQC is protective of wildlife via ingestion of contaminated prey, which is not appropriate for evaluating direct exposure of aquatic organisms to TZW contaminants.

j EPA provided TRVs for five of the chemical groups that are blended to form gasoline (EPA 2008a). Because these fractions were not quantified in Study Area samples, the average fraction of these components in gasoline was used to convert the total gasoline-range hydrocarbon concentration into gasoline fraction concentrations for comparison with the TRVs. Average fractions were derived from the literature (Fagerlund and Niemi 2003).

k Gasoline components were used as a surrogate for gasoline-range hydrocarbons. If any one component exceeded its TRV, then gasoline-range hydrocarbons was identified as an exceedance.

ACR – acute-to-chronic ratio
AWQC – ambient water quality criteria
COPC – contaminant of potential concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
EPA – US Environmental Protection Agency
HQ – hazard quotient
NV – no value
PAH – polycyclic aromatic hydrocarbon
SVOC – semivolatile organic compound
total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT, and 4,4′-DDT)
TRV – toxicity reference value
TZW – transition zone water
VOC – volatile organic compound

As noted in Section 6.5.4, because the AWQC for 4,4′-DDT is based on protection of birds, risk estimates for aquatic receptors based on this TRV are associated with substantial uncertainty. An alternative TRV protective of aquatic organisms (including benthic invertebrates) was developed in this BERA, using methods consistent with those used for AWQC derivation. The alternative water TRV for DDx compounds was calculated as 0.011 µg/L. This alternative TRV is the appropriate metric for evaluating direct exposure of aquatic organisms and was used as the basis for risk characterization.

Uncertainties Associated with Effects Data

TRVs were selected from regulatory standards (state WQS) and criteria (national AWQC), as well as other published effects thresholds (e.g., Tier II, lowest chronic value [LCV] from Suter and Tsao (1996)) following an agreed-upon hierarchy (see Attachment 10). Where available, the TRVs are based on WQS or AWQC and are assumed to have less uncertainty than TRVs based on other sources, although it is also important to take into account the relevance of the determinative receptor and pathway for each TRV. As an example, the chronic DDT AWQC (0.001 µ/L) was selected to be protective of brown pelican reproduction via a fish ingestion pathway. A criterion derived for the protection of invertebrates from direct contact with water using data included in the DDT AWQC document would be 1 to 2 orders of magnitude higher.

The LCVs were most often applied when regulatory values were not available. TRVs for sodium, potassium, and magnesium were based on LCVs, which were derived from toxicity data for a daphnid (a water column species) and may not accurately characterize effects on benthic invertebrates, fish, amphibians, and plants. In the case of sodium and potassium, the Tier II value was cited by Suter and Tsao (1996) as being below commonly occurring ambient values and not appropriate for use as a screening value. In addition, TRVs based on LCVs may inaccurately estimate risks to benthic receptors because these values are based on a limited number of studies and species.

The TRVs for four VOCs (i.e., 1,2,4-trimethylbenzene, 1,3,5-trimethylbenzene, isopropylbenzene, and chloroethane) are uncertain because they are based on surrogates (ethylbenzene and 1,1-dichloroethane). No toxicological data were available for these COPCs, and the surrogate TRVs may over- or underestimate toxicity to benthic invertebrates. The TRV for perchlorate is uncertain because it was calculated from an acute value using an estimated relationship between acute and chronic responses.

The AWQC that was the source of the iron TRV is based on a site receiving acid mine drainage, and derivation of the AWQC was not consistent with later methods for deriving criteria (Suter and Tsao 1996).

6.6.5 Risk Characterization

This section presents the risk estimates for invertebrates based on the TZW LOE. An HQ calculation was used to quantify TZW risk estimates. HQs were derived for all COPCs using Equation 6-1. The EPC and TRV are represented by TZW chemical concentrations. The following subsections present the risk characterization results and discussion.

6.6.5.1 Risk Characterization Results

Individual HQs were calculated for all COPCs across all TZW samples. Summary results for each of the nine areas are tallied in Table 6-41 and presented graphically in Figures 6-22 through 6-28.

- **ARCO** – At the ARCO site, seven COPCs have HQs ≥ 1 based on detected concentrations, including four metals (i.e., barium, iron, manganese, zinc), two

PAHs (benzo(a)anthracene, benzo(a)pyrene), and gasoline-range hydrocarbons.[68] Of these COPCs, metals HQs are generally the highest, specifically iron and barium. One COPC (cadmium) has an HQ ≥ 1 in two samples with DLs greater than the TRV; detected concentrations did not exceed the TRV.

- **Arkema Facility's Acid Plant Area –** At the Arkema facility's acid plant area, 10 COPCs have HQs ≥ 1 based on detected concentrations: five metals (barium, iron, magnesium, manganese, and sodium), one PAH (naphthalene), 4,4′-DDT, total DDx, chlorobenzene, and chloroform. HQs are highest for barium and total DDx , although total DDx HQs are based on whole-water samples (i.e., including particulate material).[69] Two COPCs (i.e., carbon disulfide and perchlorate) were not detected in this area, but each had a DL greater than its TRV.
- **Arkema Facility's Chlorate Plant Area –** Ten COPCs have HQs ≥ 1 at the facility's chlorate plant area based on detected concentrations: eight metals (barium, cadmium, iron, magnesium, manganese, nickel, potassium, and sodium), chloroform, and perchlorate. Barium and manganese HQs are the highest. This is the only area where perchlorate was detected.
- **Mobil Oil –** Thirteen COPCs have HQs ≥ 1 at the Mobil Oil site based on detected concentrations: six metals (barium, beryllium, cadmium, iron, lead, and manganese), six PAHs (benzo(a)anthracene, benzo(a)pyrene, benzo(g,h,i)perylene, fluorene, indeno(1,2,3-cd)pyrene, and phenanthrene), and gasoline-range hydrocarbons.[70] Barium, iron, manganese, and gasoline-range hydrocarbons HQs are the highest.
- **Gasco –** At the Gasco site, 31 COPCs have HQs ≥ 1 in TZW: five metals (barium, iron, lead, manganese, and nickel), 16 PAHs (2-methylnaphthalene, acenaphthene, anthracene, benzo(a)anthracene, benzo(a)pyrene, benzo(b)fluoranthene, benzo(g,h,i)perylene, benzo(k)fluoranthene, chrysene, dibenzo(a,h)anthracene, fluoranthene, fluorene, indeno(1,2,3-cd)pyrene, naphthalene, phenanthrene, and pyrene), one SVOC (dibenzofuran), seven VOCs (benzene, carbon disulfide, ethylbenzene, isopropylbenzene, o-xylene, toluene, and total xylenes), cyanide, and gasoline-range hydrocarbons.[71] Of the PAHs, naphthalene, benzo(a)pyrene, and benzo(a)anthracene have the highest HQs.

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[68] The HQs for both gasoline-range aliphatic hydrocarbons C4 through C6 and gasoline-range aliphatic hydrocarbons C10 through C12 were ≥ 1; however, these were counted as one COPC in the tally.

[69] Dissolved concentrations tended to be several orders of magnitude lower and most constituents of DDx were undetected. However, detected concentrations and DLs exceeded the TRV.

[70] The HQs for both gasoline-range aliphatic hydrocarbons C4 through C6 and gasoline-range aliphatic hydrocarbons C10 through C12 were ≥ 1; however, these were counted as one COPC in the tally.

[71] The HQs for several fractions of gasoline-range aliphatic hydrocarbons were ≥ 1; however, these were counted as one COPC in the tally.

VOC HQs are highest for ethylbenzene and total xylenes. The cyanide HQ is higher than that of any other COPCs at the Gasco site. Gasco is one of two TZW sampling sites where cyanide was detected (Siltronic is the other).

- **Gunderson –** Four COPCs have HQs ≥ 1 at the Gunderson site based on detected concentrations: three metals (barium, iron, and manganese), and one VOC (chloroethane). The maximum HQ for the site is for iron.
- **Kinder Morgan –** At Kinder Morgan, five COPCs have HQs ≥ 1: three metals (barium, iron, and manganese), one PAH (benzo(a)anthracene), and gasoline-range hydrocarbons. Maximum HQs are highest for manganese and gasoline-range hydrocarbons.
- **Rhône-Poulenc –** Thirteen COPCs have HQs ≥ 1 at the Rhône-Poulenc site: nine metals (barium, beryllium, cadmium, copper, iron, lead, magnesium, manganese, and nickel), two SVOCs (1,2-dichlorobenzene, 1,4-dichlorobenzene), and one VOC (chlorobenzene). Barium, manganese, and iron have the highest maximum HQs.
- **Siltronic –** Thirty-seven COPCs have HQs ≥ 1 at the Siltronic site, including five metals (barium, cobalt, iron, manganese, and vanadium), 16 PAHs (2-methyl-naphthalene, acenaphthene, anthracene, benzo(a)anthracene, benzo(a)pyrene, benzo(b)fluoranthene, benzo(g,h,i)perylene, benzo(k)fluoranthene, chrysene, dibenzo(a,h)anthracene, fluoranthene, fluorene, indeno(1,2,3-cd)pyrene, naphthalene, phenanthrene, and pyrene), one SVOC (dibenzofuran), 13 VOCs (1,1-dichloroethene, 1,2,4-trimethylbenzene, 1,3,5-trimethylbenzene, benzene, carbon disulfide, cis-1,2-dichloroethene, ethylbenzene, isopropylbenzene, m,p-xylene, o-xylene, toluene, total xylenes, and trichloroethene), cyanide, and gasoline-range hydrocarbons.[72] Benzo(a)pyrene, benzo(a)anthracene, naphthalene, and trichloroethene have the highest HQs at this property.
- **Willbridge –** At the Willbridge site, six COPCshave HQs ≥ 1 based on detected concentrations: five metals (barium, beryllium, cadmium, iron, manganese), and gasoline-range hydrocarbons. Iron and manganese have the highest maximum HQs for this location. Two COPCs (i.e., benzo(a)anthracene and benzo(a)pyrene) each had a DL greater than their respective TRVs. Otherwise, these two PAHs were not detected at the Willbridge site.

[72] The HQs for several fractions of gasoline-range aliphatic hydrocarbons are ≥ 1; however, these were counted as a single exceedance in the tally.

Table 6-41. TZW COPCs with HQs ≥ 1 in Individual Samples by Area

	Number of Samples with HQs ≥ 1 (Maximum HQ)										
		Arkema									
COPC	ARCO	Acid Plant Area	Chlorate Plant Area	Mobil Oil	Gasco	Gunderson	Kinder Morgan	Rhône-Poulenc	Siltronic	Willbridge
Metals										
Barium (total)	7 of 7 (73)	8 of 8 (610)	10 of 10 (1,100)	11 of 11 (88)	8 of 8 (86)	9 of 9 (68)	8 of 8 (31)	10 of 10 (170)	13 of 13 (57)	9 of 9 (86)
Beryllium (total)	0 of 7	0 of 8	0 of 10	1 of 11 (1.8)	0 of 8	0 of 9	0 of 8	1 of 10 (1.7)	0 of 13	1 of 9 (2)
Cadmium (dissolved)	0[a] of 5	0 of 4	3 of 6 (2.6)	1 of 12 (1.1)	0 of 4	0 of 2	0[a] of 3	5[b] of 7 (5.8)	0 of 6	1 of 6 (1.5)
Cobalt (total)	NA	NA	NA	NA	NA	NA	NA	NA	3 of 13 (3.6)	NA
Copper (dissolved)	0 of 5	NA	NA	0 of 12	0 of 4	0 of 2	0 of 3	1 of 7 (1.3)	0 of 6	0 of 6
Iron (total)	7 of 7 (75)	7 of 8 (110)	6 of 10 (250)	11 of 11 (110)	8 of 8 (130)	9 of 9 (91)	8 of 8 (49)	10 of 10 (98)	26 of 26 (180)	9 of 9 (120)
Lead (dissolved)	0 of 5	0 of 4	0 of 6	1 of 12 (3)	2 of 4 (1.7)	0 of 2	0 of 3	1 of 7 (2.8)	0 of 6	0 of 6
Magnesium (total)	0 of 7	4 of 8 (7)	1 of 10 (3.8)	0 of 11	0 of 8	0 of 9	0 of 8	3 of 10 (2.2)	0 of 26	0 of 9
Manganese (total)	7 of 7 (52)	8 of 8 (94)	10 of 10 (550)	11 of 11 (150)	8 of 8 (130)	9 of 9 (43)	8 of 8 (72)	10 of 10 (130)	26 of 26 (84)	8 of 9 (110)
Nickel (dissolved)	0 of 5	0 of 4	1 of 6 (1.6)	0 of 12	1 of 4 (1.1)	0 of 2	0 of 3	1[b] of 7 (1.1)	0 of 6	0 of 6
Potassium (total)	0 of 7	0 of 8	2 of 10 (3.7)	0 of 11	0 of 8	0 of 9	0 of 8	0 of 10	0 of 13	0 of 9
Sodium (total)	0 of 7	1 of 8 (14)	10 of 10 (55)	0 of 11	0 of 8	0 of 9	0 of 8	0 of 10	0 of 13	0 of 9

Table 6-41. TZW COPCs with HQs ≥ 1 in Individual Samples by Area

COPC	ARCO	Arkema: Acid Plant Area	Arkema: Chlorate Plant Area	Mobil Oil	Gasco	Gunderson	Kinder Morgan	Rhône-Poulenc	Siltronic	Willbridge
	Number of Samples with HQs ≥ 1 (Maximum HQ)									
Vanadium (total)	NA	NA	NA	NA	NA	NA	NA	NA	6 of 13 (19)	NA
Zinc (dissolved)	1 of 5 (14)	0 of 4	0 of 6	0 of 12	0 of 4	0 of 2	0 of 3	0 of 7	0 of 6	0 of 6
PAHs										
2-Methylnaphthalene	0 of 12	NA	NA	0 of 21	8 of 12 (40)	NA	0 of 11	NA	3 of 19 (17)	0 of 14
Acenaphthene	0 of 12	NA	NA	0 of 21	4 of 12 (5.2)	NA	0 of 11	NA	20 of 32 (17)	0 of 14
Anthracene	0 of 12	NA	NA	0 of 21	10 of 12 (13)	NA	0 of 11	NA	18 of 32 (87)	0 of 14
Benzo(a)anthracene	1[c] of 12 (5.6)	NA	NA	5[c] of 21 (8.5)	9 of 12 (120)	NA	2 of 11 (2.9)	NA	14[d] of 32 (1,200)	0[a, b,] of 14
Benzo(a)pyrene	2[c] of 12 (15)	NA	NA	5[c] of 21 (25)	9 of 12 (210)	NA	0[a, c] of 11	NA	18[b] of 32 (2,700)	0[a, b,] of 14
Benzo(b)fluoranthene	0 of 12	NA	NA	0 of 21	3 of 12 (3.1)	NA	0 of 11	NA	10 of 32 (49)	0 of 14
Benzo(g,h,i)perylene	0 of 12	NA	NA	1 of 21 (1.1)	3 of 12 (7.3)	NA	0 of 11	NA	9 of 32 (66)	0 of 14
Benzo(k)fluoranthene	0 of 12	NA	NA	0 of 21	3 of 12 (3.1)	NA	0 of 11	NA	7 of 32 (14)	0 of 14
Chrysene	0 of 12	NA	NA	0 of 21	3 of 12 (2.2)	NA	0 of 11	NA	7 of 32 (17)	0 of 14
Dibenzo(a,h)anthracene	0 of 12	NA	NA	0 of 21	1 of 12 (1.2)	NA	0 of 11	NA	7 of 32 (13)	0 of 14

Table 6-41. TZW COPCs with HQs ≥ 1 in Individual Samples by Area

	Number of Samples with HQs ≥ 1 (Maximum HQ)									
		Arkema								
COPC	ARCO	Acid Plant Area	Chlorate Plant Area	Mobil Oil	Gasco	Gunderson	Kinder Morgan	Rhône-Poulenc	Siltronic	Willbridge
Fluoranthene	0 of 12	NA	NA	0 of 21	3 of 12 (2.8)	NA	0 of 11	NA	8 of 32 (17)	0 of 14
Fluorene	0 of 12	NA	NA	3 of 21 (1.5)	10 of 12 (7.9)	NA	0 of 11	NA	23 of 32 (28)	0 of 14
Indeno(1,2,3-cd)pyrene	0 of 12	NA	NA	1 of 21 (1.2)	3 of 12 (9.8)	NA	0 of 11	NA	9 of 32 (61)	0 of 14
Naphthalene	0 of 12	2[b] of 9 (2.2)	0 of 10	0 of 21	6 of 12 (260)	0 of 9	0 of 12	0 of 10	23 of 60 (1,100)	0 of 14
Phenanthrene	0 of 12	NA	NA	5 of 21 (2.4)	10 of 12 (13)	NA	0 of 11	NA	21 of 32 (57)	0 of 14
Pyrene	0 of 12	NA	NA	0 of 21	3 of 12 (3.2)	NA	0 of 11	NA	8 of 32 (15)	0 of 14
SVOCs										
1,2-Dichlorobenzene	0 of 7	0 of 9	0 of 10	0 of 11	0 of 8	0 of 9	0 of 9	5 of 10 (46)	0 of 54	0 of 9
1,4-Dichlorobenzene	0 of 7	0 of 5	0 of 6	0 of 11	0 of 8	0 of 9	0 of 9	2 of 10 (16)	0 of 54	0 of 9
Dibenzofuran	0 of 12	NA	NA	0 of 21	1 of 12 (2.2)	NA	0 of 11	NA	2 of 19 (2)	0 of 14
Pesticides										
4,4′-DDT	NA	3[d] of 12 (160)	NA	NA	NA	NA	NA	0 of 2	NA	NA
Total DDx	NA	8[b] of 12 (280)	NA	NA	NA	NA	NA	2 of 2 (19)	NA	NA

Table 6-41. TZW COPCs with HQs ≥ 1 in Individual Samples by Area

	Number of Samples with HQs ≥ 1 (Maximum HQ)									
		Arkema								
COPC	ARCO	Acid Plant Area	Chlorate Plant Area	Mobil Oil	Gasco	Gunderson	Kinder Morgan	Rhône-Poulenc	Siltronic	Willbridge
VOCs										
1,1-Dichloroethene	0 of 7	0 of 9	0 of 10	0 of 11	0 of 8	0 of 9	0 of 9	0 of 10	2 of 54 (1.6)	0 of 9
1,2,4-Trimethylbenzene	NA	NA	NA	NA	NA	NA	NA	NA	11 of 41 (9.6)	NA
1,3,5-Trimethylbenzene	NA	NA	NA	NA	NA	NA	NA	NA	5 of 41 (3)	NA
Benzene	0 of 7	0 of 9	0 of 10	0 of 11	3 of 8 (4.2)	0 of 9	0 of 9	0 of 10	6 of 54 (30)	0 of 9
Carbon disulfide	0 of 7	0[a, b] of 9	0 of 10	0 of 11	3 of 8 (870)	0 of 9	0 of 9	0 of 10	1[d] of 54 (1.3)	0 of 9
Chlorobenzene	0 of 7	2 of 9 (190)	0 of 10	0 of 11	0 of 8	0 of 9	0 of 9	1 of 10 (3.3)	0 of 54	0 of 9
Chloroethane	0 of 7	0 of 9	0 of 10	0 of 11	0 of 8	1 of 9 (3.4)	0 of 9	0 of 10	0 of 54	0 of 9
Chloroform	0 of 7	1 of 9 (21)	3[c] of 10 (7.9)	0 of 11	0 of 8	0 of 9	0 of 9	0 of 10	0 of 54	0 of 9
cis-1,2-Dichloroethene	0 of 7	0 of 9	0 of 10	0 of 11	0 of 8	0 of 9	0 of 9	0 of 10	5 of 54 (110)	0 of 9
Ethylbenzene	0 of 7	0 of 9	0 of 10	0 of 11	3 of 8 (11)	0 of 9	0 of 9	0 of 10	12 of 54 (57)	0 of 9
Isopropylbenzene	0 of 7	0 of 9	0 of 10	0 of 11	2 of 8 (1.5)	0 of 9	0 of 9	0 of 10	8 of 54 (2)	0 of 9
m,p-Xylene	0 of 7	0 of 9	0 of 10	0 of 11	0 of 8	0 of 9	0 of 9	0 of 10	3 of 54 (4.4)	0 of 9
o-Xylene	0 of 7	0 of 9	0 of 10	0 of 11	3 of 8 (3.6)	0 of 9	0 of 9	0 of 10	9 of 54 (12)	0 of 9

Table 6-41. TZW COPCs with HQs ≥ 1 in Individual Samples by Area

	Number of Samples with HQs ≥ 1 (Maximum HQ)									
		Arkema								
COPC	**ARCO**	**Acid Plant Area**	**Chlorate Plant Area**	**Mobil Oil**	**Gasco**	**Gunderson**	**Kinder Morgan**	**Rhône-Poulenc**	**Siltronic**	**Willbridge**
Toluene	0 of 7	0 of 9	0 of 10	0 of 11	4 of 8 (2.9)	0 of 9	0 of 9	0 of 10	7 of 54 (18)	0 of 9
Total xylenes	0 of 7	0 of 9	0 of 10	0 of 11	3 of 8 (8.5)	0 of 9	0 of 9	0 of 10	10 of 54 (34)	0 of 9
Trichloroethene	0 of 7	0 of 9	0 of 10	0 of 11	0 of 8	0 of 9	0 of 9	0 of 10	2 of 54 (1,900)	0 of 9
Petroleum Hydrocarbons										
Gasoline-range aliphatic hydrocarbons C4-C6	1 of 9 (1.1)	NA	NA	3 of 15 (1.2)	5 of 10 (7.3)	NA	0 of 10	NA	6 of 15 (2.0)	0 of 9
Gasoline-range aliphatic hydrocarbons C6-C8	0 of 9	NA	NA	0 of 9	4 of 10 (4.3)	NA	0 of 10	NA	3 of 15 (1.2)	0 of 9
Gasoline-range aliphatic hydrocarbons C8-C10	0 of 9	NA	NA	0 of 9	0 of 10	NA	0 of 10	NA	0 of 15	0 of 9
Gasoline-range aliphatic hydrocarbons C10-C12	6 of 9 (85)	NA	NA	6 of 15 (85)	9 of 10 (540)	NA	3 of 10 (6.9)	NA	9 of 15 (150)	3 of 9 (3.8)
Gasoline-range aromatic hydrocarbons C8-C10	0 of 9	NA	NA	0 of 15	3 of 10 (2.7)	NA	0 of 10	NA	0 of 15	0 of 9
Other Contaminants										
Cyanide	NA	NA	NA	NA	8[b] of 8 (4,400)	NA	NA	NA	26[b] of 26 (130)	NA
Perchlorate	NA	0[a, b] of 9	5 of 10 (19)	NA	NA	0 of 2	NA	NA	NA	NA

a Only samples with undetected concentrations have HQs ≥ 1.
b One additional sample had a DL greater than the TRV.
c An additional two to three samples had DLs greater than the TRV.

[d] An additional four or more non-detect samples had DLs greater than the TRV.

COPC – contaminant of potential concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
DL – detection limit
HQ – hazard quotient
NA – not analyzed
PAH – polycyclic aromatic hydrocarbon
SVOC – semivolatile organic compound
total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT, and 4,4′-DDT)
TRV – toxicity reference value
TZW – transition zone water
VOC – volatile organic compound

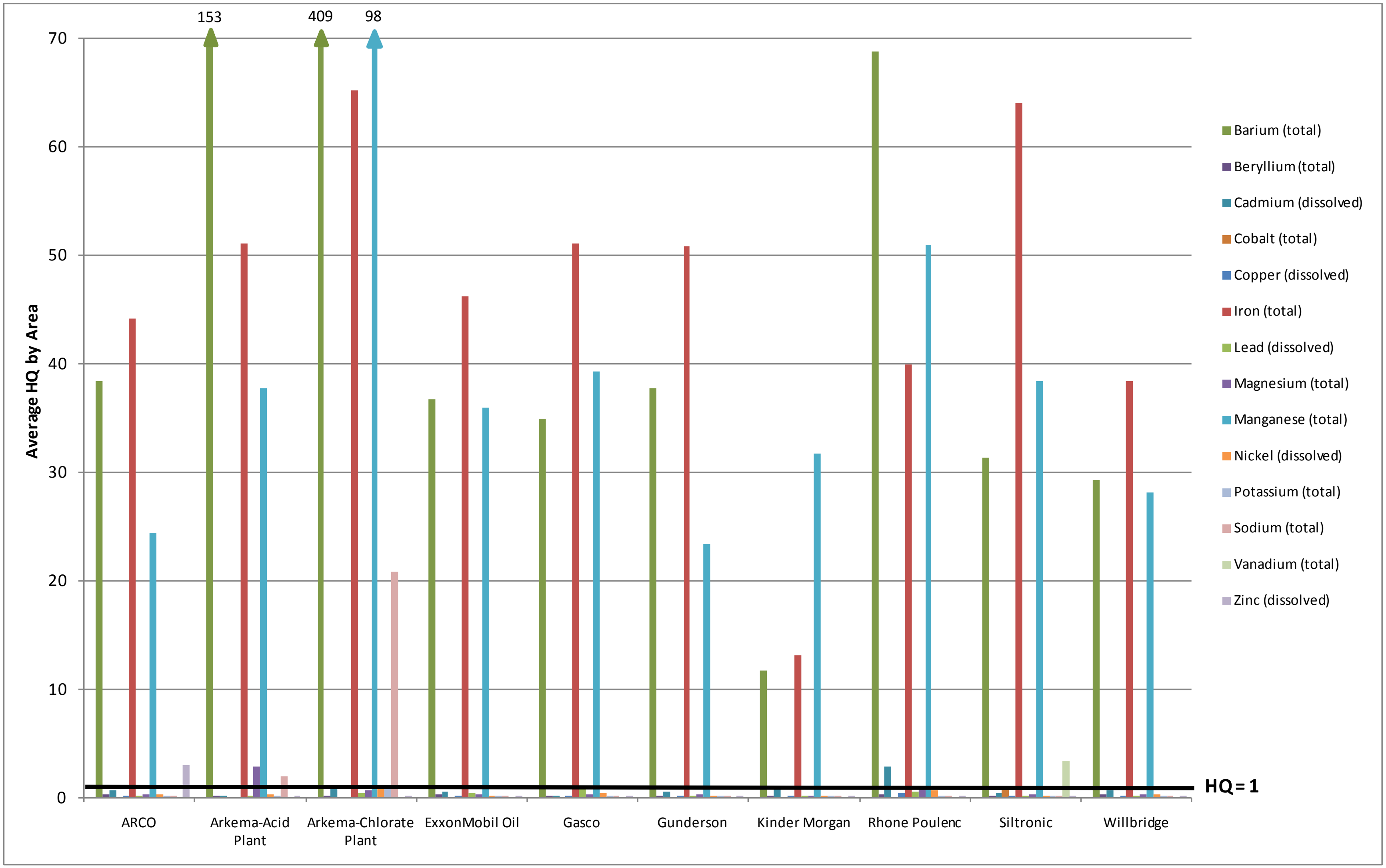

Figure 6-22. TZW Metal COPCs by Area

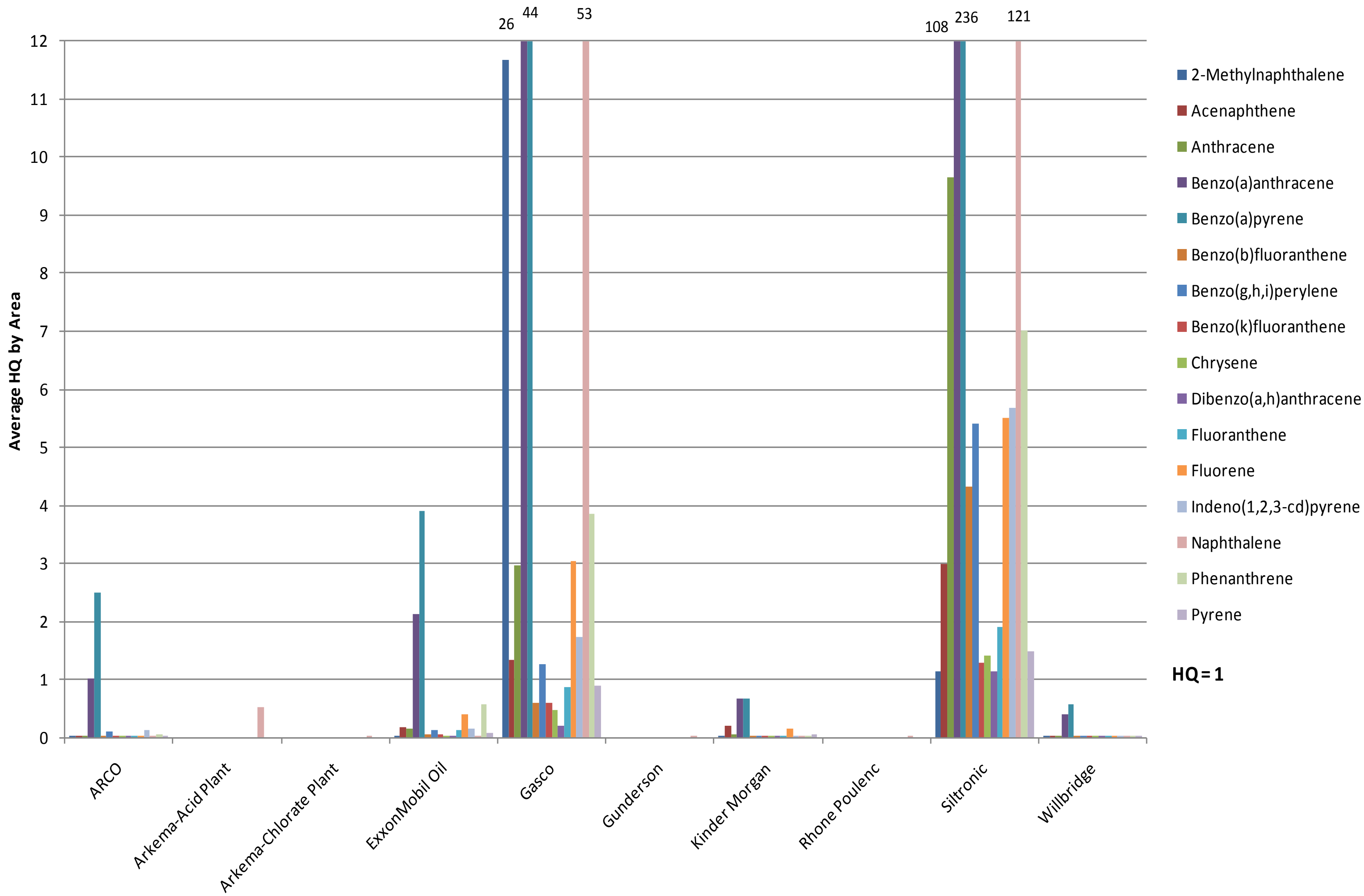

Figure 6-23. TZW PAH COPCs by Area

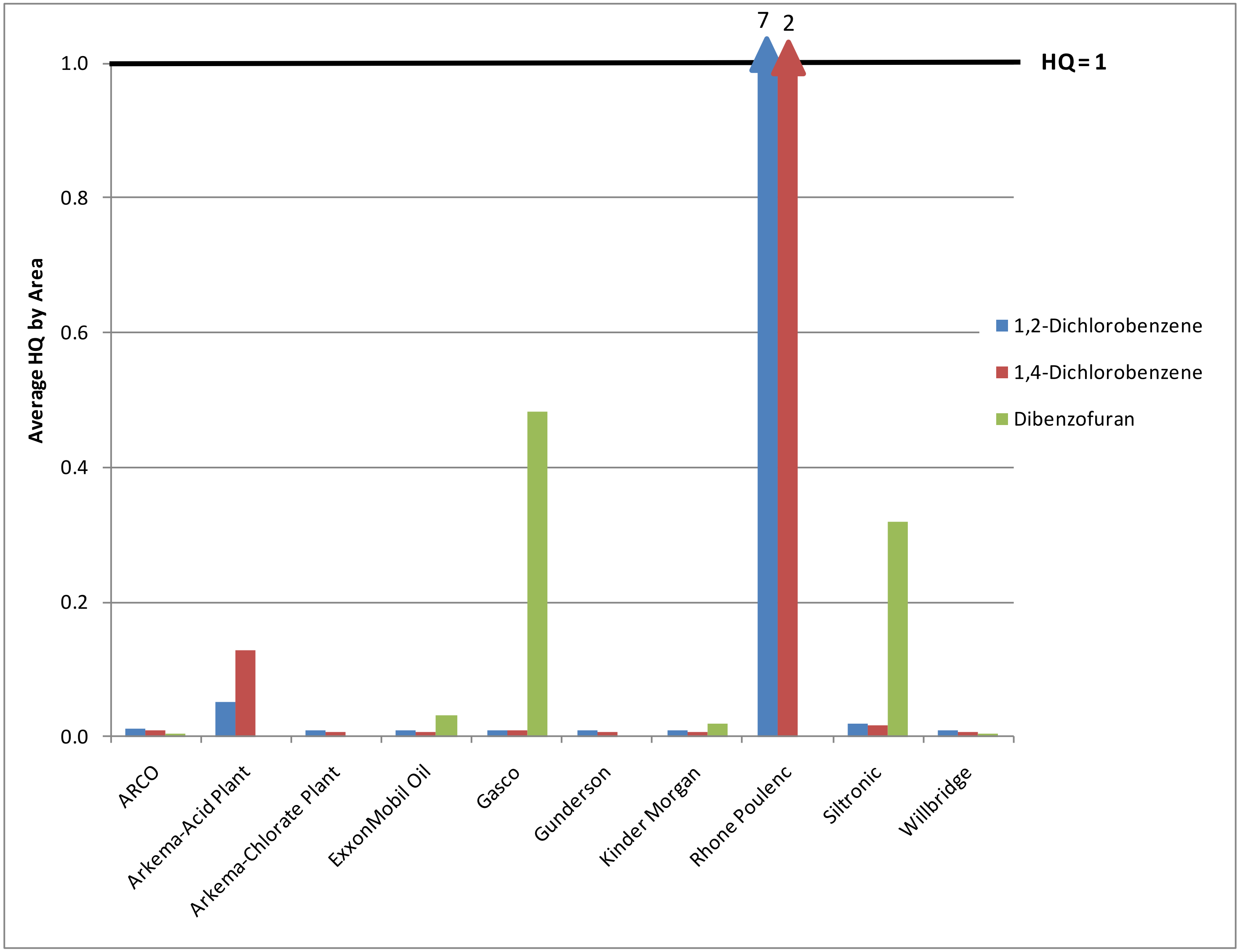

Figure 6-24. TZW SVOC COPCs by Area

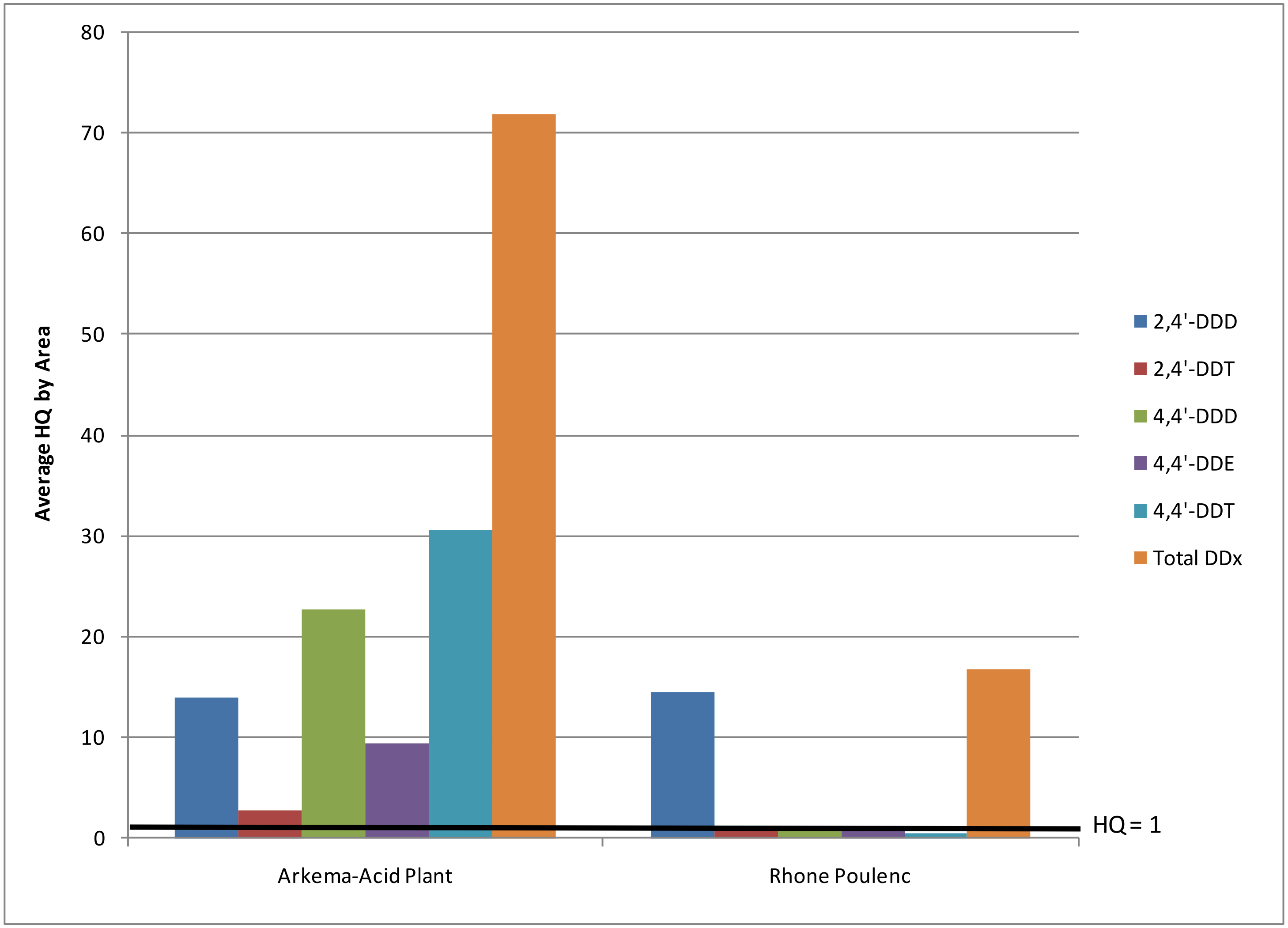

Figure 6-25. TZW DDT COPCs by Area

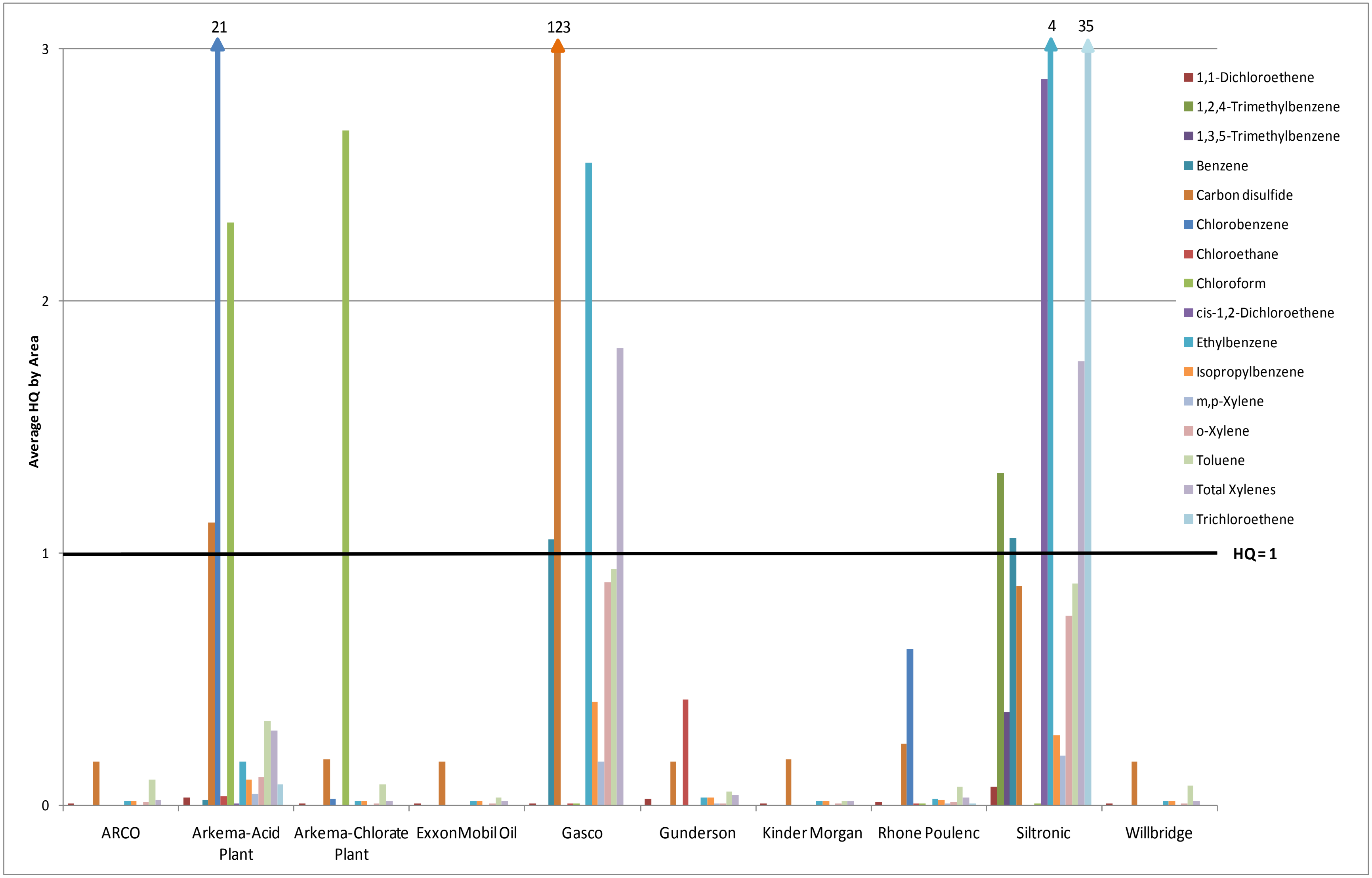

Figure 6-26. TZW VOC COPCs by Area

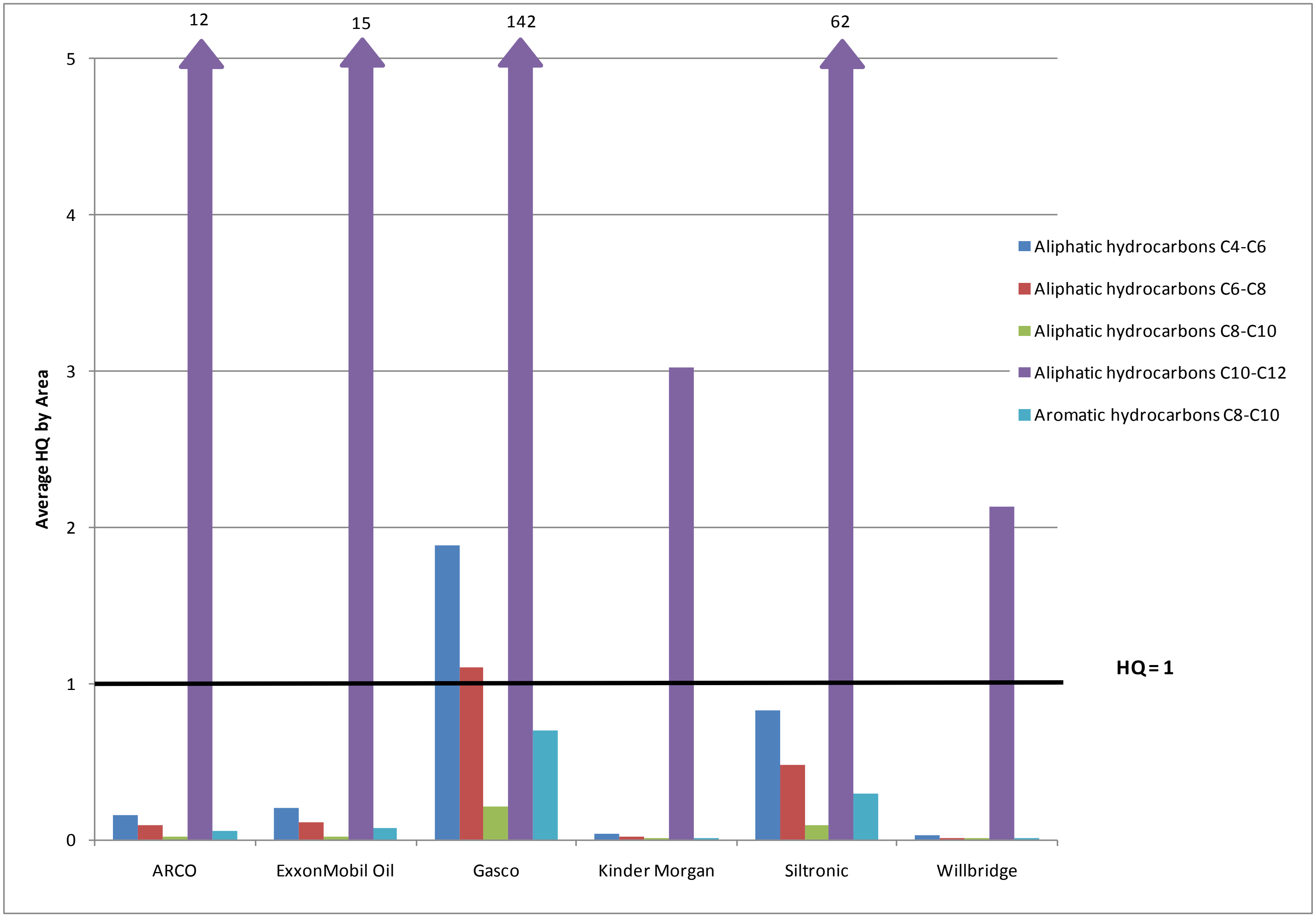

Figure 6-27. TZW TPH fraction COPCs by Area

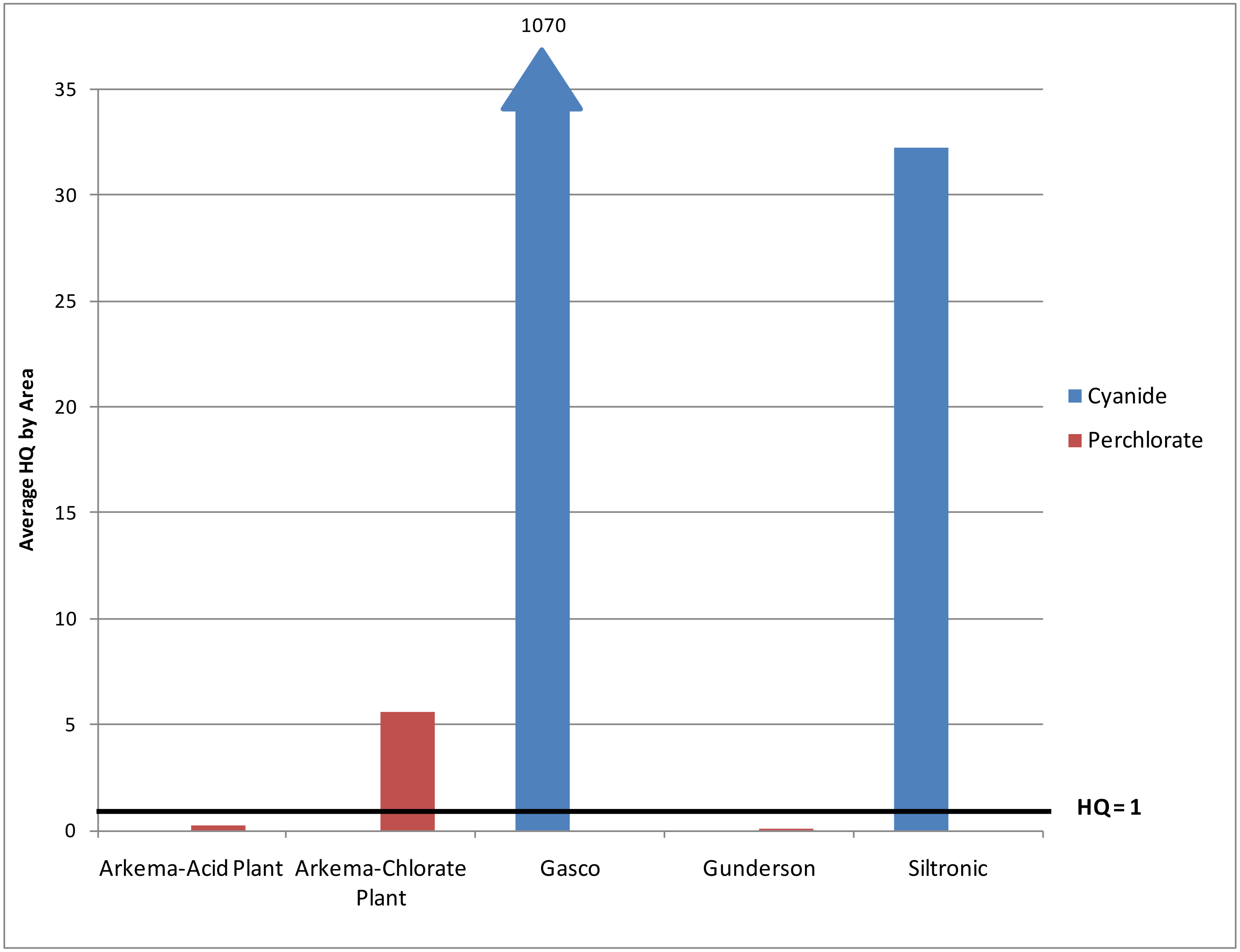

Figure 6-28. TZW COPCs by Area

The HQs presented in Table 6-41 for 4,4′-DDT and total DDx are based on an alternative TRV (0.011 µg/L) that was developed in this BERA using methods consistent with those used for AWQC derivation (see Section 6.5.4). This alternative TRV is considered more appropriate than the AWQC (0.0010 µg/L), which is based on the protection of brown pelican reproduction. HQs based on the AWQC value (0.0010 µg/L) would be an order of magnitude higher than those presented in Table 6-41.

The uncertainties associated with the TZW data as representative EPC data for benthic organisms is discussed in Section 6.6.3.3. As discussed in Section 6.6.3.3, actual water EPCs are probably much lower due to feeding habits, burrowing behavior, avoidance of low oxygen levels at TZW sample depths, and low food content in sediments at the depth that TZW was collected. Assuming a ventilation rate of about 10% (reported for filter-feeding clams (Winsor et al. 1990)), TZW HQs presented in Table 6-41 would be reduced by an order of magnitude. HQs would be < 1 for several metals (i.e., beryllium, cadmium, cobalt, copper, lead, magnesium, nickel, and potassium), dibenzofuran, several VOCs (i.e., 1,1-dichloroethene, 1,2,4-trimethylbenzene, 1,3,5-trimethylbenzene, chloroethene, isopropylbenzene, and m,p-xylene), and three individual gasoline-range fractions. HQs would be still greater than 1, but less than 10 for several additional metals, most individual PAHs, two SVOCs, several VOCs, and perchlorate.

6.6.5.2 Evaluation of Naturally Occurring Metals

Although there are many anthropogenic sources of metals, almost all of the metals measured in TZW are also common crustal elements. Barium, iron, and manganese are among the most common metals associated with sediments. These common metals are also associated with the highest HQs identified in the risk characterization, but there is substantial uncertainty that their source is anthropogenic.

The contribution of geochemical processes in sediments to the concentrations of selected metals in TZW was extensively evaluated in Appendix C2 of the draft final RI (Integral et al. 2011). Concentrations of iron and manganese in TZW are not well-correlated to potential anthropogenic source materials (i.e., petroleum hydrocarbons), suggesting that factors other than sediment contamination (e.g., naturally occurring organic materials) are contributing to concentrations measured in the TZW. Geochemical processes are also likely responsible for some percentage of the measured barium concentrations in TZW, in addition to the contribution from migration of upland groundwater to the river. Aluminum was not included in the geochemical evaluation, but a background surface water concentration (established in Section 7.0 of the draft final RI (Integral et al. 2011)) is available to provide some context for TZW (since surface water is a component of shallow TZW). An upper-bound (UPL) background concentration for aluminum was 1,485 µg/L. The majority of the TZW values were below this concentration.

6.6.5.3 COIs For Which Risks Cannot Be Quantified

COIs for which risks to benthic invertebrates cannot be quantified based on TZW data are listed in Table 6-42. These COIs are chemicals for which no TRV is available or for which the maximum DL exceeds a TRV, but detected values do not.

Table 6-42. Benthic Invertebrate TZW COIs with No Available TRV or with DL Exceeding SL TRV

COI	Rationale for Why Risks Cannot Be Quantitatively Evaluated
Metals	
Aluminum	The AWQC chronic criterion for aluminum was used to identify aluminum as a COPC. However, as per agreement with EPA, the AWQC criterion is not applicable to waters with circumneutral pH, such as those in the Study Area, precluding further evaluation of aluminum.
Calcium	Risk to benthic invertebrates based on TZW data unknown; no water threshold available
Selenium	Risk to benthic invertebrates based on TZW data unknown; 26% of non-detected samples exceed water threshold, but no detected concentration > SL TRV
Titanium	Risk to benthic invertebrates based on TZW data unknown; no water threshold available
Dioxins/Furans	
Individual congeners other than 2,3,7,8-TCDD[a]	Risk to benthic invertebrates based on TZW data unknown; no water threshold available
Petroleum Hydrocarbons	
Residual-range hydrocarbons	Risk to benthic invertebrates based on TZW data unknown; no water threshold available
Diesel-range hydrocarbons	Risk to benthic invertebrates based on TZW data unknown; no water threshold available
Total diesel-residual hydrocarbons	Risk to benthic invertebrates based on TZW data unknown; no water threshold available
Total petroleum hydrocarbons	Risk to benthic invertebrates based on TZW data unknown; no water threshold available
VOCs	
Styrene	Risk to benthic invertebrates based on TZW data unknown; 1% of non-detected samples exceed water threshold, but no detected concentration > SL TRV

[a] TZW dioxin and furan congeners COIs with no water threshold available include the following: 1,2,3,4,6,7,8-heptachlorodibenzo-*p*-dioxin, 1,2,3,4,7,8,9-heptachlorodibenzofuran, 1,2,3,4,7,8-hexachlorodibenzofuran, 1,2,3,6,7,8-hexachlorodibenzofuran, 1,2,3,7,8-pentachlorodibenzofuran, 2,3,4,7,8-pentachlorodibenzofuran, and 2,3,7,8-tetrachlorodibenzofuran.

AWQC – ambient water quality criteria
COI – contaminant of interest
COPC – contaminant of potential concern
DL – detection limit
EPA – US Environmental Protection Agency
VOC – volatile organic compound
SL – screening level
TCDD – tetrachlorodibenzo-*p*-dioxin
TRV – toxicity reference value
TZW- transition zone water
VOC – volatile organic compound

6.6.5.4 Summary of TZW Risk Evaluation

TZW COPCs with HQs ≥ 1 for benthic invertebrates are listed in Table 6-43. The results of the TZW LOE are integrated with those from other LOEs to determine risk conclusions for benthic invertebrates in Section 6.7.

Table 6-43. TZW COPCs with HQs ≥ 1

Contaminants of Potential Concern	
Metals	
Barium	Magnesium
Beryllium	Manganese
Cadmium	Nickel
Cobalt	Potassium
Copper	Sodium
Iron	Vanadium
Lead	Zinc
PAHs	
2-Methylnaphthalene	Chrysene
Acenaphthene	Dibenzo(a,h)anthracene
Anthracene	Fluoranthene
Benzo(a)anthracene	Fluorene
Benzo(a)pyrene	Indeno(1,2,3-cd)pyrene
Benzo(b)fluoranthene	Naphthalene
Benzo(g,h,i)perylene	Phenanthrene
Benzo(k)fluoranthene	Pyrene
SVOCs	
1,2-Dichlorobenzene	Dibenzofuran
1,4-Dichlorobenzene	
Pesticides	
4,4′-DDT	Total DDx
VOCs	
1,1-Dichloroethene	cis-1,2-Dichloroethene
1,2,4-Trimethylbenzene	Ethylbenzene
1,3,5-Trimethylbenzene	Isopropylbenzene
Benzene	Toluene
Carbon disulfide	Trichloroethene
Chlorobenzene	m,p-Xylene
Chloroethane	o-Xylene
Chloroform	Total xylenes
TPH	
Gasoline-range hydrocarbons[a, b]	

Table 6-43. TZW COPCs with HQs ≥ 1

Contaminants of Potential Concern	
Other Contaminants	
Cyanide	Perchlorate

[a] Gasoline-range hydrocarbons were evaluated as five components.
[b] Gasoline-range hydrocarbons is listed here but not included in the count of COPCs with HQs ≥ 1.

CERCLA – Comprehensive Environmental Response, Compensation, and Liability Act
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
PAH – polycyclic aromatic hydrocarbon
SVOC – semivolatile organic compound
total DDx – sum of all six DDT isomers (2,4′-DDD; 4,4′-DDD; 2,4′-DDE; 4,4′-DDE; 2,4′-DDT; and 4,4′-DDT)
TPH – total petroleum hydrocarbons
TZW – transition zone water
VOC – volatile organic compound

6.7 BENTHIC RISK CONCLUSIONS AND UNCERTAINTY

This section presents overall conclusions regarding Study Area risks to benthic macroinvertebrates. Risks were assessed using empirical and predicted sediment toxicity, empirical and predicted exceedances of TRVs for benthic tissue residues, surface water quality, and TZW. Because sediment, tissue, surface water, and TZW represent different routes of exposure, each medium was assessed separately. The results were then integrated to create an overall portrayal of potential benthic risks. COPCs exceeding their respective TRVs were identified for each medium or pathway.

The benthic assessment endpoints are expressed at the population and community levels, but the measurement endpoints are determined at the organism level. Therefore, conclusions about unacceptable risk to populations and communities can be drawn only by extrapolating from potential effects on individual organisms (i.e., exceedance of effect thresholds). The risk conclusions for population- and community-level risks were reached by evaluating the WOE.

The initial evaluation leading to the WOE was relatively simple and was based on a visual presentation:

- First, empirical L2 and L3 toxicity test results were mapped.
  - The maximum level of toxicity across all four endpoints was used to represent the toxicity level at a sampling location.
  - The number of endpoints identifying the maximum level of toxicity was tracked graphically.
- Predicted toxicity at chemistry-only sampling locations was added to the maps based on both benthic models (LRM and FPM).
  - Whether or not the models agreed was tracked graphically.

- Empirical bioaccumulation data at locations where a COPC exceeded the tissue-residue TRV (i.e., HQ ≥ 1) were added to the map.
  - Individual chemicals or chemical groups (i.e., metals) contributing to the exceedance of the TRV were tracked graphically.
- Samples that were predicted to exceed a tissue TRV were also displayed.
  - Predictions were made only for chemicals that showed a relationship between tissue and sediment concentrations and were considered applicable to all benthic receptors.
  - Interpolations were based on chemical concentrations exceeding a sediment threshold back-calculated from sediment-tissue regression equations or mechanistic models.
- Lastly, TZW sampling locations that exceeded surface water TRVs were identified on the map.

Surface water TRV exceedances were not displayed on the maps but were considered in the evaluation of all LOEs when framing risk conclusions.

Three factors were considered in a WOE to extrapolate from organism-level effect threshold exceedances to population and community-level risk conclusions:

- Spatial extent of exceedances of SQVs or TRVs
- Magnitude and frequency of organism-level effect thresholds exceedances
- Quality and relevance of the organism-level effect thresholds as predictors of population- and community-level risks

A summary of the benthic invertebrate COPCs is provided next. The COPCs identified by the benthic LOEs have the potential to, but do not necessarily pose unacceptable risk to the benthic community; evaluation of the spatial scale of the contaminant distribution, magnitude of the contaminant concentrations, frequency of exceedances, and certainty and relevance of the individual TRVs (i.e., the WOE framework introduced previously) need to be taken into account to assess community- or population-level risks.

6.7.1 Summary of Benthic Invertebrate COPCs

Table 6-44 presents a compilation of the benthic invertebrate COPCs, by LOE, that result in an exceedance factor or HQ ≥ 1. Eighty-three COPCs (individual chemicals, sums, or totals) were identified as posing potentially unacceptable risk to the benthic community based on exceedance of the site-specific SQVs (two different models), tissue TRVs, surface water TRVs, and TZW TRVs.[73] Of these, 44 COPCs have HQs ≥ 1 based on only

[73] Eighty-five COPCs were identified as posing potentially unacceptable risk to the benthic community based on exceedance of the site-specific SQVs. However two of these are not included in the COPC count (gasoline-range hydrocarbons and diesel-range hydrocarbons); these chemical groups are used instead to address the uncertainty in benthic community risk based on PAHs.

one LOE: 28 for TZW, 8 for the FPM, 7 for the LRM, and 1 for tissue residues. Several other contaminants (petroleum, ammonia, and sulfides in bulk sediment) exceeded screening levels but are not included in this count, although they may contribute to potentially unacceptable risk.

Risks to the benthic community could not be evaluated for COIs with no TRVs or where DLs exceeded TRVs but detected values did not. Additionally, some sediment contaminants were not evaluated because no site-specific SQV could be derived. COIs that could not be evaluated are listed in Table 6-6 for sediment, Table 6-28 for tissue, Table 6-35 for surface water, and Table 6-42 for TZW.

Table 6-44. Benthic Contaminants Exceeding an Effect Threshold (SQV, pMax, TRV) with HQs ≥ 1

	Line of Evidence					
	Predicted Toxicity		Tissue Residue			
COPC	FPM SQV Max HQ	LRM Max HQ and (pMax)	Empirical Max HQ[a]	Predicted Max HQ	Surface Water Max HQ	TZW Max HQ
Metals						
Arsenic			1.5			
Barium						1,100
Beryllium						2
Cadmium	13					5.8
Chromium	17	2.7 (0.97)				
Cobalt						3.6
Copper	5	5.3 (0.98)	2.6			1.3
Iron						250
Lead		53 (1)				3
Magnesium						7
Manganese						550
Mercury	280	3.5 (0.94)				
Nickel						1.6
Potassium						3.7
Silver	8.6	4.9 (0.94)				
Sodium						55
Vanadium						19
Zinc			2.2		1.1	14
Butyltins						
Monobutyltin					1.2b	
Tributyltin ion		11 (0.96)	11	149		
PAHs						

Table 6-44. Benthic Contaminants Exceeding an Effect Threshold (SQV, pMax, TRV) with HQs ≥ 1

	Line of Evidence					
	Predicted Toxicity		Tissue Residue			
COPC	FPM SQV Max HQ	LRM Max HQ and (pMax)	Empirical Max HQ[a]	Predicted Max HQ	Surface Water Max HQ	TZW Max HQ
2-Methylnaphthalene		140 (1)				40
Acenaphthene		6.6 (0.79)				17
Acenaphthylene		2.8 (0.82)				
Anthracene		210 (0.99)				87
Benzo(a)anthracene		4.6 (0.85)			10	1,200
Benzo(a)pyrene					14	2,700
Benzo(b)fluoranthene		4 (0.84)				49
Benzo(g,h,i)perylene		3.8 (0.84)				66
Benzo(k)fluoranthene		3.2 (0.84)				14
Chrysene		4.3 (0.85)				17
Dibenzo(a,h)anthracene		4.2 (0.85)				13
Fluoranthene		4.1 (0.83)				17
Fluorene		9.1 (0.93)				28
Indeno(1,2,3-cd)pyrene		3.8 (0.84)				61
Naphthalene					50	1,100
Phenanthrene		4.2 (0.76)				57
Pyrene		3.3 (0.82)				15
Total HPAHs	7	3.9 (0.83)				
Total LPAHs	1,500	18 (0.96)				
Total PAHs		20 (0.98)				
PCBs						
Total PCBs	62	12 (0.93)	7.5	20	<1[c]	
SVOCs						
1,2-Dichlorobenzene						46
1,4-Dichlorobenzene						16
Benzyl alcohol	6.8					
Carbazole	29	46 (0.98)				
Dibenzofuran	180	9.5 (0.95)				2.2
Phthalates						
BEHP			2.8		2.3	
Dibutyl phthalate		2.8 (0.83)				
Phenols						
4-Methylphenol	31					

Table 6-44. Benthic Contaminants Exceeding an Effect Threshold (SQV, pMax, TRV) with HQs ≥ 1

	Line of Evidence					
	Predicted Toxicity		Tissue Residue			
COPC	FPM SQV Max HQ	LRM Max HQ and (pMax)	Empirical Max HQ[a]	Predicted Max HQ	Surface Water Max HQ	TZW Max HQ
Phenol		5.2 (0.93)				
Pesticides						
2,4'-DDD		6.8 (0.95)				
4,4'-DDD		5.9 (0.93)	1.2			
4,4'-DDE		4.5 (0.89)				
4,4'-DDT		10 (0.90)			< 1[c]	160
beta-HCH	1.9					
cis-Chlordane		39 (0.99)				
delta-HCH	4.1	2.5 (0.80)				
Dieldrin	17					
Endrin	1.5					
Endrin ketone	11					
Sum DDD	27	6.2 (0.92)				
Sum DDE	2.8	12 (0.96)				
Sum DDT	1.6					
Total DDx		8 (0.94)	3.2	10	1.8	280
Total endosulfan	110					
VOCs						
1,1-Dichloroethene						1.6
1,2,4-Trimethylbenzene						9.6
1,3,5-Trimethylbenzene						3
Benzene						30
Carbon disulfide						870
Chlorobenzene						190
Chloroethane						3.4
Chloroform						21
cis-1,2-Dichloroethene						110
Ethylbenzene					1.6	57
Isopropylbenzene						2
m,p-Xylene						4.4
o-Xylene						12
Toluene						18
Total xylenes						34

Table 6-44. Benthic Contaminants Exceeding an Effect Threshold (SQV, pMax, TRV) with HQs ≥ 1

	Line of Evidence					
	Predicted Toxicity		Tissue Residue			
COPC	FPM SQV Max HQ	LRM Max HQ and (pMax)	Empirical Max HQ[a]	Predicted Max HQ	Surface Water Max HQ	TZW Max HQ
Trichloroethene					4.1	1,900
Petroleum Hydrocarbons[d]						
Diesel-range hydrocarbons		17 (1)				
Gasoline-range hydrocarbons						540[e]
Other Contaminants[d]						
Cyanide						4,400
Perchlorate						19

a Based on accumulation in one or more benthic receptors.
b Exceedance based on monobutyltin, as a surrogate for TBT.
c Based on alternative TRV relevant to benthic invertebrate receptors.
d Not included in the total count of COPCs with HQs ≥ 1; used to evaluate uncertainty associated with PAH risks.
e Based on estimated exceedance of C10-C12 aliphatic hydrocarbon fraction.

BEHP – bis(2-ethylhexyl) phthalate
CERCLA – Comprehensive Environmental Response, Compensation, and Liability Act
COPC – contaminant of potential concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
FPM—floating percentile model
HCH – hexachlorocyclohexane
HPAH – high-molecular-weight polycyclic aromatic hydrocarbon
HQ – hazard quotient
LPAH – low-molecular-weight polycyclic aromatic hydrocarbon
LRM—logistic regression model
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
SQV – sediment quality value
SVOC – semivolatile organic compound
TBT – tributyltin
total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT, and 4,4′-DDT)
TZW—transition zone water
VOC – volatile organic compound

6.7.2 Weight of Evidence

Maps 6-28a and 28b present the locations where one or more LOEs indicate a potential risk to benthic organisms. These maps reflect both results of the empirical toxicity tests and predicted risks. The predictions are based on contaminant concentrations that exceed the site-specific FPM or LRM SQVs in sediment and on contaminant concentrations that exceed TRVs in invertebrate tissue, surface water, and TZW.

A further summary of the spatial distribution, frequency, magnitude, and relevance for each LOE is provided in Table 6-45. This table provides general descriptions of the locations associated with benthic risk displayed in Maps 6-28a and 28b. These areas are further discussed in the benthic conclusions.

Community or population risks were assessed on the basis of spatial extent, frequency and magnitude of exceedance, and relevance according to the following rules:

- Spatial extent of the toxic bioassay results and exceedance of site-specific SQVs or TRVs (tissue, water, TZW) was considered directly related to degree of risk to the community.
- Contiguous (or nearby) samples where one or more LOE indicates risk to benthic organisms are identified as potential benthic community risk areas.
- Isolated locations indicating risk, with no adjacent samples confirming a risk, are not indicative of potential benthic community risk.
- Frequency and magnitude of exceedance when organism-level effect thresholds are exceeded (e.g., HQ $\geq$ 1) correlate directly with degree of risk.
- Any one endpoint (e.g., *Hyalella* survival) indicating L3 (high toxicity) effects based on empirical bioassay results represents a potential risk to benthic organisms. When spatially aggregated, such findings represent a potential benthic community risk area.
- Contaminants exceeding L3 SQVs indicate potential risk to benthic organisms. When spatially aggregated, such findings represent significant benthic community risk, except that a high degree of toxicity predicted by a single chemical or using a single LOE is considered uncertain.
- Two or more L2 (moderate toxicity) effects, whether measured or predicted by an exceedance of a SQV, represent a potential risk to benthic organisms, albeit at a lower magnitude than L3. Spatial aggregation is required to represent community-level risk. Single exceedance of one L2 SQV or bioassay threshold is not indicative of risk.
- Actual or predicted exceedance of a tissue-residue TRV represents a potential risk.
- Exceedance of a surface water TRV represents a potential risk if it occurs under either of two conditions:
  - In the vicinity of a known or likely pathway for discharge to sediment or TZW
  - In more than one survey
- Exceedance of a TZW TRV represents a potential risk, except as noted:
  - Common crustal elements found in sediment (specifically barium, iron, and manganese) do not pose risk to benthic invertebrates.
  - HQs < 10 do not represent benthic risk because invertebrate physiology and behavior effectively isolate organisms’ exposure to anoxic porewater and because, within the biologically active zone, bioturbation and resulting surface

roughness entrain overlying surface water (and dilute porewater concentrations of COPCs) (see Section 6.6.3.3 for further discussion).

- Quality and relevance of the organism-level effect thresholds affect confidence in predicted population- and community-level risk:
  - Measured or predicted L3 toxicity is considered relatively certain, with low error rates, except for chromium, 4-methylphenol, and total endosulfan. Because these three contaminants contribute false positives to the predictions of toxicity, their results are considered uncertain. Risk predicted solely on the basis of these three contaminants is not used to define benthic community risk.
  - L2 toxicity identified in either empirical bioassay results or LRM and FPM predictions are less certain because of higher error rates.
- Empirical bioassay results can override predicted toxicity due to chemicals (but not bioaccumulation) at a given location:
  - Bioaccumulation of metals does not necessarily represent risk to benthic organisms or the community. Most benthic organisms can either detoxify or sequester metals, precluding deleterious effect from exposure. In addition, benthic tissue residues of inorganic metals were not correlated with sediment concentrations.
  - The TRV for TBT bioaccumulation is uncertain in that the TRV is one-fourth of the tissue-residue threshold that is protective of benthic invertebrate prey of juvenile salmonids (Meador et al. 2002a) and includes effects that are not relevant to the benthic community.

Table 6-45. Summary of Potential Benthic Community Risk Based on Individual Lines of Evidence

Line of Evidence/COPC	Number and Spatial Extent of Exceedances	Magnitude, Max HQ (Max Probability[a])	Relevance
Empirical Toxicity			Direct measure of toxicity to benthic organisms.
L2 (four endpoints)	47, throughout Study Area, within the channel, eastern, and western shores; typically isolated. Very few Level 2 exceedances above RM 9 and below RM 3.5.	NA	Low error rates with respect to correctly classifying magnitude, with the exception of *Hyalella* biomass.
L3 (four endpoints)	46, largely between RM 6.1 and RM 7.4, west. Localized toxicity in International Slip, and limited areas along western shore downstream of RM 6, between RM 8.5 and RM 9.1, west, Willamette Cove, and at the mouth of Swan Island Lagoon.	NA	Low error rates with respect to correctly classifying magnitude.
Predicted Toxicity – FPM			Predictions of toxicity relatively certain above L3 SQVs. Relevant to community/population where exceedances are spatially aggregated.
Metals			
Cadmium	9, primarily at RM 4. Single exceedances at RM 8.1, RM 8.8, and RM 9.1.	13	
Chromium	94, throughout the Study Area.	17	SQV based solely on *Hyalella* biomass endpoint. Chromium is one of several chemicals contributing false positives to this SQV set.
Copper	8, as isolated locations except for two adjacent locations at mouth of Swan Island Lagoon	5	
Mercury	79, throughout Study Area between RM 2.8 and RM 10.5. Max HQ in Willamette Cove.	280	SQV based on *Hyalella* biomass is less than the AET and therefore may contribute to false predictions of toxicity
Silver	16, between RM 4.6 and RM 9.2: small, localized areas near RM 4.6, RM 5.7, and in Swan Island Lagoon.	8.6	
PAHs			
Total HPAHs	11, between RM 5.6 and RM 6.4, west, within LPAH footprint.	7	

Table 6-45. Summary of Potential Benthic Community Risk Based on Individual Lines of Evidence

Line of Evidence/COPC	Number and Spatial Extent of Exceedances	Magnitude, Max HQ (Max Probability[a])	Relevance
Total LPAHs	157, between RM 4.6 and RM 11.3. Potentially significant areas at RM 4.3 to RM 4.7, east; RM 5.5 to RM 5.7, east; and RM 5.1 to RM 6.9, west. Max HQ at RM 5.7, west.	1,500	SQVs based on survival endpoints are less than their respective AET and therefore may contribute to false predictions of toxicity
PCBs			
Total PCBs	55, throughout Study Area. Potentially significant areas at RM 2.1 to RM 2.4, east; near RM 3.7, east; in Swan Island Lagoon; RM 8.8 to RM 9.2, west; and near RM 11.3, east. Max HQ at RM 8.8.	62	SQV based on *Chironomous* biomass is less than the AET and therefore may contribute to false predictions of toxicity
SVOCs			
Benzyl alcohol	7, with isolated exceedances between RM 3.7 and RM 11.3.	6.8	
Carbazole	22, between RM 4.7 and RM 6.5, entirely within LPAH footprint.	29	SQV based on *Hyalella* biomass is less than the AET and therefore may contribute to false predictions of toxicity
Dibenzofuran	69, between RM 2.8 and RM 10.9; all but three locations fall within LPAH footprint.	180	SQVs based on biomass endpoints are less than their respective AET and therefore may contribute to false predictions of toxicity
Phenols			
4-Methylphenol	173, throughout Study Area.	31	4-Methylphenol is one of several chemicals contributing false positives to this SQV set.
Pesticides			
beta-HCH	7, primarily between RM 6.2 and RM 6.9, west. Isolated exceedances at RM 2.4, RM 5.7, and RM 8.5	1.9	
delta-HCH	16, at isolated locations between RM 2.7 and RM 9.3.	4.1	SQV based on *Hyalella* biomass is less than the AET and therefore may contribute to false predictions of toxicity
Dieldrin	4, at isolated locations between RM 8.3 and RM 8.8, including Swan Island Lagoon. Max HQ at RM 8.8, west.	17	

Table 6-45. Summary of Potential Benthic Community Risk Based on Individual Lines of Evidence

Line of Evidence/COPC	Number and Spatial Extent of Exceedances	Magnitude, Max HQ (Max Probability[a])	Relevance
Endrin	7, between RM 6.1 and RM 7.3.	1.5	
Endrin ketone	6, at isolated locations between RM 3.7 and RM 9.1. Max HQ at RM 8.8, (co-occurs with dieldrin max HQ).	11	
Sum DDD	48, between RM 5.5 and RM 8.8, primarily west. Potentially significant area between RM 6.8 and RM 7.4. Max at RM 7.4, west.	27	SQVs based on *Chironomus* survival and biomass endpoints are less than their respective AET and therefore may contribute to false predictions of toxicity
Sum DDE	5, at isolated locations between RM 6.8 and RM 8.8 (overlaps with sum DDD area). Max HQ at RM 8.8 (co-occurs with dieldrin max)	2.8	
Sum DDT	3, between RM 7.3 and RM 7.4; overlaps with sum DDD area.	1.6	
Total endosulfan	33, at isolated locations between RM 4 and RM 11.3. Max HQ at RM 7.3	110	SQV based solely on *Chironomus* biomass. Total endosulfan is one of several chemicals contributing false positives to this SQV set.
Predicted Toxicity – LRM			Predictions of toxicity relatively certain above L3 pMax. Relevant to community/population where exceedances spatially aggregated.
Metals			
Chromium	14, at isolated locations throughout Study Area	2.7 (0.97)	
Copper	9, at isolated locations between RM 5.8 and RM 11.1 plus a cluster at mouth of Swan Island Lagoon	5.3 (0.98)	
Lead	21, at isolated locations between RM 4.4 and RM 11.1, with potentially significant area RM 4.4 to RM 4.6, east.	53 (1)	
Mercury	21, at isolated locations between RM 4.4 and RM 10.5.	3.5 (0.94)	

Table 6-45. Summary of Potential Benthic Community Risk Based on Individual Lines of Evidence

Line of Evidence/COPC	Number and Spatial Extent of Exceedances	Magnitude, Max HQ (Max Probability[a])	Relevance
Silver	88, at typically isolated locations between RM 2.9 and RM 9.1.	4.9 (0.94)	
Butyltins			
Tributyltin ion	5, at RM 3.7, RM 8.1, 8.2 and 8.9. Three locations associated with mouth of Swan Island Lagoon	11 (0.96)	
PAHs			
2-Methyl-naphthalene	32, between RM 2.8 and RM 6.5; most co-occur with anthracene exceedances RM 5.4 to RM 6.4.	140 (1)	
Acenaphthene	12, between RM 5.4 and RM 6.4, west; all co-occur with anthracene exceedances.	6.6 (0.79)	
Acenaphthylene	10, between RM 5.4 and RM 6.4, west; all co-occur with anthracene exceedances.	2.8 (0.82)	
Anthracene	56, between RM 2.8 and RM 6.7, mostly RM 5 to RM 6.5, west.	210 (0.99)	
Benzo(a)-anthracene	12, between RM 5.4 and RM 6.4, west; all co-occur with anthracene exceedances.	4.6 (0.85)	
Benzo(b)-fluoranthene	9, between RM 5.4 and RM 6.3, west; all co-occur with anthracene exceedances.	4 (0.84)	
Benzo(g,h,i)-perylene	11, between RM 5.4 and RM 6.3, west; all co-occur with anthracene exceedances.	3.8 (0.84)	
Benzo(k)-fluoranthene	17, between RM 4.6 and RM 6.4; most co-occur with anthracene exceedances RM 5.4 to RM 6.4, west.	3.2 (0.84)	
Chrysene	12, between RM 5.4 and RM 6.4, west; all co-occur with anthracene exceedances.	4.3 (0.85)	
Dibenzo(a,h)-anthracene	10, between RM 4.7 and RM 6.3; most co-occur with anthracene exceedances, west.	4.2 (0.85)	
Fluoranthene	11, between RM 5.4 and RM 6.4, west; all co-occur with anthracene exceedances.	4.1 (0.83)	

Table 6-45. Summary of Potential Benthic Community Risk Based on Individual Lines of Evidence

Line of Evidence/COPC	Number and Spatial Extent of Exceedances	Magnitude, Max HQ (Max Probability[a])	Relevance
Fluorene	23, between RM 2.8 and RM 6.7; all co-occur with anthracene exceedances (primarily west side of river).	9.1 (0.93)	
Indeno(1,2,3-cd)pyrene	11,between RM 5.4 and RM 6.3, west; all co-occur with anthracene exceedances.	3.8 (0.84)	
Phenanthrene	13, between RM 2.8 and RM 6.4; all co-occur with anthracene exceedances (primarily on west side of river).	4.2 (0.76)	
Pyrene	11, between RM 5.4 and RM 6.4, west; all co-occur with anthracene exceedances.	3.3 (0.82)	
Total HPAHs	12, between RM 5.4 and RM 6.4, west; all co-occur with anthracene exceedances.	3.9 (0.83)	
Total LPAHs	27, between RM 2.8 and RM 6.7; all co-occur with anthracene exceedances (primarily on west side of river).	18 (0.96)	
Total PAHs	35, between RM 2.8 and RM 6.5; all co-occur with anthracene exceedances (primarily on west side of river).	20 (0.98)	
PCBs			
Total PCBs	3, isolated at RM 3.7, RM 8.3, and RM 8.8.	12 (0.93)	
SVOCs			
Carbazole	30, between RM 2.8 and RM 7.2; overlaps with anthracene exceedances up to RM 6.5. Co-occur with PAHs except at 3 locations RM 6.8 to RM 7.2	46 (0.98)	
Dibenzofuran	19,between RM 2.8 and RM 6.8; overlaps with anthracene exceedances up to RM 6.7. Co-occur with PAHs except at several locations RM 6.7 to RM 6.8	9.5 (0.95)	

Table 6-45. Summary of Potential Benthic Community Risk Based on Individual Lines of Evidence

Line of Evidence/COPC	Number and Spatial Extent of Exceedances	Magnitude, Max HQ (Max Probability[a])	Relevance
Phthalates			
Dibutyl phthalate	3, at RM 3.7, RM 5.8, and RM 7.4.	2.8 (0.83)	
Phenols			
Phenol	8, between RM 3.7 and RM 10.5	5.2 (0.93)	
Pesticides			
2,4'-DDD	14, between RM 6.5 and RM 7.4, west, with 13 between RM 7.1 and RM 7.4	6.8 (0.95)	
4,4'-DDD	10, between RM 6.3and RM 8.8, west, with all but 3 between RM 7.1 and RM 7.4.	5.9 (0.93)	
4,4'-DDE	6, between RM 7.2 and RM 8.8, west, with all but 1 between RM 7.1 and RM 7.4.	4.5 (0.89)	
4,4'-DDT	14, between RM 7.1 and RM 7.4, west.	10 (0.9)	
cis-Chlordane	14, between RM 6.4 and RM 8.8, west. Half are between RM 7.1 and RM 7.4; otherwise isolated.	39 (0.99)	
delta-HCH	7, between RM 2.7 and RM 7.4; typically isolated, but 3 exceedances between RM 7.1 and RM 7.4, west	2.5 (0.8)	
Sum DDD	13, between RM 6.3 and RM 8.8, west, with all but 3 between RM 7.1 and RM 7.4	6.2 (0.92)	
Sum DDE	10, between RM 6.8 and RM 8.8, west, with all but 4 o between RM 7.1 and RM 7.4.	12 (0.96)	
Total DDx	13, between RM 7.2 and RM 8.8, west, with all but 1between RM 7.1 and RM 7.4.	8 (0.94)	
Petroleum Hydrocarbons[b]			
Diesel range hydrocarbons	35, between RM 2.8 and RM 8.8, mostly RM 6.2 and RM 8.8, west. Isolated exceedances RM 2.8, RM 3.9, and RM 4.8.	17 (1)	

Table 6-45. Summary of Potential Benthic Community Risk Based on Individual Lines of Evidence

Line of Evidence/COPC	Number and Spatial Extent of Exceedances	Magnitude, Max HQ (Max Probability[a])	Relevance
Empirical Tissue Residue			Direct measure of tissue residues under field conditions.
Metals			Tissue-residue effects in invertebrates are uncertain. Because many species can detoxify or sequester metals, thresholds are not reliable as predictive tools.
Arsenic	2, in samples from RM 3.7, east and RM 7.4 (east) under laboratory exposure conditions	1.5	Limited number of toxicological studies (5) was available for derivation of arsenic TRV.
Copper	32, in samples collected throughout the Study Area	2.6	Tissue residues within range that can be bioregulated.
Zinc	34, in samples collected throughout the Study Area	2.2	Tissue residues within range that can be bioregulated.
Butyltins			
Tributyltin ion	1, in sample from the mouth of Swan Island Lagoon	11	Weak correlation between sediment and tissue. TRV is one-fourth the invertebrate tissue-residue threshold that is protective of juvenile salmonids consuming benthic invertebrates. Bioavailability tends to be low when shipyards are source.
PCBs			
Total PCBs	8, in samples from RM 2.3, east; RM 3.7, east; Willamette Cove; RM 6.9, west; Swan Island Lagoon; and RM 11.3, east	7.5	Reasonable relationship between sediment concentrations and tissue residues. Tissue residues exceed TRV in known sediment source areas.
Phthalates			
BEHP	1, in sample from RM 8.8, east, under laboratory exposure conditions	2.8	Limited toxicological studies available to derive TRV.
Pesticides			
4,4'-DDD	1, in sample from RM 6.9, west, under laboratory exposure conditions	1.2	Reasonable relationship between sediment concentrations and tissue residues. Tissue residue exceeds TRV in known sediment source area.
Total DDx	2, in samples from RM 6.9 to RM 7.2, west, under laboratory exposure conditions	3.2	Reasonable relationship between sediment concentrations and tissue residues. Tissue residue exceeds TRV in known sediment source area.

Table 6-45. Summary of Potential Benthic Community Risk Based on Individual Lines of Evidence

Line of Evidence/COPC	Number and Spatial Extent of Exceedances	Magnitude, Max HQ (Max Probability[a])	Relevance
Predicted Tissue Residue			Models developed for chemicals with significant relationship between tissue and sediment concentrations.
Butyltins			
Tributyltin ion	27, predicted near RM 3.7, east; RM 5.7, east; RM 6.2, east; RM 7.4, east; and mouth of Swan Island Lagoon	149	Correlation between sediment and tissue driven by one high point in laboratory-exposed worm. No other receptor displayed a significant relationship between sediment and tissue concentrations, making predictions uncertain. TRV is one-fourth the invertebrate tissue-residue threshold that is protective of juvenile salmonids consuming benthic invertebrates and TRV includes endpoints not relevant to benthic community in the LWR.
PCBs			
Total PCBs	20, predicted near RM 2.2, east; RM 3.7, east; Swan Island Lagoon, and RM 11.3, east	20	Model accurately predicts tissue concentrations within a factor of 4.5. Spatial interpolations affected by density of sampling locations.
Pesticides			
Total DDx	15, predicted for RM 7.2 to RM 7.6, west	10	Model accurately predicts tissue concentrations within a factor of 3.8. Spatial interpolations affected by density of sampling locations.
Surface Water			
Metals			
Zinc	1, during low flow condition at RM 9.7, west.	1.1	
Butyltins			
Monobutyltin	1, during high flow conditions at RM 11.	1.2	TRV is based on TBT; monobutyltin known to be less toxic.
PAHs			
Benzo(a)-anthracene	2, at RM 6.1 and RM 6.2, east in two different events.	10	

Table 6-45. Summary of Potential Benthic Community Risk Based on Individual Lines of Evidence

Line of Evidence/COPC	Number and Spatial Extent of Exceedances	Magnitude, Max HQ (Max Probability[a])	Relevance
Benzo(a)pyrene	3, during three different events; one at RM 6.1 and two at RM 6.2, east.	14	
Naphthalene	10, between RM 6.4 and RM 6.8, east.	50	
PCBs			
Total PCBs	0	< 1	Alternative TRV based on relevant fish and invertebrate receptors.
Phthalates			
BEHP	2, during high flow conditions at RM 4 and in Willamette Cove.	2.3	
Pesticides			
4,4'-DDT	0	< 1	Alternative TRV based on relevant fish and invertebrate receptors.
Total DDx	1, during low-flow conditions at RM 2, east.	1.8	Alternative TRV based on relevant fish and invertebrate receptors.
VOCs			
Ethylbenzene	1,during low flow conditions at RM 6.4, west	1.6	
Trichloroethene	1,during low-flow conditions at RM 6.4, west	4.1	
TZW			Benthic organisms limit exposure to anoxic porewater through a variety of mechanisms.
Metals			
Barium	93, at all 10 TZW investigation areas	1,100	Barium is a natural crustal element; site concentrations do not appear to be linked to anthropogenic sources.
Beryllium	3, at Mobil Oil, Rhone-Poulenc, and Willbridge	2	HQ within a range likely to be reduced by biologically mediated entrainment of overlying water.
Cadmium	10, at Arkema chlorate plant, Mobil Oil, Rhone-Poulenc, and Willbridge	5.8	HQ within a range likely to be reduced by biologically mediated entrainment of overlying water.

Table 6-45. Summary of Potential Benthic Community Risk Based on Individual Lines of Evidence

Line of Evidence/COPC	Number and Spatial Extent of Exceedances	Magnitude, Max HQ (Max Probability[a])	Relevance
Cobalt	3, at Siltronic	3.6	HQ within a range likely to be reduced by biologically mediated entrainment of overlying water.
Copper	1, at Rhone-Poulenc	1.3	HQ within a range likely to be reduced by biologically mediated entrainment of overlying water.
Iron	100, at all 10 TZW investigation areas	250	Iron a natural crustal element; site concentrations do not appear to be linked to anthropogenic sources. TRV not toxicity-based
Lead	4, at Mobil Oil, Gasco, and Rhone-Poulenc	3	HQ within a range likely to be reduced by biologically mediated entrainment of overlying water.
Magnesium	8, at Arkema (chlorate and acid plants) and Rhone-Poulenc	7	HQ within a range likely to be reduced by biologically mediated entrainment of overlying water.
Manganese	105, at all 10 TZW investigation areas	550	Manganese is a natural crustal element; site concentrations do not appear to be linked to anthropogenic sources. TRV was considered within natural range.
Nickel	3, at Arkema chlorate plant, Gasco, and Rhone-Poulenc	1.6	HQ within a range likely to be reduced by biologically mediated entrainment of overlying water.
Potassium	2, at Arkema chlorate plant	3.7	HQ within a range likely to be reduced by biologically mediated entrainment of overlying water.
Sodium	11, at Arkema chlorate and acid plants	55	HQ above a range likely to be reduced by biologically mediated entrainment of overlying water.
Vanadium	6, at Siltronic	19	HQ above a range likely to be reduced by biologically mediated entrainment of overlying water.
Zinc	1, at ARCO	14	HQ above a range likely to be reduced by biologically mediated entrainment of overlying water.
PAHs			
2-Methyl-naphthalene	8, at Siltronic and Gasco	40	HQ above a range likely to be reduced by biologically mediated entrainment of overlying water.

Table 6-45. Summary of Potential Benthic Community Risk Based on Individual Lines of Evidence

Line of Evidence/COPC	Number and Spatial Extent of Exceedances	Magnitude, Max HQ (Max Probability[a])	Relevance
Acenaphthene	24, at Siltronic and Gasco	17	HQ above a range likely to be reduced by biologically mediated entrainment of overlying water.
Anthracene	28, at Siltronic and Gasco	87	HQ above a range likely to be reduced by biologically mediated entrainment of overlying water.
Benzo(a)-anthracene	31, at Siltronic, Gasco, ARCO, Mobil Oil, and Kinder Morgan	1,200	HQ above a range likely to be reduced by biologically mediated entrainment of overlying water.
Benzo(a)pyrene	34, at Siltronic, Gasco, ARCO, and Mobil Oil	2,700	HQ above a range likely to be reduced by biologically mediated entrainment of overlying water.
Benzo(b)-fluoranthene	13, at Siltronic and Gasco	49	HQ above a range likely to be reduced by biologically mediated entrainment of overlying water.
Benzo(g,h,i)-perylene	13, at Siltronic, Gasco and Mobil Oil	66	HQ above a range likely to be reduced by biologically mediated entrainment of overlying water.
Benzo(k)-fluoranthene	10, at Siltronic and Gasco	14	HQ above a range likely to be reduced by biologically mediated entrainment of overlying water.
Chrysene	10, at Siltronic and Gasco	17	HQ above a range likely to be reduced by biologically mediated entrainment of overlying water.
Dibenzo(a,h)-anthracene	8, at Siltronic and Gasco	13	HQ above a range likely to be reduced by biologically mediated entrainment of overlying water.
Fluoranthene	11, at Siltronic and Gasco	17	HQ above a range likely to be reduced by biologically mediated entrainment of overlying water.
Fluorene	36, at Siltronic, Gasco, and Mobil Oil	28	HQ above a range likely to be reduced by biologically mediated entrainment of overlying water.
Indeno(1,2,3-cd)-pyrene	13, at Siltronic, Gasco, and Mobil Oil	61	HQ above a range likely to be reduced by biologically mediated entrainment of overlying water.
Naphthalene	31, at Siltronic, Gasco, and Arkema acid plant)	1,100	HQ above a range likely to be reduced by biologically mediated entrainment of overlying water.
Phenanthrene	36, at Siltronic, Gasco and Mobil Oil	57	HQ above a range likely to be reduced by biologically mediated entrainment of overlying water.

Table 6-45. Summary of Potential Benthic Community Risk Based on Individual Lines of Evidence

Line of Evidence/COPC	Number and Spatial Extent of Exceedances	Magnitude, Max HQ (Max Probability[a])	Relevance
Pyrene	11, at Siltronic and Gasco	15	HQ above a range likely to be reduced by biologically mediated entrainment of overlying water.
SVOCs			
1,2-Dichloro-benzene	5, at Rhone-Poulenc	46	HQ above a range likely to be reduced by biologically mediated entrainment of overlying water.
1,4-dichloro-benzene	2, at Rhone-Poulenc	16	HQ above a range likely to be reduced by biologically mediated entrainment of overlying water.
Dibenzofuran	3, at Siltronic and Gasco	2.2	HQ within a range likely to be reduced by biologically mediated entrainment of overlying water.
Pesticides			
4,4'-DDT	3, at Arkema acid plant	160	HQ above a range likely to be reduced by biologically mediated entrainment of overlying water.
Total DDx	10, at Arkema acid plant and Rhone-Poulenc	280	HQ above a range likely to be reduced by biologically mediated entrainment of overlying water.
VOCs			
1,1-Dichloroethene	2, at Siltronic	1.6	HQ within a range likely to be reduced by biologically mediated entrainment of overlying water.
1,2,4-Trimethyl-benzene	11, at Siltronic	9.6	HQ within a range likely to be reduced by biologically mediated entrainment of overlying water.
1,3,5-Trimethyl-benzene	5, at Siltronic	3	HQ within a range likely to be reduced by biologically mediated entrainment of overlying water.
Benzene	9, at Siltronic and Gasco	30	HQ above a range likely to be reduced by biologically mediated entrainment of overlying water.
Carbon disulfide	4, at Siltronic and Gasco	870	HQ above a range likely to be reduced by biologically mediated entrainment of overlying water.
Chlorobenzene	3, at Arkema acid plant and Rhone-Poulenc	190	HQ above a range likely to be reduced by biologically mediated entrainment of overlying water.

Table 6-45. Summary of Potential Benthic Community Risk Based on Individual Lines of Evidence

Line of Evidence/COPC	Number and Spatial Extent of Exceedances	Magnitude, Max HQ (Max Probability[a])	Relevance
Chloroethane	1, at Gunderson	3.4	HQ within a range likely to be reduced by biologically mediated entrainment of overlying water.
Chloroform	4, at Arkema chlorate and acid plants	21	HQ above a range likely to be reduced by biologically mediated entrainment of overlying water.
cis-1,2-Dichloroethene	5, at Siltronic	110	HQ above a range likely to be reduced by biologically mediated entrainment of overlying water.
Ethylbenzene	15, at Siltronic and Gasco	57	HQ above a range likely to be reduced by biologically mediated entrainment of overlying water.
Isopropylbenzene	10, at Siltronic and Gasco	2	HQ within a range likely to be reduced by biologically mediated entrainment of overlying water.
m,p-Xylene	3, at Siltronic	4.4	HQ within a range likely to be reduced by biologically mediated entrainment of overlying water.
o-Xylene	12, at Siltronic and Gasco	12	HQ above a range likely to be reduced by biologically mediated entrainment of overlying water.
Toluene	11, at Siltronic and Gasco	18	HQ above a range likely to be reduced by biologically mediated entrainment of overlying water.
Total xylenes	13, at Siltronic and Gasco	34	HQ above a range likely to be reduced by biologically mediated entrainment of overlying water.
Trichloroethene	2, at Siltronic	1,900	HQ above a range likely to be reduced by biologically mediated entrainment of overlying water.
Petroleum Hydrocarbons[b]			
Gasoline-Range Hydrocarbons			
C4-C6 aliphatic hydrocarbons	15, at ARCO, Mobil Oil, Gasco, and Siltronic	7.3	HQ within a range likely to be reduced by biologically mediated entrainment of overlying water.
C6-C8 aliphatic hydrocarbons	7, at Siltronic and Gasco	4.3	HQ within a range likely to be reduced by biologically mediated entrainment of overlying water.
C8-C10 aliphatic hydrocarbons	0	0	

Table 6-45. Summary of Potential Benthic Community Risk Based on Individual Lines of Evidence

Line of Evidence/COPC	Number and Spatial Extent of Exceedances	Magnitude, Max HQ (Max Probability[a])	Relevance
C10-C12 aliphatic hydrocarbons	36, at ARCO, Mobil Oil, Gasco, Siltronic, Kinder Morgan, and Willbridge	540	HQ above a range likely to be reduced by biologically mediated entrainment of overlying water.
C8-C10 aromatic hydrocarbons	3, at Gasco	2.7	HQ within a range likely to be reduced by biologically mediated entrainment of overlying water.
Other Contaminants[a]			
Cyanide	34, at Siltronic and Gasco	4,400	HQ above a range likely to be reduced by biologically mediated entrainment of overlying water.
Perchlorate	5, at Arkema chlorate plant	19	HQ above a range likely to be reduced by biologically mediated entrainment of overlying water.

[a] Maximum probability of toxicity as predicted by the LRM.

[b] Not regulated under CERCLA.

BEHP – bis(2-ethylhexyl) phthalate
CERCLA – Comprehensive Environmental Response, Compensation, and Liability Act
COPC – contaminant of potential concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
dw – dry weight
FPM—floating percentile model
HCH – hexachlorocyclohexane
HPAH – high-molecular-weight polycyclic aromatic hydrocarbon
HQ – hazard quotient
L2 – Level 2 (moderate toxicity)
L3 – Level 3 (high toxicity)
LPAH – low-molecular-weight polycyclic aromatic hydrocarbon
LRM – logistic regression model
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
RM – river mile
SQV – sediment quality value
SVOC – semivolatile organic compound
total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT, and 4,4′-DDT)
TRV – toxicity reference value
TZW – transition zone water
VOC – volatile organic compound

6.7.2.4 Qualitative Assessment of Benthic Community Health

Quantitative benthic community structure was not evaluated as a LOE in the BERA. However, twenty-one benthic samples were collected during the Round 2 field investigations to assess the types of organisms present and their abundance for potential use in evaluating tissue residues in field collected organisms(Integral et al. 2004a). Samples were collected between RM 2 and RM 10. The benthic community tended to be dominated by various species oligochaetes, chironomid midges, the clam *Corbicula* spp. and the amphipod *Corophium* spp. Oligochaetes and chironomids are generally interpreted to be pollution-tolerant taxa, although this tolerance was originally based on a response to organic enrichment (Hilsenhoff 1987). Tolerance in these small invertebrates is typically derived from opportunistic behaviors (e.g., small size, short duration to maturity, high egg production) that allow them to exploit temporally or spatially variable resources (e.g., habitat, food) or respond to periodic perturbations. Given the physically dynamic nature of the LWR, the presence of these species may represent a naturally occurring community. This is borne out by a brief review of historical benthic data collected between RM 17 and RM 25, in the Multnomah Channel and the mouth of the Willamette River (Tetra Tech 1993). These data showed that the both the upriver and downstream reaches of the LWR (relative to the Study Area) are dominated by many pollution-tolerant species including oligochaetes, chironomids, and *Corbicula* spp. The amphipod *Corophium* spp. was also dominant in the downstream reach, likely reflecting the influence of the Columbia River. The upstream-most sampling locations sampled by the LWG during Round 2 (between RM 9 and RM 10) were similar in community structure to the historical upstream locations(i.e., dominated by pollution-tolerant taxa).

The distribution of benthic community successional stages documented in sediment profile images was revisited (see Section 2) as part of assessing the overall health of the benthic community. As discussed in Section 2.2.1, successional stage responds to the temporal and spatial stability and habitat quality of the environment in which the benthic organisms reside. Stage 3 communities are considered healthy mature communities typified by larger, deeper-burrowing, longer-lived organisms that reflect the overall lack of environmental perturbation whether physical, chemical, or biological. Stage 1 communities are the earlier colonizers that follow disturbance events or reflect long-term perturbation of an environment. The health of these early communities is likely dependent on the type of disturbance that restarted successional development (i.e., chemical versus physical). Stage 2 communities indicate recovery from some type of disturbance.

In natural environments, Stage 3 communities are the typical or expected stage in fine-grained, depositional environments; by contrast, Stage 1 is the typical or anticipated stage in erosional, frequently disturbed, or physically unstable (e.g., highly depositional or steeply sloped) environments. In organically enriched environments, benthic communities may resemble Stage 2 or even Stage 1 if the degree of enrichment is sufficient. Where chemical contamination is sufficient to affect benthic communities, Stage 2 or Stage 1 communities might also be expected.

The SPI image analysis (SEA 2002) included classification of the physical regime at the point where the image was taken based on grain size; small-scale stratigraphy within the sediment column; boundary roughness at the sediment-water interface; and the presence of bedforms, ripples, or other surface features indicative of sediment transport. Physical classifications are related to the likely transport regime and are defined as erosional, depositional, highly depositional, mixed, and unknown (because of debris or other factors that degrade the image or interfere with the analysis). The information about the physical regime and the community successional stage was used to identify areas where the presence of early successional benthic community stages may result from non-physical factors.

To evaluate the relationship between successional stage and physical regime, biological and physical information was used to reclassify individual SPI locations into expected and unexpected community responses, as presented in Table 6-46.

Table 6-46. Classification of Community Response to Physical Transport Regime

	Physical Regime			
Biological Community	**Highly Depositional**	**Depositional**	**Erosional/ Transport**	**Mixed**
Early (Stage 1)	Expected	Worse than expected; may indicate non-physical factors are affecting community	Expected	Expected
Transitional (Stage 2)	Expected	Worse than expected; may indicate non-physical factors are affecting community	Expected	Expected
Mature (Stage 3)	Better than expected, meets overall goal of community health	Expected	Better than expected, meets overall goal of community health	Better than expected, meets overall goal of community health

Once individual locations were reclassified, the evaluation focused on the locations where community response was worse than expected (potentially due to non-physical factors). Specifically, the evaluation focused on locations where Stage 1 and 2 communities were present in fine-grained depositional environments. An attempt was made to identify other factors influencing the community, including bottom slopes and sediment chemistry. SPI successional stage, grain size type, physical regime, slope, and expected/unexpected community stage classifications are provided in Attachment 4.

Where successional stage of the benthic community could be determined,[74] approximately 48% of the sediment profile image locations within the Study Area were classified as Stage 3 and 49% as Stage 1, with the remaining 3% classified as Stage 2. As shown in Table 2-2, 43 of the early or transitional type communities were located in depositional areas, which was considered an unexpected and possibly deleterious condition. Of those, seven locations within the Study Area occurred in areas with bottom slopes greater than 20%, resulting in a reclassification to “expected” because of likely instability in the bottom substrate. Notes from the SPI analysis suggest that five other locations in the Study Area may have been disturbed or were indicative of historical dredge cuts; at these locations the lower successional stage may be explained in light of past disturbance. The remaining 31 locations where the successional stage did not appear to match the sediment type and physical regime were examined to determine whether they occurred in areas where effects on benthic organisms were predicted or measured. The majority (19) of unexpected community stages occurred between RM 5.0 and RM 9.0; seven occurred in the lower Study Area (below RM 5.0) and five occurred in the upper Study Area (above RM 9.0). Map 6-29 shows locations of the unexpected community successional stages and the benthic risk areas defined by exceedance of L3 SQVs. The reach between RM 5.0 and RM 9.0 also contains the greatest combined area of predicted benthic invertebrate toxicity, based on empirical and predicted sediment toxicity, empirical and predicted tissue-residues, and TZW LOEs. This overlap suggests possible chemical toxicity, among other potential factors, as the reason for the presence of lower successional stages in this reach.

Spatial and temporal variability in the estimates of areas that may be toxic, the classification of community successional stage, and estimates of habitat stability combine to create uncertainty in the analysis of the SPI data. However, the data suggest that the physical environment in the Study Area can explain the condition of the benthic community throughout most parts of this stretch of the river. In over 90% of the images evaluated, the successional stage matched the expected community based on the physical regime, when slope was included as a habitat characteristic. These qualitative results suggest that overall, the benthic community is typical of a large river system that is strongly influenced by physical processes. Potential impacts from sediment contamination appear to be limited to nearshore depositional areas that have received historical releases of contamination.

6.7.3 Risk Conclusions

Toxicity to individual organisms is the primary LOE used to define areas of benthic risk in the Portland Harbor Study Area, although the incidence of toxicity was low. Seven to 26% of empirical bioassays, depending on endpoint, were considered toxic by a L3 response. Predictions of L3 toxicity ranged from 16 to 24% of the chemistry-only samples in the Study Area, depending on SQV model (LRM or FPM) and endpoint. Limited areas were predicted to represent a bioaccumulative risk from either PCBs or DDx. Risks from

[74] Successional community stage could not be determined where debris or hard substrates prevented camera penetration.

water-borne contaminants only occurred in areas with known or likely pathways for discharge to sediment or TZW. Risks from exposure to contaminants in TZW were limited to areas with known or likely pathways for contaminated groundwater discharge.

Benthic risk areas were identified in the following Study Area locations. LOEs contributing to this assessment are discussed for each area:

- RM 2.2 to 2.3, east – Benthic community risk primarily from bioaccumulation of PCBs (adjacent prediction of toxicity overridden by non-toxic empirical bioassay)
- International Terminals – Risks demonstrated by empirical toxicity, accompanied by empirical and predicted PCB bioaccumulation
- RM 4.0 to 4.2, west – Risks based on empirical and predicted toxicity
- RM 4.4 to 4.6, east – Risks based on toxicity predicted by both FPM and LRM; area bounded by locations of non-toxic empirical bioassay results
- RM 4.9, west – Risks based on empirical and predicted toxicity
- RM 5.2 to RM 6.0, west – Risks based on empirical and predicted toxicity, supported by exceedances of TZW TRVs in one area
- RM 6.2 to RM 7.2, west – Risks based on empirical and predicted toxicity (primarily from PAHs), supported by TZW exceedance of TRVs in several areas
- RM 7.2 to 7.4, west – Risks based on empirical and predicted toxicity, supported by TZW exceedance of TRVs in several areas and empirical or predicted bioaccumulation of DDx
- Willamette Cove – Risks based on empirical or predicted toxicity and bioaccumulation of PCBs
- Mouth of Swan Island Lagoon – Risks based on empirical toxicity, empirical TBT accumulation at one location, and predictions of PCB bioaccumulation
- RM 8.8 to 9.2, west – Risks based on empirical and predicted toxicity and on empirical accumulation of BEHP at one location and predictions of PCB bioaccumulation
- RM 11, east – Risks based primarily on predictions of PCB bioaccumulation

In summary, the general conclusions for the benthic risk assessment are as follows:

- Potentially unacceptable benthic risks are highly associated with shoreline areas, slips, and areas of elevated sediment chemical concentrations and represent approximately 7% of the total Study Area.
- Qualitative evidence from the SPI analysis suggests that the benthic community structure is largely physically controlled, with limited areas of potential chemical toxicity.PCBs are the most widespread bioaccumulative COPC associated with

potentially unacceptable benthic risk. DDx compounds are associated with bioaccumulative risk in more localized areas.

- Exceedance of SQVs for metals indicating potentially unacceptable risk to the benthic community occurred primarily between RM 5.6 and RM 9.2 of the Study Area, typically on a small scale (e.g., several samples).
- Exceedance of site-specific SQVs for PAHs indicating potentially unacceptable risk to the benthic community is associated with known source areas between RM 5.0 and RM 7.0 on the west side of the river.
- Exceedance of SQVs for pesticides indicating potentially unacceptable risk to the benthic community appears to be limited to areas on the west side of the river between RM 6.2 and RM 8.8.

7.0 FISH RISK ASSESSMENT

This section presents the draft final BERA for fish in the Study Area. For fish receptors, multiple exposure routes were evaluated for the media by which fish may be exposed to sediment contaminants (see Figure 3-2).

To address the different ways fish may be exposed to sediment contaminants, nine receptors representing four general feeding guilds were evaluated:

- Invertivorous fish – sculpin, peamouth, and juvenile Chinook salmon[75]
- Omnivorous fish – largescale sucker, carp,[76] and pre-breeding white sturgeon
- Piscivorous fish – smallmouth bass and northern pikeminnow
- Detritivorous fish – Pacific lamprey ammocoetes

Four primary quantitative LOEs were used to characterize risks to fish:

- The tissue-residue LOE, wherein COPC residues in composite tissue samples of the receptor species were compared with tissue-residue TRVs
- The dietary LOE, wherein estimated dietary COPC doses were compared with dietary-dose TRVs
- The surface water LOE, wherein surface water COPC concentrations were compared with water concentrationTRVs
- The TZW LOE, wherein shallow TZW concentrations were compared with water concentration TRVs

Direct exposure of benthic fish to PAHs in sediment was also evaluated as a semi-quantitative LOE per EPA's Problem Formulation (Attachment 2). This evaluation included an assessment of the apparent health of pre-breeding sturgeon.

HQs were calculated by dividing medium-specific (i.e., tissue, sediment, water) EPCs by their respective effects thresholds; these HQs were used in the risk characterization. Several factors had to be considered when estimating EPCs for the dietary LOE, including feeding rates, foraging areas, and diets. Based on these data, dietary-dose TRVs (in mg/kg bw/day) were converted to receptor-specific threshold tissue and sediment concentrations (i.e., TTCs and TSCs, respectively, in mg/kg) to facilitate direct comparison to media concentrations.

[75] Juvenile Chinook salmon and Pacific lamprey ammocoetes are special status species (i.e., federally threatened and an Oregon State sensitive species of special concern to tribes, respectively) and were evaluated at the organism level. All other fish receptors were evaluated at the population level.

[76] Per EPA's Problem Formulation (Attachment 2), carp were selected as a surrogate fish receptor for the evaluation of dioxins and dioxin-like PCB congeners only.

In general, the selected TRVs express the lowest-observed-adverse-effect levels, which represent the threshold of exposure where effects have been observed to occur in relatively sensitive species. TRVs were selected from published studies and approved by EPA for use in the BERA. LOAEL TRVs provide a basis for evaluating whether exposure concentrations are at or above a level that cause a significant adverse effect on survival, growth, or reproduction of organisms in experimentally exposed laboratory populations relative to control populations.

This approach follows the conventional practice in ecological risk assessment of using organism-level TRVs defined in this manner to evaluate the potential for effects on populations. No explicit fish population modeling was included in this BERA. Rather, the BERA assesses whether COPCs occur at concentrations that have been shown to affect the survival, growth, or reproduction of aquatic organisms in the laboratory. If so, then the COPC is identified as posing potential risk to fish populations, triggering a semi-quantitative risk characterization that considers the spatial extent and magnitude of organism-level TRV exceedances, and the quality and relevance of the organism-level TRV as a predictor of a population or community level effects.

As documented in EPA's Problem Formulation (Attachment 2), receptors that are threatened, endangered, otherwise protected under federal laws, or of particular cultural significance (i.e., special status species including juvenile Chinook salmon and Pacific lamprey ammocoetes) were evaluated at an organism level by comparing EPCs to the NOAEL. NOAELs represent the highest experimental exposure level at which no adverse effects were observed.

All of the fish COPCs identified through the SLERA and refined screening process were evaluated in this assessment. Risk characterization was a winnowing process that allowed proportionally more effort to be focused on the COPC-receptor combinations with the potential for unacceptable risk, incorporating principles (screening and iterative refinement) of ecological risk assessment (EPA 1997). For all but the TZW LOE, the risk evaluation occurred in two or, in the case of the dietary LOE, three steps, progressing from more conservative to more realistic estimates of exposure. For the TZW LOE, data were sufficient only to conduct the first step. For each step, if risk estimates indicated a potentially unacceptable risk, then the exposure assumptions were developed in greater detail taking into account receptor-specific appropriate exposure areas and, for the dietary LOE, multi-species diets.

- **Step 1 –** HQs were calculated for all COPCs on a sample-by-sample basis. This calculation was performed in accordance with the methods described in the EPA's Problem Formulation (Attachment 2). This analysis was used to identify the sampling locations within the Study Area that contribute a greater proportion of exposure.
- **Step 2 –** Because individual samples do not represent "a conservative estimate of the average chemical concentration in an environmental medium… for each exposure unit within the site" (as per EPA guidance on calculating EPCs (EPA

2002a)), HQs based on individual composite samples are not appropriate estimates of risks to receptor populations in the Study Area. HQs were therefore calculated over relevant exposure scales based on species-specific foraging range assumptions. The foraging range assumptions and resulting foraging areas used for exposure analyses and risk characterization for each fish receptor are presented in Table 7-1. EPCs were calculated considering all pertinent data from throughout the specified foraging areas. For the dietary LOE, an additional conservative assumption was included in Step 2, wherein HQs were calculated separately for each prey species that the receptor might consume.

- **Step** 3 **–** For the dietary LOE, HQs reflect the diversity of prey items that the receptor is assumed to ingest. Risk conclusions for each LOE are based on the final step (i.e., Step 1 for TZW, Step 2 for tissue residue and surface water, and Step 3 for the dietary LOE).[77]

Table 7-1. Summary of Fish Receptor-Specific Exposure Areas

Receptor	Exposure Scale[a]	Exposure Area
Largescale sucker, juvenile white sturgeon, juvenile Chinook salmon, peamouth, Pacific lamprey ammocoetes	Site-wide	RM 1.9 to RM 11.8 (Study Area)
Sculpin	0.1 mile	Individual sampling locations
Smallmouth bass, northern pikeminnow	1-mile increments of the Study Area	RM 1.5 to 2.5; RM 2.5 to 3.5; RM 3.5 to 4.5; RM 4.5 to 5.5; RM 5.5 to 6.5; RM 6.5 to 7.5; RM 7.5 to 8.5; Swan Island Lagoon, RM 8.5 to 9.5; RM 9.5 to 10.5; RM 10.5 to RM 11

[a] The rationale for the exposure scale is presented in Attachment 13.

RM – river mile

[77] As agreed to between EPA and LWG on October 15, 2010 (LWG 2010).

Uncertainty Associated with Juvenile White Sturgeon Exposure Scale

A site-wide exposure scale of the 10-mile Study Area was assumed for the juvenile white sturgeon; however, there is uncertainty associated with this assumption. Some studies suggest that sturgeon can show strong site fidelity (Veinott et al. 1999), while other studies indicate that individual sturgeon can have large ranges (DeVore and Grimes 1993). Juvenile (pre-breeding) white sturgeon were targeted for collection from the Study Area based on the assumption that they would have the longest exposure time, longer than other life stages because sturgeon are migratory.

Although none of the Round 3 pre-breeding sturgeon caught from the Study Area were PIT-tagged, one legal-sized sturgeon collected and analyzed from the Study Area in March 2007 as part of Round 3 sampling had been previously tagged with a spaghetti wire tag by WDFW. The age of this tagged sturgeon based on a pectoral fin ray sample was 7 years old. Per WDFW (2007), the sturgeon was originally tagged on June 6, 2006, at Rocky Point, which is located along the west shore of Grays Bay near the Pacific/Wahkiakum counties border on the Washington side of the Columbia River. The initial tagging location was approximately 72 miles from the location where the sturgeon was collected in the Study Area, supporting a much larger home range than that assumed for juvenile sturgeon (the 10-mile stretch of the Study Area).

The risk characterization methods for each fish LOE are presented in more detail in the following subsections: Section 7.1, tissue-residue assessment; Section 7.2, dietary assessment; Section 7.3, surface water assessment; Section 7.4, TZW assessment; and Section 7.5 assessment of direct exposure to PAHs in sediment.

Section 7.6 presents the overall risk conclusions for fish, including the identification of potentially unacceptable risks and the general uncertainty analysis for the fish risk assessment. Specific uncertainties associated with each LOE are discussed in the individual LOE sections (7.1 through 7.5). Section 7.6 provides the final determination of the ecological significance of risk associated with each COPC considering the spatial extent and magnitude of HQs ≥ 1, the key uncertainties in the exposure assessment, the key uncertainties in the effects characterization, and a qualitative WOE analysis of the various LOEs.

Figure 7-1 presents a flowchart of the fish risk assessment section organization.

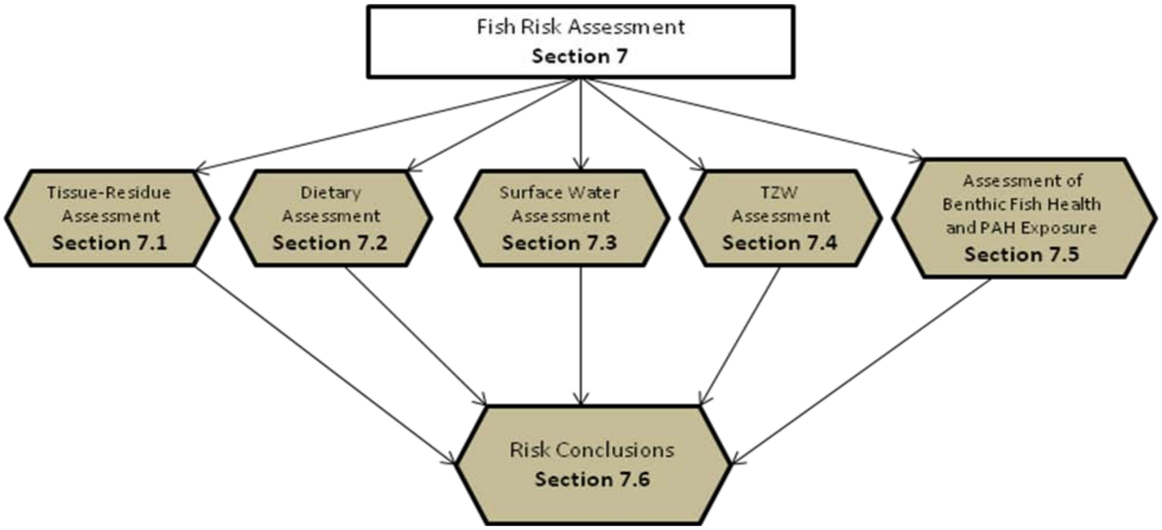

Figure 7-1. Overview of Fish Risk Assessment Section Organization

7.1 TISSUE-RESIDUE ASSESSMENT

Tissue-residue assessment is one of the LOEs used to evaluate risks to fish receptors. The term tissue residue refers to a COPC concentration in a tissue, in this case, composite samples of whole-body fish tissue. The tissue-residue LOE incorporates multiple exposure pathways from contaminated sediment to fish (i.e., dietary ingestion, direct contact with water and sediment).

The details of the tissue-residue assessment are presented as follows:

- Section 7.1.1 presents the general approach.
- Section 7.1.2 lists the COPCs identified for all receptors evaluated in the tissue-residue LOE.
- Section 7.1.3 describes how exposure concentrations were derived. Exposure data in this assessment are represented by measured or predicted chemical concentrations in tissue. All tissue chemical concentrations and calculated UCLs are presented in Attachment 4. The development of the BSARs used to predict tissue concentrations are presented in Attachment 8.
- Section 7.1.4 summarizes the effects data. Effects data in this assessment are represented by baseline tissue NOAELs and LOAELs. Additional details on the selected tissue-residue TRVs are presented in Attachment 9.
- Section 7.1.5 presents the risk characterization results and associated uncertainties. COPCs with HQs ≥ 1 are further assessed in the fish risk conclusions section (Section 7.6).

Figure 7-2 shows how the fish tissue-residue assessment section is organized.

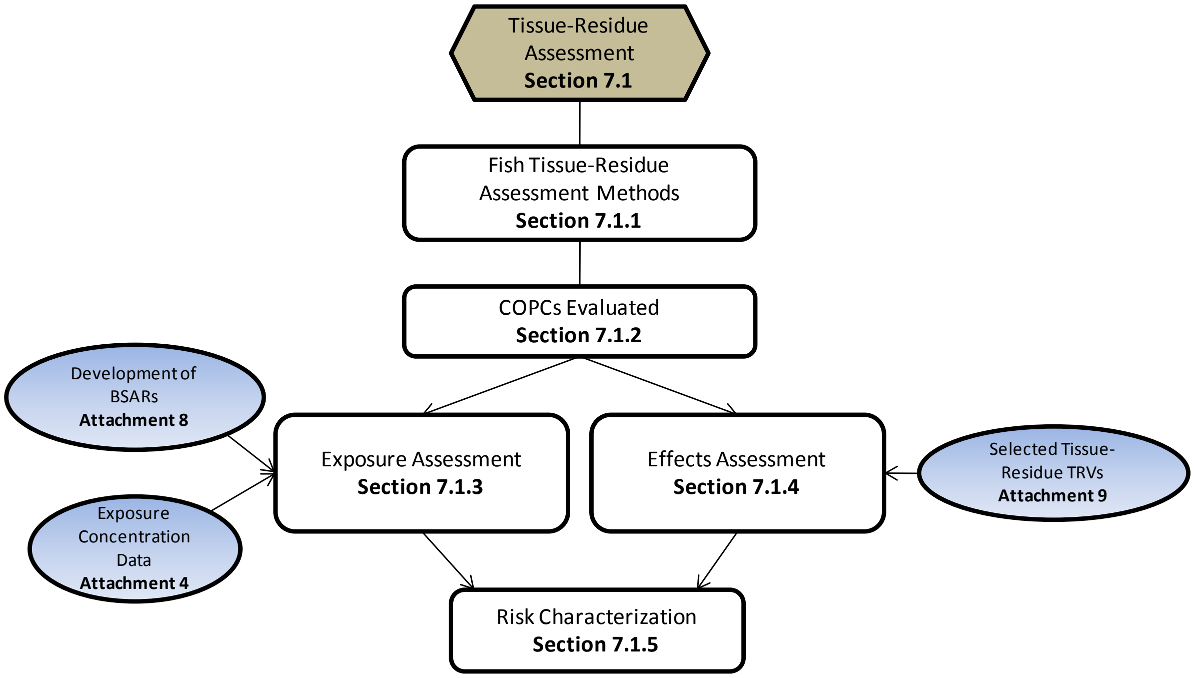

Figure 7-2. Overview of Fish Tissue Assessment Section Organization

7.1.1 Fish Tissue-Residue Assessment Methods

Receptor-specific fish tissue-residue COPCs were identified in the SLERA and refined screen using screening-level TRVs (Attachment 5). For these COPCs, HQs were calculated by comparing chemical concentrations in composite samples of whole-body fish tissue to baseline tissue TRVs. Analysis of tissue-residue risks occurred in two steps, progressing from more conservative to more realistic estimates of exposure and risk:

- **Step 1** – HQs were calculated on a sample-by-sample basis.
- **Step 2 –** For receptor-COPC pairs resulting in a Step 1 maximum HQ ≥ 1, HQs were calculated by receptor-specific exposure area (Table 7-1).

Potentially unacceptable risks were identified for those COPCs with Step 2 HQs ≥ 1.[78] Quantitative risk results (i.e., magnitude and spatial distribution of HQs ≥ 1); uncertainties about exposures, effects, and risk; and comparisons to background concentrations are presented in the tissue-residue LOE risk characterization (Section 7.1.5). WOE across LOEs is discussed in the risk conclusions for fish (Section 7.6).

[78] As agreed to between EPA and LWG on October 15, 2010 (LWG 2010).

7.1.2 COPCs Evaluated

Receptor-COPC pairs were identified in the SLERA and refined screen (Attachment 5). Table 7-2 presents the fish tissue COPCs. All receptor-COPC pairs shown in the table were evaluated, with the exception of aluminum and BEHP; baseline TRVs could not be established for either COPC. Aluminum was screened in as a COPC in the SLERA and refined screen because tissue residues in five fish receptors (i.e., largescale sucker, juvenile white sturgeon, peamouth, sculpin, and Pacific lamprey) exceeded the aluminum screening-level TRV.[79] However, per EPA (EPA 2008e), aluminum was not further evaluated as a tissue COPC because of the lack of a reliable effect threshold value for the BERA. Tissue concentrations of BEHP were compared with a NOAEL TRV and tissue concentrations of aluminum and BEHP in Study Area fish tissue were compared with upriver tissue data; results are discussed in the uncertainty analysis (Section 7.1.5.2.2).

Table 7-2. Fish Tissue-Residue COPCs

	Receptor							
COPC	Large-scale Sucker	Juvenile White Sturgeon	Juvenile Chinook Salmon	Pea-mouth	Sculpin	Small-mouth Bass	Northern Pike-minnow	Pacific Lamprey
Metals								
Aluminum[a]	X	X		X	X			X
Antimony						X		
Cadmium						X		
Chromium	X	X						
Copper					X			X
Lead				X		X		
Mercury							X	
Zinc			X					
Phthalates								
BEHP[a]	X				X	X		
PCBs								
Total PCBs	X				X	X	X	
Pesticides								
4,4′-DDD[b]	X		X		X	X		X
4,4′-DDT[b]					X			

[79] The aluminum SL TRV was based on the 5th percentile of LOAELs reported in Dyer et al. (2000).

Table 7-2. Fish Tissue-Residue COPCs

	Receptor							
COPC	Large-scale Sucker	Juvenile White Sturgeon	Juvenile Chinook Salmon	Pea-mouth	Sculpin	Small-mouth Bass	Northern Pike-minnow	Pacific Lamprey
beta-HCH					X			
Total DDx	X				X	X	X	

[a] Aluminum and BEHP were identified as COPCs based on the SLERA and refined screen; however, baseline HQs were not calculated because baseline TRVs could not be identified for aluminum or BEHP (Attachment 9).

[b] 4,4′-DDD and 4,4′-DDT were identified as COPCs in the SLERA (Attachment 5). These DDT metabolites were further evaluated in this assessment as total DDx and not as individual metabolites.

BEHP – bis(2-ethylhexyl) phthalate
COPC – contaminant of potential concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
HQ – hazard quotient
HCH – hexachlorocyclohexane
PCB – polychlorinated biphenyl
SLERA – screening-level ecological risk assessment
total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT and 4,4′-DDT)
TRV – toxicity reference value

Eight COIs could not be evaluated by the fish tissue-residue LOE because no toxicological data associated with tissue residues were available (Table 7-3). These COIs include two butyltins that are breakdown products of TBT (i.e., monobutyltin and dibutyltin). A tissue-residue TRV is available for TBT, which was identified as a COPC and which is the most toxic butyltin (EPA 1991). Risks due to mono- and dibutyltin are assumed to be lower than those of TBT, which are described in this section.

Table 7-3. Fish Tissue COIs with No Screening-Level TRVs

COI	
Metals	
Manganese	
Butyltins	
Monobutyltin ion[a]	Dibutyltin ion[a]
SVOCs	
Benzoic acid	Bis(2-chloroethoxy) methane
Benzyl alcohol	
Phenols	
4-Chloro-3-methylphenol	4-Nitrophenol

[a] Because TBT is the most toxic butyltin, risks due to mono- and dibutyltin are assumed to be lower than those of TBT.

COI – contaminant of interest
SVOC – semivolatile organic compound
TBT – tributyltin
TRV – toxicity reference value

TRVs were exceeded by at least one DL for 10 COIs whose risk was not evaluated: BEHP, butyl benzyl phthalate, dibutyl phthalate, diethyl phthalate, dimethyl phthalate, hexachlorobutadiene, endrin, and alpha-, beta-, and delta-HCH (see Table 5-5). These 10 tissue COIs were not retained as COPCs for at least one fish receptor because no detected concentrations exceeded baseline TRVs.

Uncertainties Associated with Metals in the Tissue-Residue Approach

Eight inorganic metals were identified as COPCs for fish. The utility of the tissue-residue approach for inorganic metals has been questioned because the uptake, distribution, and disposition of inorganic metals are typically species-specific and governed by highly specific biochemical processes (evolved at least in part to regulate internal concentrations of essential metals) that alter the metal form and involve facilitated or active transport. For example, some organisms take up metal and sequester it into "storage" compartments in chemical forms that have little toxicological potency, whereas other organisms actively excrete excess metals (EPA 2007e). In general, non-essential metals are not actively regulated, at least not to the same degree as essential metals like copper and zinc. However, non-essential metals may still be regulated to various degrees because the mechanisms for regulating essential metals are not metal-specific (Phillips and Rainbow 1989). Cadmium and lead are not essential metals (Luoma and Rainbow 2008), but fish are considered partial regulators of cadmium and lead, meaning that these fish tend to use a combination of active regulation and storage (Phillips and Rainbow 1989). Aluminum is likewise a non-essential metal, as it has no known biological function, yet aluminum concentrations in fish appear to be regulated. Cleveland et al. (1991), for example, found that aluminum concentrations in brook trout initially increased during a 56-day exposure to a nominal aluminum concentration of 200 µg/L, but then declined during the remainder of the exposure period. Given that aluminum is a gill toxicant, and concentrations in fish have been shown to peak and then decline over long-term continuous aluminum exposures, whole-body aluminum concentration is not a strong indicator of potential aluminum toxicity in fish. Differences in regulation of tissue concentrations of metals by fish create difficulties when interpreting the toxicological significance of whole-body residues and increase the uncertainty when extrapolating across different exposure routes, durations, and species. Consideration of these processes suggests that inorganic metal tissue residues are a weak LOE. The uncertainty of this LOE for inorganic metals is considered in the evaluation of COPCs across multiple LOEs in Section 7.6.

7.1.3 Exposure Assessment

EPCs were derived to be conservative estimates of average COPC concentrations in whole-body tissue residues for each exposure unit within the Study Area, per EPA guidance (EPA 2002a).

7.1.3.1 Empirical Tissue EPCs

This section presents a summary of the methods used to derive EPCs from empirical data on tissue residues. Uncertainties associated with these values are also noted. Tissue EPCs were estimated from detected COPC concentrations in composite samples collected from the Study Area. Two steps were followed:

- **Step 1 –** EPCs were calculated from tissue concentrations in composite samples collected from throughout the Study Area. Maps 4-6 through 4-13 present the composite sample locations for all of the fish receptors. COPC concentration data for each composite sample are presented in Attachment 4.

- **Step 2 –** For receptor-COPC pairs resulting in a Step 1 maximum HQ ≥ 1, EPCs were then calculated as the UCL within the receptor-specific exposure areas (Table 7-1). Where data were insufficient to allow calculation of a UCL, the maximum concentration was used to represent the EPC. Because sculpin tissue samples were composited over an area roughly equal to the sculpin exposure scale, individual composite sample values were used as EPCs.

Tissue EPCs based on a UCL were calculated using ProUCL Version 4.0 software (EPA 2007f). EPA's ProUCL software tests the goodness of fit for a given dataset and then computes the appropriate 95th UCL. 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 was performed, the software conducted an analysis of the data to determine the most appropriate UCL and made a recommendation, which was then used as the EPC for the risk calculations. A minimum of six detected concentrations is required to derive a UCL (EPA 2007f). Attachment 4 presents the summary statistics (i.e., minimum, maximum, and mean COPC concentrations), distribution types, ProUCL-recommended UCLs, and tissue EPCs for each COPC. Tissue data used to calculate UCLs were collected under EPA-approved quality assurance project plans (QAPPs) for the purpose of exposure calculation in this BERA (as described in Section 4). Under EPA guidance (EPA 2007f), both composite and discrete samples are appropriate for calculation of UCLs using ProUCL software.

7.1.3.2 Predicted Tissue EPCs

Sculpin EPCs for locations without empirical tissue-residue data (Map 4-10) were predicted from surface sediment concentrations using a site-specific bioaccumulation model.[80] The predictive models used in the BERA are the same models used to generate risk-based PRGs for the FS. The models are presented in the draft bioaccumulation modeling report for the Portland Harbor RI/FS (2009b).

A mechanistic model was used to predict total PCB, pesticide, and dioxin and furan concentrations.[81] The mechanistic model is appropriate only for persistent hydrophobic organic chemicals (Arnot and Gobas 2004). Site-specific statistical bioaccumulation models (BSARs) were used for sculpin tissue COPCs that were not suitable for the mechanistic model and that met appropriate regression analysis assumptions (statistically significant positive slope [$p \leq 0.05$] and $r^2 \geq 0.30$).

[80] Per the EPA Problem Formulation (Attachment 2), tissue chemical concentrations were to be predicted for both sculpin and smallmouth bass using localized sediment chemical concentrations in those areas where tissue data were not collected. However, no tissue chemical concentrations were predicted for smallmouth bass because samples were available to represent each home range (1-mile segment) within the Study Area (Map 4-11).

[81] Because dioxins and furans were not tissue COPCs for sculpin (or any other fish receptor), the mechanistic model was not used to predict dioxin and furan sculpin tissue concentrations.

Table 7-4 presents the sculpin COPCs and the selected models used to predict tissue concentrations. Total PCBs, total DDx, and beta-HCH were modeled mechanistically. The only two sculpin COPCs that were not modeled mechanistically were copper and BEHP. Neither met the statistical criteria for BSAR development. Copper and BEHP data showed no relationship between co-located sediment and sculpin tissue concentrations. This lack of relationship suggests that the organisms are bioregulating their tissue residues (e.g., for copper, an essential metal), that the source of the COPC is not limited to local sediments, or both. In the absence of either an empirical relationship between co-located sediment and tissue concentrations, or a mechanistic basis for relating the two, no sediment COPC bioaccumulation models could be developed for copper and BEHP. The predicted sculpin tissue EPCs for total PCBs, total DDx, and beta-HCH are presented in Attachment 4.

Table 7-4. Sculpin COPCs and Selected Models Used to Predict Tissue Concentrations

Sculpin COPC	Tissue Concentration Predicted?	Selected Model
Metals		
Copper	No[a]	NA
Phthalates		
BEHP	No[a]	NA
PCBs		
Total PCBs	Yes	Mechanistic model
Pesticides		
beta-HCH	Yes	Mechanistic model
Total DDx	Yes	Mechanistic model

[a] Site-specific BSARs were not selected for these COPCs because these COPCs did not meet the appropriate BSAR analysis assumptions (Windward 2009b), did not have a statistically significant positive slope ($p \leq 0.05$), or had an $r^2 \geq 0.30$.

BEHP – bis(2-ethylhexyl) phthalate
BSAR – biota-sediment accumulation regression
COPC – contaminant of potential concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
HCH – hexachlorocyclohexane
NA – not applicable
total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT and 4,4′-DDT)

7.1.4 Effects Assessment

This section presents the tissue TRVs that were selected to characterize effects for fish receptor-COPC pairs and their uncertainties. These TRVs are used in combination with EPCs to characterize risk (Section 7.1.5).

Tissue TRVs were derived in cooperation with EPA based on the agency's August 5, 2008, revised TRV methodology (EPA 2008g). Details and decisions between LWG and EPA regarding the development of tissue TRVs are presented in Attachment 9. TRVs were derived by compiling and reviewing acceptable existing toxicological studies for all fish tissue COPCs. Acceptable tissue TRV studies included all fish toxicological studies in which measurements were made of tissue-residue contaminant concentrations in whole-body tissues that were associated with measured adverse effects on survival, growth, and reproduction. Studies reporting adverse effects on behaviors were included if the behavior could be related to a survival, growth, or reproductive endpoint or if otherwise directed by EPA (2008b, 2009d). Effect thresholds from the acceptable studies were compiled and used to develop TRVs as follows.

If fewer than five acceptable toxicity studies were available for a given COPC, literature-based LOAEL and NOAEL values were selected. The lowest effect-level tissue chemical concentration was selected as the fish tissue-residue LOAEL. A NOAEL was then derived from the same study that yielded the selected LOAEL. Methods for NOAEL derivation are provided in Attachment 9.

If five or more acceptable toxicity studies were available for a given COPC, an SSD was developed using the LOAEL data, and a 5th and 10th percentile LOAEL were selected. An SSD displays effect threshold data as a plot of the toxicity data for each species on the x-axis and the cumulative probability on the y-axis. Details on the derivation of SSDs are presented in Attachment 9.

Tenth percentile and literature-based LOAELs were used to assess risk to the fish populations. Juvenile Chinook salmon and Pacific lamprey ammocoetes risks were evaluated using 5th percentile LOAEL or literature-based NOAEL TRVs. The selection of low percentiles in acceptable studies of effects is consistent with AWQC (Stephan et al. 1985) and other precedents in the field of ecotoxicology for protection of special-status species (e.g., Meador et al. 2002b). A final check of the derived 5th percentile TRVs was conducted to ensure that LOAELs were not reported for Chinook salmon or lamprey at lower concentrations.

Table 7-5 presents the selected TRVs. TRVs based on SSDs were developed for five of the COPCs (cadmium, copper, mercury, total PCBs, and total DDx). For an additional four COPCs (chromium, lead, zinc, and beta-HCH), limited toxicological data were available (i.e., four or fewer studies), and TRVs were based on the lowest thresholds reported in the reviewed literature. TRVs based on limited toxicological data are highly uncertain and may contribute to either an over- or underestimation of risks associated with a given tissue residue in Study Area fish. Baseline TRVs could not be developed for two COPCs (BEHP and aluminum). No data were identified for aluminum. In the only toxicological study identified for BEHP, fathead minnows were exposed to BEHP for 56 days, with no adverse effects on growth or survival reported at a tissue residue of 9.6 µg/g ww (Mehrle and Mayer 1976). As agreed to by LWG and EPA (EPA 2008b), no acceptable literature-based LOAEL could be derived from this study, and as an uncertainty analysis, BEHP concentrations were compared with only the literature-based NOAEL reported in this study (≥ 9.6 µg/g ww).

In the derivation of TRVs based on SSDs, an ACR was applied to all tissue-residue LOAELs from studies lasting 30 days or less and based on a survival endpoint. ACRs were applied under the assumption that concentrations required to elicit acute mortality are generally higher than those that reduce growth or reproduction. The contaminant-specific ACRs were those reported in Raimondo et al. (2007), as determined from an extensive dataset of 456 same-species pairs of acute and chronic data from multiple species and chemicals. While these ACRs are an improvement on generic safety factors, their application to all mortality studies is conservative; the ACRs were calculated as the ratio of the LC50 or EC50 to a chronic NOAEL or the MATC, with MATC being the geometric mean of the NOAEL and the LOAEL determined from growth, reproduction, or survival endpoints.

Several LOAELs (four cadmium, three copper, one mercury, seven DDx, and nine PCB LOAELs) were excluded from TRV derivation, partly because contaminant residues were measured in eggs or embryos rather than whole-body fish. These studies were excluded because egg or embryo data are not directly comparable to the contaminant concentration data for fully formed fish that were used to characterize receptor exposure in the Study Area. In many cases, LOAELs based on egg residues are lower than those based on more mature fish.

From reported ratios of chemical concentrations in maternal adult fish to those in unfertilized eggs (Niimi 1983), it is possible to estimate the concentrations in adult fish that would produce eggs at specified concentrations. However, extrapolation of effect thresholds in eggs to whole-body fish based on these relationships is higly uncertain because so few studies reporting such ratios are available and because the ratios are likely to vary considerably with factors such as chemical, species, lipid content, moisture content, and experimental conditions. Inclusion of egg-residue data in SSDs without use of an egg-to-adult correction faction would have likely resulted in lower TRVs; the resulting degree of uncertainty, however, was judged too great for use in this BERA (see Attachment 9 for detailed documentation of decisions between LWG and EPA regarding the development of tissue TRVs).

In some cases, studies in which fish were exposed to contaminants in the field or in the laboratory to environmental media collected from a contaminated site were not considered when deriving TRVs. In the case of PCBs and DDTs, inclusion of these studies would have resulted in lower TRVs. These studies were not included because fish under field conditions are exposed to complex mixtures; isolating effects due to (and identifying the corresponding effect threshold for) any given component of the mix cannot be performed reliably at present. Additionally, controlling for other factors unique to the field location makes extrapolation from that site to the Portland Harbor Study Area highly uncertain.

Table 7-5. Selected Fish Whole-Body Tissue TRVs

COPC	TRV (mg/kg ww) 5th Percentile LOAEL or Literature-Based NOAEL	10th Percentile or Literature-Based LOAEL	Derivation	Key Uncertainties
Metals				
Antimony	NC	1.1	LOAEL was derived from Doe et al. (1987).	Only one acceptable toxicity study was identified. Mortality LOAEL from a 30-day study was divided by a default ACR of 8.3. Use of tissue LOE for inorganic metals is uncertain. TRV may over- or underestimate risks.
Cadmium	0.17	0.22	LOAELs are based on 5th and 10th percentile SSD, respectively.	Use of tissue LOE for inorganic metals is uncertain. TRVs may over- or underestimate risks. Studies reporting adverse effects associated with egg or embryo residues were not included in SSD (see discussion following this table).
Chromium	NC	44.1	LOAEL was derived from Roling et al. (2006).	Only one acceptable toxicity study was identified. Use of tissue LOE for inorganic metals is uncertain. TRV may over- or underestimate risks.
Copper	2.8	3.1	LOAELs are based on 5th and 10th percentile SSD, respectively.	SSD was derived from only five studies; fish actively regulate tissue copper concentrations and therefore TRVs may over- or underestimate risks. Six toxicity studies were eliminated from SSD because LOAEL was below the nutritional sufficiency threshold (2.2 mg/kg ww) for some but not all species. Studies reporting adverse effects associated with egg or embryo residues were not included in SSD (see discussion following this table).
Lead	NC	4.0	LOAEL was derived from Holcombe et al. (1976).	There was limited toxicity data and an insufficient number of studies for SSD. Use of tissue LOE for inorganic metals is uncertain. TRV may over- or underestimate risks.
Mercury	0.45	0.53	LOAELs are based on 5th and 10th percentile SSD, respectively.	Studies reporting adverse effects associated with egg or embryo residues were not included in SSD (see discussion following this table).
Zinc	34	36	NOAEL and LOAEL were derived from Spehar (1976).	Only two acceptable toxicity studies were identified; fish actively regulate tissue zinc concentrations and therefore, TRVs may over- or underestimate risks.
Phthalates				
BEHP	>9.6	NA	NOAEL was derived from Mehrle and Mayer (1976).	There were insufficient toxicity data to derive a LOAEL. Only one acceptable unbounded NOAEL was identified; unbounded NOAEL cannot conclusively indicate unacceptable risk.

Table 7-5. Selected Fish Whole-Body Tissue TRVs

COPC	TRV (mg/kg ww) 5th Percentile LOAEL or Literature-Based NOAEL	TRV (mg/kg ww) 10th Percentile or Literature-Based LOAEL	Derivation	Key Uncertainties
PCBs				
Total PCBs	0.42	0.93	LOAELs are based on 5th and 10th percentile SSD, respectively.	The TRV may overestimate PCB toxicity because the lowest five LOAELs included in the SSD are associated with significant uncertainty (see Attachment 9); studies reporting adverse effects associated with egg or embryo residues were not included in SSD (see discussion following this table).
Pesticides				
beta-HCH	NC	0.20	LOAEL is based on surrogate (gamma-HCH); LOAEL was derived from Schimmel et al. (1977).	LOAEL was based on a surrogate; only three acceptable toxicity studies were identified; TRVs may over- or underestimate risks.
Total DDx	0.77	1.6	LOAELs are based on 5th and 10th percentile SSD, respectively.	Studies reporting adverse effects associated with field exposures and egg or embryo residues were not included in the SSD; field studies were generally below the identified 10th percentile TRV and inclusion without correction for extrapolating from eggs to adults would have resulted in lower TRVs (see discussion following this table).

ACR – acute-to-chronic ratio
BEHP – bis(2-ethylhexyl) phthalate
COPC – contaminant of potential concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
HCH – hexachlorocyclohexane LOAEL – lowest-observed-adverse-effect level
LOE – line of evidence
NA – not applicable (no acceptable TRV was derived)
NC – not calculated (NOAEL not needed for risk evaluation
NOAEL – no-observed-adverse-effect level
PCB – polychlorinated biphenyl
SSD – species sensitivity distribution
total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT and 4,4′-DDT)
TRV – toxicity reference value
ww – wet weight

Effects from exposure to multiple chemicals that share the same mode of toxic action and other environmental stressors in the Study Area that could result in additive, synergistic, or antagonistic effects were not factored into the effects assessment. Because the combined effects of complex chemical mixtures and other stressors occurring in the environment have not been sufficiently studied, effects of this uncertainty on predicted risk are unknown.

TRVs were derived from laboratory studies in which fish were exposed to a single chemical or, in the case of PCBs and DDTs, to well-defined chemical mixtures under controlled conditions. Specific chemical exposures associated with toxicity can be determined from such studies. By controlling for natural sources of variability, laboratory studies do not address potential implications associated with mixtures of contaminants or the interaction of chemical toxicity with other stressors that occur in the natural environment. A number of studies in which fish from contaminated sites were raised in the laboratory have been conducted to investigate potential adverse effects associated with PCBs and DDTs (Berlin et al. 1981; Hopkins et al. 1969; Mac et al. 1985). Other studies have exposed fish to field-collected contaminated sediments to investigate potential adverse effects associated with specific mixtures of site chemicals(e.g., Roberts et al. 1989). Such studies incorporate conditions and exposure scenarios that provide insight into risks associated with specific sites and the chemical mixtures present at those sites. However, these studies were not used to derive TRVs in this BERA because adverse effects observed in organisms from studies at other sites may be attributed to the presence of multiple chemicals not present in the Study Area or to other uncontrolled environmental factors.

Uncertainty associated with the selected 10th percentile LOAEL for total PCBs (0.93 mg/kg ww) arises from uncertainty associated with several studies included in the TRV derivation process and, to a lesser extent, from uncertainty in the selected statistical distribution of the effects data. The total PCB SSD was derived from 19 literature-based LOAELs (Attachment 9). Five of the lowest LOAELs used in the SSD are associated with significant uncertainty, as summarized in Table 7-6. Because these studies report lower LOAELs than are observed in other PCB studies, they result in a more conservative (i.e., lower) TRV. If these studies had not been included, the 10th percentile TRV would have increased from 0.93 to 3.6 mg/kg ww.

Table 7-6. Summary of Lowest PCB LOAELs and Associated Uncertainties

Study	LOAEL (mg/kg ww)	Endpoint	Uncertainties
Hugla and Thome (1999)	0.52	Reduced fecundity of barbel	Fish were reared under elevated temperatures to alter their reproduction; the number of fish used in the experiment was not clearly reported; and statistical analyses appear to be based on an incorrect number of treatment levels.
Fisher et al. (1994)	1.1	Reduced body weight in live fry of Atlantic salmon	Fish were exposed only during the egg stage, so subsequent tissue concentrations measured in sac fry are lower because of dilution through growth; magnitude of growth reduction, although statistically significant, was very small (82 mg or 10% difference from controls) and the ecological relevance of this effect is questionable.
Berlin et al. (1981)	1.5	Increased fry mortality in lake trout	Fish used in the experiment originated from the Great Lakes at a time when PCB, DDT, and dioxin contamination was widespread; egg PCB and DDT residues measured prior to initiation of experiment were 7.6 and 3.8 µg/g, respectively. Tissue residues and adverse effects were not measured at the same time. Significant excess mortality occurred at days 57 to 96 and, to a lesser extent, days 97 to 136, but tissue residue was not measured until the end of the 176-day experiment, at which time the tissue residue was lower than at the beginning of the experiment (i.e., the initial tissue concentration due to maternal transfer of PCBs obtained from Great Lakes exposure was higher than the final tissue concentration.)
Broyles and Noveck (1979)	3.6	Increased mortality in Chinook salmon	Fish used in the experiment originated from the Great Lakes at a time when PCB, DDT, and dioxin contamination was widespread. Egg PCB residues, which were not measured, were estimated to be 3 to 11 mg/kg. LOAEL is based on measured ^{14}C-labeled PCB 153, which did not account for the tissue burden in fry resulting from maternal transfer.
	9.2	51 – 87% mortality in lake trout	

DDT – dichlorodiphenyltrichloroethane
LOAEL – lowest-observed-adverse-effect level
PCB – polychlorinated biphenyl
ww – wet weight

The statistical distribution of the total PCB SSD was uncertain because @RISK returned three statistical models with a similarly good fit. The lognormal distribution was selected because it had a more even distribution of residuals; however, Pearson 6 and log-logistic models had similarly good fits. The log-logistic model resulted in nearly the same 10th percentile TRV as the selected log-normal model, so selection of this distribution would not have changed risk predictions. If the Pearson 6 distribution had been selected, the 10th percentile TRV would have decreased from 0.93 to 0.76 mg/kg ww (see Attachment 9). The implications of these uncertainties are discussed in the risk characterization section (Section 7.1.5.1).

7.1.5 Risk Characterization

This section presents the risk characterization for the tissue-residue LOE for fish. An HQ approach was used to quantify risk estimates (Equation 6-1) following the two-step risk characterization process described in Section 7.1.1. The EPC and TRV are represented by tissue-residue concentrations expressed as mg/kg ww.Section 7.1.5.1 presents the risk characterization results and uncertainty evaluation for each fish receptor. Section 7.1.5.2 presents an evaluation of non-Study Area concentrations. Section 7.1.5.3 presents the COIs for which risks cannot be quantified. Section 7.1.5.4 presents a summary of the tissue-residue LOE. Section 7.1.5.5 presents the uncertainty evaluation of non-target ecological receptors. Results of the tissue-residue LOE, along with those from other LOEs were evaluated together in the fish risk conclusions section (Section 7.6) considering the relative strengths and uncertainties of each LOE.

7.1.5.1 Risk Characterization Results and Uncertainty Evaluation

The HQ results from the Step 1 are presented in Table 7-7. Receptor-COPC pairs with HQs ≥ 1 based on individual samples (Step 1) were retained for further evaluation in Step 2.[82]

[82] Per EPA (Attachment 2), carp was selected as a surrogate receptor only for the evaluation of dioxins and dioxin-like PCB congeners. However, dioxins and dioxin-like PCB congeners (as TEQ) did not screen in as a COPC in the SLERA and refined screen (Attachment 5). Therefore, carp were not further evaluated in this risk assessment.

Table 7-7. Number of Composite Fish Tissue Samples with HQs ≥ 1

COPC	Largescale Sucker	Juvenile White Sturgeon	Juvenile Chinook Salmon	Peamouth	Pacific Lamprey	Sculpin	Smallmouth Bass	Northern Pikeminnow
	Number of Composite Fish Tissue Samples with HQs ≥ 1 (Maximum HQ)[a]							
	Large-Home-Range Fish					**Small-Home-Range Fish**		
Metals								
Antimony	NA	NA	NA	NA	NA	NA	**1 of 32 (5.4)**	NA
Cadmium	NA	NA	NA	NA	NA	NA	0 of 32 (0.91)	NA
Chromium	0 of 6 (0.063)	0 of 15 (0.91)	NA	NA	NA	NA	NA	NA
Copper	NA	NA	NA	NA	**4 of 4 (2.2)**	**3 of 38 (2.3)**	NA	NA
Lead	NA	NA	NA	**1 of 4 (2.7)**	NA	NA	**2 of 32 (280)**	NA
Mercury	NA	NA	NA	NA	NA	NA	NA	0 of 6 (0.91)
Zinc	NA	NA	0 of 15 (0.98)	NA	NA	NA	NA	NA
PCBs								
Total PCBs	**2 of 6 (2.2)**	NA	NA	NA	NA	**4 of 38 (9.4); [90[b] of 1,100 (111)][c]**	**9 of 32 (7.1)**	**2 of 6 (2.0)**
Pesticides								
beta-HCH	NA	NA	NA	NA	NA	0 of 38 (0.048); [0[b] of 1,084 (0.038)]	NA	NA
Total DDx	0 of 6 (0.42)	NA	0 of 15 (0.38)	NA	0 of 6 (0.16)	**1 of 38 (1.9); [29[b] of 1,128 (21)]**	0 of 32 (0.91)	0 of 6 (0.48)
BEHP	0 of 6[d] (0.31)	NA	NA	NA	NA	**1 of 38 (2.9)[d]**	**2 of 32 (9.1)[d]**	NA

a HQs based on LOAELs for all fish except for juvenile Chinook salmon and Pacific lamprey HQs based on NOAELs.
b HQs based on predicted sculpin tissue concentrations using the mechanistic model and individual surface sediment samples.
c An additional 10 predicted samples had DLs that were greater than the TRV. The maximum HQ based on a DL is 14.5.
d No BEHP LOAEL was identified; HQs based on the only NOAEL identified.

BEHP – bis(2-ethylhexyl) phthalate
COPC – contaminant of potential concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
DL – detection limit
HCH – hexachlorocyclohexane
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
NA – not applicable; not a receptor-COPC pair
NOAEL – no-observed-adverse-effect level
PCB – polychlorinated biphenyl
total DDx - sum of all six DDT isomers (2,4′-DDD; 4,4′-DDD; 2,4′-DDE; 4,4′-DDE; 2,4′-DDT; and 4,4′-DDT)
TRV – toxicity reference value

Bold identifies HQs ≥ 1.

Five COPCs (cadmium, chromium, mercury, zinc, and beta-HCH) were not further evaluated for any fish receptor beyond Step-1 because they never exceeded their respective baseline TRVs in any fish sample. Antimony, copper, lead, total PCBs, and total DDx tissue concentrations in at least one receptor exceeded their respective baseline TRVs, and each was analyzed in Step 2 for at least one receptor. BEHP was evaluated in Step-2 for sculpin and smallmouth bass because tissue concentrations exceeded a NOAEL (no LOAEL TRV was available).

7.1.5.1.1 Large-Home-Range Fish

Step 2 HQs for large-home-range fish are presented in Table 7-8 and discussed below.

Table 7-8. Site-Wide Tissue HQs for Large-Home-Range Fish

		Largescale Sucker			Peamouth			Pacific Lamprey		
COPC	**Unit (ww)**	**Site-wide EPC**	**10th Percentile LOAEL TRV**	**HQ (Range of HQs)[a] unitless**	**Site-wide EPC**	**Literature-Based LOAEL TRV**	**HQ (Range of HQs)[a] unitless**	**Site-wide EPC**	**5th Percentile LOAEL TRV**	**HQ (Range of HQs)[a] unitless**
Metals										
Copper	mg/kg	NA	NA	NA	NA	NA	NA	6.2[b]	2.8	**2.2 (1.1 – 2.2)**
Lead	mg/kg	NA	NA	NA	10.6[b]	4.0	**2.7 (< 0.1 – 2.7)**	NA	NA	NA
PCBs										
Total PCBs	µg/kg	1,498[c]	930	**1.6 (0.1 – 2.2)**	NA	NA	NA	NA	NA	NA

a Based on individual composite samples.
b EPC is based on maximum concentration; insufficient data were available to derive site-wide UCL.
c EPC is based on site-wide UCL concentration.

COPC – contaminant of potential concern
EPC – exposure point concentration
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
NA – not applicable (not a receptor-COPC pair)
PCB – polychlorinated biphenyl
TRV – toxicity reference value
UCL – upper confidence limit on the mean
ww – wet weight

Bold identifies HQs ≥ 1.

Largescale Sucker

Of the COPCs identified for largescale sucker, one (total PCBs) has an HQ ≥ 1 based on individual composite samples (Table 7-7; Map 7-1). The total PCB HQ based on the site-wide EPC is 1.6 (Table 7-8).

As discussed in Section 7.1.4 two quantifiable sources of uncertainty are associated with the PCBs TRV. One uncertainty is related to the inclusion of uncertain LOAELs in the TRV derivation; their exclusion would have resulted in a higher TRV. If the uncertain studies (Table 7-6) had not been included in the TRV derivation, the PCBs HQ would have decreased from 1.6 to < 1.

A second, lesser uncertainty is the selection of the statistical distribution of the SSD. If a different statistical distribution had been selected (leading to a lower TRV), the site-wide HQ would have increased from 1.6 to 2.0.

Tissue data for characterizing largescale sucker exposure are considered adequate because there are a sufficient number of composite tissue samples from throughout the Study Area to calculate EPCs based on UCLs for all COPCs.

Peamouth

Of the COPCs identified for peamouth, one (lead) has an HQ ≥ 1 based on individual composite samples (Table 7-7; Map 7-2). Because insufficient data were available to calculate a site-wide UCL for lead concentrations in peamouth, the maximum concentration was used as the site-wide EPC in Step 2. The site-wide HQ is 2.7 (Table 7-8). Given the available data, the use of the maximum concentration as the EPC is considered the most appropriate estimator of the average, but it is less certain than a UCL based on a larger dataset, and might over- or underestimate the average lead concentration in peamouth. Additionally, as discussed in Section 7.1.4, data available to derive a lead TRV using the SSD approach were insufficient, leading to a highly uncertain TRV. Furthermore, as discussed in Section 7.1.2 the tissue-residue LOE is relatively weak because differences in regulation of inorganic metals by fish create difficulties in interpreting the toxicological significance of whole-body tissue residues.

Pacific Lamprey Ammocoetes

Of the COPCs identified for Pacific lamprey ammocoetes, one (copper) has an HQ ≥ 1 based on individual composite samples (Table 7-7; Map 7-3). Because the data available for calculation of a site-wide UCL for copper concentrations in lamprey were insufficient, the maximum concentration was used as the EPC. The site-wide HQ is 2.2 (Table 7-8). As discussed above for peamouth, use of the maximum concentration as the EPC might over- or underestimate risk.

The LWG has conducted toxicity tests on lamprey ammocoetes to establish their relative sensitivity to toxicants associated with six different modes of action. One of the tested chemicals was copper. Lamprey ammocoetes were found to be relatively insensitive to copper toxicity, with an LC50 falling at the 46^{th} percentile of the SSD, approximately an order of magnitude higher than the 5^{th} percentile of the LC50 distribution. This finding suggests that use of the 5^{th} percentile LOAEL as the TRV overestimates risk to lamprey ammocoetes. This supposition is uncertain, however, because the toxicity tests were for acute waterborne exposure.

The tissue-residue LOE is relatively weak because fish actively regulate tissue copper concentrations. Differences in regulation of inorganic metals by fish create difficulties in interpreting the toxicological significance of whole-body tissue residues.

7.1.5.1.2 Small-Home-Range Fish

Sculpin

Of the COPCs identified for sculpin, three (copper, total PCBs, and total DDx) have 10th percentile LOAEL HQs ≥ 1 based on individual composite tissue samples (Table 7-7; Map 7-4). One BEHP sample exceeded the NOAEL; however, no LOAEL TRV was identified. BEHP is discussed in the uncertainty analysis along with other contaminants for which no suitable TRVs could be identified. The relevant exposure area for sculpin was considered roughly equivalent to the sampling area of sculpin. Therefore, sculpin tissue data were evaluated only on a sample-by-sample basis. The spatial distribution of estimated risks is discussed below:

- **Total PCBs –** Four composite tissue samples (11% of all samples) have a total PCB HQ ≥ 1, one each in the following locations: RM 2.3 and RM 2.4 on the east side of the Study Area, Willamette Cove, and RM 11.3 on the east side of the Study Area (Map 7-4). Ninety samples whose concentrations were predicted rather than measured (8% of all such samples) have HQs ≥ 1. Map 7-4 presents the NN-interpolated[83] areas where predicted total PCB tissue concentrations result in HQs ≥ 1. Predicted HQs ≥ 1 occur in the same locations where tissue samples have HQs ≥ 1, except International Slip, Swan Island Lagoon, and RM 8.8 to RM 9.1 on the west side and additional isolated sediment sample locations (see Attachment 4, Part D). In general, the areas of predicted HQs ≥ 1 are near or at the same locations of HQs < 1 for composite sculpin samples in which total PCBs were detected (Map 7-4). Empirical data better represent tissue concentrations in Study Area sculpin than do predicted tissue concentrations.

  As discussed in the effects section (Section 7.1.4) two quantifiable sources of uncertainty are associated with the PCBs TRV. One uncertainty is due to inclusion of uncertain LOAELs in the TRV derivation; their exclusion would have resulted in a higher TRV. If the uncertain studies had not been included in the SSD, HQs for all samples but one from RM 11 would fall from ≥ 1 to < 1. The RM 11 sample would fall from 9.6 to 2.4. The number of samples with predicted HQs ≥ 1 would be reduced from 90 of 1,100 to 29 of 1,100.

  If a different statistical distribution had been selected (leading to a lower TRV), the maximum HQ would have increased from 9.4 to 12 and the number of samples with predicted HQs ≥ 1 would have increased from 90 of 1,100 to 111 of 1,100.

[83] The NN-interpolation algorithm is built into ArcGIS software. It has the advantage over other spatial statistical algorithms of being fully defined, so once the dataset and grid have been established, any GIS analyst applying the algorithm will get the same interpolation result. Other spatial statistical methods require the analyst to use professional judgment to fully define the interpolation algorithm, which introduces subjectivity into the analysis and confounds reproducibility.

- **Total DDx –** In one composite tissue sample (3% of all sculpin composite samples) and 29 samples whose concentrations were predicted (3% of all such samples[84]) total DDx HQs are ≥ 1. The composite sample with an HQ ≥ 1 was collected from RM 7.3 on the west side of the Study Area (Map 7-4). Map 7-4 presents the NN-interpolated areas where predicted total DDx tissue concentrations result in HQs ≥ 1 from approximately RM 7.1 to RM 7.4 on the west side of the Study Area, and additional isolated sediment sample locations between RM 6.3 and 8.8 (see Attachment 4, Part D).
- **Copper –** Three tissue samples (8% of all samples) have HQs ≥ 1, one each at RM 5.5 on the east side, RM 10.3 on the west side, and RM 11.3 on the east side of the Study Area. HQs range from 1.1 to 2.3 (Map 7-4). Copper concentrations in sculpin tissue could not be predicted from sediment because no relationship between copper concentrations in tissue and sediment were identified (see Section 7.1.3.2).
- **BEHP –** One of the 38 sculpin samples had a BEHP concentration greater than the literature-based NOAEL (no LOAEL was available), with a NOAEL HQ of 2.9 at the mouth of Swan Island Lagoon. The remaining 37 sculpin samples throughout the Study Area were less than the NOAEL TRV for BEHP. To evaluate differences between exposure at the site and upriver, Study Area BEHP tissue data were compared with available upriver fish tissue data (upriver sculpin data were not available) (see Section 7.1.5.2.2). BEHP concentrations were detected in Study Area and upstream brown bullhead, lamprey, and smallmouth bass. Concentrations in Study Area brown bullhead and lamprey were similar to those in upriver samples; concentrations in some Study Area smallmouth bass were higher than those in upstream samples. Because a tissue-residue LOAEL was not available, risks from BEHP are uncertain.

Smallmouth Bass

Of the COPCs identified for smallmouth bass, three (antimony, lead, and total PCBs) tissue concentrations in individual composite samples result in 10th percentile LOAEL HQs ≥ 1 (Table 7-7).[85] The locations of individual samples with HQs ≥ 1[86] are presented on Map 7-5. HQs for these COPCs were calculated for 1-mile exposure areas throughout the Study Area (Table 7-9). Insufficient data were available to calculate

[84] Eleven of the 29 predicted tissue samples with HQs ≥ 1 are based on N-qualified sediment data (i.e., the analyst believed that the result was due to analytical interference from a chemical other than the target analyte).

[85] Two BEHP samples exceeded the NOAEL; however, no LOAEL TRV was identified. BEHP is discussed in the uncertainty analysis along with other chemicals for which no suitable TRVs could be identified.

[86] HQs were calculated and are presented on a sample-by-sample basis because the compositing area of smallmouth bass was generally within a 1-mile area, which is consistent with the species' assumed exposure area.

exposure area UCLs; therefore, the maximum individual composite value (by RM) was used to assess risk to smallmouth bass.

Table 7-9. Smallmouth Bass 1-Mile Exposure Area-Specific Tissue 10th Percentile LOAEL HQ

	LOAEL HQ [a] (Range of HQs)[b]		
Exposure Area	**Antimony**	**Lead**	**Total PCBs**
RM 1.5 to RM 2.5	0.0052[c]	0.0025[c]	**1.5**[c]
RM 2.5 to RM 3.5	0.0053 (0.00091 – 0.0053)	0.45 (0.0015 – 0.45)	0.84 (0.22 – 0.84)
RM 3.5 to RM 4.5	0.0052 (0.00091 – 0.0052)	0.014 (0.0013 – 0.014)	**1.6** (0.31 – **1.6**)
RM 4.5 to RM 5.5	0.0053 (0.00091 – 0.0053)	0.0028 (0.0015 – 0.0028)	0.42 (0.29 – 0.42)
RM 5.5 to RM 6.5	0.0053 (0.00091 – 0.0053)	0.0043 (0.0023 – 0.0043)	0.73 (0.29 – 0.73)
RM 6.5 to RM 7.5	0.0053 (0.00091 – 0.0053)	0.0084 (0.0013 – 0.0084)	**2.2** (0.12 – **2.2**)
RM 7.5 to RM 8.5	0.013 (0.0027 – 0.013)	0.0022 (0.0012 – 0.0022)	0.97 (0.31 – 0.97)
RM 8.5 to RM 9.5	0.0052 (0.0027 – 0.0052)	0.0088 (0.002 – 0.0088)	1.0 (0.38 – 1.0)
Swan Island Lagoon	0.0027 (0.0018 – 0.0027)	0.076 (0.0019 – 0.076)	**5.3** (**1.2 – 5.3**)
RM 9.5 to RM 10.5	**5.4** (0.034 – **5.4**)	**280** (**1.7** – **280**)	0.87 (0.67 – 0.87)
RM 10.5 to RM 11.8	0.0053 (0.0046 – 0.0053)	0.033 (0.028 – 0.033)	**7.1** (0.57 – **7.1**)

[a] The 10th percentile LOAEL HQ was calculated using the maximum concentration available from within each exposure area.
[b] Based on individual composite samples.
[c] Range of HQs is not presented because only one value was available for this exposure area.
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
PCB – polychlorinated biphenyl
RM – river mile
Bold identifies HQs ≥ 1.

The spatial distribution of estimated risk is discussed below:

- **Antimony –** One smallmouth bass sample had a concentration of 5.9 mg/kg ww, resulting in an HQ ≥ 1 between RM 9.5 and RM 10.5; the calculated HQ is 5.4. The single composite sample is an outlier for both antimony and lead suggesting that a fish in the sample might have swallowed a fishing sinker. Antimony can be mixed with lead as a hardener for lead-based products (ATSDR 1992). For example, one fish tackle supplier notes that fishing sinkers contain 94% lead and 6% antimony for hardness and color (Blue Ocean Tackle 2011).The antimony tissue concentrations detected in all of the other samples (n = 31) ranged from 0.001 to 0.04, with HQs ≤ 0.03.

- **Lead –** Two smallmouth bass samples have HQs ≥ 1 between RM 9.5 and RM 10.5; calculated HQs are 280 and 1.7. The HQ of 280 (1,100 mg/kg ww) is a statistical outlier. The lead tissue concentrations detected in all other samples (n = 31) ranged from 0.0048 to 6.8 mg/kg ww, with HQs ranging from < 1 to 1.7.
- **Total PCBs –** Total PCB HQs are ≥ 1 in smallmouth bass samples from several locations across the Study Area, with HQs ranging from 1.5 to 7.1 in five exposure areas.

  As discussed above for sculpin two quantifiable sources of uncertainty could have resulted in a higher or lower PCBs TRV. If some of the uncertain studies had been excluded from the SSD and the higher TRV had been used, HQs for all but two exposure areas, would have been reduced from ≥ 1 to < 1. The HQs for Swan Island Lagoon for exposure area RM 10.5 to RM 11.8 would have been 1.4 and 1.8, respectively. If a different statistical distribution had been selected (leading to a lower TRV), the number of samples with an HQ ≥ 1 would have increased from 9 of 32 to 14 of 32 and the maximum HQ would have increased from 7.1 to 8.7.
- **BEHP –** Two of the 38 smallmouth bass samples had a BEHP concentration greater than the NOAEL TRV (no LOAEL was available). The NOAEL HQs are 3.3 and 9.1 in these two samples, which were collected from the exposure area between RM 3.5 and RM 4.5 during Round 1 sampling. In smallmouth bass composites collected during Round 3 in the same sampling area, BEHP concentrations were not detected.

Northern Pikeminnow

Of the COPCs identified for northern pikeminnow, one (total PCBs) had tissue concentrations in individual composite samples that resulted in 10th percentile LOAEL HQs ≥ 1 (Table 7-7; Map 7-6). HQs for these COPCs were calculated for 1-mile exposure areas throughout the Study Area (Table 7-10). Insufficient data were available to calculate exposure area UCLs; therefore the maximum concentration (by river mile) was used to assess risk to northern pikeminnow.

Table 7-10. Northern Pikeminnow 1-Mile Exposure Area-Specific Tissue 10th Percentile LOAEL HQs

	LOAEL HQ[a] (Range of HQs)[b]
Exposure Area	Total PCBs
RM 1.5 to RM 2.5	0.77[c]
RM 2.5 to RM 3.5	0.77 (0.40 – 0.77)
RM 3.5 to RM 4.5	ND
RM 4.5 to RM 5.5	0.47[c]

Table 7-10. Northern Pikeminnow 1-Mile Exposure Area-Specific Tissue 10th Percentile LOAEL HQs

Exposure Area	LOAEL HQ[a] (Range of HQs)[b] Total PCBs
RM 5.5 to RM 6.5	0.47[c]
RM 6.5 to RM 7.5	**2.0**[c]
RM 7.5 to RM 8.5	**1.1**[c]
RM 8.5 to RM 9.5	**1.1**[c]
Swan Island Lagoon	0.84[c]
RM 9.5 to RM 10.5	ND
RM 10.5 to RM 11.8	ND

[a] The 10th percentile LOAEL HQ was calculated using the maximum concentration available from within each exposure area.
[b] Based on individual composite samples.
[c] Range of HQs is not presented because only one value was available for this exposure area.
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
ND – no data
PCB – polychlorinated biphenyl
RM – river mile
Bold identifies HQs ≥ 1.

The spatial distribution of the estimated risk is discussed below:

- **Total PCBs –** The total PCBs 10th percentile LOAEL HQ in northern pikeminnow samples range from 1.1 to 2.0 in three exposure areas (between RM 6.5 and RM 9.5).

  As discussed above for sculpin two quantifiable sources of uncertainty could have resulted in a higher or lower PCBs TRV. If some of the uncertain studies had been excluded from the SSD and the higher TRV had been used, all samples would have been reduced from ≥ 1 to < 1. If a different statistical distribution had been selected (leading to a lower TRV), the number of samples with an HQ ≥ 1 would have increased from two of six to three of six, and the maximum HQ would have increased from 2.0 to 2.5.

7.1.5.2 Evaluation of Non-Study Area Concentrations

This section evaluates non-Study Area tissue data, including data from just above and below the Study Area (RM 11.8 to RM 15.3 and RM 0 to RM 1.9, respectively), and data from the upriver reach (above RM 15.3). These data were evaluated per EPA's Problem Formulation (Attachment 2).

7.1.5.2.1 Tissue Data from the Downstream and Downtown Reaches

Per EPA (2008j), data collected from just outside the boundaries of the Study Area were also evaluated for this BERA. Two composite samples of sculpin tissue were available from the downstream reach (RM 0 to RM 1.9) [87], and two were available from the downtown reach (RM 11.8 to RM 15.3).[88] The non-Study Area sediment and tissue chemistry data from these reaches are presented in Attachment 4.

A CFD of sculpin tissue concentrations within the Study Area, the downstream reach, and the downtown reach were plotted for sculpin COPCs (i.e., total PCBs, total DDx, and copper) with tissue 10th percentile LOAELs (Figures 7-3 through 7-5). For total PCBs and total DDx, tissue concentrations in sculpin collected from the downstream reach and the downtown reach are less than the respective 10th percentile LOAELs, resulting in HQs < 1; inclusion of these samples in the fish risk assessment would not have resulted in identification of additional sampling locations with potentially unacceptable risk associated with these contaminants. In one sculpin sample collected from the downstream reach, the copper concentration was slightly greater than the tissue 10th percentile LOAEL (HQ = 1.2). Three samples from the Study Area also exceeded the copper 10th percentile LOAEL. Inclusion of the downstream samples in the fish risk assessment would have resulted in a greater spatial extent of fish tissue samples slightly exceeding the copper TRV. As discussed above, however, a high degree of uncertainty is associated with the tissue LOE when assessing risk to fish from copper.

[87] Samples were collected from the west and east bank at approximately RM 1.5, which provides limited spatial coverage of the 2-mile downstream reach.

[88] Samples were collected from the west and east bank at approximately RM 12, which provides limited spatial coverage of the 3.5-mile downtown reach.

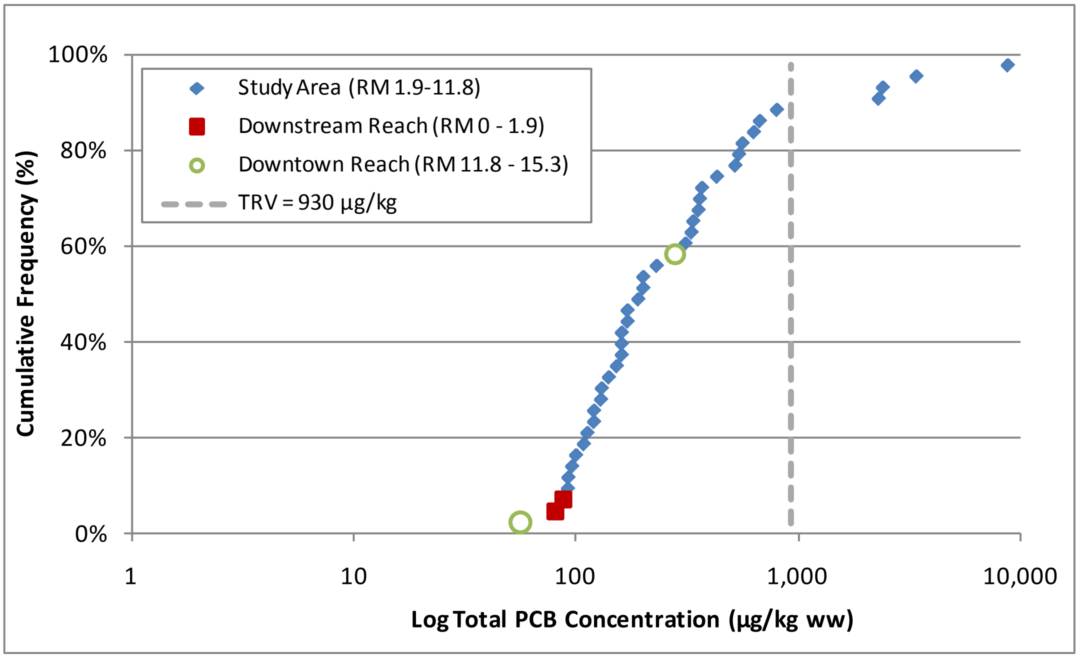

Figure 7-3. CFD of Sculpin Total PCB Concentrations

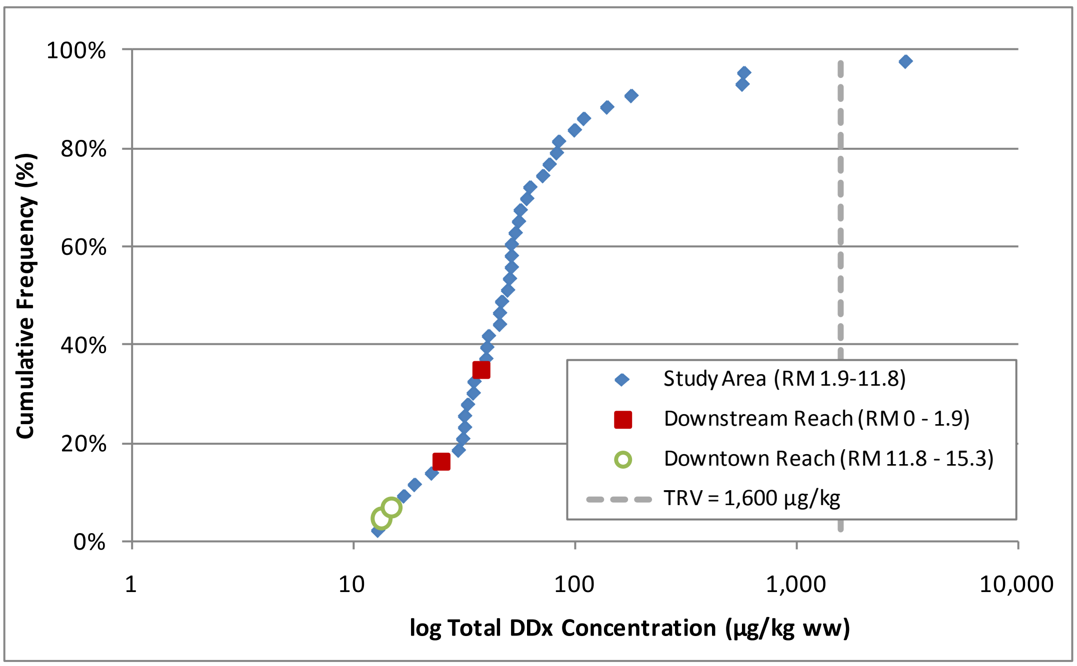

Figure 7-4. CFD of Sculpin Total DDx Concentrations

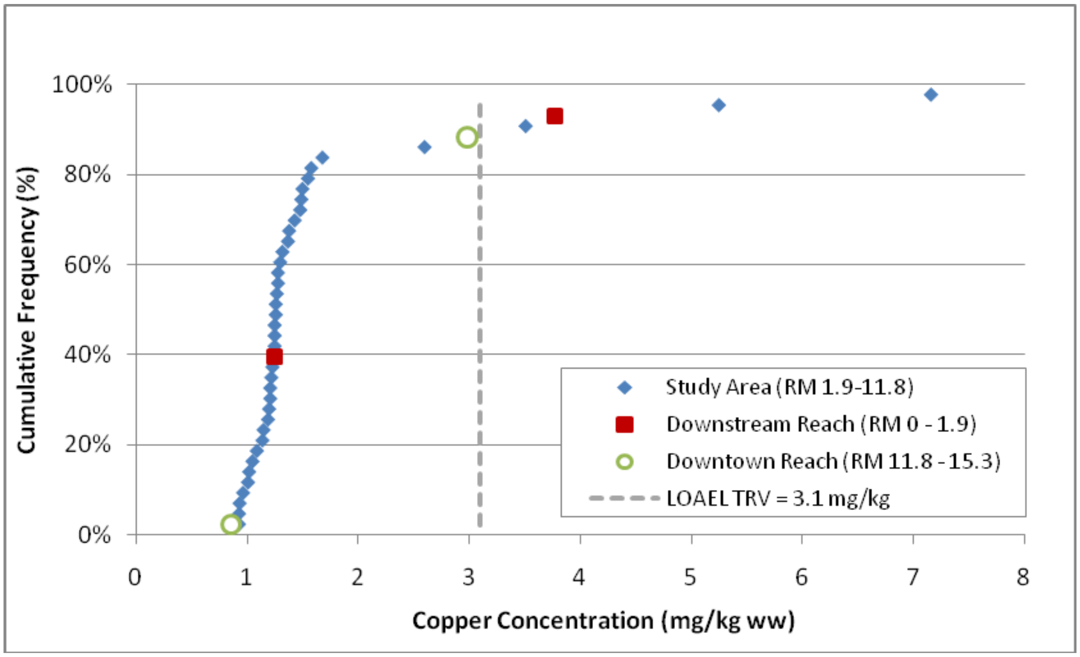

Figure 7-5. CFD of Sculpin Copper Concentrations

7.1.5.2.2 Tissue Data from the Upriver Reach

Upriver tissue residue data were available for three COPCs (aluminum, BEHP, and copper). Baseline TRVs, and therefore HQs, could not be calculated for aluminum and BEHP because toxicological data were insufficient. Copper concentrations in tissues of sculpin and lamprey from the Study Area exceed the corresponding TRVs. This section presents information on the sample data (species,composite sizes, fish weights and lengths) and then compares upriver and Study Area tissue residues.

Available Upriver Tissue Data

Tissue data for four fish receptor species (from RM 15.3 to RM 28.4) were collected from the upriver reach by LWG between June 2002 and September 2006. The non-Study Area sediment and tissue chemistry data from these reaches are presented in Attachment 4. Table 7-11 summarizes the number of whole-body tissue composite samples, the number of fish in each composite, and the weights and lengths of fish species collected in the Study Area and at upriver locations. This information can be used to compare the Study Area and upriver samples. Juvenile Chinook salmon, smallmouth bass, brown bullhead, and lamprey were the only species collected from both the Study Area and at upriver locations.

Table 7-11. Lengths and Weights of Fish Collected for Whole-Body Composite Tissue Samples from the Study Area and Upriver Locations

Species (year collected)	Location	No. of Composite Tissue Samples	No. of Fish per Composite	Weight (kg) Avg	Weight (kg) Min	Weight (kg) Max	Length (cm) Avg	Length (cm) Min	Length (cm) Max
Juvenile Chinook salmon (2002)	Study Area	2	15	0.012	0.0085	0.016	9.9	8.8	11
		1	14						
		1	13						
		1	12						
		1	11						
	Upriver	1	12	0.014	0.012	0.018	11	10	12
Juvenile Chinook salmon (2005)	Study Area	9	30	0.0024	0.0012	0.0063	6.5	5.1	9.0
		1	27						
		1	24						
		1	21						
	Upriver	3	30	0.0028	0.0012	0.0057	6.9	5.3	8.8
Smallmouth bass (2002)	Study Area	14	5	0.37	0.17	1.2	28	22	43
		1	4						
		1	2						
		1	1						
	Upriver	6	5	0.68	0.13	1.9	33	23	46
Brown bullhead (2002)	Study Area	6	5	0.25	0.14	0.38	27	22	30
	Upriver	3	5	0.23	0.15	0.40	26	23	31
Lamprey ammocoete (2005)	Study Area	1	7	0.0014	0.00010	0.0053	8.5	4.5	13
		1	28						
		1	19						
	Upriver	2[a]	44	0.0011	0.00010	0.0035	8.0	3.5	14
		1	49						
Lamprey macropthalmia (2005)	Study Area	2	6	0.0040	0.0027	0.0052	13	11	15
	Upriver	1	9	0.0032	0.0024	0.0044	12	11	14

[a] Two samples were created from a post-homogenization split.

More fish were collected from the Study Area than from upriver locations, with the exception of lamprey. The sample average weight and length of lamprey, both ammocoetes and macropthalmia, were slightly greater in the Study Area than in upriver locations. The composite samples sizes were inconsistent for both ammocoetes (7 to 49 individuals per composite) and and macropthalmia (six to nine individuals per composite). These inconsistencies are due to the low numbers of ammocoetes captured during sampling within the Study Area.

Brown bullhead is the only species whose sizes and number of individuals per composite were similar in both the Study Area and upriver locations.

In 2002, the juvenile Chinook collected upriver were slightly larger than those collected in the Study Area. The composite samples sizes were inconsistent, ranging from 11 to 15 fish per composite.

The juvenile Chinook collected in 2005 were similar based on average weight, average length, and number of fish per composite. The number of fish per composite was generally consistent between the Study Area and upriver; however 3 of the 12 composite samples from the Study Area consisted of fewer than the typical 30 fish per composite, because of the inability to collect more.

The smallmouth bass collected in the upriver reach were larger than those collected in the Study Area. The number of fish per composite was generally consistent between the Study Area and upriver reach; however, 3 of the 17 composite samples from the Study Area had fewer than the typical five fish per composite.

Comparison of Upriver and Study Area Tissue Concentrations

BEHP concentrations in Study Area and upriver fish tissue are presented in Table 7-12. BEHP was detected in Study Area and upriver brown bullhead, lamprey, and smallmouth bass. Concentrations in brown bullhead and lamprey were similar in Study Area and upriver samples,but mean concentrations in Study Area smallmouth bass were higher than those in upriver samples.

Table 7-12. BEHP Concentrations in Study Area and Upriver Fish Tissue

	Study Area					Upriver				
		Concentration (µg/kg ww)					Concentration (µg/kg ww)			
Species	DF	Min Detect	Mean Detect	Max Detect	RL Range	DF	Min Detect	Mean Detect	Max Detect	RL Range
Brown bullhead	1 of 6	2,700	2,700	2,700	98 – 100	1 of 3	3,000	3,000	3,000	99
Juvenile Chinook salmon	0 of 11	NA	NA	NA	95 – 860	2 of 4	140	140	140	81 – 120
Lamprey	1 of 1	170	170 J	170	NA	4 of 4	120	137	160 J	NA

Table 7-12. BEHP Concentrations in Study Area and Upriver Fish Tissue

	Study Area					Upriver				
		Concentration (µg/kg ww)					Concentration (µg/kg ww)			
Species	**DF**	**Min Detect**	**Mean Detect**	**Max Detect**	**RL Range**	**DF**	**Min Detect**	**Mean Detect**	**Max Detect**	**RL Range**
Smallmouth bass	6 of 32	44	21,000	87,000	66 – 1,300	1 of 6	4,800	4,800	4,800	99 – 550

BEHP – bis(2-ethylhexyl) phthalate
DF – detection frequency
J – estimated concentration
RL – reporting limit
NA – not applicable
ww – wet weight

Differences in number of samples from upstream and Study Area fish and low detection frequency (for all species except lamprey) make comparison of COPC concentrations somewhat uncertain. Difference in fish sizes can affect the bioconcentration of biomagnifying organic chemicals, but body size does not generally affect tissue burdens of SVOCs since they do not typically persist in fish tissues and are detected rarely or not at all (EPA 2009e). Like SVOCs, metals concentrations in tissue are generally not affected by body size because organisms are capable of bioregulating inorganic metals. Although bioaccumulation and trophic transfer of metals occur, biomagnification is rare, except for organometallic compounds such as methylmercury (EPA 2007e).

Aluminum concentrations in the Study Area (RM 1.9 to RM 11.8) were generally similar to or less than those in the upriver reach (RM 15.3 to RM 28.4) (Figure 7-6). Background aluminum concentrations were also compared with Study Area concentrations in sediment and surface water (Attachment 11); background data for all three LOEs are discussed in the fish risk conclusions section (Section 7.6).

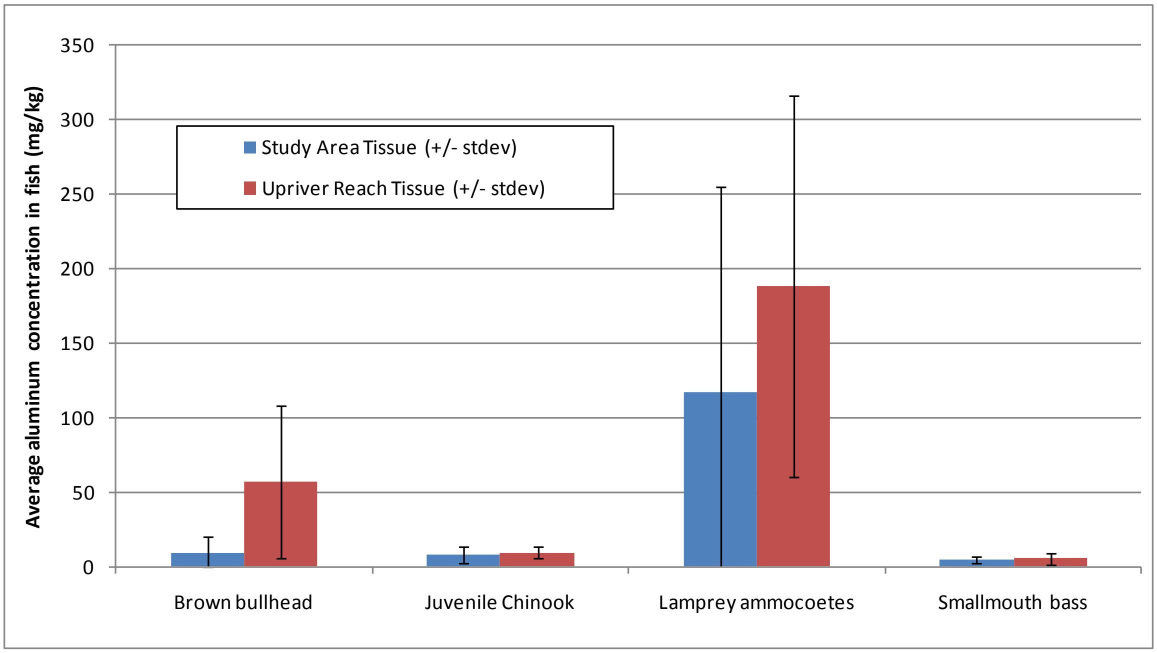

Figure 7-6. Aluminum Concentrations in Fish Tissue

For lamprey ammocoetes the 5th percentile copper LOAEL HQ is 2.2. Sculpin 10th percentile LOAEL HQs range from 1.1 to 2.3 (among four samples). Average copper concentrations in brown bullhead, juvenile Chinook salmon, lamprey ammocoetes, and smallmouth bass were similar in tissues collected from the upriver reach and the Study Area, as shown in Figure 7-7. Elevated concentrations of copper (i.e., greater than the 5th percentile and 10th percentile LOAEL TRVs of 2.8 and 3.1 mg/kg ww, respectively) were also present in tissues from upriver reach lamprey and greater than those in lamprey from the Study Area. No sculpin data were available from the upriver reach.

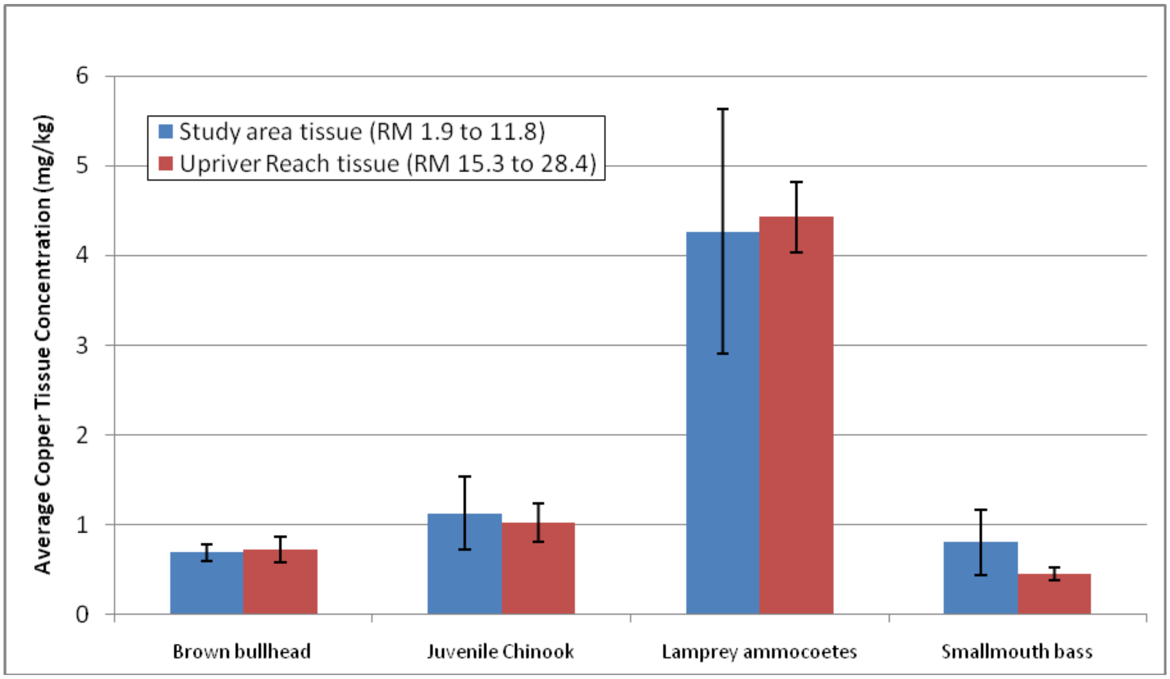

Figure 7-7. Comparision of Study Area and Upriver Reach Copper Tissue Concentrations in Fish

7.1.5.3 COIs for Which Risks Cannot Be Quantified

COIs for which risks to fish cannot be quantified based on tissue data are listed in Table 7-13. These COIs include contaminants for which no TRV is available and those whose maximum DL exceeded a TRV when detected values did not.

Table 7-13. Fish Tissue COIs with No Available TRV or with DLs Exceeding Screening-Level TRVs

COI	Rationale for Why Risks Cannot Be Quantitatively Evaluated
Metals	
Manganese	Risk to fish based on tissue-residue LOE unknown; no tissue TRV available.
Butyltins	
Monobutyltin ion Dibutyltin ion	Risk to fish based on tissue-residue LOE unknown; no tissue TRV available. However, TBT, for which a TRV is available, is the most toxic butyltin; risks due to mono- and dibutyltin are assumed to be lower than those of TBT.
SVOCs	
Benzoic acid	Risk to fish based on tissue-residue LOE unknown; no tissue TRV available.
Benzyl alcohol	Risk to fish based on tissue-residue LOE unknown; no tissue TRV available.
Bis(2-chloroethoxy) methane	Risk to fish based on tissue-residue LOE unknown; no tissue TRV available.
Hexachlorobutadiene	Risk to sculpin based on tissue–residue LOE are unknown; 57% of non-detected sculpin tissue samples had DLs > screening-level TRV, but no detected concentrations > screening-level TRV.
Phthalates	
BEHP	Risk to juvenile Chinook salmon based on tissue data are unknown; 36% of non-detected juvenile Chinook salmon tissue samples had DLs > screening-level TRV, but contaminant was never detected in juvenile Chinook salmon tissue.
Butyl benzyl phthalate	Risk to juvenile Chinook salmon based on tissue data are unknown; 57% of non-detected juvenile Chinook salmon tissue samples had DLs > screening-level TRV, but no detected concentrations > screening-level TRV.
Dibutyl phthalate	Risk to largescale sucker, juvenile Chinook salmon, sculpin, and smallmouth bass based on tissue data are unknown; 50% of non-detected largescale sucker, 67% of non-detected juvenile Chinook salmon, 63% of non-detected sculpin, and 45% of non-detected smallmouth bass tissue samples had DLs > screening-level TRV, but no detected concentrations > screening-level TRV (contaminant was never detected in largescale sucker, sculpin, and smallmouth bass tissue).
Diethyl phthalate	Risk to largescale sucker, juvenile Chinook salmon, and lamprey ammocoetes based on tissue data are unknown; 17% of non-detected largescale sucker, 27% of non-detected juvenile Chinook salmon, and 100% (n=1) of lamprey ammocoete tissue samples had DLs > screening-level TRV, but no detected concentrations > screening-level TRV (contaminant was never detected in largescale sucker, juvenile Chinook salmon, and lamprey ammocoete tissue).

Table 7-13. Fish Tissue COIs with No Available TRV or with DLs Exceeding Screening-Level TRVs

COI	Rationale for Why Risks Cannot Be Quantitatively Evaluated
Phenols	
4-Chloro-3-methylphenol	Risk to fish based on tissue data unknown; no tissue TRV available.
4-Nitrophenol	Risk to fish based on tissue data unknown; no tissue TRV available.
Pesticides	
Endrin	Risk to largescale sucker, sculpin, and smallmouth bass based on tissue data are unknown; 17% of non-detected largescale sucker, 9% of non-detected sculpin, and 12% of non-detected smallmouth bass tissue samples had DLs > screening-level TRV, but no detected concentrations > screening-level TRV.
alpha-HCH	Risk to largescale sucker, sculpin, smallmouth bass, and northern pikeminnow based on tissue data are unknown; 17% of non-detected largescale sucker, 6% of non-detected sculpin, 18% of non-detected smallmouth bass, and 67% of non-detected northern pikeminnow tissue samples had DLs > screening-level TRV, but no detected concentrations > screening-level TRV (contaminant was never detected in largescale sucker or northern pikeminnow tissue).
beta-HCH	Risk to largescale sucker, smallmouth bass, and northern pikeminnow based on tissue data are unknown; 17% of non-detected largescale sucker, 29% of non-detected smallmouth bass, and 67% of non-detected northern pikeminnow tissue samples had DLs > screening-level TRV, but no detected concentrations > screening-level TRV (contaminant was never detected in largescale sucker or northern pikeminnow tissue).
delta-HCH	Risk to largescale sucker, sculpin, smallmouth bass, and northern pikeminnow based on tissue data are unknown; 17% of non-detected largescale sucker, 6% of non-detected sculpin, 9% of non-detected smallmouth bass, and 67% of non-detected northern pikeminnow tissue samples had DLs > screening-level TRV, but no detected concentration > screening-level TRV (contaminant was never detected in largescale sucker, smallmouth bass, or northern pikeminnow tissue).

BEHP – bis(2-ethylhexyl) phthalate
COI – contaminant of interest
DL – detection limit
HCH – hexachlorocyclohexane
LOE – line of evidence
SVOC – semivolatile organic compound
TBT – tributyltin
TRV – toxicity reference value

7.1.5.4 Summary of Fish Tissue LOE

One or more receptor species have HQs ≥ 1 for five fish tissue COPCs. Large-home-range fish have three COPCs with HQs ≥ 1: largescale sucker for total PCBs; peamouth for lead, and Pacific lamprey ammocoete for copper. Small-home-range fish have five COPCs with HQs ≥ 1: sculpin for copper, total PCBs, and total DDx; smallmouth bass for antimony and lead; and northern pikeminnow for total PCBs. Results of the tissue-residue LOE are integrated with those of other LOEs to determine risk conclusions for fish in Section 7.6.

7.1.5.5 Evaluation of Non-Target Ecological Receptors

Per EPA (2008j), the fish species not identified as ecological receptors of concern (brown bullhead and black crappie) were evaluated as part of the fish tissue uncertainty assessment. With the same methods used to derive fish tissue COPCs for other fish receptors in the SLERA and refined screen (Attachment 5), one COPC was identified for black crappie and two COPCs were identified for brown bullhead (Table 7-14).

Table 7-14. Non-Target Ecological Receptor COPCs

COPC	Black Crappie	Brown Bullhead
Metals		
Aluminum	X	
Phthalates		
BEHP		X
PCBs		
Total PCBs		X

BEHP – bis(2-ethylhexyl) phthalate
COPC – contaminant of potential concern
PCB – polychlorinated biphenyl

Aluminum could not be evaluated because no TRV was available. No COPCs other than aluminum were identified for black crappie; therefore, black crappie was not further evaluated.

BEHP and total PCBs were identified as tissue-residue COPCs for brown bullhead. Attachment 4 presents a summary of BEHP and total PCB concentrations in brown bullhead tissue samples. The total PCB concentration of each composite sample was compared with the fish 10th percentile LOAEL. In only one of the six brown bullhead samples does total PCBs exceed the fish 10th percentile LOAEL (HQ = 1.8). The total PCBs site-wide UCL concentration of 1,400 µg/kg is also greater than the 10th percentile LOAEL (HQ = 1.5). No LOAEL was available for BEHP; however, concentrations of BEHP in brown bullhead from the Study Area and upriver were generally below DLs (detected in one Study Area and one upriver sample; maximum DLs were 2,700 and

3,000 µg/kg ww, respectively) and were all less than the BEHP literature-based NOAEL (9,600 µg/kg ww).

HQs for total PCBs are ≥ 1 for small-home-range fish receptors (sculpin, smallmouth bass, and northern pikeminnow) and one large-home-range fish receptor (largescale sucker), with 10th percentile LOAEL HQs higher than those for brown bullhead. Therefore, the selected fish receptors are protective of black crappie and brown bullhead.

7.2 DIETARY ASSESSMENT

The dietary-dose assessment was one LOE for evaluating risks from exposure to metabolized and regulated chemicals (i.e., inorganic metals[89] and PAHs) by all fish receptors except Pacific lamprey. Dietary exposure to contaminants from the Study Area was not evaluated as a relevant pathway for early life stage (i.e., ammocoete and macropthalmia) lamprey, which feed on suspended detritus.

Receptor-specific fish dietary COPCs were identified in the SLERA and refined screen using screening-level dietary TRVs (Attachment 5). These COPCs were evaluated by comparing diet-based toxicity thresholds to chemical concentrations in prey tissue and incidentally ingested sediment. Toxicity thresholds were expressed as concentrations in tissue and in sediment that were back-calculated from dietary-dose thresholds using receptor-specific exposure assumptions.

Uncertainties Associated with Metals in the Dietary Approach

Two inorganic metals (cadmium and copper) were identified as COPCs for fish. The utility of the dietary approach for inorganic metals has been questioned because the uptake and toxicity of inorganic metals by fish can vary widely depending on digestive physiology (e.g., gut residence time), nutritional quality of the food, distribution and chemical form of the metal in prey tissue, and environmental conditions under which toxicity was evaluated (e.g., temperature) (EPA 2007e). The dietary toxicity threshold dose estimated for the most sensitive laboratory toxicity tests provides a conservative estimate of the dietary exposure potentially associated with toxicity to receptor species of fish consuming prey from the Study Area. For this reason EPA recommends such comparison "only for conservatively screening for exposure and potential risks to consumers (i.e., in cases where whole-body residues in prey are below dietary toxic thresholds). For more definitive assessments, further research is needed to quantify the bioavailability and effects of inorganic dietary metals" (EPA 2007e). The uncertainty of this LOE for inorganic metals is considered in the evaluation of COPCs across multiple LOEs in Section 7.6.

The details of this assessment are presented as follows:

- Section 7.2.1 describes the methods used to assess dietary risks to fish.
- Section 7.2.2 summarizes the COPCs identified for all receptors evaluated in the dietary risk assessment.

[89] Per EPA (EPA 2008f), mercury was included as a COPC in the dietary-dose evaluation, although it is not a metabolized or regulated chemical.

- Section 7.2.3 presents an overview of the assumptions used to derive exposure concentrations. Exposure data in this assessment are represented by COPC concentrations in prey tissue and sediment samples. The rationale for exposure assumptions is presented in Attachment 13. All dietary exposure data (i.e., tissue and sediment concentrations) and calculated UCLs are presented in Attachment 4.
- Section 7.2.4 summarizes the effects data. In this assessment, effects data are represented by EPA-recommended NOAEL and LOAEL dietary-dose TRVs. Details and uncertainties associated with the selected TRVs for fish dietary COPCs are presented in Attachment 13. The comprehensive literature search process is presented in Attachment 14.
- Section 7.2.5 presents the risk characterization results and associated uncertainties. Results of the dietary LOE are further assessed along with the other LOEs in the fish risk conclusions (Section 7.6). The individual sample-by-sample and dietary component assessment are presented in Attachment 12.

Figure 7-8 shows how the fish dietary assessment section is organized.

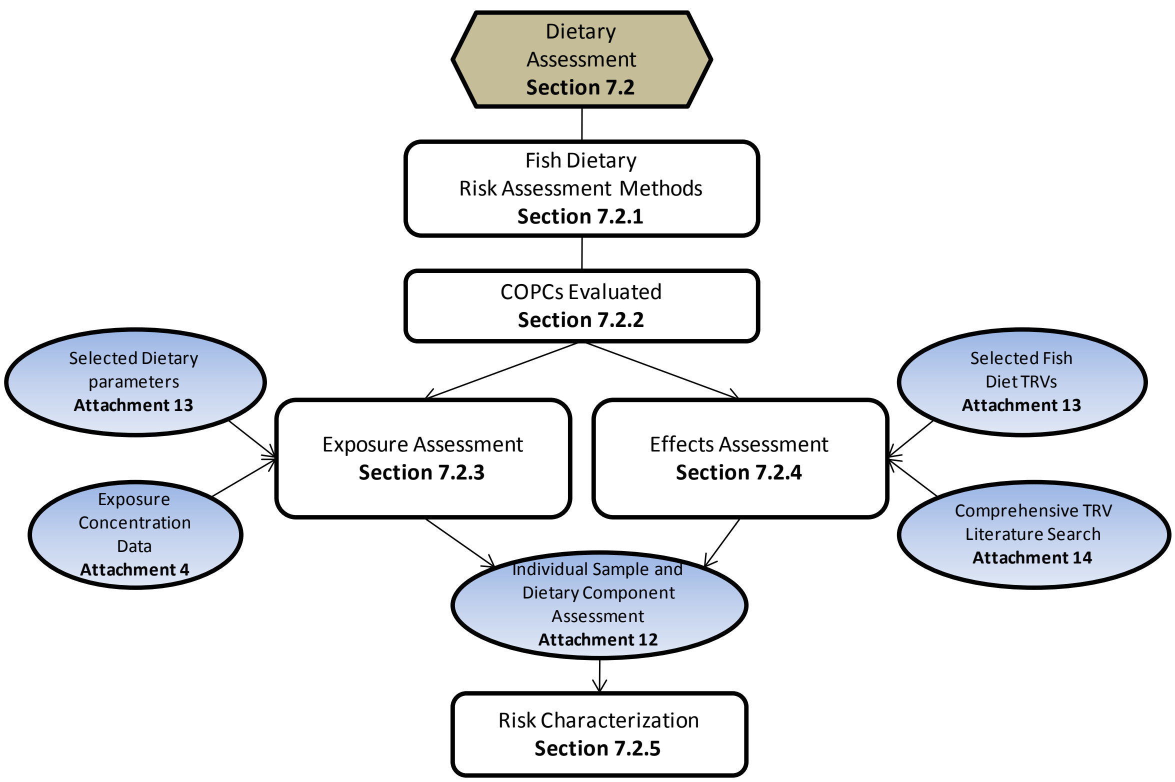

Figure 7-8. Overview of Fish Dietary Assessment Section Organization

7.2.1 Fish Dietary Risk Assessment Methods

Dietary HQs were calculated by comparing COPC concentrations in prey items and in sediment to receptor-specific toxicity thresholds. Back-calculated from dietary-dose thresholds using receptor-specific exposure assumptions, these thresholds are expressed as TTCs (mg/kg ww) in prey and as TSCs (mg/kg dw) in sediment.

EPA's Problem Formulation (Attachment 2) specified that TTCs and TSCs be back-calculated from dietary-dose TRVs using receptor-specific parameters (i.e., body weight, prey ingestion rate, incidental sediment ingestion rate [SIR], and site use) compiled from general and region-specific literature. With the back-calculation approach, tissue and sediment concentrations can be compared directly to TTCs and TSCs. This process allows for simple direct comparisons to COPC concentration data for these exposure media. The alternative would be to forward-calculate dietary-dose estimates using the same exposure parameters and equations. The forward-calculation method does not allow direct comparison to exposure media concentration data because exposure and effects data are expressed in terms of the amount of toxicant consumed per unit body weight per day (e.g., mg/kg bw/day). The two methods are mathematically equivalent (i.e., they give the same answers).

To derive TTCs and TSCs, dietary-dose TRVs (expressed in terms of the amount of toxicant consumed per unit body weight per day) were first derived from the literature using the following equation:

$$TRV_{diet} = \frac{(FIR \times C_{diet})}{BW}$$ **Equation 7-1**

Where:

TRV_{diet} = dietary-dose toxicity reference value (mg/kg bw/day)

FIR = food ingestion rate (kg food/day) as measured and reported in the toxicity study

C_{diet} = contaminant concentration in diet (NOAEL or LOAEL) as measured and reported in the toxicity study (mg/kg ww)

BW = body weight (kg) as measured and reported in the toxicity study

Next, the following equations were used to develop receptor-specific TTCs and TSCs from the dietary-dose TRVs:

$$TTC = \frac{TRV_{diet}}{\left(\frac{FIR}{BW}\right)}$$ **Equation 7-2**

Where:

TTC = threshold tissue concentration (mg/kg ww)

TRV_{diet} = dietary-dose toxicity reference value (mg/kg bw/day)

FIR = receptor-specific food ingestion rate (kg ww food/day)

BW = receptor-specific body weight (kg)

And:

$$TSC = \frac{TRV_{diet}}{\left(\frac{SIR}{BW}\right)} \qquad \textbf{Equation 7-3}$$

Where:

TSC	=	threshold sediment concentration (mg/kg dw)[90]
TRV_{diet}	=	dietary-dose toxicity reference value (mg/kg bw/day)
SIR	=	receptor-specific incidental sediment ingestion rate (kg dw sediment/day)
BW	=	receptor-specific body weight (kg)

TTC and TSC HQs were then calculated independently for prey and sediment, respectively and summed to calculate the total dietary HQ as illustrated in the following equations:

$$HQ_{TTC} = \frac{EPC_{prey}}{TTC} \text{ and } HQ_{TSC} = \frac{EPC_{sediment}}{TSC} \qquad \textbf{Equation 7-4}$$

$$HQ_{total} = HQ_{TTC} + HQ_{TSC} \qquad \textbf{Equation 7-5}$$

Where:

HQ_{total}	=	total hazard quotient based on prey tissue and incidental sediment ingestion.
HQ_{TTC}	=	threshold tissue concentration-based hazard quotient
HQ_{TSC}	=	threshold sediment concentration-based hazard quotient
EPC_{prey}	=	exposure point concentration for a given prey item
$EPC_{sediment}$	=	exposure point concentration for sediment
TTC	=	threshold tissue concentration
TSC	=	threshold sediment concentration

As described in Section 7.0, analysis of dietary risks to fish occurred in three steps, progressing from more conservative to more realistic estimates of exposure and risk:

- **Step 1 –** The derivation of HQs on an sample-by-sample basis for each composite sample of individual prey species and of sediment.
- **Step 2 –** The derivation of HQs over a relevant exposure area for individual prey species and sediment.
- **Step 3 –** The derivation of HQs over a relevant exposure area accounting for the ingestion of multiple prey species

[90] The TSC applies only to the incidental ingestion of sediment.

HQs in the first two steps were calculated per EPA (2008j), as outlined in EPA's Problem Formulation (Attachment 2). The HQ results from the first two steps were used to narrow the list of COPCs for evaluation in the third step. For Steps 1 and 2, total HQs were calculated as the sum of the TSC HQ and the maximum TTC HQ. For Step 3, to account for the ingestion of multiple prey species, dietary portions were assigned to each prey species for a given receptor. Dietary portions were used to derive total HQs using the following equation:

$$HQ_{total} = \left(\sum_{i=1}^{n} HQ_i F_i\right) + (HQ_{sed}) \qquad \textbf{Equation 7-6}$$

Where:

HQ_{total} = hazard quotient within a relevant exposure area based on multiple prey and sediment ingestion

HQ_i = hazard quotient within a relevant exposure area based on particular prey species

F_i = portion of particular prey species in the diet

n = number of dietary items

HQ_{sed} = hazard quotient within a relevant exposure area based on incidental ingestion of sediment

The Evaluation of HQs on a Sample-by-Sample Basis

Dietary risks were evaluated in Step 1 on a prey sample-by-prey sample basis per EPA direction. These results are presented in Attachment 12. This evaluation is consistent with the screening and iterative refinement procedures. A sample-by-sample analysis can be used to screen out COPCs when no individual prey sample HQs is ≥ 1. If individual sample HQs are ≥ 1, the appropriate course of action is to refine exposure assumptions before drawing risk conclusions because fish (other than sculpin) forage over relatively large areas and typically feed on multiple species. Sample-level evaluations do not represent population-level effects.

Therefore, the risk characterization of fish is based on risk estimates in which diets are composed of multiple prey species within a relevant exposure scale. Risk conclusions for fish receptors are ultimately based on these more realistic exposure assumptions.

Dietary risk conclusions were based on Step 3. [91] The results of the dietary LOE are further evaluated in the risk conclusions for fish (Section 7.6) along with results from the other LOEs in light of the magnitude, spatial distribution, and frequency of HQs; the underlying uncertainties of exposure and effects data; and agreement of HQs across LOEs (where applicable).

7.2.2 COPCs Evaluated

Receptor-COPC pairs were identified in the SLERA and refined screen (Attachment 5). Table 7-15 presents the fish dietary COPCs. Each COPC was evaluated, with the

[91] As agreed to between EPA and LWG on October 15, 2010 (LWG 2010).

exception of mono-, di-, and tetrabutyltin, which could not be evaluated because no LOAEL was available from the literature. A TRV was available for TBT. Because TBT is the most toxic butyltin (EPA 1991), risks from mono-, di-, and tetrabutyltins are assumed to be lower than those of TBT.

Table 7-15. Fish Dietary-Dose COPCs

	Omnivorous Fish		Invertivorous Fish			Piscivorous Fish	
COPC	**Largescale Sucker**	**Juvenile White Sturgeon**	**Juvenile Chinook Salmon**	**Sculpin**	**Peamouth**	**Small-mouth Bass**	**Northern Pike-minnow**
Metals							
Cadmium	X	X	X	X	X	X	X
Copper	X	X	X	X	X	X	X
Mercury	X	X	X	X	X	X	X
Butyltins							
Monobutyltin ion	X	X	X	X	X	X	X
Dibutyltin ion	X	X	X	X	X	X	X
Tetrabutyltin	X	X	X	X	X		
Tributyltin ion	X	X	X	X	X	X	X
PAHs							
Benzo(a)pyrene		X					
Total PAHs	X	X		X	X		

COPC – contaminant of potential concern
PAH – polycyclic aromatic hydrocarbon

Eleven fish diet COIs could not be screened or otherwise evaluated because dietary toxicological data were not available (Table 7-16). Dietary risk to fish associated with these contaminants is unknown.

Table 7-16. Fish Dietary-Dose COIs with No Screening-Level Threshold

COI	
Metals	
Antimony	Nickel
Chromium	Thallium
Manganese	
PAHs	
1-Methylnaphthalene	Dibenzothiophene
2-Methylnaphthalene	Perylene

Table 7-16. Fish Dietary-Dose COIs with No Screening-Level Threshold

COI	
Benzo(e)pyrene	Alkylated PAHs

COI – contaminant of interest
PAH – polycyclic aromatic hydrocarbon

7.2.3 Exposure Assessment

This section presents the methods and assumptions that were used to estimate fish dietary exposures to COPCs.

7.2.3.1 Exposure Concentrations

This section presents the methods used to derive prey tissue and sediment EPCs by the dietary approach. Tissue and sediment EPCs were calculated from detected concentrations in composite samples collected from the Study Area or from tissue concentrations measured at the end of laboratory bioaccumulation tests.

As described in Section 7.0, the dietary LOE involved three steps in which the data used to represent EPCs varied:

- **Step 1 –** EPCs were first represented by COPC concentrations in composite samples of prey tissue and sediment from throughout the Study Area to evaluate dietary risks on a sample-by-sample basis. COPC concentration data for each sample are presented in the Attachment 4.
- **Step 2 –** For those receptor-COPC pairs whose sum of the maximum prey and sediment HQs resulted in a dietary HQ ≥ 1 in Step 1, prey tissue and sediment UCL EPCs were calculated for receptor-specific exposure areas (Table 7-1). Where insufficient data were available for calculation of a UCL, the maximum concentration was used to represent the EPC. This was the case for smallmouth bass and northern pikeminnow, and EPCswere represented by maximum concentrations within the 1-mile exposure areas. Because all prey and sediment tissue samples for sculpin were composited over an area roughly equal to the sculpin exposure scale, EPCs for sculpin were represented on a sample-by-sample basis.

  UCL prey tissue and sediment EPCs were calculated using ProUCL Version 4.0 software (EPA 2007f). EPA's ProUCL software tests the goodness of fit for a given dataset and then computes the appropriate 95th UCL (as described in Section 7.1.3.1). In the case where an insufficient number of detected data values was available (n < 6), the maximum concentration[92] was used to represent the

[92] When the maximum concentration was a non-detected value, half the maximum detection limit was used to represent the EPC.

EPC. Attachment 4 presents the ProUCL-recommended UCLs and selected prey tissue and sediment EPCs. EPCs based on tissue and sediment UCLs (or maximum concentrations) were used to calculate HQs using Equations 7-4 and 7-5.

Uncertainty is associated with the use of maximum concentrations to represent prey EPCs (as discussed in Section 7.1.3.1). The use of maximum concentrations in composite samples to represent prey EPCs may result in an over- or underestimate of risks to sculpin, smallmouth bass, and northern pikeminnow because the available samples may fall above or below the true population mean.

- **Step 3 –** In order to estimate dietary risks that account for the ingestion of multiple prey species, dietary portions were assigned to each prey item for a given receptor. Prey portions that were selected are based on the diets reported in regional literature studies and are presented in Section 7.2.3.2.2. EPCs based on prey portions were used to calculate HQs using Equation 7-6.

7.2.3.2 Exposure Parameters and Dietary Prey Assumptions

The following subsections present the exposure parameters used to calculate TTCs and TSCs for fish. Dietary prey assumptions used to derive tissue EPCs are also presented.

7.2.3.2.1 Exposure Parameters

Body weights, FIRs, and SIRs vary among fish receptors, as shown in Table 7-17. Details, the rationale for the selected receptor-specific exposure parameters, and uncertainties are presented in Attachment 13.

Table 7-17. Exposure Parameters Used for Fish Dietary Risk Calculations

Receptor	BW (kg)[a]	FIR (kg ww/day)[b]	% Moisture in Prey	SI (%)[c]	SIR (kg dw/day)[d]
Largescale sucker	0.79	0.040	85%[f]	8%	0.00048
Juvenile white sturgeon	7.6	0.28	85%[f]	8%; 56%[e]	0.0033; 0.023[e]
Juvenile Chinook salmon	0.012	0.0011	79%[g]	1%	0.0000024
Peamouth	0.10	0.0072	79%[g]	5%	0.000075
Sculpin	0.020	0.0017	79%[g]	5%	0.000018
Smallmouth bass	0.40	0.022	74%[h]	1%	0.000058
Northern pikeminnow	0.56	0.030	74%[h]	1%	0.000078

a Body weights are based on field-collected data (including Round 3 data).

b FIR was calculated based on the equation from Arnot and Gobas (2004): FIR (ww) = $(0.022 \times BW^{0.85}) \times (\exp^{(0.06 \times T)})$; in which exp = 2.71828 and T = 13.4°C (average of temperatures collected by ODEQ from 1995 to 2005 from a sampling location near the SP&S Railroad Bridge).

c Percentage of diet represented by incidental sediment ingestion.

d SIR = FIR × SI. The SIR was calculated as a percentage of the FIR on a dw basis. The dw FIR was calculated according to the following equation: FIR (dw) = FIR (ww) × (1 - moisture content of diet).

e Two SI scenarios (8% and 56%) were evaluated for juvenile white sturgeon.

[f] Average percent moisture of invertebrate tissue analyzed from the Study Area.

[g] Average percent moisture of invertebrate tissue (excluding laboratory-exposed clams and crayfish) analyzed from the Study Area.

[h] Average percent moisture of fish tissue analyzed from the Study Area.

BW – body weight
dw – dry weight
FIR – food ingestion rate
ODEQ – Oregon Department of Environmental Quality
SI – sediment ingestion
SIR – sediment ingestion rate
ww – wet weight

Body weights are based on the average body weights measured in individual fish from the Round 1, 2, and 3 sampling efforts. Measured FIRs were not available for the fish receptors; food ingestionwas estimated using the equation presented in Arnot and Gobas (2004) (Equation 7-7):

$$FIR = (0.022 \times BW^{0.85}) \times exp^{(0.06*T)}$$ **Equation 7-7**

Where:

FIR = food ingestion rate (kg ww/day)
BW = body weight (kg)
exp = 2.71828
T = temperature (degrees Celsius)

Temperature was assumed to be 13.4º C, the average at a sampling location within the Study Area (SP&S Railroad Bridge) from 1995 to 2005 (ODEQ 2005). The effect of changing the water temperature on calculated fish ingestion rates is further evaluated in Section 7.2.5.4.

SIRs on a dry-weight basis were calculated according to Equation 7-8; the FIR (based on wet weight) was converted to dry weight based on the average percent moisture across relevant prey (Table 7-17).

$$SIR = (FIR \times F_{solids}) \times SI$$ **Equation 7-8**

Where:

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

Measured incidental sediment ingestion portions were not available for fish receptors. Incidental sediment ingestion was estimated using best professional judgment in consultation with fish biologists who have conducted dietary studies with the receptor species (see Attachment 13 for details).

As fish habitat areas were not defined for the Study Area, habitat was not factored into the development of site use factors (SUFs) or exposure areas. All exposure areas throughout the Study Area were assumed to provide some type of fish habitat; however, differences in habitat quality throughout the Study Area could contribute to an over- or underestimation of exposure, depending on exposure concentrations associated with different habitats.

7.2.3.2.2 Dietary Prey Assumptions

Table 7-18 indicates the prey species in the BERA dataset that were used to derive prey tissue EPCs. Prey included fish and invertebrate species collected in the Study Area (i.e., largescale sucker, carp, peamouth, sculpin, northern pikeminnow, crayfish, clams, epibenthic invertebrates, and mussels) and invertebrate species that underwent laboratory bioaccumulation testing (i.e., clams and worms). Chemical concentrations of the stomach contents of juvenile white sturgeon and juvenile Chinook salmon were also available. The prey species for each receptor were selected on the basis of information from the literature. Details, rationale for the selections, and associated uncertainties are presented in Attachment 13.

Table 7-18. Receptor-Specific Prey Species Used to Derive Risk Estimates Assuming Consumption of Single Prey Items

Prey Species	Large-scale Sucker	Juvenile White Sturgeon	Juvenile Chinook Salmon	Sculpin	Pea-mouth	Small-mouth Bass	Northern Pike-minnow
Invertebrates							
Clam	X[a]	X[a]	X[a]	X[a]	X[a]		
Worm	X[b]	X[b]	X[b]	X[b]	X[b]	X[b]	X[b]
Crayfish						X	X
Mussel		X					
Epibenthic invertebrates			X		X		
Fish							
Largescale sucker							X
Carp							X
Peamouth							X
Sculpin				X	X	X	X
Northern pikeminnow							X
Stomach Contents							
Juvenile white sturgeon		X					
Juvenile Chinook salmon			X				

[a] Risk estimates were evaluated using both laboratory and field-collected clam tissue (data were evaluated separately and not combined). Laboratory concentrations for neutral organic COPCs were represented by adjusted steady-state concentrations (see Attachment 3 for details).

[b] Laboratory concentrations for neutral organic COPCs were represented by adjusted steady-state concentrations.

COPC – contaminant of potential concern

Prey species were evaluated individually (i.e., assuming the consumption of only one prey type) in the first two steps of risk characterization (see Figure 7-9).

Uncertainties Associated with Using Laboratory Bioaccumulation Testing to Represent Prey Contaminant Concentrations

Uncertainty is associated with the use of lab worm and lab clam tissue concentrations to represent prey in the fish receptor diets. Tissues were analyzed following 28-day laboratory bioaccumulation testing with field-collected sediment from the Study Area. Field and steady-state conditions might not be represented accurately by tissue contaminant concentrations in laboratory tests because of the physical manipulation of sediments and possible changes in the chemical form affecting bioavailability and uptake.

Also, depending on the hydrophobicity of the chemical, a 28-day test might be too short for tissue concentrations to reach steady state. To address that concern, clam and worm tissue concentrations of neutral organic COPCs (i.e., benzo(a)pyrene and total PAHs) were adjusted to yield theoretical steady-state concentrations, following the process described in the Inland Testing Manual (EPA and USACE 1998) (see Attachment 3). The steady-state equations (based on McFarland (1995)) and assumptions (i.e., K_{OW} values) used to predict the steady-state adjusted concentrations are uncertain in that they do not reflect laboratory test conditions, a sediment matrix, or chemical mixtures. Adjusted clam and worm tissue chemical concentrations may over- or underestimate concentrations expected in Study Area field-collected clams and worms.

Field-collected clam tissue concentrations yield more representative dietary-dose estimates than laboratory-exposed tissue concentrations.

For receptor-COPC pairs retained through the third step in the process (Figure 7-9), the dietary proportions represented by individual prey items were varied to better represent multi-species diets presented in the literature. Table 7-19 presents the prey portions assigned to each prey species to derive HQs. Details on the rationale for the selected prey portions are presented in Attachment 13. If no data were available for a given prey species, a surrogate prey species was used (e.g., if no data were available for largescale sucker, a species of similar trophic level [such as carp] was used to represent largescale sucker contaminant concentrations). The effect on HQs caused by varying prey portions was evaluated as part of the uncertainty analysis (Section 7.2.5.4). Stomach contents for juvenile Chinook salmon were not evaluated beyond the first step of risk characterization, as the maximum concentrations did not exceed the TRVs for any COPC (see Attachment 12).

Table 7-19. Receptor-Specific Prey Species and Portions Used to Derive Risk Estimates Based on Multiple-Prey Consumption

Prey Species	Largescale Sucker[a]	Juvenile White Sturgeon[a]	Juvenile Chinook Salmon[b]	Sculpin[c]	Peamouth	Smallmouth Bass	Northern Pike-minnow
Invertebrates							
Clam[d]	1.0 and 0	1.0 and 0	0.30 and 1.0	1.0 and 0	0.25		
Worm[e]	1.0 and 0	1.0 and 0	0.40 and 1.0	1.0 and 0	0.25	0.05[f]	0.25[f]
Crayfish						0.05[g]	0.30[g]
Epibenthic			0.30[h] and 1.0		0.40[h]		

Table 7-19. Receptor-Specific Prey Species and Portions Used to Derive Risk Estimates Based on Multiple-Prey Consumption

Prey Species	Largescale Sucker[a]	Juvenile White Sturgeon[a]	Juvenile Chinook Salmon[b]	Sculpin[c]	Peamouth	Smallmouth Bass	Northern Pike-minnow
invertebrates							
Fish							
Largescale sucker							0.05[i]
Carp							0.05
Peamouth							0.05[i]
Sculpin				1.0 and 0	0.10	0.90	0.25[j]
Northern pikeminnow							0.05[i]

[a] Two scenarios were evaluated for largescale sucker and juvenile white sturgeon: one based on the ingestion of clams and one based on the ingestion on worms.

[b] Two scenarios were evaluated for juvenile Chinook salmon: one based on the ingestion of 30% clams, 40% worms, and 30% epibenthic invertebrates, and one based on ingestion of epibenthic invertebrates only.

[c] Sculpin prey were each evaluated individually on a sample-by-sample basis.

[d] HQs were calculated using field clam tissue only. Tissues from field clams are more representative of field conditions in the Study Area than are tissues from laboratory-exposed clams.

[e] HQs were calculated using laboratory-exposed worms.

[f] Crayfish were used as a surrogate when no worm tissue data were available.

[g] Worms were used as a surrogate when no crayfish tissue data were available.

[h] Clams and worms were used as a surrogate when no epibenthic invertebrate tissue data were available.

[i] Sculpin were used as a surrogate when no largescale sucker, peamouth, or northern pikeminnow tissue data were available.

[j] Carp was used as a surrogate when no sculpin tissue data were available.

HQ – hazard quotient

Uncertainty Associated with Lack of Pelagic Prey

For fish receptors whose diet includes significant portions of pelagic (water column) prey species (i.e., juvenile Chinook salmon and peamouth), risk estimates are uncertain because chemistry data for these prey organisms are not available. Pelagic prey were represented by epibenthic invertebrate tissue for most COPCs. For some COPCs, contaminant concentrations in epibenthic invertebrates were not available and the pelagic prey component was represented by benthic invertebrate organisms (i.e., field clams or laboratory-exposed worms).No TBT data were available for epibenthic invertebrate tissue; therefore, the TBT risk estimates for juvenile Chinook salmon and peamouth are highly uncertain.

BCFs can be used to estimate pelagic prey concentrations; however, BCFs are generally not available for small pelagic invertebrates. EPA (2003a) reported freshwater TBT BCFs range up to 17,483 for zebra mussels and from 240 to 2,250 for several fish species (i.e., carp, guppy, goldfish, and rainbow trout). None of the species with available BCFs represent appropriate pelagic prey for selected pelagic-feeding fish receptors. The wide range of reported BCFs demonstrates the variability of TBT uptake from the water column in aquatic organisms.

7.2.3.3 Exposure Concentrations

The exposure assessment was an iterative process. If a particular COPC was not found to pose a potentially unacceptable risk to a fish receptor in Step 1 or Step 2, it was not carried forward in the process. If in Step 1 the maximum EPC represented by a single sample led to a conclusion of potentially unacceptable risk (i.e., at least one sample had an HQ ≥ 1), then the COPC was carried forward. If in Step 2 the maximum EPC for ingestion of any individual prey species plus sediment from any foraging area resulted in a total HQ ≥ 1, the COPC was analyzed in Step 3 by accounting for prey fractions. Attachment 4 provides all EPCs (expressed as tissue and sediment concentrations) for all receptor-COPC pairs for the multiple exposure steps.

7.2.4 Effects Assessment

This section presents the TRVs used to characterize effects for fish receptor-COPC pairs and the associated uncertainties. Dietary-dose TRVs (expressed as mg/kg bw/day) are based on LOAELs and NOAELs derived from the toxicological literature. Dietary-dose TRVs were used to derive receptor-specific TTCs and TSCs following the methods described in Section 7.2.1.

Per EPA's Problem Formulation (Attachment 2), LOAELs were used to assess effects on all fish receptors evaluated at the population level. As directed in EPA's Problem Formulation, fish receptors that are threatened, endangered, otherwise protected under federal law, or are of particular cultural significance were assessed using a NOAEL. These species were evaluated at the organism level, not the population level. This status applied only to juvenile Chinook salmon; therefore, NOAELs were used to assess COPC effects on juvenile Chinook salmon.

Uncertainty is associated with the use of LOAELs to assess effects on populations, as LOAELs are based on organism-level effects. The endpoints used to derive the LOAEL for each COPC are discussed below to examine the ecological significance of TRV exceedances. See Section 7.2.1 for further discussion of how LOAELs were used in risk characterization.

7.2.4.1 Selected Dietary TRVs

EPA (2008f) provided the dietary NOAELs and LOAELs for fish; these values are based on an extensive search of the available toxicological literature. The TRVs selected for each fish dietary COPC are the lowest literature-based LOAEL and NOAEL. Attachment 13 presents the details, sources, and uncertainties associated with the selected TRVs. Attachment 14 presents the LWG-recommended literature-based fish dietary TRVs for all COPCs. The fish dietary TRVs adopted for this BERA, as well as their key uncertainties, are presented in Table 7-20.

Table 7-20. Fish Dietary-Dose TRVs

COPC	TRV (mg/kg bw/day) NOAEL	LOAEL	Source	Magnitude of Effects and Key Uncertainties
Metals				
Cadmium	0.002[a]	0.01	Kim et al. (2004); Kang et al. (2005)	The selected LOAEL is for a 23% reduction in growth of rockfish after 60 days exposure. LOAEL is 2 to 3 orders of magnitude lower than both the NOAELs and LOAELs reported in other toxicological studies; NOAEL was extrapolated from the LOAEL. Relative sensitivities of rockfish in salt water and receptors in the Study Area are unknown.
Copper	0.24	0.48	Murai et al. (1981)	The selected LOAEL is for an 8% reduction in growth of catfish after 16 weeks exposure. The selected TRVs could not be replicated by other researchers in subsequent studies using similar exposures and fish of similar age (Erickson et al. 2003; Gatlin and Wilson 1986) and have been characterized as atypical in other studies of copper in fish (Lorentzen et al. 1998); selected TRVs are at or near nutritional requirements found in the literature, ranging from 0.2 to 0.3 mg/kg bw/day (Tacon 1992; Lim et al. 2008). Relative sensitivities of catfish and Study Area receptors are unknown.
Mercury	0.005	0.013	Matta et al. (2001)	The selected LOAEL is for a 48% reduction in survival of male mummichog after 42 days exposure. LOAEL is associated with aggressive behavior and may be associated with aquaria confinement, with unknown significance for wild populations. Relative sensitivities of mummichog and Study Area receptors are unknown.
Butyltins				
TBT	0.030	0.15	Nakayama et al. (2005)	The selected LOAEL is for a 32% reduction in hatchability and 8% reduction in swim-up-success of medaka fry after 21 days exposure. Effect was not dose-responsive, as the next higher dose had a 14% reduction in hatchability and 3% reduction in swim-up success relative to controls. Relative sensitivity of medaka and Study Area receptors is unknown.
PAHs				
Benzo(a)pyrene	0.66	1.4	Rice et al. (2000)	The selected LOAEL is for a 0.7% reduction in the daily growth rate of English sole. In the second trial of this experiment, no significant effects on growth rate were observed. Limited toxicity data were available. Relative sensitivities of English sole in salt water and receptors in the Study Area are unknown.
Total PAHs	6.1	18	Meador et. al. (2006)	The selected LOAEL is for a 9% reduction in the dry weight of juvenile Chinook salmon after 53 days exposure. TRVs are based on exposure to a PAH mixture designed to resemble the field PAH mixture in the Duwamish River, Seattle and therefore may not represent PAH concentrations in the Study Area[b]; limited toxicity data were available.

[a] NOAEL was extrapolated from the LOAEL using a UF of 5. The use of UFs, as required by EPA, adds a high degree of conservatism and may overestimate risks. The use of UFs is highly uncertain and not recommended by LWG for use in determining risks for making risk management decisions (Chapman et al. 1998).

[b] Fourteen of the sixteen PAHs included in the Study Area total PAH sum were included as part of the field PAH mixture in Meador et. al. (2006); seven additional PAHs were also included.

bw – body weight
COPC – contaminant of potential concern
EPA – US Environmental Protection Agency
LOAEL – lowest-observed-adverse-effect level
LWG – Lower Willamette Group
NOAEL – no-observed-adverse-effect level
PAH – polycyclic aromatic hydrocarbon
TBT – tributyltin
TRV – toxicity reference value
UF – uncertainty factor

Fish Dietary TRV Uncertainties

Dietary-dose TRVs for fish were calculated from toxicological studies using the reported chemical exposure concentrations in food, fish body weight, and fish FIR. Fish feeding rates and body weights were reported to only a limited extent in the toxicological studies; when toxicological studies did not report fish body weight or ingestion rate, these values were derived from other literature sources. Wildlife dose-based TRVs are frequently used in ERAs, and standard ingestion rates and body weights are available. For fish, however, the dietary-dose-based approach is not commonly used in ecological risk assessment, and limited data are available to calculate dietary-dose TRVs. The effect of this uncertainty on risk estimates is unknown.

7.2.4.2 Back-Calculated TTCs and TSCs for Fish

Once dietary TRVs were selected, receptor-specific TTCs and TSCs were back-calculated using receptor-specific parameters for body weight, prey ingestion rate, and incidental SIR, following the methods described in Section 7.2.1. For all fish dietary receptor-COPC pairs, the results are shown in Table 7-21 (TTCs) and Table 7-22 (TSCs).

Table 7-21. Calculated Prey TTCs for Fish Receptor-COPC Pairs

	Prey Threshold Tissue Concentration (mg/kg ww)													
	Largescale Sucker		Juvenile White Sturgeon		Juvenile Chinook Salmon		Peamouth		Sculpin		Smallmouth Bass		Northern Pikeminnow	
COPC	NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL
Metals														
Cadmium	0.0395	0.198	0.0543	0.271	0.0218	0.109	0.0278	0.139	0.0235	0.118	0.0364	0.182	0.0373	0.187
Copper	4.74	9.48	6.51	13	2.62	5.24	3.33	6.67	2.82	5.65	4.36	8.73	4.48	8.96
Mercury	0.0988	0.257	0.136	0.353	0.0545	0.142	0.0694	0.181	0.0588	0.153	0.0909	0.236	0.0933	0.243
Butyltins														
Tributyltin ion	0.593	2.96	0.814	4.07	0.327	1.64	0.417	2.08	0.353	1.76	0.545	2.73	0.56	2.80
PAHs														
Benzo(a)pyrene	NA	NA	17.9	38.0	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Total PAHs	120	356	166	489	NA	NA	84.7	250	71.8	212	NA	NA	NA	NA

COPC – contaminant of potential concern
LOAEL – lowest-observed-adverse-effect level
NA – not applicable (not a receptor-COPC pair)
NOAEL – no-observed-adverse-effect level
PAH – polycyclic aromatic hydrocarbon
TTC – threshold tissue concentration
ww – wet weight

Table 7-22. Calculated TSCs for Fish Receptor-COPC Pairs

	Threshold Sediment Concentration (mg/kg dw)													
	Largescale Sucker		Juvenile White Sturgeon		Juvenile Chinook Salmon		Peamouth		Sculpin		Smallmouth Bass		Northern Pikeminnow	
COPC	NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL
Metals														
Cadmium	3.29	16.5	0.661 (4.61)[a]	3.3 (23.0)[a]	10	50	2.67	13.3	2.22	11.1	13.8	69	14.4	71.8
Copper	395	790	79.3 (533)[a]	159 (1,110)[a]	1,200	2,400	320	640	267	533	1,660	3,310	1,720	3,450
Mercury	8.23	21.4	1.65 (11.5)[a]	4.3 (29.9)[a]	25	65	6.67	17.3	5.56	14.4	34.5	89.7	35.9	93.3
Butyltins														
Tributyltin ion	49.4	247	69.1 (9.91)[a]	345 (49.6)[a]	150	750	40.0	200	33.3	167	207	1,030,000	215	1,080
PAHs														
Benzo(a)pyrene	NA	NA	218 (1,520)[a]	463 (3,220)[a]	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Total PAHs	10,000	29,600	2,020 (14,000)[a]	5,950 (41,500)[a]	NA	NA	8,130	24,000	6,780	20,000	NA	NA	NA	NA

[a] TSC based on assumption of 56% incidental sediment ingestion (value for 8% incidental ingestion shown in parentheses).

COPC – contaminant of potential concern
dw – dry weight
LOAEL – lowest-observed-adverse-effect level
NA – not applicable (not a receptor-COPC pair)
NOAEL – no-observed-adverse-effect level
TSC – threshold sediment concentration
PAH – polycyclic aromatic hydrocarbon

7.2.5 Risk Characterization

This section presents the fish risk characterization results for the dietary LOE. An HQ approach was used to quantify risk (Equation 6-1) following the three-step risk characterization process described in Section 7.2.1. Risk characterization results for the dietary LOE are presented in Section 7.2.5.1, with an uncertainty evaluation for each fish receptor. Section 7.2.5.2 summarizes COIs for which risks cannot be quantified. Section 7.2.5.3 summarizes risk conclusions for all fish receptors. Additional uncertainties are evaluated in Section 7.2.5.4. The relative strengths and uncertainties for all fish LOEs are evaluated together in the fish risk conclusions (Section 7.6).

7.2.5.1 Risk Characterization Results and Uncertainty Evaluation

The HQ results from the first two steps (see Section 7.2.1) are presented in Attachment 12; these results were used to narrow the list of COPCs for evaluation in the third step. The maximum HQs from Step 2 are shown in Table 7-23. Receptor-COPC pairs resulting in an HQ ≥ 1 in the Step 2 analysis of individual prey were retained for analysis in Step 3. HQs from the Step 3 analysis within relevant exposure areas are presented below by fish receptor.

Table 7-23. Maximum HQs from Step 2a

COPC	Largescale Sucker	Juvenile White Sturgeon	Juvenile Chinook Salmon	Peamouth	Sculpin[b]	Smallmouth Bass	Northern Pikeminnow
Metals							
Cadmium	0.52	0.68	**4.5**	0.74	**6.4**	**1.3**	**1.2**
Copper	**1.1**	**1.2**	**3.8**	**1.5**	**8.9**	**2.4**	**2.4**
Mercury	0.061	0.092	0.24	0.32	**5.3**	0.55	**2.0**
Butyltins							
TBT	NA	0.13	**1.1**	0.18	**1.2**	NA	NA

Note: HQs shown here are the sum of maximum tissue and sediment HQs for relevant exposure area(s). If HQs were calculated over multiple relevant exposure areas, the HQ shown is the highest combined tissue and sediment HQ.

[a] For COPCs with HQ ≥ 1 in Step 1 based on individual prey species.

[b] Sculpin were not evaluated in a second step in Attachment 12. Step 1 HQs shown are the sum of maximum site-wide tissue and sediment sample-by-sample HQs.

COPC – contaminant of potential concern

HQ – hazard quotient

NA – not applicable (HQs are < 1 for all 1-mile exposure areas based on all individual prey species)

TBT – tributyltin

Bold identifies HQs ≥ 1.

7.2.5.1.1 Large-Home-Range Fish

HQs for large-home-range fish resulting from Step 3 of the risk characterization are presented in Table 7-24 and discussed below.

Table 7-24. Large-Home-Range Fish Site-Wide LOAEL HQs from Step 3

		Total HQ[a]		
Receptor	**Prey Assumption**	**TBT**	**Cadmium**	**Copper**
Largescale sucker	100% clams	NA	NA	**1.1**
	100% worms	NA	NA	0.50
Juvenile white sturgeon	100% clams	NA	NA	**1.2** (0.80)[b]
	100% worms	NA	NA	0.73 (0.36)[b]
	Stomach contents[c]	NA	NA	0.85
Juvenile Chinook Salmon	30% clams, 40% worms, 30% epibenthic tissue	0.79[d]	**3.5**	**2.5**
	100% epibenthic tissue	ND	**1.7**	**2.4**
Peamouth	25% clam, 25% worm, 40% epibenthic tissue, 10% sculpin	NA	NA	**1.0**

[a] HQs are the sum of TTC HQ and TSC HQ, calculated using literature-based LOAELs for all receptors except juvenile Chinook salmon. HQs for juvenile Chinook salmon are calculated using NOAELs.

[b] HQ is based on 56% incidentally ingested sediment; (HQ based on 8% sediment ingestion is shown in parentheses).

[c] The total HQ shown for stomach contents is the stomach contents HQ. It has not been summed with a sediment HQ because stomach contents already account for incidental sediment ingestion.

[d] Based on 45% clams and 55% worms; no TBT data available from epibenthic tissue.

HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
ND – no data
NA – not applicable (not a receptor-COPC pair evaluated)
NOAEL – no-observed-adverse-effect level
TBT – tributyltin
TSC – threshold sediment concentration
TTC – threshold tissue concentration

Largescale Sucker

Five COPCs were identified for largescale sucker in the SLERA and refined screen.[93] One COPC (copper) has an HQ ≥ 1 in Step 2 based on individual prey components (Table 7-23). In Step 3, a multiple prey species diet was not evaluated; only individual prey were evaluated because both clams and worms are thought to be equally representative of the benthic invertebrate prey for largescale suckers (Table 7-24).

[93] Monobutyltin, dibutyltin, and tetrabutyltin were not included in this count because TBT was used as a surrogate.

The copper HQ is ≥ 1 only for clams. The copper HQ would be < 1 if exposure was averaged over both clams and worms. Sediment contributes only 10% of the total dietary HQ. A distinct pattern of copper concentrations in individual clams throughout the Study Area was not evident.

There is additional uncertainty in the selected dietary exposure parameters. As noted below in Section 7.2.5.4, assumptions about water temperature (as it affects feeding rate) and body weight could increase or decrease dietary HQs by as much as 25%.

A high degree of uncertainty is associated with the selected LOAEL TRV for copper because the 8% reduction in channel catfish growth documented by Murai et al. (1981) could not be replicated by other researchers in subsequent studies in which researchers used similar exposures and fish of similar age (Erickson et al. 2003; Gatlin and Wilson 1986); the Murai et al. (1981) study has been characterized as atypical by another study of copper in fish (Lorentzen et al. 1998) (see Attachment 14 for additional details). These subsequent studies confirm that the Murai et al. (1981) study results are anomalous. Furthermore, the selected copper dietary NOAEL and LOAEL TRVs (0.24 and 0.48 mg/kg bw/day, respectively) are at or near the high end of the range of nutritional requirements found in the literature for some but not all fish species; these nutritional requirements range between 0.2 and 0.3 mg/kg bw/day[94] (Tacon 1992; Lim et al. 2008).

The next lowest copper LOAEL and NOAEL from the toxicity studies reviewed were 2.0 and 1.0 mg/kg bw/day, respectively, for reduced growth of rockfish (Kang et al. 2005). If calculated from this next lowest LOAEL for largescale sucker, the TTC would be 39 mg/kg ww and the TSC would be 3,274 mg/kg dw. The total LOAEL HQ for a largescale sucker diet consisting solely of clams (the only diet yielding an HQ ≥ 1) would change from 1.1 using the Murai et al. (1981) data (Table 7-24) to 0.27 using the Kang et al. (2005) data. The relative sensitivities of channel catfish, rockfish, and largescale sucker to copper are unknown.

Juvenile White Sturgeon

Six COPCs were identified for juvenile white sturgeon in the SLERA and refined screen.[95] One COPC (copper) has an HQ ≥ 1 in Step 2 based on individual prey components (Table 7-23). As for largescale sucker, an analysis of a multiple prey item diet was not performed in Step 3; only stomach contents and individual prey were evaluated because both clam and worms were thought to be representative of benthic invertebrate prey for juvenile white sturgeon (Table 7-24).

The site-wide copper HQ based on ingestion of clams (assuming 56% incidental sediment ingestion) is ≥ 1. For stomach contents and for consumption of worms plus 56%

[94] Nutritional requirement estimates are based on atypical fish consumption level for aquaculture of 5% bw/day (Gatlin and Wilson 1986; Lall and Hines 1987).

[95] Monobutyltin, dibutyltin, and tetrabutyltin were not included in this count because TBT was used as a surrogate.

incidental sediment ingestion, copper HQs are < 1. When 8% sediment ingestion is assumed, HQs for all dietary items are < 1. Sediment contributes 33% of the total copper HQ for the clam/incidental sediment diet. A distinct pattern of copper concentrations in individual clams throughout the Study Area was not evident.

Several uncertainties are associated with the exposure assumptions used to derive risk estimates. The greatest uncertainty with the white sturgeon dietary assessment is the assumption that juvenile sturgeon forage only within the 10-mile reach of the Study Area. The literature and PIT-tagged sturgeon collected during Round 3 indicate that the exposure area is likely much greater than the Study Area. Juvenile sturgeon dietary exposure concentrations may be over- or underestimated, depending on the exposure to contaminants outside of the Study Area. The use of contaminant concentrations in worm tissues after 28-day laboratory bioaccumulation tests may not be representative of those in benthic invertebrates of the Study Area and may over- or underestimate risks. There is additional uncerainty in the selected dietary exposure parameters. As noted below in Section 7.2.5.4, assumptions about body weight and water temperature (as it affects feeding rate) could increase or decrease dietary HQs by as much as 25%.

A high degree of uncertainty is associated with the selected TRVs for copper, as described above for largescale sucker. Table 7-25 compares copper HQs for juvenile white sturgeon based on the LOAELs presented in Murai et al. (1981) and those in Kang et al. (2005). No HQs are ≥ 1 when using the Kang et al. (2005) toxicity data. The relative sensitivities of channel catfish, rockfish, and juvenile white sturgeon to copper are unknown.

Table 7-25. Comparison of Juvenile White Sturgeon Copper LOAEL HQs

Prey Assumption	LOAEL HQ[a]	
	Based on Murai et al. (1981)	**Based on Kang et al. (2005)**
100% Clam	**1.2**	0.28
100% Epibenthic tissue	**1.3**	0.31

[a] Sum of TTC and TSC HQs.

HQ – hazard quotient

LOAEL – lowest-observed-adverse-effect level

Bold identifies HQs ≥ 1.

Juvenile Chinook Salmon

Four COPCs were identified for juvenile Chinook salmon in the SLERA and refined screen.[96] Three COPCs (i.e., cadmium, copper, and TBT) have NOAEL HQs ≥ 1 based on individual prey components (Table 7-23). HQs for these COPCs were calculated across multiple prey species and incidental sediment ingestion using NOAEL-based TTCs and TSCs (Table 7-24).

Two of the three COPCs evaluated (cadmium, and copper) have site-wide NOAEL HQs ≥ 1 for both the multiple-prey species diet and the epibenthic invertebrate-only diet (Table 7-24). A TBT HQ could be derived using only worm and clam data because analyses for TBT were not conducted in epibenthic invertebrate tissue samples. Site-wide HQs for TBT based on the combined worm and clam diet are < 1.

Several uncertainties are associated with the exposure assumptions used to derive the risk estimates. As primarily pelagic feeders, juvenile Chinook salmon are unlikely to ingest the benthic organisms (i.e., worms and clams) that were used to represent their diet. The unrealistic substitution of worms and clams for pelagic prey probably overestimates dietary exposures to sediment-associated contaminants because the bioaccumulation of inorganic metals from contaminated sediments is generally greater for benthic organisms than for pelagic organisms (e.g., Farag et al. 1998). In addition, the use of stead-state-adjusted tissue concentrations in worms after 28-day laboratory bioaccumulation tests might not be representative of concentrations in benthic invertebrates collected directly from the Study Area. Additional uncertainty is found in the selected dietary exposure parameters. As noted in Section 7.2.5.4, assumptions about water temperature (as it affects feeding rate) and body weight could increase or decrease dietary HQs by as much as 25%. The effect of alternative but plausible assumptions about relative proportions of prey items in the diet was analyzed, as reported in Section 7.2.5.4; HQs would not change from ≥ 1 to < 1 or vice versa.

Additional uncertainties are associated with the effects data used to derive the risk estimates for each of the COPCs.

- **TBT –** The selected LOAEL is for a 32% reduction in hatchability and 8% reduction in swim-up-success of medaka fry. Selection of the NOAEL from this study is conservative because the observed effects were not dose-responsive. The relative sensitivities of medaka and juvenile Chinook salmon to TBT are unknown.
- **Cadmium –** The NOAEL used to evaluate risks to juvenile Chinook salmon was extrapolated from the LOAEL using an uncertainty factor (UF) of 5. The LOAEL for a 23% reduction in growth of rockfish in salt water following dietary exposure to cadmium is much lower than effects thresholds reported in other toxicity studies. It is 2 to 3 orders of magnitude lower than the nine NOAELs identified in

[96] Monobutyltin, dibutyltin, and tetrabutyltin were not included in this count because TBT was used as a surrogate.

the other studies reviewed (including four salmonid NOAELs) and 2 to 3 orders of magnitude lower than the four LOAELs reported in the other studies reviewed (including two salmonid LOAELs). As such, the selected TRVs are conservative because the majority of the toxicological studies reviewed (including 4 NOAELs and 2 LOAELs for salmonids) indicate that the selected TRVs overpredict cadmium toxicity to juvenile salmon. HQs based on a NOAEL TRV from any other study would be much lower than 1.

- **Copper –** A high degree of uncertainty is associated with the selected TRVs for copper, as discussed above for largescale sucker.The juvenile Chinook salmon TTC and TSC based on the NOAEL reported in Kang et al. (2005) value would be 10.9 mg/kg ww and 5,000 mg/kg dw, respectively. As shown in Table 7-26, juvenile Chinook copper HQs would be < 1 if calculated using the NOAELs presented in Kang et al. (2005) rather than Murai et al. (1981). No HQs would be ≥ 1 based on the toxicity data presented in Kang et al. (2005). Salmonid specific toxicity data indicate the juvenile Chinook are likely less sensitive than rockfish; 12 salmon-specific NOAELs were higher than the NOAEL reported in Kang et al. (2005).

Table 7-26. Comparison of Juvenile Chinook Salmon Copper NOAEL HQs

	NOAEL HQ[a]	
Prey Assumption	**Based on Murai et al. (1981)**	**Based on Kang et al. (2005)**
30% Clam, 40% worm, 30% epibenthic tissue	**2.5**	0.59
100% Epibenthic tissue	**2.4**	0.56

[a] Sum of TTC and TSC HQs.
NOAEL – no-observed-adverse-effect level
HQ – hazard quotient
Bold identifies HQs ≥1.

Peamouth

Five COPCs were identified for peamouth in the SLERA and refined screen.[97] One COPC (copper) has an HQ ≥ 1 based on individual prey components (Table 7-23). Copper HQs were calculated across multiple prey species and incidental sediment ingestion using LOAEL-based TTCs and TSCs (Table 7-24). The copper HQs based on multiple prey species for all exposure areas are ≤ 1.

Several uncertainties are associated with the exposure assumptions used to derive the risk estimates. As noted in Section 7.2.5.4, copper HQs would be ≥ 1 if clams constituted more than 67% of the peamouth's diet, an unlikely scenario given the range of bivalves reported in peamouth diets. Assumptions about water temperature (as it affects feeding

[97] Monobutyltin, dibutyltin, and tetrabutyltin were not included in this count because TBT was used as a surrogate.

rate) and body weight could increase or decrease dietary HQs by as much as 25% (see analysis in Section 7.2.5.4). A high degree of uncertainty is associated with the selected TRVs for copper, as discussed above for largescale sucker. If the higher TRV had been selected, the copper HQ for peamouth would be < 1. The relative sensitivities of catfish, rockfish, and peamouth to copper are unknown.

7.2.5.1.2 Small-Home-Range Fish

Sculpin

Five COPCs were identified for sculpin in the SLERA and refined screen.[98] In Step 1, four COPCs (i.e., cadmium, copper, mercury, and TBT) have HQs ≥ 1 based on individual prey components (Table 7-23). The relevant exposure area for sculpin was considered roughly equivalent to the sampling area of prey; therefore, sculpin were not evaluated in a second step in Attachment 12. Instead, the sculpin diet was evaluated on a sample-by-sample and individual prey basis. The prey items evaluated for sculpin were clam, lab worm, and sculpin. Sediment ingested incidentally was assumed to constitute 5% of the diet. HQs for individual prey items are presented in Table 7-27. A spatial distribution of HQs for each COPC is presented on Map 7-7 and Figures 7-9 through 7-12.

Table 7-27. Number of Sculpin Prey and Sediment Samples with LOAEL HQs ≥ 1

	Number of Samples with HQs ≥ 1 (HQ range)			
COPC	**Clam**	**Lab Worm**	**Sculpin**	**Sediment**
Metals				
Cadmium	7 of 38 (0.34 – **1.8**)	2 of 35 (0.31 – **2.2**)	0 of 38 (0.025 – 0.19)	1 of 1,348 (0.00014 – **4.2**)
Copper	38 of 38 (1.1 – **2.4**)	1 of 35 (0.32 – **3.6**)	1 of 38 (0.16 – **1.3**)	9 of 1,358 (0.0082 – **5.3**)
Mercury	0 of 35 (0.033 – 0.17)	0 of 34 (0.020 – 0.069)	0 of 39 (0.16 – 0.83)	1 of 1,345 (0.00042 – **4.5**)
Butyltins				
Tributyltin ion	0 of 34 (0.0010 – 0.30)	0 of 35 (0.00025 – 0.97)	0 of 12 (0.0010 – 0.0023)	0 of 405 ($1.1x10^{-8}$ – 0.28)

COPC – contaminant of potential concern

HQ – hazard quotient

LOAEL – lowest-observed-adverse-effect level

[98] Monobutyltin, dibutyltin, and tetrabutyltin were not included in this count because TBT was used as a surrogate.

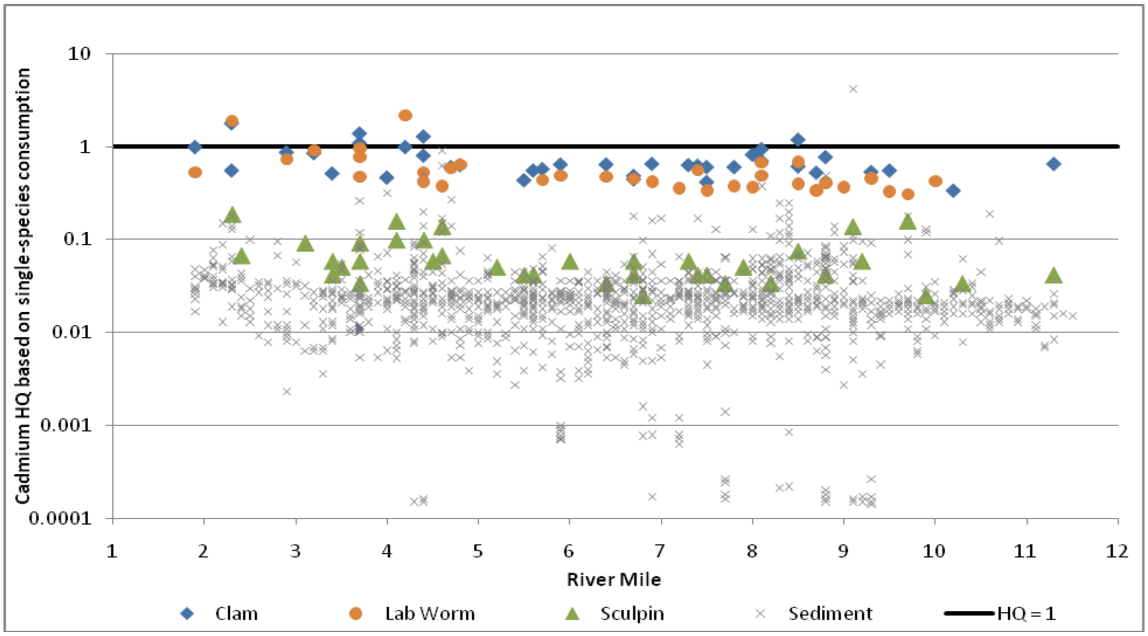

Figure 7-9. Sculpin Prey Tissue and Sediment HQs by RM for Cadmium

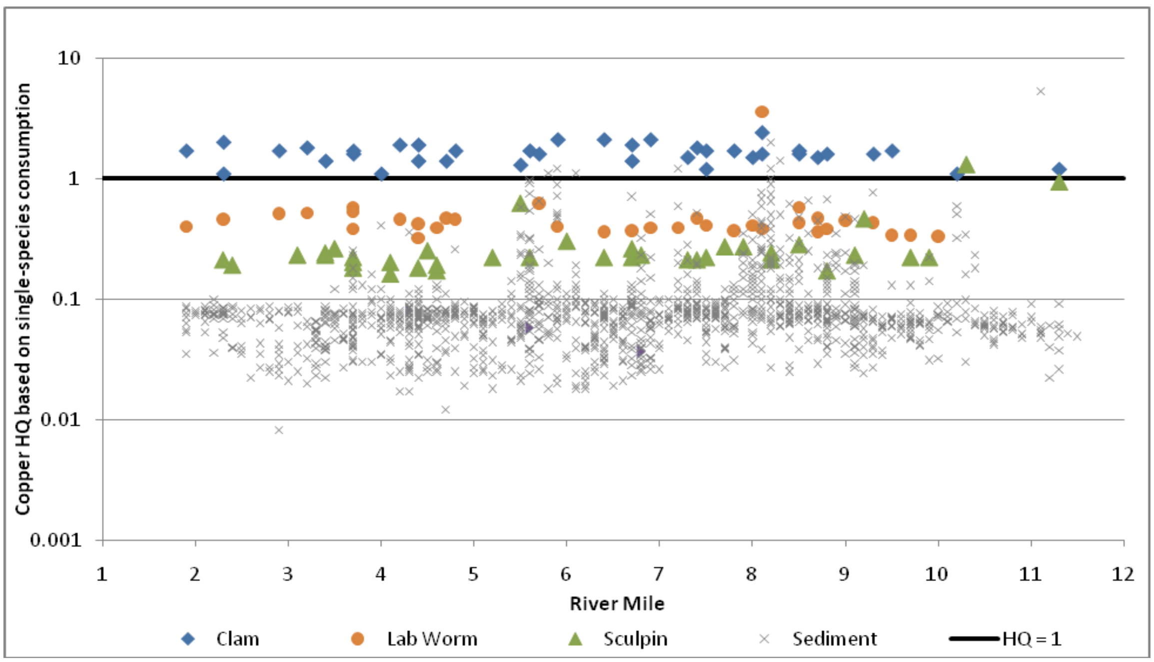

Figure 7-10. Sculpin Prey Tissue and Sediment HQs by RM for Copper

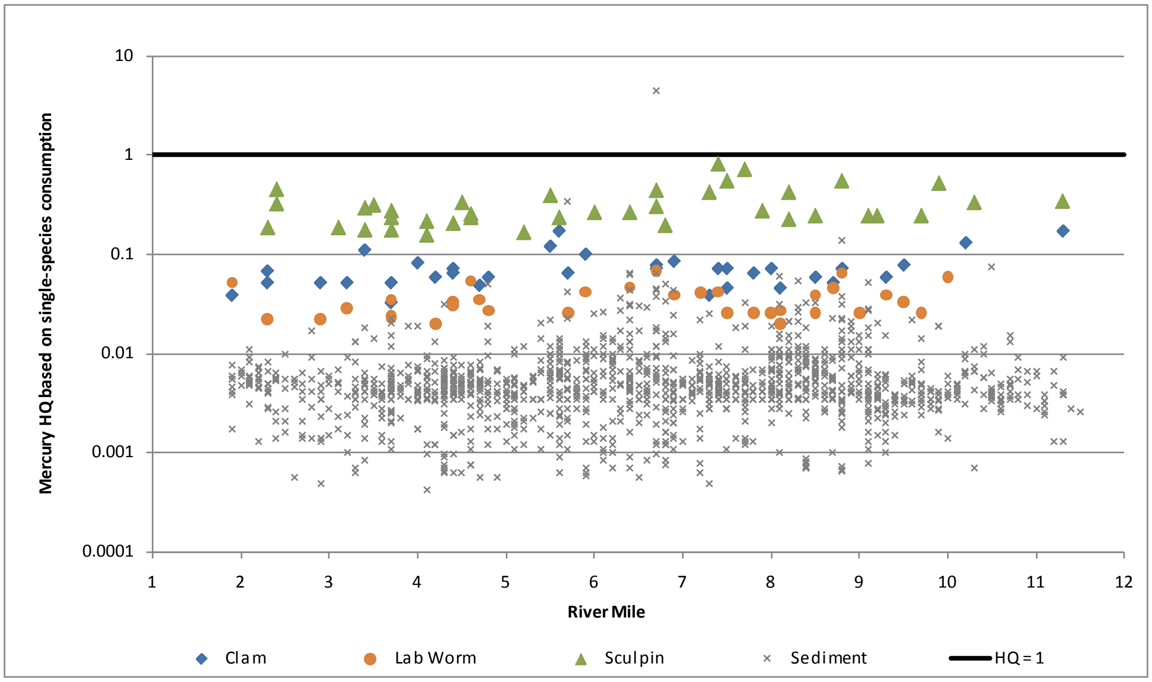

Figure 7-11. Sculpin Prey Tissue and Sediment HQs by RM for Mercury

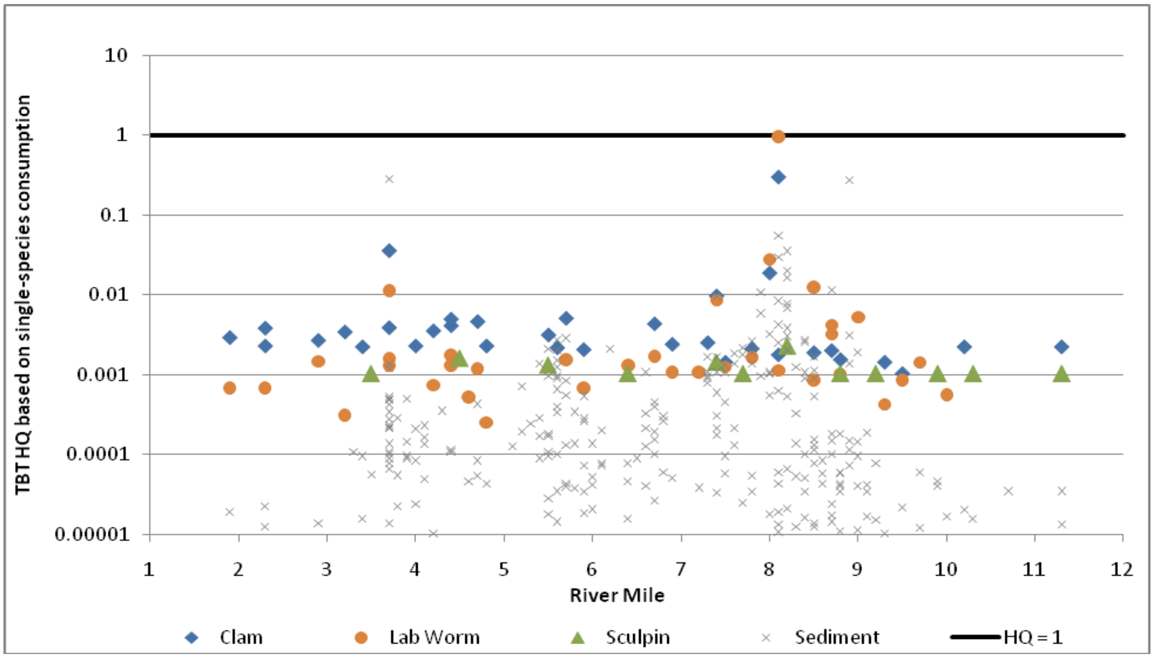

Figure 7-12. Sculpin Prey Tissue and Sediment HQs by RM for TBT

Cadmium, copper, mercury, and TBT were all identified as COPCs with HQs ≥ 1 for sculpin. The spatial distribution of risk estimates are discussed below:

- **Cadmium –** HQs are ≥ 1 for seven field clam samples, two worm samples, and one sediment sample. Samples with HQs ≥ 1 were located generally within five areas: at approximately RM 2.2 on the east side, RM 4.2 on the east side, International Slip, Slip 1, and Swan Island Lagoon (Map 7-7 and Figure 7-9). Because of the low magnitude of HQs and the low frequency of exceedance, cadmium might pose dietary risks only to sculpin that forage within these localized areas; however, sculpin that consume prey from any other locations in the Study Area are likely not at risk.
- **Copper –** HQs are ≥ 1 for all field clam samples, one worm sample, one sculpin sample, and eight sediment samples. Copper exceedances in clams are distributed site-wide (Figure 7-10; Map 7-7); however, risk conclusions for copper are highly uncertain because of uncertainties associated with the TRVs (as discussed in more detail below).
- **Mercury –** One sediment sample exceeded sculpin dietary thresholds for mercury; however, an individual sediment sample does not represent a realistic dietary exposure concentration. Sediment and tissue samples resulting in HQs close to 1 were not collected from locations near one another (Figure 7-11; Map 7-7)
- **TBT –** No individual prey item exceeded the TTC and no sediment sample exceeded the TSC. When the TTC and TSC HQs were combined, however, the total HQ based on maximum tissue and sediment concentrations has an HQ ≥ 1. Only one worm sample based on laboratory bioaccumulation testing had TBT concentrations approaching the TTC. This sample at RM 8.1 resulted in an HQ of 0.97. Three nearby sediment samples (from RM 8.1 and RM 8.2) could result in a total HQ ≥ 1; their TSC HQs range from 0.030 to 0.056 (Figure 7-12; Map 7-7). The sum of the high lab worm HQ and the maximum HQ of the three co-located sediment samples is 1.

Several uncertainties are associated with the exposure assumptions used to derive the risk estimates. The source of the primary uncertainty is the use of only single composite samples of individual prey species. Dietary composition data discussed in Attachment 13 indicates that fish likely constitute some fraction of the sculpin diet. Figures 7-9 and 7-10 indicate that a multi-species diet including fish would reduce cadmium and copper HQs to near or below 1 throughout the Study Area. As noted in Section 7.2.5.4, assumptions about water temperature (as it affects feeding rate) and body weight could increase or decrease dietary HQs by as much as 25%. In addition, the use of worm tissue concentrations from 28-day laboratory bioaccumulation tests might not be representative of benthic invertebrate field tissue contaminant concentrations in the Study Area and may over- or underestimate risks.

The following additional uncertaintiesare associated with the effects data used to derive the risk estimates for each of the COPCs.

- **TBT –** The selected reproduction LOAEL is for a 32% reduction in hatchability and 8% reduction in swim-up-success of medaka fry. Selection of this LOAEL is conservative because the observed effects were not dose-responsive. The relative sensitivities of medaka and sculpin to TBT are unknown.
- **Cadmium –** As discussed above for juvenile Chinook salmon the selected LOAEL for a 23% reduction in growth of rockfish in salt water following dietary exposure to cadmium is conservative because the majority of the toxicological studies reviewed indicate that the selected TRVs may over-predict cadmium toxicity to fish. HQs based on a LOAEL TRV from any other study would result in HQs much lower than 1. The relative sensitivities of rockfish and sculpin to cadmium are unknown.
- **Copper –** A high degree of uncertainty is associated with the selected TRVs for an 8% reduction in growth of catfish following dietary exposure to copper, as discussed above for juvenile white sturgeon. Prey-specific copper HQs for sculpin based on the LOAELs presented in Murai et al. (1981) and Kang et al. (2005) are shown in Table 7-28. None of the HQs for prey items are ≥ 1 when the Kang et al. (2005) toxicity data are used. One sediment sample results in an HQ of 1.3; however, an individual sediment sample does not represent a realistic dietary exposure concentration. The relative sensitivities of catfish, rockfish, and sculpin to copper are unknown.

Table 7-28. Comparison of Sculpin Copper LOAEL HQs for Individual Prey Items

Prey Item	Copper Concentration Range (mg/kg)	LOAEL HQ Based on Murai et al. **(1981)**		LOAEL HQ Based on Kang et al. **(2005)**	
		LOAEL TTC/TSC	LOAEL HQ Range	LOAEL TTC/TSC	LOAEL HQ Range
Clam	5.99 – 13.5 (ww)	5.65	**1.1 – 2.4**	23.5	0.25 – 0.57
Lab worm	1.83 – 20.2 (ww)	5.65	0.32 – **3.6**	23.5	0.078 – 0.86
Sculpin	0.929 – 7.16 (ww)	5.65	0.16 – **1.3**	23.5	0.040 – 0.30
Sediment	4.37 – 2,830 (dw)	533	0.0082 – **5.3**	2,220	0.0020 – **1.3**

dw – dry weight
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
Bold identifies HQs ≥ 1.
TSC – threshold sediment concentration
TTC – threshold tissue concentration
ww – wet weight

Smallmouth Bass

Four COPCs were identified for smallmouth bass in the SLERA and refined screen[99]. Two COPCs (i.e., cadmium, and copper) have HQs ≥ 1 based on individual prey components (Table 7-23).TTC-based LOAEL HQs for these COPCs were calculated across multiple prey and summed with TSC-based LOAEL HQs for incidental sediment ingestion, as shown by exposure area in Table 7-29.

Table 7-29. Smallmouth Bass 1-Mile Exposure Area LOAEL HQs Across Multiple Prey Items

	Total HQ[a]	
Exposure Area	**Cadmium**	**Copper**
RM 1.5 to RM 2.5	0.19	0.22
RM 2.5 to RM 3.5	0.10	0.27
RM 3.5 to RM 4.5	0.14	0.26
RM 4.5 to RM 5.5	0.12	0.26
RM 5.5 to RM 6.5	0.064	0.52
RM 6.5 to RM 7.5	0.062	0.29
RM 7.5 to RM 8.5	0.056	0.28
RM 8.5 to RM 9.5	0.069	0.40
Swan Island Lagoon	0.13	0.41
RM 9.5 to RM 10.5	0.11	0.87
RM 10.5 to RM 11.8	0.042	0.86

[a] Total HQ were calculated using the following prey portions: 90% sculpin, 5% crayfish, and 5% lab worms. When no lab worm data were available, crayfish were assigned a prey portion of 10%; and when no crayfish data were available, lab worms were assigned a prey portion of 10%.

HQ – hazard quotient

LOAEL – lowest-observed-adverse-effect level

RM – river mile

HQs based on multiple prey items for all exposure areas are < 1 for cadmium and copper; therefore, these COPCs are not expected to pose unacceptable risks to smallmouth bass.

Several uncertainties are associated with the exposure assumptions used to derive the risk estimates. Most of the EPCs were based on maximum concentrations (see Attachment 4). As discussed in Section 7.2.3.1, uncertainty is associated with the use of maximum concentrations to represent prey EPCs; this uncertainty may result in an over or underestimate of risk because the available samples may fall above or below the true population mean. As noted in Section 7.2.5, assumptions about body weight and water

[99] Monobutyltin, dibutyltin, and tetrabutyltin were not included in this count because TBT was used as a surrogate.

temperature (as it affects feeding rate) could increase or decrease dietary HQs by as much as 25%. In addition, concentrations in worm tissues after 28-day laboratory bioaccumulation tests might not be representative of those in benthic invertebrates collected directly from the Study Area; risk may be over- or underestimated.

Additional uncertainties are associated with the effects data used to derive the risk estimates for each of the COPCs. These uncertainties are the same as those discussed above for sculpin-cadmium and sculpin-copper. Selection of alternative TRVs would result in lower HQs.

Northern Pikeminnow

Four COPCs were identified for northern pikeminnow in the SLERA and refined screen.[100] Three COPCs (i.e., cadmium, copper, and mercury) have HQs ≥ 1 based on individual prey components (Table 7-23). LOAEL-based TTC HQs for these COPCs were calculated across multiple prey and summed with LOAEL-based TSC HQs for incidental sediment ingestion, as shown by exposure area in Table 7-30.

Table 7-30. Northern Pikeminnow 1-Mile Exposure Area LOAEL HQs Across Multiple Prey Items

	Total HQ[a]		
Exposure Area	**Cadmium**	**Copper**	**Mercury**
RM 1.5 to RM 2.5	0.43	0.56	0.16
RM 2.5 to RM 3.5	0.25	0.72	0.16
RM 3.5 to RM 4.5	0.37	0.70	0.13
RM 4.5 to RM 5.5	0.21	0.66	0.21
RM 5.5 to RM 6.5	0.19	0.85	0.23
RM 6.5 to RM 7.5	0.18	0.76	0.34
RM 7.5 to RM 8.5	0.16	0.61	0.28
RM 8.5 to RM 9.5	0.15	0.73	0.25
Swan Island Lagoon	0.23	**1.3**	0.17
RM 9.5 to RM 10.5	0.16	0.91	0.19
RM 10.5 to RM 11.8	0.12	**1.5**	0.16

[a] Total HQ was calculated using the following prey portions: 30% crayfish, 25% lab worms, 25% sculpin, 5% largescale sucker, 5% carp, 5% peamouth, and 5% northern pikeminnow. When no pikeminnow data were available, sculpin were assigned a prey portion of 30%. When no lab worm data were available, an additional 25% was added to the prey portion of crayfish.

HQ – hazard quotient
RM – river mile
LOAEL – lowest-observed-adverse-effect level
Bold identifies HQs ≥ 1.

[100] Monobutyltin, dibutyltin, and tetrabutyltin were not included in this count because TBT was used as a surrogate.

Copper HQs based on a multiple-prey diet were $\geq$ 1 in two exposure areas (1.3 in Swan Island Lagoon and 1.5 between RM 10.5 and RM 11.8).

Several uncertainties are associated with the exposure assumptions used to derive the risk estimates. Prey tissue contaminant concentrations were represented by those in worms after 28-day laboratory bioaccumulation tests, and might not be representative of those in benthic invertebrates collected directly from the Study Area; risk may be over- or underestimated. Alternative but plausible assumptions about the relative proportion of the northern pikeminnow diet contributed by the various prey items would result in copper HQs < 1 (i.e., if crayfish and lab worms collectively made up approximately < 20% of the diet; see Section 7.2.5.4 below). Reasonable alternative diets do not affect cadmium and mercury HQs. Also as noted in Section 7.2.5.4, assumptions about water temperature (as it affects feeding rate) and body weight could increase or decrease dietary HQs by as much as 25%.

Additional uncertainties are associated with the effects data:

- **Cadmium –** As discussed above for juvenile Chinook salmon, the selected LOAEL is conservative because the majority of the toxicological studies reviewed indicate that the selected TRV may overpredict cadmium toxicity to fish. HQs based on a LOAEL TRV from any other study would result in HQs much lower than 1. The relative sensitivities of rockfish and northern pikeminnow to cadmium are unknown.
- **Copper –** A high degree of uncertainty is associated with the selected TRVs for copper, as discussed above for juvenile white sturgeon. Copper HQs for northern pikeminnow based on the LOAELs presented in Murai et al. (1981) and Kang et al. (2005) are shown in Table 7-31. When the toxicity data presented in Kang et al. (2005) are used, calculated HQs are consistently < 1 in all exposure areas. The relative sensitivities of channel catfish, rockfish, and northern pikeminnow to copper are unknown.

Table 7-31. Comparison of Northern Pikeminnow Copper LOAEL HQs

	Total LOAEL HQ[a]	
Exposure Area	**Based on Murai et al. (1981)**	**Based on Kang et al. (2005)**
RM 1.5 to RM 2.5	0.56	0.14
RM 2.5 to RM 3.5	0.72	0.18
RM 3.5 to RM 4.5	0.70	0.16
RM 4.5 to RM 5.5	0.66	0.16
RM 5.5 to RM 6.5	0.85	0.20
RM 6.5 to RM 7.5	0.76	0.18

Table 7-31. Comparison of Northern Pikeminnow Copper LOAEL HQs

	Total LOAEL HQ[a]	
Exposure Area	**Based on Murai et al. (1981)**	**Based on Kang et al. (2005)**
RM 7.5 to RM 8.5	0.61	0.15
RM 8.5 to RM 9.5	0.73	0.18
Swan Island Lagoon	**1.3**	0.30
RM 9.5 to RM 10.5	0.91	0.22
RM 10.5 to RM 11.8	**1.5**	0.36

[a] Total HQ was calculated using the following prey portions: 30% crayfish, 25% lab worms, 25% sculpin, 5% largescale sucker, 5% carp, 5% peamouth, and 5% northern pikeminnow. When no pikeminnow data were available, sculpin were assigned a prey portion of 30%. When no lab worm data were available, an additional 25% was added to the prey portion of crayfish.

HQ – hazard quotient

LOAEL – lowest-observed-adverse-effect level

Bold identifies HQs ≥ 1.

7.2.5.2 COIs for Which Risks Cannot Be Quantified

Dietary risks to fish from 11 COIs could not be quantified because no dietary TRV is available. They COIs are listed in Table 7-32.

Table 7-32. Fish Dietary-Dose COIs with No Available TRVs

COI	Rationale for Why Risks Cannot Be Quantitatively Evaluated
Metals	
Antimony	Dietary risks to fish unknown; no dietary TRV available.
Chromium	Dietary risks to fish unknown; no dietary TRV available.
Manganese	Dietary risks to fish unknown; no dietary TRV available.
Nickel	Dietary risks to fish unknown; no dietary TRV available.
Thallium	Dietary risks to fish unknown; no dietary TRV available.
PAHs	
1-Methylnaphthalene	Dietary risks to fish unknown; no dietary TRV available.
2-Methylnaphthalene	Dietary risks to fish unknown; no dietary TRV available.
Benzo(e)pyrene	Dietary risks to fish unknown; no dietary TRV available.
Dibenzothiophene	Dietary risks to fish unknown; no dietary TRV available.
Perylene	Dietary risks to fish unknown; no dietary TRV available.
Alkylated PAHs	Dietary risks to fish unknown; no dietary TRV available.

COI – contaminant of interest

PAH – polycyclic aromatic hydrocarbon

TRV – toxicity reference value

7.2.5.3 Summary of Fish Diet LOE

Four dietary fish COPCs have HQs ≥ 1 in the final step of the risk analysis for at least one fish receptor: cadmium, copper, TBT, and mercury. For large-home-range fish, HQs are ≥ 1 for cadmium (juvenile chinook salmon and peamouth) and copper (largescale sucker, juvenile white sturgeon, juvenile Chinook salmon, and peamouth). For small-home-range fish, HQs are ≥ 1 for cadmium (sculpin, and northern pikeminnow), copper (sculpin and northern pikeminnow), TBT (sculpin), and mercury (sculpin). The results of the dietary LOE are integrated with those from other LOEs in Section 7.6 to determine the fish risk conclusions.

7.2.5.4 Evaluation of Additional Uncertainties

Uncertainties associated with exposure assumptions, effect thresholds (TRVs), and risk characterization methods are identified in previous subsections. This subsection examines uncertainty for additional factors of the fish dietary assessment, specifically the relative dietary contribution of multiple prey species, and the effect of water temperature on fish ingestion rates.

7.2.5.4.1 Prey Portions

Selected prey portions (Table 7-19) were based on dietary information presented in the literature. In EPA's Problem Formulation (Attachment 2), EPA requested that prey portions in multi-species diets be varied from 0 to 100%. An evaluation was conducted to determine how varying the prey portions in the diet from the defaults shown in Table 7-19 would affect risk conclusions (i.e., whether or not an HQ would change from < 1 to ≥ 1, or vice versa) for all receptor-COPC pairs in which multiple prey species were evaluated in the diet.[101]

In Step 2 of the risk characterization, HQs were calculated assuming single prey species constituted 100% of the diet (Table 7-23). The prey portion uncertainty evaluation identified the range of possible HQs when the contribution of individual prey species to the diet varied from 0 to 100%. The range of HQs was used to determine whether HQs for any COPC would change from ≥ 1 to < 1 or vice versa under different but plausible assumptions about prey portions (default portions are presented in Table 7-19). The range of plausible prey fractions were determined based on the dietary information in Attachment 13. Results of this evaluation are shown in Table 7-33.

[101] The sculpin diet was not evaluated because prey species (i.e., worms and clams) were evaluated individually; therefore, no multi-species diet could be evaluated for this receptor. The largescale sucker and juvenile white sturgeon diets were also not evaluated because prey species (i.e., worms and clams) were evaluated individually, and assigning prey portions to the selected prey species was too uncertain.

Table 7-33. Summary of Dietary Prey Portion Uncertainty Evaluation

COPC by Receptor	HQ	Exposure Area	Prey Portions Used in Risk Characterization	Uncertainty Evaluation	Does Uncertainty Evaluation Change HQ Status?
Juvenile Chinook Salmon					
TBT	0.79	All exposure areas	45% clams and 55% worms	HQs could range from 0.38 to 1.1 based on 100% ingestion of clams and lab worms, respectively.	No; HQ would be ≥1 if > 89% of diet was composed of worms; however this is not an assumption supported by the literature.
Cadmium	3.5	All exposure areas	30% clams, 40% worms, and 30% epibenthic tissue	HQs could range from 1.7 to 4.5 based on 100% ingestion of epibenthic invertebrates and clams, respectively.	No; HQ would be ≥ 1 regardless of prey portions.
Copper	2.5	All exposure areas	30% clams, 40% worms, and 30% epibenthic tissue	HQs could range from 1.6 to 3.8 based on 100% ingestion of lab worms and clams, respectively.	No; HQ would be ≥ 1 regardless of prey portions.
Peamouth					
Copper	1.0	All exposure areas	25% clams, 25% worms, 40% epibenthic tissue, and 10% sculpin	HQs could range from 0.41 to 1.5 based on 100% ingestion of sculpin and clam, respectively.	Yes; HQ would be < 1 if clam tissue constituted < 25% of diet and epibenthic invertebrate tissue did not constitute a larger fraction, which is a reasonable assumption.
Smallmouth Bass					
Copper	0.22 – 0.87	All exposure areas	90% sculpin, 5% crayfish, and 5% lab worms	More than 71% of diet would have to be represented by crayfish for HQs to be ≥ 1 in all exposure areas, or more than 42% of diet would have to be represented by lab worms for HQs to be ≥ 1 in one exposure area (Swan Island Lagoon).	Yes; HQ would be ≥ 1 if > 71% of diet was composed of crayfish, which is a reasonable assumption.[a]

Table 7-33. Summary of Dietary Prey Portion Uncertainty Evaluation

COPC by Receptor	HQ	Exposure Area	Prey Portions Used in Risk Characterization	Uncertainty Evaluation	Does Uncertainty Evaluation Change HQ Status?
Cadmium	0.042 – 0.19	All exposure areas	90% sculpin, 5% crayfish, and 5% lab worms	More than 91% of diet would have to be represented by lab worms for HQs to be ≥ 1 in two exposure areas (RM 1.5 to RM 2.5 and RM 3.5 to RM 4.5); worm prey portion of > 77% is not supported by the literature.	No; HQ would be ≥ 1 if > 77% of diet is composed of worms; however, this is not an assumption supported by the literature.
Northern Pikeminnow					
Copper	1.1 and 1.3	Swan Island Lagoon; RM 10.5 to RM 11.8	30% crayfish, 25% lab worms, 25% sculpin, 5% largescale sucker, 5% carp, 5% peamouth, and 5% northern pikeminnow	HQs could range from 0.13 to 2.4 based on 100% ingestion of northern pikeminnow and crayfish, respectively.	Yes; HQ would be < 1 if crayfish and lab worms collectively made up approximately < 20% of the diet, which is a reasonable assumption.
Cadmium	0.096 – 0.43	All exposure areas	30% crayfish, 25% lab worms, 25% sculpin, 5% largescale sucker, 5% carp, 5% peamouth, and 5% northern pikeminnow	More than 83% of diet would have to be represented by worms for HQs to be ≥ 1 in two exposure areas (RM 1.5 to RM 2.5 and RM 3.5 to RM 4.5); worm prey portion of > 83% is not supported by the literature.	No; HQ would be ≥ 1 if > 83% of diet is composed of worms; however, this is not an assumption supported by the literature.
Mercury	0.13 – 0.34	All exposure areas	30% crayfish, 25% lab worms, 25% sculpin, 5% largescale sucker, 5% carp, 5% peamouth, and 5% northern pikeminnow	More than 77% of diet would have to be represented by pikeminnow for HQs to be ≥ 1 in five exposure areas (RM 4.5 to RM 9.5); pikeminnow prey portion of > 77% is not supported by the literature.	No; HQ would be ≥ 1 if > 77% of diet is composed of pikeminnow; however, this is not an assumption supported by the literature.

[a] Crayfish made up 62% of the stomach contents of juvenile smallmouth bass (Vile and Friesen 2005) and were the dominant prey item in Study Area smallmouth bass collected during Round 1 sampling)

COPC – contaminant of potential concern
HQ – hazard quotient
RM – river mile
TBT – tributyltin

For the juvenile Chinook salmon-cadmium and -copper receptor-COPC pairs, varying the prey portions would not lower HQs to < 1 (Table 7-33).

For a few receptor-COPC pairs, the assumption that any given prey species could make up 0 to 100% of the diet results in additional receptor-COPC pairs with HQs ≥ 1.

- Juvenile Chinook salmon-TBT
- Smallmouth bass-cadmium
- Smallmouth bass-copper
- Northern pikeminnow-cadmium
- Northern pikeminnow-mercury

However, the proportion of the total diet that a given prey species would have to represent to yield an HQ ≥ 1 is unrealistic for five of these receptor-COPC pairs (all but smallmouth bass-copper) based on the diet prey portions presented in the general literature and region-specific studies (summarized in Attachment 13). Therefore, the potential for risk to these fish receptor-COPC pairs is very low.

For copper-smallmouth bass, however, the dietary proportion of crayfish (> 71%) that would yield an HQ ≥ 1 is possible. The evaluation of copper exposure to a piscivorous fish is covered by northern pikeminnow because copper-northern pikeminnow has already been identified as resulting in an HQ ≥ 1. The assumption that any given prey species could make up 0 to 100% of the diet results in two additional receptor-COPC pairs with HQs < 1. The peamouth-copper and northern pikeminnow-copper receptor-COPC pairs would not have HQ ≥ 1 under different but plausible dietary assumptions, suggesting that risks could be overestimated.

7.2.5.4.2 Water Temperature and Calculated Fish Ingestion Rates

Dietary fish ingestion rates were estimated using Equation 7-7 and assuming a temperature of 13.4 ºC. The water temperature is based on the average recorded within the Study Area (at the SP&S Railroad Bridge) from 1995 to 2005 (ODEQ 2005). Per EPA's Problem Formulation (Attachment 2), fish FIRs were also calculated using a high-end water temperature provided by EPA (16.2 ºC). Ingestion rates increased approximately 17% for food and 16% for sediment at the higher water temperature. HQs based on the latter ingestion rates would increase by approximately 16 to 17% compared with HQs based on ingestion rates calculated at the lower water temperature of 13.4 ºC. A 16 to 17% increase in HQ would not change the risk results for the fish dietary assessment (i.e., the same COPCs would exceed TRVs).

7.2.5.4.3 Fish Body Weight and Calculated Fish Ingestion Rates

Per EPA's Problem Formulation (Attachment 2), the effect of fish body weight on risk calculations was also examined. Risks were calculated using the full range of body weights of fish collected from the Study Area, instead of the average body weight. When minimum and maximum body weights for each fish receptor were assumed, calculated HQs varied by less than 25% from HQs based on mean body weight. Risk conclusions would not be affected by using the range of possible body weight assumptions.

7.3 SURFACE WATER ASSESSMENT

The surface water assessment was another LOE for evaluating risks to all fish receptors. Surface water COPCs were identified in the SLERA and refined screen using water TRVs based on AWQCs or other TRVs available in the literature (Attachment 5). In this assessment, the same water TRVs were used to evaluate baseline risks to fish.

Adult Chinook salmon are not a selected ecological receptor of concern; but per EPA's Problem Formulation (Attachment 2) the exposure of adult Chinook to metals in water was evaluated to determine if metals exposure might disrupt olfactory function in migrating populations.

Figure 7-13 describes the layout of the surface water assessment section.

- Section 7.3.1 presents the general approach.
- Section 7.3.2 lists the COPCs evaluated in the surface water LOE.
- Section 7.3.3 describes how exposure concentrations were derived. Exposure data in this assessment are represented by COPC concentrations in surface water samples. All surface water contaminant concentrations and calculated UCLs are presented in Attachment 4.
- Section 7.3.4 summarizes the effects data. The water TRVs used in this assessment are the same as those used for the SLERA and refined screen. An evaluation of whether the water TRVs are protective of lamprey is also presented based on results of a sensitivity study of the lamprey toxicity tests that were conducted for the BERA. Water effect thresholds related to avoidance behavior in migrating salmonids are also presented. Details on the development of the water TRVs are presented in Attachment 10. Details on results of the lamprey toxicity tests are presented in Attachment 15.
- Section 7.3.5 presents the risk characterization results and associated uncertainties. These COPCs are further assessed in the fish risk conclusions (Section 7.6).

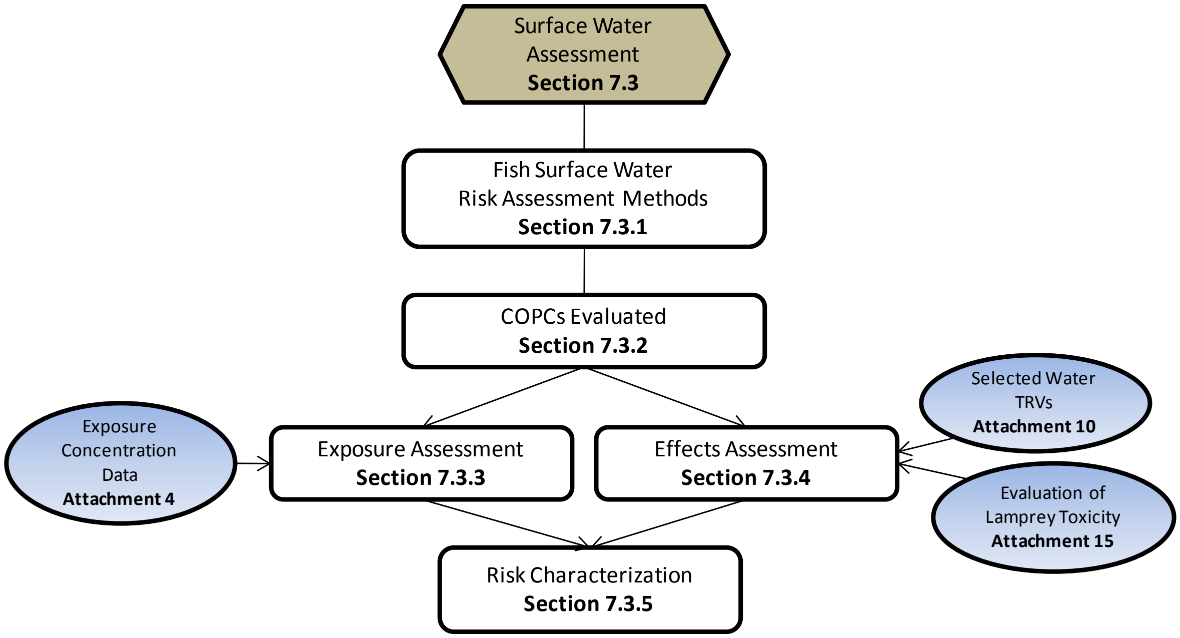

Figure 7-13. Overview of Fish Surface Water Assessment Section Organization (revised)

7.3.1 Fish Surface Water Risk Assessment Methods

Surface water HQs were calculated by comparing COPC concentrations in water samples to chronic water TRVs. These TRVs were developed from water quality criteria and literature-based TRVs, according to a hierarchy articulated in Attachment 10. Because of concern about whether the selected TRVs would be protective of lamprey, lamprey water toxicity tests were conducted (Windward and Integral 2008) for six contaminants representing six different modes of toxicity, and the resultant data were compared with data for other species. This analysis provided some assurance that the selected TRVs were protective of lamprey. Additionally, metals water toxicity data were reviewed to ensure that the selected TRVs would protect the reproductive requirement of olfactory function in salmon.

The analysis of surface water risks to fish progressed from more conservative to more realistic estimates of exposure and risk, as follows:

- **Step 1 –** HQs were calculated on a sample-by-sample basis. This step was the same as the assessment conducted for benthic invertebrates (Section 6.5).
- **Step 2 –** For those ROC-COPC pairs resulting in a maximum HQ ≥ 1 in Step 1, HQs were then calculated within the receptor-specific exposure areas (Table 7-1).

Potentially unacceptable risks were identified based on those COPCs that resulted in HQs $\geq$ 1 in Step 2.[102] The quantitative risk results (i.e., magnitude, spatial distribution, and frequency of HQs), the seasonal and sampling method patterns of HQs, and underlying uncertainties of exposure and effects data are presented in the risk characterization (Section 7.3.5) for each receptor. The relative strengths and uncertainties for all fish LOEs are evaluated together in the risk conclusions for fish (Section 7.6).

7.3.2 COPCs Evaluated

Eleven of the 14 surface water COPCs identified in the SLERA and refined screen (Attachment 5; Table 5-4) are evaluated in the BERA. Three individual DDT metabolites identified in the SLERA (2,4′-DDD, 2,4′-DDT, and 4,4′-DDD) were evaluated as part of total DDx and were not evaluated individually; 4,4′-DDT was evaluated both individually and as total DDx because the TRV for DDx is based on 4,4′-DDT. All other COPCs were evaluated in this assessment.

Five surface water COIs were not evaluated in the SLERA and refined screen because no toxicological data were available (Table 5-6). Four of these COIs (4-chloroaniline, aniline, 2,4-DB, and MCPP) were infrequently detected and were detected in isolated areas at different times. The risks to fish from these contaminants in surface water are unknown because of the absence of toxicological data. Surface water thresholds are unavailable for individual dioxins and furans other than 2,3,7,8-TCDD. Dioxins and furans are evaluated as total dioxin/furan TEQ and total TEQ, toxicity-weighted sums based on the relative toxicity of each congener to 2,3,7,8-TCDD using TEFs based on their common mechanism for toxicity. Based on this evaluation, total dioxin/furan TEQ and total TEQ were not identified as COPCs in the SLERA.

Aluminum was not identified as a COPC as per agreement with EPA because the AWQC were developed based on toxicity data from acidic waters and are not applicable to the Study Area. Aluminum concentrations in background surface water and sediment were evaluated to identify local sources of aluminum contamination within the Study Area, if any (Section 6.5.5.3). Like aluminum, zinc is naturally occurring in the environment, and background zinc concentrations were also evaluated.

In addition, one COI (2,4′-DDE) was not retained as a COPC in the refined screen because no measured concentration exceeded the TRV (although at least one DL exceeded a TRV). However, 2,4′-DDE was evaluated as a component of total DDx.

7.3.3 Exposure Assessment

This section presents the methods that were used to estimate fish exposures to COPCs in surface water. Surface water EPCs developed for comparison to AWQC and literature-derived TRVs are discussed in Section 7.3.3.1. Surface water exposure concentrations

[102] As agreed to between EPA and LWG on October 15, 2010 (LWG 2010) .

used to evaluate olfactory function in migrating populations of adult Chinook salmon are presented in Section 7.3.3.2.

7.3.3.1 Surface Water EPCs

Surface water sampling was conducted following EPA QAPPs for the purpose of characterizing ecological risks. Basic information about surface water sampling events and general trends in COPC concentrations are presented in the benthic risk assessment (Section 6.5.3). All surface water data collected using different sampling methods were used to develop EPCs for fish. The surface water data were collected with reasonable coverage to characterize the spatial and temporal variability in the magnitude and extent of COPC concentrations in water throughout the Study Area. The one exception is that non-LWG near-bottom peristaltic samples were collected during only one sampling event in May 2005 at one location (at RM 6.5 on the west bank of the Study Area). Analyses for two COPCs (i.e., ethylbenzene and trichloroethene) were conducted only in those samples; therefore, risk to fish from exposure to these COPCs are highly uncertain.

Uncertainty Associated with Surface Water Sampling Methods

Surface water sampling methods varied across sampling events, creating some uncertainty about data comparability. Surface water samples were collected both as single-point samples and as transect samples (vertical and/or horizontal) using two types of sampling methods (XAD and peristaltic pump). Samples were collected over seven sampling events, and not all surface water locations were sampled during each event. Surface water transect samples are representative of a wider range of conditions surface water concentrations because the technique collects more water over a greater area and longer period. Horizontal transects were only sampled at five locations within the Study Area (at RM 2.0, RM 4.0, RM 6.3, RM 11, and at the mouth of Multnomah Channel) and thus are limited spatially. The evaluation of transect, single-point, XAD, and peristaltic samples provides a larger dataset for estimating fish surface water EPCs. The advantage of having more data at least partially offsets the disadvantage of adding unquantified uncertainty about data comparability across sampling methods.

Surface water EPCs were first calculated on a sample-by-sample basis. This step was the same as the assessment conducted for benthic invertebrates (Section 6.5.3). EPCs were then calculated as the UCL within the receptor-specific exposure areas (Table 7-1) for those receptor-COPC pairs with a maximum HQ $\geq$ 1. If insufficient data were available for derivation of a UCL concentration (fewer than six detected concentrations were available), the HQ was based on the maximum water concentration. Such use of maximum concentrations may result in an over- or underestimate of risks because the available samples may fall above or below the true population mean.

For sculpin, EPCs were represented by individual samples or, for VOCs and naphthalene, by samples collected within a very limited spatial scale.[103] For smallmouth bass and northern pikeminnow, EPCs were represented by UCLs (or maximum concentrations

[103] Because of the small spatial scale over which ethylbenzene and trichloroethene samples were collected (over approximately a 0.15-mile stretch), VOC EPCs for sculpin were represented by UCLs (or maximum concentrations) based on these spatially limited data, which were considered roughly equivalent to a relevant exposure scale for sculpin. In addition, naphthlene results from samples collected over this 0.15-mile stretch were also treated as a single EPC.

where UCLs could not be derived) calculated within 1-mile exposure areas. For large-home-range and migratory fish receptors (i.e., largescale sucker, juvenile white sturgeon, juvenile Chinook salmon, peamouth, and Pacific lamprey ammocoetes), EPCs were represented by site-wide UCLs.

Site-wide UCLs were calculated for all surface water COPCs that were sampled with reasonable temporal and spatial coverage throughout the Study Area (see Map 4-15).[104] COPC surface water concentration data for all individual samples, and for calculated site-wide and 1-mile exposure area UCL concentrations, are presented in Attachment 4.

At locations where both XAD and peristaltic samples were collected and analyzed for organic COPCs, the results of the peristaltic samples (i.e., the low-resolution results) were removed from the dataset used to derive UCLs. These data were removed because COPC concentrations from XAD samples are based on high-resolution analyses with lower DLs and higher accuracy.[105]

Surface water EPCs based on a UCL were calculated using ProUCL Version 4.0 software (EPA 2007f). EPA's ProUCL software tests the goodness of fit for a given dataset and then computes the appropriate 95th UCL (as described in Section 7.1.3.1). Attachment 4 presents the summary statistics (i.e., minimum, maximum, and mean COPC concentrations), distribution types, and ProUCL-recommended UCLs for all surface water EPCs.

[104] Surface water samples that were analyzed for ethylbenzene and trichloroethene were collected as part of a non-LWG sampling event only at locations primarily offshore of the Siltronic property along an approximately 0.15-mile length of the river and were not included in the Study Area-wide or 1-mile exposure scale evaluation. The primary purpose for collecting these surface water data was to characterize attenuation rates as VOCs migrated from groundwater, through TZW, and into surface water. In addition, these data are unlikely to accurately characterize concentrations in the 1-mile exposure area from RM 5.5 to 6.5. For example, data collection was biased in that samples were only collected from areas where groundwater discharge to surface water was suspected. Also, data were collected on one side of the river over a small portion of the 1-mile exposure area and are therefore unlikely to represent concentrations throughout the entire 1-mile exposure area.

[105] An analysis of uncertainty associated with exclusion of peristaltic samples collocated with XAD samples is presented in Section 7.3.5.

As discussed in Section 7.1.3.1, uncertainty is associated with the use of maximum concentrations to represent surface water EPCs. For evaluating risks to smallmouth bass and northern pikeminnow, only limited data for some surface water COPCs were available to derive the surface water UCL concentrations within 1-mile exposure areas. When a limited number of samples is available, ProUCL will not estimate a confidence interval on the mean; instead the maximum sampled concentration is used to estimate the UCL. The use of maximum concentrations to represent surface water EPCs may result in an over- or underestimate of risk. EPCs based on maximum concentrations are further discussed in the risk characterization section.

There is uncertainty in the exposure data in that surface water samples were collected over the course of several sampling events, which differed in duration and methods. UCLs calculated from these data do not account for the longer exposure durations represented by some samples. The UCLs are meant to represent spatially and temporally averaged concentrations. Samples collected by methods that are less representative of spatially and temporally averaged concentrations are likely to introduce error into the EPCs because they are overrepresented in the UCL calculation (i.e., they are given the same weight as integrated samples that are better estimators of the mean). Relatively high concentrations in the less representative samples would bias the UCL calculation to overestimate the EPC, and relatively low concentrations in the less representative samples would bias the UCL calculation to underestimate the EPC.

Samples were collected at a spatial scale that approximates the 1-mile exposure areas for northern pikeminnow and smallmouth bass (see Figure 4-15). However, due to limited replication at this scale, maximum concentrations were often used as the EPC. Unquantified uncertainty is associated with use of maximum concentrations to represent some EPCs for these receptors (because the sample size is too small to quantitatively estimate a confidence interval) (EPCs are presented in Attachment 4). This uncertainty is partly mitigated by the presence in the dataset of many integrated surface water samples.

Because water samples were collected at a spatial scale that is larger than sculpin exposure areas, COPC concentrations in some places where sculpin may be exposed are not uniquely characterized. A large number of samples was collected from throughout the Study Area, and the available data suitably characterize the extent and magnitude of surface water contamination. Because surface water flows through the river, the sampled locations reflect the spatial and temporal variability in contaminant concentrations throughout the Study Area; thus, the surface water data are not likely to either over- or underestimate exposure for small-home-range fish.

Sampling did not consider differences in habitat quality for large-home-range fish througout the Study Area. UCLs might over- or underpredict EPCs in the preferred habitats of the fish receptors if for some reason water quality in the preferred habitat differs from that represented by the surface water data.

7.3.3.2 Exposure of Migrating Adult Chinook Salmon

Per EPA's Problem Formulation (Attachment 2), the exposure of adult Chinook salmon to metals concentrations in water was evaluated to determine whether olfactory function might be disrupted in migrating populations. Impaired olfactory function and avoidance response behaviors have been associated with water-borne metals. Olfactory function and avoidance toxicity studies of salmonids exposed to individual metals in water have focused on copper and cobalt, with the bulk of the work conducted on copper. Since analyses for cobalt were not conducted in Study Area surface water samples, only copper was evaluated in this assessment.

Surface water concentrations of dissolved copper in the Study Area ranged from 0.37 to 2.39 µg/L. Adult Chinook salmon (from both spring and fall runs) are exposed to these copper concentrations as they migrate from the ocean through the Study Area, to their upstream spawning grounds. The migration of adult spring Chinook salmon through the Multnomah Channel and the Willamette main stem downstream from Willamette Falls peaks in mid- to late April and is mostly complete by July (ODFW 2001; Schreck et al. 1994a). In 2001, 85 to 95% of spring Chinook salmon returning to the Willamette Basin were raised in hatcheries (ODFW 2001), although wild populations of spring Chinook salmon also migrate through the Study Area to spawn upstream, above Willamette Falls and in the Clackamas River. The migration speed and distance of adult spring Chinook is variable and may be related to the time of migration (Schreck et al. 1994a).

7.3.4 Effects Assessment

This section presents the effects thresholds used to evaluate surface water risks to fish receptors. Section 7.3.4.1 presents the water TRVs, which were used to evaluate all fish receptors. Section 7.3.4.2 summarizes results of the lamprey ammocoete toxicity testing conducted to determine whether the TRVs were protective of lamprey. Section 7.3.4.3 presents effects thresholds associated with impaired olfactory function in salmonids.

7.3.4.1 Water TRVs

Surface water concentrations were compared with effects thresholds in the risk characterization. Per agreement with EPA (2008f), chronic water TRVs were developed for all surface water COPCs based on the hierarchy detailed in Attachment 10. Section 6.5.4 and Table 6-32 present the water TRVs developed for all surface water COPCs.

Because the selected AWQC-derived values for total PCBs (0.014 µg/L) and 4,4′-DDT (0.001 µg/L) were based on protection of mammals and birds, respectively, risk estimates for aquatic receptors based on these TRVs are uncertain. Therefore, alternative criteria protective of fish and invertebrates were developed using methods consistent with those for AWQC derivation. The alternative water TRV for DDx compounds was calculated as 0.011 µg/L. The alternative water TRV for total PCBs was calculated as 0.19 µg/L. Derivation of these alternative TRVs is described in Section 6.5.4 and Attachment 10. For evaluating direct exposure of aquatic organisms to water, the alternative TRVs are considered more appropriate than the AWQC-based TRVs because the total PCBs AWQC is based on the protection of mink via ingestion of contaminated prey, and the

4,4′-DDT AWQC is based on the protection of brown pelican via ingestion of contaminated prey.

Two sets of water HQs were derived for total PCBs, 4,4′-DDT, and total DDx: one using the selected AWQC-based TRVs and one using the alternative water TRVs. Because the the alternative TRVs are more appropriate for assessing risk to fish, they were used to determine risk conclusions.

7.3.4.2 Evaluation of Water Toxicity Thresholds for Lamprey Ammocoetes

A sensitivity study of lamprey ammocoetes (*Lampetra* sp.) was conducted in response to a request from EPA (2006b) tohelp assess whether the water TRVs used in this risk evaluation are protective of lamprey survival and growth at the organism level.

With the exception of sea lamprey studies conducted during the development of a lampricide for use in the Great Lakes, lamprey species have not been widely studied by aquatic toxicologists. To narrow this data gap, acute toxicity tests of six contaminants (pentachlorophenol, copper, aniline, diazinon, naphthalene, and lindane) were conducted on field-collected lamprey ammocoetes from four uncontaminated Oregon coastal streams. These six contaminants were selected to represent the range of toxic modes of action. Comparing the sensitivity of lamprey ammocoetes to the sensitivity of standard test species, for representative chemicals from these six classes of toxiciants, provided insight into whether the TRVs, which are derived from data for standard test species, would be protective of lamprey ammocoetes.

The relative sensitivity of lamprey ammocoetes was evaluated using an SSD, which displays available toxicity data as a plot of the LC50 for each species on the x-axis and the cumulative probability (estimated fraction of species with lower LC50s) on the y-axis. The measured lamprey LC50s were compared with LC50s for all other aquatic species for which toxicological data were available. The study confirmed that across the tested modes of action, rainbow trout or other salmonids were at least as sensitive as lamprey ammocoetes. Table 7-34 compares the lamprey LC50s derived from the toxicity tests to the range of LC50s for other species and presents the lamprey LC50 percentile of the SSD. Of the six test contaminants, lamprey ammocoetes were most sensitive to pentachlorophenol, which represented the oxidative phosphorylation uncoupler mode of action (the same mode of action as commercial lampricides). However, even in the case of pentachlorophenol, the ammocoetes were not more sensitive than rainbow trout. The sensitivity of lamprey ammocoetes to copper was around the average for other aquatic species. For the other four contaminants (i.e., aniline, diazinon, naphthalene, and lindane), the sensitivity of lamprey ammocoetes fell at the upper end of the SSDs, indicating that lamprey ammocoetes were less sensitive than most tested aquatic species.

Table 7-34. Summary of Lamprey LC50s Compared with LC50s of Other Aquatic Species

Analyte	LC50 (µg/L) Lamprey	LC50 (µg/L) Other Aquatic Species	Percentile of SSD
Pentachlorophenol	31	4.4 – 11,260	15
Copper	46	2.7 – 107,860	46
Aniline	430,000	126 – 477,900	90
Diazinon	8,900	0.38 – 11,640	72
Naphthalene	10,000[a]	2,000 – 6,600	NA
Lindane	>2,680[b]	1 – 22,500	NA

[a] The LC50 for naphthalene was estimated at 10,000 µg/L, based on 50% mortality in the test highest concentration.
[b] An LC50 could not be derived for lindane, with 12.5% mortality in the highest test concentration of 2,680 µg/L.
LC50 – concentration that is lethal of 50% of an exposed population
NA – not available
SSD – species sensitivity distribution

Because lamprey were found to be as sensitive as or less sensitive than rainbow trout and other salmonids for all of the six contaminants tested, the aquatic toxicological thresholds (i.e., water TRVs) are likely to be sufficiently conservative for this measurement endpoint. Further details of the sensitivity study are presented in Attachment 15.

7.3.4.3 Olfactory Function and Avoidance Behavior Effects in Salmonids

The water TRVs for metals were based on EPA AWQC or Tier II values. As discussed previously, the typical endpoints used to derive chronic AWQC include survival, growth, and reproduction. Although the ability of metals to induce avoidance behavior in fish has been known for decades (Sprague 1964), data on the effects of metals on avoidance behavior have not been used by EPA to lower AWQC because EPA has concluded that current AWQC are protective of this behavioral endpoint. In part due to the listing of several populations of Pacific salmon as threatened or endangered in the Pacific Northwest, an increasing number of studies have evaluated the effects of copper on olfactory function in salmon and other fish. Olfactory impairment is a physiological response that may be considered an indicator of a potential organism-level behavioral response. Olfactory function in fish plays a major role in mediating behaviors important for both survival and reproduction, such as juvenile imprinting on home waters, predator avoidance, and adult migration and homing (Baldwin et al. 2003). Concern has been expressed that AWQC, for copper at least, might not be protective against olfactory impairment in juvenile salmon (McIntyre et al. 2008). In juvenile salmon, copper at sufficiently high concentrations might impair the ability of the fish to detect an odorant that serves as an alarm queue and helps the fish avoid predation (Baldwin et al. 2003). The physiological endpoint of olfactory impairment is expected to be a more sensitive endpoint than the organism-level behavioral endpoint because a fish has to smell an odorant before the odorant can elicit a behavioral response (DeForest et al. 2011b). Since olfactory impairment is a physiological endpoint, it is not a measurement endpoint that is directly linked to the assessment endpoints for fish that directly relate to survival, reproduction, and growth.

The following first summarizes the toxicological studies on the effects of metals on olfactory function and avoidance behavior in salmonids. It is followed by a summary of whether existing metals criteria are protective against olfactory impairment and behavioral effects.

7.3.4.3.1 Effects of Metals on Olfactory Function and Behavioral Avoidance Response

As noted above, the olfactory system in fish plays a variety of important roles. Many of the earlier studies on metal toxicity, and recent studies as well, evaluated the effects of a metal, or mixture of metals, on the behavioral response of fish when exposed. In a typical study design, fish are placed in an exposure system, such as a Y-maze, where they have the option of choosing between control water and metal-containing water. These studies can be used to define metals concentrations that fish might avoid, all else equal. Typically, depending on the concentration, the presence of metals in water produces avoidance behavior (Hansen et al. 1999c; Atchison et al. 1987). This has been confirmed in field studies that have shown cases where distinct spatial gradients of metals occur (e.g., near point source discharges) and fish may use their sense of smell to avoid contamination (Saunders and Sprague 1967). Similarly, Baldwin et al. (2003) reported that the avoidance response due to the presence of metals in water might disrupt migration patterns or prevent fish from inhabiting areas that would otherwise offer productive habitat.

The exposure of salmonids to individual metals in laboratory studies at sublethal concentrations has been shown to produce avoidance behavior. A literature review of laboratory studies that evaluated behavioral avoidance of metals by salmonids was conducted. In the reviewed literature, salmonids displayed avoidance behavior at copper concentrations ranging over several orders of magnitude, from 0.10 to 88 µg/L (Table 7-35).

Table 7-35. Thresholds for Effects of Copper on Olfactory Function and Avoidance Behavior in Fish

Species	Copper Concentration Effect Level (µg/L)[a]	Water Hardness (mg $CaCO_3$/L)	Source
Rainbow trout	0.1	89.5	Folmar (1976)[b]
Chinook salmon	0.8 – 22.5[c]	25.3	Hansen et al. (1999c)
Rainbow trout	1.6 – 88[d]	25.3	Hansen et al. (1999c)
Juvenile coho	2[e]	120	Sandahl et al. (2007)
Atlantic salmon	2.3	20	Sprague et al. (1964)
Rainbow trout	4.4	23.0 – 27.0	Giattina et al.(1982)
Juvenile coho	4.4[f]	120	Sandahl et al. (2004)
Coho salmon	< 6.4	30.5	Rehnberg and Schreck (1986)[b]
Rainbow trout	8	90	Hara et al (1976)[b]

Table 7-35. Thresholds for Effects of Copper on Olfactory Function and Avoidance Behavior in Fish

Species	Copper Concentration Effect Level (µg/L)[a]	Water Hardness (mg $CaCO_3$/L)	Source
Juvenile coho	13[g]	100	Baldwin et al. (2003)
Rainbow trout	< 22[h]	61.8 – 64.0	Saucier et al (1991)[b]
Brown trout	55	157.8	Baldigo and Baudanza (2001)[b]
Rainbow trout	70	112.4	Black and Birge (1980)[b]

a Effect level is the lowest reported concentration at which avoidance behavior was observed for a given species in each toxicological study reviewed.

b This study has high uncertainty as minimal methodological details are provided and the copper concentration of 0.1 µg/L is the nominal copper concentration added to the test dilution water (the copper concentration in the test dilution water was not reported).

c Benchmark concentrations estimating percent loss of olfactory function were calculated for juvenile coho following a 7-day exposure to 0, 5, 10, or 20 µg/L copper. It was estimated that fish would exhibit 20% loss of olfactory function at 4.4 µg/L copper, 50% loss at 11.1 µg/L copper, and 90% loss at 20 µg/L copper.

d Rainbow trout failed to avoid water with concentrations above the acutely lethal concentration of 180 µg/L (Hansen et al. 1999c).

e Laboratory study evaluated olfactory function by measuring predator avoidance behavior triggered by nonspecific chemical alarm pheromones. The study estimated that 2 µg/L copper produced a 40% reduction in olfactory function based on measurements of electro-olfactograms after fish were exposed to copper concentrations (0, 5, 10, or 20 µg/L) for 30 minutes.

f Fish demonstrated a 20% loss of olfactory function following a 7-day exposure at 4.4 µg /L. Fish exposed to 11.1 µg/L experienced a 50% loss of olfactory function, and fish exposed to 20 µg/L experienced a 90% loss of olfactory function.

g Fish experienced a 50% loss of olfactory function when exposed to 13 µg/L copper for 30 minutes.

h Copper concentrations were measured as Cu^{2+}.

Hardness affects copper bioavailability; therefore, avoidance behavior studies conducted with a water hardness range similar to that of the LWR (i.e., 22.5 to 54.5 mg $CaCO_3$/L) are more pertinent to this assessment. Baldwin et al. (2003) and McIntyre et al. (2008) demonstrated that hardness has a marginal influence on the effects of copper on olfactory function and might not be an important factor in avoidance behavior, but dissolved OC in concentrations ≥ 6 mg/L reduced the effects of copper on salmon olfactory function (McIntyre et al. 2008).

Impairment of the olfactory system after exposure to metals can be temporary, depending on the concentration and duration of the metals exposure. Partial recovery of Chinook salmon olfactory cells after short-term exposure (30 minutes) to a copper concentration of 25 µg/L occurs within 60 minutes (Hansen et al. 1999a). Sandahl et al. (2006) found that the chum salmon (*Oncorhynchus keta*) olfactory system can recover from short-term (4-hour) exposures to copper concentrations of 3 to 58 µg/L within 1 day. However, the death of olfactory cells has been shown to occur following exposures lasting more than 4 hours (Hansen et al. 1999b; Julliard et al. 1996). Although it is possible for fish to regenerate olfactory cells, this regrowth takes from 8 to 42 days (Hansen et al. 1999a), during which time the fish is not receiving important information usually conferred by those cells. Accordingly, olfactory cells in salmon appear to recover, at least partially, following short-term exposures (≤ 4 hours) to low and moderate copper concentrations, while longer exposures, or exposures to higher copper concentrations, may result in olfactory cell death or a much longer olfactory cell recovery time.

7.3.4.3.2 Relevance of Avoidance Response to Migration

Salmonid populations with healthy salmon runs survive in environments with dissolved copper concentrations that range from 2 to 23 µg/L (e.g., Copper River, Alaska) (Brooks 2004). Adult and juvenile salmonids survive and successfully navigate in both the Copper River and Sacramento River (California), which frequently have dissolved copper concentrations greater than 2 µg/L, suggesting that olfactory inhibition may be minimal for salmonid populations that are genetically adapted or physiologically acclimated to elevated copper levels. Conversely, other studies have suggested that physiological acclimation to dissolved copper concentrations above 2 µg/L causes the loss of avoidance behavior at higher copper concentrations (Brooks 1998; Hansen et al. 1999c) with the result that salmonids might not avoid lethal concentrations (however, copper concentrations would have to be many-fold greater than AWQC to be lethal to salmon). A field study conducted by Saunders and Sprague (1967) reported that Atlantic salmon (*Salmo salar*) migrating upstream avoided areas containing copper and zinc, and the estimated threshold for avoidance of copper ranged from approximately 17 to 21 µg/L.

7.3.4.3.3 Protectiveness of AWQC Relative to Olfactory Impairment and Behavioral Avoidance

The water TRV for copper is hardness-based, meaning that copper toxicity is modified depending on the hardness of the water (increasing hardness tends to reduce copper toxicity). Other water chemistry parameters, such as dissolved OC and pH, also influence the bioavailability and, hence, toxicity of copper (Santore et al. 2001). EPA's currently recommended AWQC for copper are based on the biotic ligand model (BLM), which accounts for the pH and concentrations of dissolved OC, hardness, alkalinity, and several ions in calculating criteria concentrations. Briefly, largely using the data from McIntyre et al. (2008) on the effects of copper on olfactory inhibition in coho salmon, Meyer and Adams (2010) parameterized an olfactory-based BLM that allows for the calculation of IC20 values for olfactory impairment. DeForest et al. (2011a) subsequently applied the olfactory-based BLM from Meyer and Adams (2010) to estimate IC20 values for olfactory impairment for 133 stream sites in the western United States. They found that hardness-based chronic copper criteria were lower than predicted IC20 values for olfactory impairment in 129 of the 133 (97%) site waters. In a review of behavioral toxicity endpoints for aquatic organisms exposed to copper relative to hardness-based copper AWQC (Shephard 2008, 2010; Shephard and Zodrow 2009), 3 of 146 behavioral LOECs for copper were found to be less than corresponding hardness-based copper AWQC (with one of the studies (Folmar 1976) having high uncertainty). Accordingly, the hardness-based copper AWQC was protective of behavioral effects in 98 to 99% of the behavioral toxicity studies conducted to date.

Accordingly, it is reasonable to assume that the copper TRV used in this BERA (the AWQC) is protective against effects of copper on olfactory function and avoidance behavior in fish. Most of the studies that have evaluated the effects of metals on olfactory impairment and behavioral avoidance relative to AWQC have focused on copper. The conclusions from this summary are assumed to equally apply to other metals, although this wider applicability is uncertain from the data available to-date.

7.3.5 Risk Characterization

This section presents the risk estimates for fish based on the surface water LOE. An HQ approach was used to quantify risk estimates following the two-step risk characterization process described in Section 7.3.1. HQs were derived for all COPCs using Equation 6-1, in which the EPC and TRV represent surface water concentrations. Section 7.3.5.1 presents the risk characterization results and uncertainty evaluation for each fish receptor. Section 7.3.5.2 presents the results of the evaluation of avoidance behavior in migrating salmonids. Section 7.3.5.3 presents an evaluation of background concentrations. Results of the surface water LOE are integrated with those from other LOEs and uncertainties in the fish risk conclusions section (Section 7.6).

7.3.5.1 Risk Characterization Results and Uncertainty Evaluation

The HQs from the sample-by-sample comparison of EPCs with corresponding TRVs are presented in Table 7-36. HQs are ≥ 1 for all COPCs, except 4,4′-DDT and total PCBs. HQs for 4,4′-DDT and total PCBs are < 1 when calculated based on the alternative TRVs but ≥ 1.0 when calculated based on AWQC. All COPCs, including 4,4′-DDT and total PCBs, were retained for subsequent evaluation at the scale of the receptor-specific exposure areas as presented in the following subsections.

Table 7-36. Number of Individual Surface Water Samples with HQs ≥ 1

COPC	Number of EPCs with HQs ≥ 1 (Maximum HQ)	Percentage of Samples with HQs ≥ 1
Metals		
Zinc (dissolved)	1of 167 (**1.1**)	< 1%
Butyltins		
Monobutyltin	1 of 167 (**1.2**)	< 1%
PAHs		
Benzo(a)anthracene	2 of 245 (**10**)	< 1%
Benzo(a)pyrene	3 of 245 (**14**)	1.2%
Naphthalene	10 of 268 (**50**)	3.7%
Phthalates		
BEHP	2[a] of 190 (**2.3**)	1.1%

Table 7-36. Number of Individual Surface Water Samples with HQs ≥ 1

COPC	Number of EPCs with HQs ≥ 1 (Maximum HQ)	Percentage of Samples with HQs ≥ 1
PCBs		
Total PCBs	0 of 160 (0.089)[b]	< 1%
Pesticides		
4,4′-DDT	0 of 170 (0.43)[c]	< 1%
Total DDx	1 of 170 (**1.8**)[c, d]	< 1%
VOCs		
Ethylbenzene	1 of 23 (**1.6**)	3.7%
Trichloroethene	1 of 23 (**4.1**)	3.7%

[a] An additional two samples had DLs that were greater than the TRV. The maximum HQ based on a DL is 1.4 for BEHP.

[b] Maximum HQ and the number/percentage of EPCs with HQs ≥ 1 presented in the table are based on the alternative TRV. Two of 160 samples had total PCB concentrations greater than the AWQC total PCB TRV of 0.014 µg/L, which is specific to protection of mink via consumption of contaminated prey (maximum HQ = 1.2).

[c] Maximum HQ and the number/percentage of EPCs with HQs ≥ 1 presented in the table is based on the alternative TRV. Nineteen of170 and 35 of 170 samples had 4,4′-DDT and total DDx concentrations, respectively, greater than the AWQC 4,4′-DDT TRV of 0.001 µg/L, which is based on protection of birds (maximum HQs are 4.7 and 20, respectively). An additional four samples had DLs that were greater than the AWQC TRV. The maximum HQ based on a DL is 1.6 for both 4,4′-DDT and total DDx.

[d] The only sample resulting in an HQ ≥ 1 is based on N-qualified data, indicating that the elevated concentration was likely due to analytical interference from a different chemical.

AWQC – ambient water quality criteria
BEHP – bis(2-ethylhexyl) phthalate
COPC – contaminant of potential concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
DL – detection limit
EPC – exposure point concentration
HQ – hazard quotient
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT and 4,4′-DDT)
TRV – toxicity reference value
VOC – volatile organic compound

Bold identifies HQs ≥ 1.

7.3.5.1.1 Large-Home-Range Fish

HQs are the same for all large-home-range fish including largescale sucker, juvenile white sturgeon, juvenile Chinook salmon, peamouth, and Pacific lamprey ammocoetes. HQs calculated based on site-wide UCLs are presented in Table 7-37. Site-wide HQs are < 1 for all COPCs. No site-wide UCLs were derived for VOCs (ethylbenzene and trichloroethene) because of the limited spatial extent of the data.

Table 7-37. Summary of Site-Wide Surface Water UCL HQs

COPC	UCL Concentration (ng/L)	Water TRV (ng/L)	HQ
Metals			
Zinc	2,500	36,500	0.068
Butyltins			
Monobutyltin ion	4.3	72	0.059
PAHs			
Benzo(a)anthracene	6.9	27	0.26
Benzo(a)pyrene	9.5	14	0.68
Naphthalene	3.18[a]	12,000	0.00027
Phthalates			
BEHP	540	3,000	0.18
PCBs			
Total PCBs	1.8	190 (14)[b]	0.0095 (0.13)[b]
Pesticides			
4,4′-DDT	0.67	300 (1)[b]	0.0022 (0.67)[b]
Total DDx	1.6	300 (1)[b, c]	0.0053 (**1.6**)[b, c]

[a] Naphthalene UCL is based on results of all surface water samples except those collected between RM 6.4 and RM 6.5 from the non-LWG sampling event; the average naphthalene concentration from the non-LWG sampling event (106 µg/L) was treated as a single result in the UCL calculation.

[b] TRVs and HQs are presented as alternative TRV (the AWQC TRV and HQ are shown in parentheses).

[c] Criterion for 4,4′-DDT used to evaluate total DDx.

AWQC – ambient water quality criteria
BEHP – bis(2-ethylhexyl) phthalate
COPC – contaminant of potential concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
HQ – hazard quotient
LWG – Lower Willamette Group
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT and 4,4′-DDT)
TRV – toxicity reference value
UCL – upper confidence limit on the mean

Bold identifies HQs ≥ 1.

Risk from total PCBs, 4,4′-DDT, and total DDx was evaluated based on alternative TRVs (as described in Section 7.3.4.1). When risks are calculated using the alternative TRVs, all HQs are < 1, except one sample results in a total DDx HQ ≥ 1. All PCBs concentrations were also below the LCV for fish (0.098 µg/L) reported in the PCBs AWQC criteria document (EPA 1980c). The AWQC TRVs for total PCBs and 4,4′-DDT are based on protection of mammals and birds, respectively via ingestion of contaminated prey; the alternative TRVs are more realistic for aquatic organisms directly exposed to surface water, including the large-home-range fish in the Study Area.

There is uncertainty in the exposure data in that surface water samples were collected over the course of several sampling events, which differed in duration and methods. UCLs calculated from these data are likely to overrepresent data from the sampling events of shorter duration and may over- or underpredict the true time-weighted average and associated risks, depending on the representativeness of the data from these shorter-term sampling events. Having been collected from several locations throughout the Study Area, the surface water samples characterize spatial variability in COPC concentrations in water at this exposure scale. However, because the sampling design did not account for differences in habitat quality for large-home-range fish througout the Study Area, UCLs may over- or underpredict risks, depending on the degree to which COPC concentrations represent conditions in the preferred habitats of the receptors.

When samples for analysis of organic chemicals were collected from the same location using both XAD and peristaltic sampling methods, EPCs were calculated using only the XAD samples. Data from surface water samples collected at the same location using both methods were compared with evaluate the uncertainty associated with exclusion of the peristaltic data (see Section 6.5.5.2.2). In only two of the 30 locations where co-located samples were collected did COPCs detected in peristaltic samples exceed the TRV when those in XAD samples did not. This difference was observed for total DDTs and 4,4′-DDT. Inclusion of these peristaltic data in the UCLs would not have changed the HQs for large-home-range fish.

7.3.5.1.2 Small-Home-Range Fish

Sculpin

The risk characterization for sculpin, except for ethylbenzene and trichloroethene, is based on the individual surface water sample HQs (Table 7-36). HQs are ≥ 1 for at least one surface water sample for all COPCs, except 4,4′-DDT and total PCBs. HQs based on the alternative TRV for 4,4′-DDT, and total PCBs are < 1 in all but one sample, whose total DDx HQ = 1.8. In some samples HQs are ≥ 1when calculated based on AWQC. A detailed discussion of HQs on a sample-by-sample basis is presented in Section 6.5.5, and sample locations where HQs are ≥ 1 are presented on Map 6-27.

As discussed in Section 7.3.3.1, ethylbenzene and trichloroethene samples were collected over a small area (a stretch of approximately 0.15 mile), although it is a reasonably relevant exposure scale for sculpin. Because of the limited spatial extent of these data, ethylbenzene and trichloroethene risks were evaluated only for sculpin. VOC EPCs for sculpin were represented by UCLs (or maximum concentrations) based on these spatially limited data. Naphthlene results from samples collected over this 0.15-mile stretch were also treated as a single EPC. The sculpin HQs for naphthalene, ethylbenzene, and trichloroethene are presented in Table 7-38. Based on these results, ethylbenzene screens out as a COPC for sculpin.

Table 7-38. Number of Sculpin Surface Water Naphthalene, Ethylbenzene, and Trichloroethene EPCs with HQs ≥ 1

COPC	Number of EPCs with HQs ≥ 1 (Maximum HQ)	Percentage of EPCs with HQs ≥ 1
PAHs		
Naphthalene	1 of 246 (**14**)	0.41%
VOCs		
Ethylbenzene	0 of 1 (0.32)[a]	0%
Trichloroethene	1 of 1 (**4.1**)[b]	100%

[a] HQ is based on a single EPC derived from a UCL of samples collected as part of a non-LWG sampling event from approximately RM 6.4 to RM 6.5 along an approximately 0.15- mile length of the river on the west bank.

[b] HQ is based on a single EPC represented as the maximum concentration (UCL could not be derived) of samples collected as part of a non-LWG sampling event from approximately RM 6.4 to RM 6.5 along an approximately 0.15- mile length of the river on the west bank.

COPC – contaminant of potential concern
DL – detection limit
EPC – exposure point concentration
HQ – hazard quotient
LWG – Lower Willamette Group
PAH – polycyclic aromatic hydrocarbon
RM – river mile
TRV – toxicity reference value
VOC – volatile organic compound
UCL – upper confidence limit on the mean

Bold identifies HQs ≥ 1.

There is uncertainty in the exposure data in that surface water samples were collected over the course of several sampling events, which differed in duration and methods. Because samples were collected at a larger spatial scale than sculpin home ranges, concentrations of COPCs in some areas where sculpin may be exposed are not characterized. However, the magnitude of exposure in these areas is not likely greater than that in the areas characterized.

The single trichloroethene HQ ≥ 1 is based on the maximum concentration and likely overestimates risk to sculpin. Trichloroethene was detected in only 2 of 23 samples, all from the west bank between RM 6.4 and RM 6.5 (0.61 and 194 µg/L). Trichloroethene was not detected above the DL (0.2 µg/L) in any other sample.

Sculpin HQs for naphthalene are < 1 for all samples, except those collected between RM 6.4 and RM 6.5; the HQ based on the UCL concentration (173 µg/L) from these data is 14. Additional uncertainty arises because the TRV was extrapolated from a fish acute LC50. No samples exceeded the only chronic value for fish (619 µg/L) reported in Suter and Tsao (1996).

Total DDx concentrations in one sample (W001 located at RM 2.0) exceeded the TRV (HQ based on alternative TRV = 1.8); however, this result was N-qualified, indicating the elevated concentration was likely due to analytical interference from a different chemical. As discussed in Section 7.3.4.1, a high degree of uncertainty is associated with risk estimates for 4,4′-DDT, and total DDx when calculated based on the AWQC-derived TRV for 4,4′-DDT. The AWQC TRV is based on protection of individual brown pelicans through ingestion of contaminated prey. When 4,4′-DDT and total DDx HQs are calculated using the alternative TRV based on protection of aquatic organisms directly exposed to surface water, all but one HQ is < 1.

There is additional uncertainty in HQs for total DDx and 4,4′-DDT because nearly one-third of total DDx and 4,4′-DDT exceedances are based on N-qualified data (31% for total DDx and 32% for 4,4′-DDT). The qualification indicates that the analyst believed the result was due to analytical interference from a chemical other than the target analyte. The highest non-N-qualified HQ based on the AWQC TRV is 9.8; this sample was collected at RM 7.2 during a low-flow event (Figure 6-19).

When samples for analysis of organic chemicals were collected from the same location using both XAD and peristaltic sampling methods, EPCs were calculated using only the XAD samples. Data from surface water samples collected at the same location using both methods were compared to evaluate the uncertainty associated with exclusion of the peristaltic data (See Section 6.5.5.2.2). In only two of the 30 locations where co-located samples were collected did COPCs detected in peristaltic samples exceed the TRV when those in XAD samples did not. This difference was observed for total DDTs and 4,4′-DDT. Inclusion of these peristaltic data would have identified two additional locations (W027 and W031) with DDx HQs ≥ 1 based on the AWQC TRV. HQs at these two locations are < 1 based on the more appropriate alternative TRV, which is based on toxicity to aquatic organisms directly exposed to surface water rather than on toxicity to brown pelican via ingestion of contaminated prey.

As discussed in Section 7.3.4.1, a high degree of uncertainty is associated with risk estimates for total PCBs calculated using the AWQC-derived TRV. The AWQC TRV is based on protection of individual mink through ingestion of contaminated prey. When total PCBs HQs are calculated using the alternative TRV based on protection of aquatic organisms directly exposed to surface water, all HQs are < 1. All PCB concentrations were also below the LCV for fish (0.098 µg/L) reported in the PCBs AWQC criteria document (EPA 1980c).

The single monobutyltin TRV exceedance (HQ = 1.2) is uncertain. The monobutyltin TRV likely overestimates risk because it is based on a TRV for a surrogate-TBT, which is the most toxic of the butyltins (EPA 1991).

Uncertainty in the HQs for benzo(a)anthracene and benzo(a)pyrene is present because the Tier II chronic TRVs are based on benthic invertebrate toxicity data. As discussed in Section 7.3.4.1, a TRV based on benthic toxicity may over- or underestimate risks to fish.

Smallmouth Bass and Northern Pikeminnow

Six COPCs (i.e., zinc, monobutyltin, benzo(a)anthracene, benzo(a)pyrene, naphthalene, and BEHP) have HQs ≥ 1 in at least one 1-mile exposure area for smallmouth bass and northern pikeminnow (Table 7-39).

Table 7-39. Surface Water 1-Mile Exposure Area HQs

Exposure Area	Zinc	Monobutyltin ion	Benzo(a) anthracene	Benzo(a) pyrene	Naphthalene	BEHP	Total PCBs[a]	4,4′-DDT[a]	Total DDx[a]
RM 1.5 to RM 2.5	0.052	0.35[b,c]	0.020	0.028	0.0037[b,c]	0.53[c]	0.017 (0.23)	0.0056 (0.061)	0.69 **(7.6)**
RM 2.5 to RM 3.5	0.056[b,c]	0.24[b,c]	0.072	0.13	0.0043[b,c]	0.43[c]	0.0029 (0.039)	0.0077 (0.085)	0.039 (0.43)
RM 3.5 to RM 4.5	0.058	0.25[c]	0.11	0.12	0.0041[b,c]	**2.3[c]**	0.0312 (0.42)	0.048 (0.53)	0.11 **(1.2)**
RM 4.5 to RM 5.5	0.13[c]	0.28[b,c]	0.29[c]	0.54	0.00050[b,c]	0.073[b,c]	0.0066 (0.089)[b,c]	0.023 (0.26)[b]	0.023 (0.26)[b,c]
RM 5.5 to RM 6.5	0.062	0.28[c]	**4.7**	**7.6**	**2.8**[d]	0.73[c]	0.0033 (0.045)	0.16 **(1.7)**	0.19 **(2.0)**
RM 6.5 to RM 7.5	0.049	0.24[b,c]	0.063	0.18	0.00052	0.51	0.040 (0.55)	0.19 **(2.1)**	0.59 **(6.5)**
RM 7.5 to RM 8.5	0.18[c]	0.039[b,c]	0.14[b]	0.31[b]	0.0013[b,c]	0.68[b,c]	0.0067 (0.091)[b,c]	0.63 (0.69)	0.17 **(1.9)[c]**
RM 8.5 to RM 9.5	0.13[c]	0.076[b,c]	0.14[b,c]	0.31[b]	0.013[c]	0.93[c]	0.0069 (0.094)[b,c]	0.26 **(2.9)**	0.31 **(3.4)[c]**
Swan Island Lagoon	0.089	0.26[c]	0.010[c]	0.43	0.0027[b,c]	0.32[b,c]	0.0057 (0.078)	0.0053 (0.058)	0.019 (0.21)
RM 9.5 to RM 10.5	**1.2**	0.21[c]	0.14[b,c]	0.31[b]	0.003[c]	0.33[c]	0.021 (0.29)	0.43 **(4.7)**	0.30 **(3.4)[c]**
RM 10.5 to RM 11.8	0.060[c]	**1.2[c]**	0.015	0.035	0.0033[b,c]	0.37[c]	0.0023 (0.032)	0.017 (0.19)	0.076 (0.84)[e]

[a] HQ based on alternative TRV; HQ in parentheses based on AWQC TRV.
[b] Maximum concentration is based on one-half DL (where one-half DL > maximum detected concentration or where COPC is not detected).
[c] Data were insufficient to calculate a UCL so EPC was based on the maximum concentration.
[d] Naphthalene UCL is based on results of all surface water samples collected between RM 5.5 and RM 6.5, except samples collected between RM 6.4 and RM 6.5 from the non-LWG sampling event; the average naphthalene concentration from the non-LWG sampling event (106 µg/L) was treated as a single result in the UCL calculation.

AWQC – ambient water quality criteria
BEHP – bis(2-ethylhexyl) phthalate
COPC – contaminant of potential concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
Bold identifies HQs ≥ 1.

DL – detection limit
EPC – exposure point concentration
HQ – hazard quotient
LWG – Lower Willamette Group
PCB – polychlorinated biphenyl
RM – river mile

total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT and 4,4′-DDT)
TRV – toxicity reference value
UCL – upper confidence limit on the mean

When 4,4′-DDT, and total DDx HQs are calculated using the alternative TRVs based on protection of aquatic organisms directly exposed to surface water, all HQs are < 1. When the AWQC TRVs were used, HQs in some exposure areas were ≥ 1. Total PCBs HQs are < 1 based on both the alternative TRV and the AWQC TRV. As discussed above for large-home-range fish, the AWQC TRVs for total PCBs and 4,4′-DDT are based on protection of mammals and birds, respectively, via ingestion of contaminated prey; the alternative TRVs better reflect toxicity to directly exposed aquatic organisms, including northern pikeminnow and smallmouth bass. As discussed above for sculpin, there were two locations where exclusion of peristaltic samples could have affected HQs. Inclusion of peristaltic data would have identified two additional sampling locations (W027 at RM 3 and W031 at RM 6.1) with DDx HQs ≥ 1 based on the AWQC TRV; however, UCL HQs are < 1 and HQs for individual samples are < 1 based on the alternative TRV.

The uncertainties associated with monobutyltin discussed above for sculpin likely result in an overestimate of risk. Uncertainties for benzo(a)anthracene, benzo(a)pyrene, and naphthalene discussed above for sculpin also apply to smallmouth bass and northern pikeminnow. Benzo(a)anthracene, benzo(a)pyrene, and naphthalene HQs are ≥ 1 for only one sampling location (W012) between RM 6.4 and RM 6.5 (see Map 6-27).

There is uncertainty in the exposure data in that surface water samples were collected over the course of several sampling events, which differed in duration and methods, as discussed above for large-home-range fish. Additional uncertainty is associated with risks from all COPCs from at least some exposure areas because the EPCs are based on maximum concentrations (see Table 7-39). As discussed in the exposure section (Section 7.3.3.1), the use of maximum concentrations to represent EPCs may result in an over- or underestimate of risk.

The spatial extent of surface water samples generally includes several locations within each 1-mile exposure area for northern pikeminnow and smallmouth bass and therefore should account for some spatial variability at this exposure scale. However, because sampling design did not account for differences in habitat use throughout the Study Area, risk may be over- or underestimated, depending on the degree to which COPC concentrations used to calculate HQs represent those in preferred habitats.

7.3.5.2 Evaluation of Olfactory-Associated Migration Effects

Per EPA’s Problem Formulation (Attachment 2), the exposure of adult Chinook salmon to metals concentrations in water was evaluated to determine if these concentrations might elicit a behavioral avoidance response that could potentially disrupt migrating populations. Copper concentrations associated with salmonid avoidance response behavior were compared with copper concentrations measured in surface water at the Study Area.

Concentrations of dissolved copper at the Study Area ranged from 0.37 to 2.39 µg/L. This site-wide range of copper concentrations is at the low end of the range of copper concentrations associated with salmonid avoidance response in laboratory studies (Table 7-35). Excluding the uncertain Folmar (1976) study, where copper avoidance by rainbow trout was observed at a nominal added copper concentration of 0.1 µg/L (the total copper concentration was not reported), the lowest copper concentrations associated with statistically significant avoidance were observed at copper concentrations of 0.7 and 2.8 µg/L for Chinook salmon, but not at copper concentrations ≥1.6 µg/L for rainbow trout (Hansen et al. 1999c). Accordingly, the avoidance data are equivocal for Chinook salmon copper concentrations within the Study Area-wide range. Field studies show inconclusive results on whether copper concentrations < 2 µg/L result in impaired olfactory function in fish. Only the highest concentration detected in the Study Area (2.39 µg/L at W023 in January 2006) was greater than 2 µg/L. All other samples were < 2 µg/L. Therefore, it is highly unlikely that copper concentrations in the Study Area would cause a change in migratory behavior. The fact that juvenile and adult salmon passage through the LWR has been observed demonstrates the absence of direct evidence thatsurface water copper concentrations in the Study Area are affecting migration. Furthermore, these copper concentrations are typical of regional levels; based on summary statistics from the USGS National Stream Quality Accounting Network program, water copper concentrations at six sites along the Columbia River Basin from 1996 to 2003 ranged from < 1 µg/L to 6.7 µg/L (USGS 2006). Therefore, the pathway associated with reproductive impairment is incomplete and the potential risk from migratory impairment due to copper avoidance is insignificant.

7.3.5.3 Evaluation of Background Concentrations

Aluminum was not identified as a COPC because no acceptable TRV was identified. Background concentrations were established as part of the RI (see Section 7.0 of the draft final RI (Integral et al. 2011)). Background and Study Area concentrations in sediment and surface water are compared in Attachment 11. The Study Area UCL concentration of aluminum (460 µg/L) was approximately one-third as great as the background UCL and UPL concentrations (1,278 and 1,485 µg/L, respectively). The Study Area UCL sediment aluminum concentration (24,375 mg/kg dw) was similar to the background sediment UCL and UPL (24,877 and 33,842 mg/kg dw, respectively). Based on these comparisons, it was concluded that any potential risk to fish in the Study Area from aluminum cannot be attributed to sources from within the Study Area and that because aluminum is a naturally occurring element with no apparent site-specific sources along the Study Area (historical regional sources of aluminum have included aluminum smelting at Troutdale and the Dalles), it is unlikely that aluminum poses unacceptable risk to fish.

Zinc is also a naturally occurring crustal element in the environment. A background water concentration could not be established because of a limited number of data points (see Attachment 11). The Study Area UCL concentration of zinc (2.5 µg/L) was greater than the highest zinc concentration detected in background[106] (range of 1.4 to 2.2 µg/L). The Study Area UCL sediment zinc concentration (164 mg/kg dw) was greater than the background sediment UCL and UPL (79 and 110 mg/kg dw, respectively). These data indicate that zinc concentrations are elevated above background and that zinc concentrations in the Study Area cannot be attributed solely to background. This is as would be expected in a river within an urbanized basin.

7.3.5.4 COIs for Which Risks Cannot Be Quantified

COIs for which risks to fish cannot be quantified based on surface water data are listed in Table 7-40. TRVs are not available for any of the contaminants listed.

Table 7-40. Fish Surface Water COIs with No Available TRV

COI	Rationale for Why Risks Cannot be Quantitatively Evaluated
Metals	
Aluminum	The AWQC chronic criterion for aluminum was used to identify aluminum as a COPC. However, as per agreement with EPA, the AWQC criterion is not applicable to waters with circumneutral pH, such as those in the Study Area, precluding further evaluation of aluminum.
SVOCs	
4-Chloroaniline	Risk to fish based on surface water data unknown; no water threshold available.
Aniline	Risk to fish based on surface water data unknown; no water threshold available.
Herbicides	
2,4-DB	Risk to fish based on surface water data unknown; no water threshold available.
MCPP	Risk to fish based on surface water data unknown; no water threshold available.

2,4-DB – 4-(2,4-dichlorophenoxy)butyric acid
COI – contaminant of interest
MCPP – methylchlorophenoxypropionic acid
TRV – toxicity reference value
SVOC – semivolatile organic compound

7.3.5.5 Summary of Surface Water LOE

No COPCs with HQs ≥ 1 were identified for large-home-range fish. Eight surface water COPCs with HQs ≥ 1 were identified for small-home-range fish: zinc, monobutyltin, benzo(a)anthracene, benzo(a)pyrene, naphthalene, BEHP, total DDx, and trichloroethene.

[106] Zinc concentrations were detected in only 3 of 22 surface water samples included in the background dataset (see Section 7.0 of the RI).

With one minor exception, all HQs calculated using the alternative TRVs (based on effects in fish) for 4,4′-DDT and DDx are < 1.[107] When calculated using the 4,4′-DDT AWQC-based TRV, HQs for 4,4′-DDT and total DDx are ≥ 1 for both small-home-range and large-home-range fish. The AWQC-based TRV for 4,4′-DDT reflects protection of piscivorous birds via ingestion of contaminated prey; the alternative TRV for protection of directly exposed aquatic organisms is more reflective of risks to fish. Total PCBs HQs for some exposure areas are also ≥ 1 for small-home-range fish when calculated using the AWQC-based TRV, which reflects protection of mammals via ingestion of contaminated prey. The alternative TRV for protection of directly exposed aquatic organisms is more reflective of risks to fish. No total PCBs HQs are ≥ 1 when calculated using the alternative TRV. Results of the surface water LOE are integrated with those of other LOEs to determine risk conclusions for fish in Section 7.6.

7.4 TZW ASSESSMENT

The TZW[108] assessment is an additional LOE that was used to estimate risks to benthic fish (represented in the BERA by sculpin and lamprey ammocoetes) that might be exposed to TZW. A similar evaluation for benthic invertebrates is presented in Section 6.6. This section generally summarizes that previous section.

The TZW samples evaluated in this assessment were collected primarily during a 2005 sampling effort focusing on areas offshore of nine upland sites with known or likely pathways for discharge of upland contaminated groundwater to the Study Area. Sampling locations were selected at each of the nine study sites based on results of a groundwater discharge mapping field effort. The RI Appendix C2 presents the process used to select these sites per agreement with EPA, ODEQ, and LWG. The findings of the discharge mapping effort were considered in conjunction with relevant site data (e.g., hydrogeology, surface sediment texture delineation, distribution of COIs in upland groundwater and sediments) to identify zones of possible contaminated groundwater discharge. The TZW sampling locations selected for each site focused primarily on the zones of possible groundwater plume discharge, based on the groundwater pathway assessment (GWPA) discharge mapping effort. Additional sampling locations were specified to provide comparative data for TZW quality outside of the potential discharge zones (Integral et al. 2011).

Because the primary objective of RI groundwater pathway assessment was to evaluate whether transport pathways from upland contaminated groundwater plumes to the river were complete, TZW target analyte lists varied from site to site and were derived primarily based on the COIs in the upland groundwater plumes. Therefore, not all COIs in sediments were analyzed in TZW samples. As described in Sections 4.4.3.1 and 6.1.5.2

[108] For the purpose of the BERA, TZW is the porewater associated with sediment matrix within the top 38 cm of the sediment column. TZW is composed of some percentage of both groundwater and surface water.

of the draft final RI (Integral et al. 2011), there also may be other contaminated groundwater plumes in the Study Area that may be discharging into river sediments where TZW samples have not been collected.

The remainder of this section is organized as follows:

- Section 7.4.1 describes the general approach used to assess risks to benthic fish from TZW.
- Section 7.4.2 summarizes the TZW COPCs evaluated. Some COPCs were not evaluated because no toxicity thresholds were available.
- Section 7.4.3 explains how exposure concentrations were estimated and describes uncertainties in those estimates. All TZW contaminant concentrations are presented in Attachment 4.
- Section 7.4.4 summarizes the effects data. Details on the development of the water TRVs are presented in Attachment 10.
- Section 7.4.5 presents the risk characterization results and associated uncertainties.

Figure 7-14 shows how the TZW evaluation is organized.

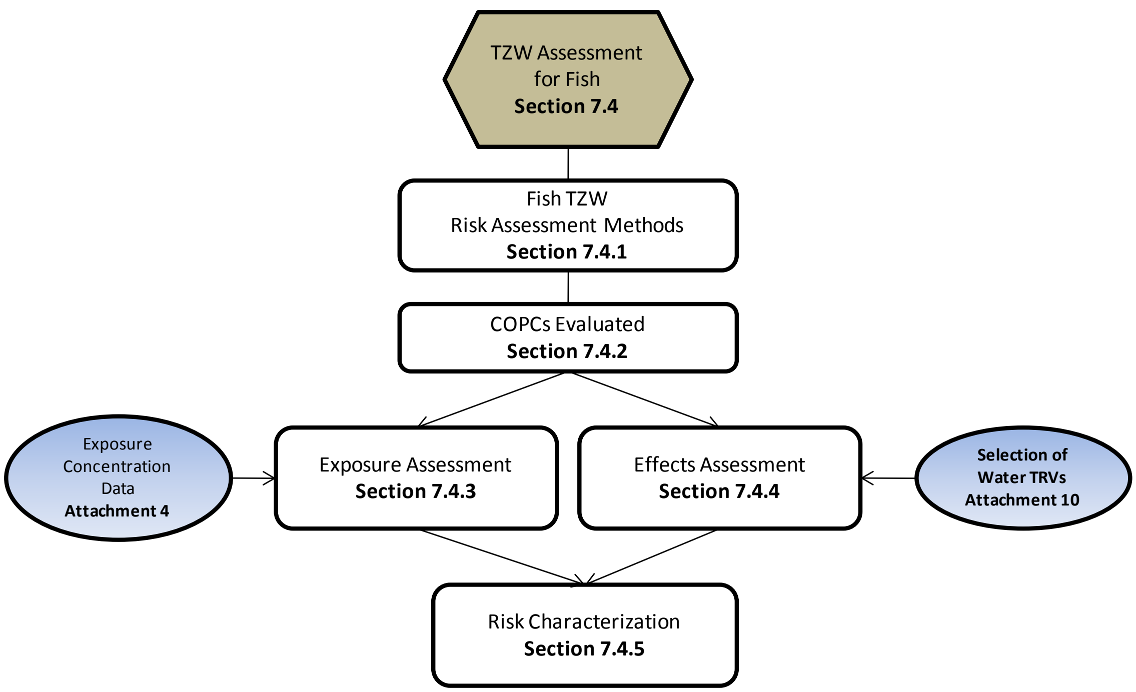

Figure 7-14. Overview of TZW Section Organization

7.4.1 Fish TZW Risk Assessment Methods

As described in Section 6.6.1, TZW HQs were calculated by comparing COPC concentrations in individual TZW samples to chronic water TRVs developed according to a hierarchy of water quality criteria and literature-based TRVs articulated in Attachment 10. Potentially unacceptable risks were identified by COPCs that resulted in HQs≥ 1. Exposure data, effects data, and the quantitative risk results (i.e., magnitude, spatial distribution, and frequency of HQs) are presented in the following sections. The relative strengths and uncertainties for all fish LOEs are evaluated together in the risk conclusions for fish (Section 7.6).

7.4.2 COPCs Evaluated

Fifty-four of the 58 TZW COPCs identified in the SLERA and refined screen (Attachment 5) are evaluated in the BERA. Four individual DDT metabolites identified in the SLERA (2,4′-DDD, 2,4′-DDT, 4,4′-DDD, and 4,4′-DDE) were evaluated as total DDx and were not evaluated individually; 4,4′-DDT was evaluated both individually and as total DDTs because the TRV for DDTs is based on 4,4′-DDT.

Table 6-41 presents the detected TZW COPCs by site.

Seven TZW COIs, could not be evaluated because no toxicological data were available from which to develop water TRVs (Table 7-41). The risks to fish receptors associated with exposure to these contaminants in TZW are therefore unknown. By agreement with EPA, aluminum was not identified as a COPC because its AWQC was developed based on toxicity data from acidic waters and so is not applicable to the circumneutral waters of the Study Area. Aluminum concentrations in background surface water and sediment were evaluated to determine if there may be a local source of aluminum contamination within the Study Area (Section 6.5.4.3).

Table 7-41. TZW COIs Without Screening-Level Benchmarks

COIs	
Metals	
Aluminum	Titanium
Calcium	
Petroleum Hydrocarbons	
Residual-range hydrocarbons	Total diesel-residual hydrocarbons
Diesel-range hydrocarbons	Total petroleum hydrocarbons

COI – contaminant of interest
TPH – total petroleum hydrocarbons
TZW – transition zone water

Surface water thresholds are unavailable for individual dioxins and furans other than 2,3,7,8-TCDD. Dioxins and furans are evaluated as total dioxin/furan TEQ and total TEQ, toxicity-weighted sums based on the relative toxicity of each congener to 2,3,7,8-TCDD using TEFs based on their common mechanism for toxicity. Based on this evaluation, total dioxin/furan TEQ and total TEQ were not identified as COPCs in the SLERA.

In addition, two tissue COIs (selenium and styrene) were not retained as COPCs because no detected concentrations exceeded TRVs (although at least one DL exceeded a TRV). The TZW LOE therefore was not used in assessing potential risks to fish receptors from these contaminants(see Table 5-5).

7.4.3 Exposure Assessment

This section presents the TZW exposure concentrations used to assess potential risks to fish. An overview of the sampling methods for all LWG- and non-LWG-collected TZW data used in this assessment is presented in the benthic risk assessment (Section 6.6.3.1). TZW sampling locations used in this assessment are presented on Map 4-15.

7.4.3.1 TZW EPCs

TZW EPCs in this assessment are represented by TZW concentrations in all individual TZW samples collected in the Study Area collected regardless of sampling method or depth sampled.[109] TZW concentrations were compared with water TRVs to characterize risks to benthic fish via exposure to TZW. A summary of the chemicals detected in shallow TZW and the range of concentrations is presented in the benthic risk assessment (Section 6.6.3.2). All TZW data, by site, are presented in Attachment 4.

7.4.3.2 Uncertainty Associated with Ecological Exposure to TZW

Section 6.6.3.3 discusses the uncertainties associated with the benthic invertebrate exposure data for the TZW assessment, including the key uncertainty of the degree to which the collected TZW samples are representative of exposure conditions for benthic fish. The same uncertainties are relevant to fish living in or on the sediment (i.e., sculpin and lamprey ammocoetes).

As is the case for most benthic invertebrates, the water column exposure pathway for benthic fish is more important than theTZW pathway. Lamprey ammocoetes burrow in the sediment in J-shaped burrows, where they filter feed by putting their mouth at the sediment surface (Kostow 2002a). While ammocoetes may occasionally draw food from sediments when their burrow openings are closed or during movement within their burrows, food is drawn primarily from the water (Moore and Mallatt 1980). The potential for exposure to TZW is even less for non-burrowing benthic fish, such as sculpin. This species may feed at the very surface of the sediment, preying on benthic and epibenthic invertebrates, small fish, and fish eggs. As adults, large sculpin can burrow as deep as 14 inches (36 cm) into gravel to forage (Wydoski and Whitney 1979). Nonetheless, sculpin have less exposure to sediment than burrowing organisms because they do not live within the sediment.

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[109] All TZW samples evaluated in this BERA were within the 0 – 38 cm depth; however, the depth of the different sampling equipment used to collect TZW (i.e., using peeper, Trident® probe, and Geoprobe) varied.

Given their feeding habits and the low oxygen levels at the depths represented by the TZW samples, benthic fish, whether burrowers (lamprey ammocoetes) or benthic feeders (sculpin), have relatively low exposure to porewater compared with surface water. Thus, the representativeness of the COPC concentrations in shallow TZW for purposes of estimating exposure and subsequent risks to benthic fish is questionable and conservative, to an uncertain degree.

7.4.4 Effects Assessment

TZW contaminant concentrations were compared with the effects thresholds as part of the risk characterization process. At the direction of EPA (2008f), chronic water TRVs were developed for all TZW COPCs according to the hierarchy detailed in Attachment 10. Chronic water TRVs were developed through a review of WQS, criteria, published benchmarks, and toxicity data. The selected TRVs were approved by EPA for use in the BERA. Criteria for metals COPCs were hardness-adjusted when appropriate. If the published criteria for individual metals were based on dissolved concentrations, then the dissolved sample result was compared with the dissolved criterion; otherwise the total concentration for both the sample and criterion were used. Table 6-40 presents the TRVs for all TZW COPCs and their sources. These values were developed based on the sensitivities of fish and invertebrate species and are considered protective of all aquatic receptors, including benthic invertebrates.

As noted in Section 7.3.4.1, because the selected AWQC for 4,4′-DDT is based on protection of birds via ingestion of contaminated prey, risk estimates for aquatic receptors based on this TRV are associated with substantial uncertainty. An alternative TRV protective of aquatic organisms (including fish) was developed in this BERA, using methods consistent with those used for AWQC derivation. The alternative water TRV for DDx compounds was calculated as 0.011 µg/L. For evaluating direct exposure of aquatic organisms to water, this alternative TRV is considered more appropriate than the AWQC (0.0010 µg/L), which is based on the protection of brown pelican via ingestion of contaminated prey.Two sets of water HQs were derived: one using the selected DDT water TRV (0.0010 µg/L) and one using the alternative water TRV (0.011 µg/L). Because the the alternative TRVs are more appropriate for assessing risk to fish, they were used to determine risk conclusions.

To address some of the uncertainties associated with the representativeness of water effects data for Pacific lamprey, site-specific acute toxicity tests were conducted with lamprey ammocoetes as part of the BERA (Windward and Integral 2008) (see Section 7.3.4.2). These tests were conducted with six chemicals selected to represent a range of toxic modes of action. The results of the test showed that lamprey in general were as sensitive as or less sensitive than other fish. Therefore, the selected TRVs are not likely to underpredict risk but may overpredict risk. Lamprey-specific LOAELs for mortality for these chemicals were as follows: 30 µg/L for pentachlorophenol, 43 µg/L for copper, 13 µg/L for diazinon, 630 µg/L for aniline, 10 µg/L for naphthalene, and > 2.7 µg/L for lindane. These LOAELs were used in an uncertainty analysis to evaluate risk to lamprey from contaminants with HQs ≥ 1 that have similar modes of action.

> **Uncertainties Associated with Effects Data**
>
> TRVs were selected from regulatory standards (state WQS) and criteria (national AWQC), as well as other published effects thresholds (e.g., Tier II, LCV from Suter and Tsao (1996)) following an agreed-upon hierarchy (see Attachment 10). Where available, the TRVs are based on WQS or AWQC and are assumed to have less uncertainty than TRVs based on other sources, although it is also important to take into account the relevance of the determinative receptor and pathway for each TRV. As an example, the chronic DDT AWQC (0.001 µ/L) was selected to be protective of brown pelican reproduction via a fish ingestion pathway. A criterion derived for the protection of invertebrates from direct contact with water using data included in the DDT AWQC document would be 1 to 2 orders of magnitude higher.
>
> The LCVs were most often applied when regulatory values were not available. TRVs for sodium, potassium, and magnesium were based on LCVs, which were derived from daphnid toxicity data and may not accurately characterize effects on benthic invertebrates, fish, amphibians, and plants. In the case of sodium and potassium, the Tier II value was cited by Suter and Tsao (1996) as being below commonly occurring ambient values and not appropriate for use as a screening value. In addition, TRVs based on LCVs may inaccurately estimate risks to benthic receptors because these values are based on a limited number of studies and species.
>
> The TRVs for four VOCs (i.e., 1,2,4-trimethylbenzene, 1,3,5-trimethylbenzene, isopropylbenzene, and chloroethane) are uncertain because they are based on surrogates (ethylbenzene and 1,1-dichloroethane). No toxicological data were available for these COPCs, and the surrogate TRVs may over- or underestimate toxicity to benthic fish.
>
> The TRV for perchlorate is uncertain because it was calculated from an acute value using an estimated relationship between acute and chronic responses.
>
> The AWQC that was the source of the iron TRV is based on a site receiving acid mine drainage, and derivation of the AWQC was not consistent with later methods for deriving criteria (Suter and Tsao 1996).

7.4.5 Risk Characterization

This section presents the risk estimates for benthic fish based on the TZW LOE. It also includes an evaluation of naturally occurring metals, and a list of COIs that could not be evaluated.

7.4.5.1 Risk Characterization Results

Individual HQs were calculated for all COPCs across all TZW samples. The frequencies with which individual samples have HQs ≥ 1 are shown in Table 7-42. For total DDx and 4,4′-DDT, the HQs are based on the alternative TRV (0.011 µg/L) developed for the BERA (see Section 7.3.4.1). This alternative TRV is considered more appropriate than the AWQC (0.0010 µg/L), which is based on the protection of brown pelican via ingestion of contaminated prey. HQs based on the AWQC value (0.0010 µg/L) would be an order of magnitude higher than those presented in Table 7-42.

The uncertainties associated with the TZW data as representive exposure data for benthic fish is dicussed in Section 7.4.3. Actual exposure of benthic organisms to TZW is likely lower than represented by TZW COPC concentrations because of feeding habits, burrowing behavior, and low oxygen levels at TZW sample depths. If the ventilation rate for fish is assumed to be similar to the 10% that has been reported for filter-feeding clams (Winsor et al. 1990), the TZW HQs presented in Table 6-41 would be reduced by an order of magnitude. HQs would be < 1 for several metals (i.e., beryllium, cadmium, cobalt, copper, lead, magnesium, nickel, and potassium), dibenzofuran, several VOCs (i.e., 1,1-dichloroethene, 1,2,4-trimethylbenzene, 1,3,5-trimethylbenzene, chloroethene, isopropylbenzene, and m,p-xylene), and three inidividual gasoline-range fractions. HQs would remain ≥ 1 but < 10 for several additional metals, most individual PAHs, two SVOCs, several VOCs, and perchlorate.

The point-by-point assessment of TZW is not representative of the site-wide exposure scale that is more appropriate for the evaluation of lamprey ammocoetes. Thus, a high degree of uncertainty is associated with the evaluation of TZW for the potential risks to lamprey ammocoetes or other benthic fish with site-wide exposure scales.With TZW samples having been collected preferentially from locations where TZW is most likely to carry contaminants from upland sites, risks to fish from exposure to TZW are likely to be overestimated. Because petroleum compounds are not CERCLA contaminants gasoline-range hydrocarbons are not included in the final count of contaminants posing potentially unacceptable risk; these contaminants may nonetheless pose risk to benthic fish.

Table 7-42. TZW COPCs with HQs ≥ 1 in Individual Samples by Area

COPC	Number of Samples with HQs ≥ 1 (Maximum HQ)										
		Arkema									
	ARCO	Acid Plant Area	Chlorate Plant Area	Mobil Oil	Gasco	Gunderson	Kinder Morgan	Rhône-Poulenc	Siltronic	Willbridge
Metals										
Barium (total)	7 of 7 (73)	8 of 8 (610)	10 of 10 (1,100)	11 of 11 (88)	8 of 8 (86)	9 of 9 (68)	8 of 8 (31)	10 of 10 (170)	13 of 13 (57)	9 of 9 (86)
Beryllium (total)	0 of 7	0 of 8	0 of 10	1 of 11 (1.8)	0 of 8	0 of 9	0 of 8	1 of 10 (1.7)	0 of 13	1 of 9 (2)
Cadmium (dissolved)	0[a] of 5	0 of 4	3 of 6 (2.6)	1 of 12 (1.1)	0 of 4	0 of 2	0[a] of 3	5[b] of 7 (5.8)	0 of 6	1 of 6 (1.5)
Cobalt (total)	NA	NA	NA	NA	NA	NA	NA	NA	3 of 13 (3.6)	NA
Copper (dissolved)	0 of 5	NA	NA	0 of 12	0 of 4	0 of 2	0 of 3	1 of 7 (1.3)	0 of 6	0 of 6
Iron (total)	7 of 7 (75)	7 of 8 (110)	6 of 10 (250)	11 of 11 (110)	8 of 8 (130)	9 of 9 (91)	8 of 8 (49)	10 of 10 (98)	26 of 26 (180)	9 of 9 (120)
Lead (dissolved)	0 of 5	0 of 4	0 of 6	1 of 12 (3)	2 of 4 (1.7)	0 of 2	0 of 3	1 of 7 (2.8)	0 of 6	0 of 6
Magnesium (total)	0 of 7	4 of 8 (7)	1 of 10 (3.8)	0 of 11	0 of 8	0 of 9	0 of 8	3 of 10 (2.2)	0 of 26	0 of 9
Manganese (total)	7 of 7 (52)	8 of 8 (94)	10 of 10 (550)	11 of 11 (150)	8 of 8 (130)	9 of 9 (43)	8 of 8 (72)	10 of 10 (130)	26 of 26 (84)	8 of 9 (110)
Nickel (dissolved)	0 of 5	0 of 4	1 of 6 (1.6)	0 of 12	1 of 4 (1.1)	0 of 2	0 of 3	1[b] of 7 (1.1)	0 of 6	0 of 6
Potassium (total)	0 of 7	0 of 8	2 of 10 (3.7)	0 of 11	0 of 8	0 of 9	0 of 8	0 of 10	0 of 13	0 of 9

Table 7-42. TZW COPCs with HQs ≥ 1 in Individual Samples by Area

	Number of Samples with HQs ≥ 1 (Maximum HQ)									
		Arkema								
COPC	**ARCO**	**Acid Plant Area**	**Chlorate Plant Area**	**Mobil Oil**	**Gasco**	**Gunderson**	**Kinder Morgan**	**Rhône-Poulenc**	**Siltronic**	**Willbridge**
Sodium (total)	0 of 7	1 of 8 (14)	10 of 10 (55)	0 of 11	0 of 8	0 of 9	0 of 8	0 of 10	0 of 13	0 of 9
Vanadium (total)	NA	NA	NA	NA	NA	NA	NA	NA	6 of 13 (19)	NA
Zinc (dissolved)	1 of 5 (14)	0 of 4	0 of 6	0 of 12	0 of 4	0 of 2	0 of 3	0 of 7	0 of 6	0 of 6
PAHs										
2-Methylnaphthalene	0 of 12	NA	NA	0 of 21	8 of 12 (40)	NA	0 of 11	NA	3 of 19 (17)	0 of 14
Acenaphthene	0 of 12	NA	NA	0 of 21	4 of 12 (5.2)	NA	0 of 11	NA	20 of 32 (17)	0 of 14
Anthracene	0 of 12	NA	NA	0 of 21	10 of 12 (13)	NA	0 of 11	NA	18 of 32 (87)	0 of 14
Benzo(a)anthracene	1[c] of 12 (5.6)	NA	NA	5[c] of 21 (8.5)	9 of 12 (120)	NA	2 of 11 (2.9)	NA	14[d] of 32 (1,200)	0[a, b,] of 14
Benzo(a)pyrene	2[c] of 12 (15)	NA	NA	5[c]21 (25)	9 of 12 (210)	NA	0[a, c] of 11	NA	18[b] of 32 (2,700)	0[a, b,] of 14
Benzo(b)fluoranthene	0 of 12	NA	NA	0 of 21	3 of 12 (3.1)	NA	0 of 11	NA	10 of 32 (49)	0 of 14
Benzo(g,h,i)perylene	0 of 12	NA	NA	1 of 21 (1.1)	3 of 12 (7.3)	NA	0 of 11	NA	9 of 32 (66)	0 of 14
Benzo(k)fluoranthene	0 of 12	NA	NA	0 of 21	3 of 12 (3.1)	NA	0 of 11	NA	7 of 32 (14)	0 of 14
Chrysene	0 of 12	NA	NA	0 of 21	3 of 12 (2.2)	NA	0 of 11	NA	7 of 32 (17)	0 of 14

Table 7-42. TZW COPCs with HQs ≥ 1 in Individual Samples by Area

	Number of Samples with HQs ≥ 1 (Maximum HQ)										
		Arkema									
COPC	ARCO	Acid Plant Area	Chlorate Plant Area	Mobil Oil	Gasco	Gunderson	Kinder Morgan	Rhône-Poulenc	Siltronic	Willbridge
Dibenzo(a,h)anthracene	0 of 12	NA	NA	0 of 21	1 of 12 (1.2)	NA	0 of 11	NA	7 of 32 (13)	0 of 14
Fluoranthene	0 of 12	NA	NA	0 of 21	3 of 12 (2.8)	NA	0 of 11	NA	8 of 32 (17)	0 of 14
Fluorene	0 of 12	NA	NA	3 of 21 (1.5)	10 of 12 (7.9)	NA	0 of 11	NA	23 of 32 (28)	0 of 14
Indeno(1,2,3-cd)pyrene	0 of 12	NA	NA	1 of 21 (1.2)	3 of 12 (9.8)	NA	0 of 11	NA	9 of 32 (61)	0 of 14
Naphthalene	0 of 12	2[b] of 9 (2.2)	0 of 10	0 of 21	6 of 12 (260)	0 of 9	0 of 12	0 of 10	23 of 60 (1,100)	0 of 14
Phenanthrene	0 of 12	NA	NA	5 of 21 (2.4)	10 of 12 (13)	NA	0 of 11	NA	21 of 32 (57)	0 of 14
Pyrene	0 of 12	NA	NA	0 of 21	3 of 12 (3.2)	NA	0 of 11	NA	8 of 32 (15)	0 of 14
SVOCs										
1,2-Dichlorobenzene	0 of 7	0 of 9	0 of 10	0 of 11	0 of 8	0 of 9	0 of 9	5 of 10 (46)	0 of 54	0 of 9
1,4-Dichlorobenzene	0 of 7	0 of 5	0 of 6	0 of 11	0 of 8	0 of 9	0 of 9	2 of 10 (16)	0 of 54	0 of 9
Dibenzofuran	0 of 12	NA	NA	0 of 21	1 of 12 (2.2)	NA	0 of 11	NA	2 of 19 (2)	0 of 14
Pesticides										
4,4′-DDT	NA	2[de] of 12 (6.0)	NA	NA	NA	NA	NA	0 of 2	NA	NA

Table 7-42. TZW COPCs with HQs ≥ 1 in Individual Samples by Area

	Number of Samples with HQs ≥ 1 (Maximum HQ)									
		Arkema								
COPC	ARCO	Acid Plant Area	Chlorate Plant Area	Mobil Oil	Gasco	Gunderson	Kinder Morgan	Rhône-Poulenc	Siltronic	Willbridge
Total DDx	NA	4[be] of 12 (10)	NA	NA	NA	NA	NA	0 of 2	NA	NA
VOCs										
1,1-Dichloroethene	0 of 7	0 of 9	0 of 10	0 of 11	0 of 8	0 of 9	0 of 9	0 of 10	2 of 54 (1.6)	0 of 9
1,2,4-Trimethylbenzene	NA	NA	NA	NA	NA	NA	NA	NA	11 of 41 (9.6)	NA
1,3,5-Trimethylbenzene	NA	NA	NA	NA	NA	NA	NA	NA	5 of 41 (3)	NA
Benzene	0 of 7	0 of 9	0 of 10	0 of 11	3 of 8 (4.2)	0 of 9	0 of 9	0 of 10	6 of 54 (30)	0 of 9
Carbon disulfide	0 of 7	0[a, b] of 9	0 of 10	0 of 11	3 of 8 (870)	0 of 9	0 of 9	0 of 10	1[d] of 54 (1.3)	0 of 9
Chlorobenzene	0 of 7	2 of 9 (190)	0 of 10	0 of 11	0 of 8	0 of 9	0 of 9	1 of 10 (3.3)	0 of 54	0 of 9
Chloroethane	0 of 7	0 of 9	0 of 10	0 of 11	0 of 8	1 of 9 (3.4)	0 of 9	0 of 10	0 of 54	0 of 9
Chloroform	0 of 7	1 of 9 (21)	3[c] of 10 (7.9)	0 of 11	0 of 8	0 of 9	0 of 9	0 of 10	0 of 54	0 of 9
cis-1,2-Dichloroethene	0 of 7	0 of 9	0 of 10	0 of 11	0 of 8	0 of 9	0 of 9	0 of 10	5 of 54 (110)	0 of 9
Ethylbenzene	0 of 7	0 of 9	0 of 10	0 of 11	3 of 8 (11)	0 of 9	0 of 9	0 of 10	12 of 54 (57)	0 of 9
Isopropylbenzene	0 of 7	0 of 9	0 of 10	0 of 11	2 of 8 (1.5)	0 of 9	0 of 9	0 of 10	8 of 54 (2)	0 of 9
m,p-Xylene	0 of 7	0 of 9	0 of 10	0 of 11	0 of 8	0 of 9	0 of 9	0 of 10	3 of 54 (4.4)	0 of 9

Table 7-42. TZW COPCs with HQs ≥ 1 in Individual Samples by Area

	Number of Samples with HQs ≥ 1 (Maximum HQ)									
		Arkema								
COPC	ARCO	Acid Plant Area	Chlorate Plant Area	Mobil Oil	Gasco	Gunderson	Kinder Morgan	Rhône-Poulenc	Siltronic	Willbridge
o-Xylene	0 of 7	0 of 9	0 of 10	0 of 11	3 of 8 (3.6)	0 of 9	0 of 9	0 of 10	9 of 54 (12)	0 of 9
Toluene	0 of 7	0 of 9	0 of 10	0 of 11	4 of 8 (2.9)	0 of 9	0 of 9	0 of 10	7 of 54 (18)	0 of 9
Total xylenes	0 of 7	0 of 9	0 of 10	0 of 11	3 of 8 (8.5)	0 of 9	0 of 9	0 of 10	10 of 54 (34)	0 of 9
Trichloroethene	0 of 7	0 of 9	0 of 10	0 of 11	0 of 8	0 of 9	0 of 9	0 of 10	2 of 54 (1,900)	0 of 9
Petroleum Hydrocarbons[f]										
Gasoline-range aliphatic hydrocarbons C4-C6	1 of 9 (1.1)	NA	NA	3 of 15 (1.2)	5 of 10 (7.3)	NA	0 of 10	NA	6 of 15 (2.0)	0 of 9
Gasoline-range aliphatic hydrocarbons C6-C8	0 of 9	NA	NA	0 of 9	4 of 10 (4.3)	NA	0 of 10	NA	3 of 15 (1.2)	0 of 9
Gasoline-range aliphatic hydrocarbons C8-C10	0 of 9	NA	NA	0 of 9	0 of 10	NA	0 of 10	NA	0 of 15	0 of 9
Gasoline-range aliphatic hydrocarbons C10-C12	6 of 9 (35)	NA	NA	6 of 15 (85)	9 of 10 (540)	NA	3 of 10 (6.9)	NA	9 of 15 (150)	3 of 9 (3.8)
Gasoline-range aromatic hydrocarbons C8-C10	0 of 9	NA	NA	0 of 15	3 of 10 (2.7)	NA	0 of 10	NA	0 of 15	0 of 9
Other Contaminants										
Cyanide	NA	NA	NA	NA	8[b] of 8 (4,400)	NA	NA	NA	26[b] of 26 (130)	NA
Perchlorate	NA	0[a, b] of 9	5 of 10 (19)	NA	NA	0 of 2	NA	NA	NA	NA

a Only samples with non-detected concentrations have HQs ≥ 1.

b One additional sample had a DL greater than the TRV.

[c] An additional two to three samples had DLs greater than the TRV.

[d] An additional four or more non-detect samples had DLs greater than the TRV.

[e] HQ presented is based on the alternative 4,4′-DDT TRV for protection of directly exposed aquatic organisms. HQ based on unfiltered concentration; dissolved concentrations tended to be several orders of magnitude lower and most constituents of DDx were undetected.

[f] Because petroleum compounds are not CERCLA contaminants, gasoline-range hydrocarbons are not included in the final count of contaminants posing potentially unacceptable risk; they are included here because they may nonetheless contribute to risk.

COPC – contaminant of potential concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
DL – detection limit
HQ – hazard quotient
NA – not analyzed
PAH – polycyclic aromatic hydrocarbon
SVOC – semivolatile organic compound
total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT and 4,4′-DDT)
TRV – toxicity reference value
TZW – transition zone water
VOC – volatile organic compound

7.4.5.2 Evaluation of Naturally Occurring Metals

Although there are many anthropogenic sources of metals, almost all of the metals measured in TZW are also common crustal elements in sediment. Aluminum, barium, iron, and manganese are among the most common metals associated with sediments and were detected in all TZW samples. These common metals were also associated with the highest HQs identified in the risk characterization, but there is substantial uncertainty that these metals represent anthropogenic sources.

The contribution of geochemical processes in sediments to the concentrations of selected metals in TZW was extensively evaluated in Appendix C2 of the draft final RI (Integral et al. 2011). Concentrations of iron and manganese in TZW are not well-correlated to potential anthropogenic source materials (i.e., petroleum hydrocarbons), suggesting that factors other than contamination in the sediment (e.g., naturally occurring organic materials) are contributing to concentrations measured in the TZW. Geochemical processes are also likely contributing to the concentrations of barium in TZW, rather than migration of upland groundwater to the river.

Aluminum was not included in the geochemical evaluation, but a background surface water concentration (established in Section 7.0 of the draft final RI [Integral et al. 2011]) is available to provide some context for TZW (since surface water is a component of shallow TZW). An upper-bound (UPL) background concentration for aluminum was 1,485 µg/L. The majority of the TZW values were below this concentration.

7.4.5.3 Evaluation of Lamprey Toxicity Tests

Acute toxicity tests conducted with Pacific lamprey as part of the BERA can be used to further evaluate the risk characterization for lamprey (Andersen et al. 2010). Toxicity data were collected for six chemicals representing a variety of toxic modes of action (pentachlorophenol, copper, aniline, diazinon, naphthalene, and lindane). The objective of the tests was to determine whether surface water TRVs based on standard test species would be protective of Pacific lamprey (Andersen et al. 2010). The relative sensitivity of lamprey ammocoetes was evaluated using an SSD based on available toxicity data for aquatic species, including fish, invertebrates, and amphibians (Andersen et al. 2010).

When compared to other species, Pacific lamprey ammocoetes exhibited average sensitivity to copper (46th percentile), but relative insensitivity to diazinon and aniline (72nd and 90th percentile, respectively). LC50s could not be calculated for naphthalene and lindane because of low mortality at the highest test concentrations; these results indicate that lamprey ammocoetes are less sensitive than other species. Pentachlorophenol was relatively toxic to lamprey, which is not unexpected because its toxic mode of action is similar to that of trifluormethy-4-nitrophenol, a chemical commonly used to control sea lamprey(Hubert 2003).

For five of the six chemicals tested, Pacific lamprey are as sensitive as or less sensitive than other species tested. These findings can be extrapolated, with some uncertainty, to suggest that the same pattern applies to other COPCs for which there are no lamprey-specific data.

Only two contaminants included in these tests—copper and naphthalene—are COPCs for lamprey (Table 7-42). The TZW HQs for copper and naphthalene are 1.3 and 1,100, respectively. Because of its relatively high HQ, napthalene was evaluated in more detail using the Pacific lamprey toxicity data. The acute NOAEL and LOAEL for lamprey were 5,300 and 10,000 µg/L, respectively; an acute LC50 could not be calculated because of low mortality in the test (Andersen et al. 2010). A lamprey-specific chronic toxicity value was derived from the acute data using an ACR consistent with the methods used to derive ambient water quality criteria (Stephan et al. 1985). One ACR for fish was available from the Tier II document (Suter and Tsao 1996): 12.8 for fathead minnow. This ACR is consistent with a median ACR of 12.1 for nonpolar polyaromatic narcotic chemicals (including PAHs) derived from 32 values for both fish and aquatic invertebrates (Raimondo et al. 2007). With the ACR of 12.8 for fathead minnow, the chronic LOAEL for Pacific lamprey was calculated as 781 µg/L, which is greater than the surface water TRV of 12 µg/L by a factor of 65. When calculated with the lamprey-specific TRV, the HQ for naphthalene in TZW is 17, compared to the HQ of 1,100 presented in Table 7-42 (i.e., lower by a factor of 65).

The TRVs for many of the contaminants with HQs ≥ 10 were derived as Tier II values, calculated by Suter and Tsao (1996) for contaminants lacking the minimum species diversity requirements for deriving AWQC. Because each value was designed to protect a wide range of species, assumptions were made to estimate toxicity thresholds when data were available for only a small number of species. Generally, a Tier II chronic value was calculated by dividing the lowest genus mean acute value (GMAV) by two factors: one factor ranging from 3.6 to 242 to account for the narrow range of species tested and another factor to convert acute data to chronic sensitivity. For example, for naphthalene the lowest GMAV of 1,600 µg/L for rainbow trout was divided by a factor of 8.6 to account for the lack of taxa diversity and then by an ACR of 16, resulting in a chronic value of 12 µg/L. This value is greater than the Pacific lamprey-specific chronic value by a factor of 65, as noted above. Therefore, for Tier II contaminants the use of the lowest available acute value for any genus, along with a factor to account for the lack of toxicity data for a range of species, generally results in very low chronic toxicity values that are likely to overpredict risk to Pacific lamprey.

The TRVs for the remainder of the contaminants with HQs ≥ 10 were either AWQC values (zinc and cyanide) or were derived using the same methods as those used to derive AWQC (PAHs and perchlorate) (EPA 1985, 1987, 2003c; Dean et al. 2004). These values were calculated using a wide variety of species. Unlike the Tier II values, the final acute value was derived as the 5% level of all the GMAVs, and is thus is expected to be protective of 95% of the species. Based on the lamprey acute toxicity test results described above, lamprey appear to be of average or lower sensitivity compared to other species, indicating that the derivation of TRVs using the AWQC methodology is uncertain for lamprey and may overpredict risk.

7.4.5.4 COIs for Which Risks Cannot Be Quantified

COIs for which risks to benthic fish cannot be quantified from TZW data are listed in Table 7-43. These are contaminants for which no TRV is available or whose maximum DL exceeds a TRV, but whose detected concentrations do not.

Table 7-43. Fish TZW COIs with No Available TRV or with DLs Exceeding Screening-Level TRVs

COI	Rationale for Why Risks Cannot be Quantitatively Evaluated
Metals	
Aluminum	The AWQC chronic criterion for aluminum was used to identify aluminum as a COPC. However, as per agreement with EPA, the AWQC criterion is not applicable to waters with circumneutral pH, such as those in the Study Area, precluding further evaluation of aluminum.
Calcium	Risk to benthic fish based on TZW data unknown; no water threshold available.
Selenium	Risk to benthic fish based on TZW data unknown; 26% of non-detected samples exceed water threshold, but no detected concentration > screening-level TRV.
Titanium	Risk to benthic fish based on TZW data unknown; no water threshold available.
Petroleum	
Residual-range hydrocarbons	Risk to benthic fish based on TZW data unknown; no water threshold available.
Diesel-range hydrocarbons	Risk to benthic fish based on TZW data unknown; no water threshold available.
Total diesel-residual hydrocarbons	Risk to benthic fish based on TZW data unknown; no water threshold available.
Total petroleum hydrocarbons	Risk to benthic fish based on TZW data unknown; no water threshold available.
VOCs	
Styrene	Risk to benthic fish based on TZW data unknown; 1% of non-detected samples exceed water threshold, but no detected concentration > screening-level TRV

AWQC – ambient water quality criteria
COI – contaminant of interest
COPC – contaminant of potential concern
DL – detection limit
EPA – US Environmental Protection Agency
TRV – toxicity reference value
TZW – transition zone water
VOC – volatile organic compound

7.4.5.5 Summary of TZW Risk Evaluation

Fifty-three TZW COPCs with HQs ≥ 1 were identified for benthic fish (Table 7-44). [110] The relative strengths and uncertainties for all fish LOEs are evaluated together in the risk conclusions for fish, Section 7.6.

Table 7-44. TZW COPCs with HQ ≥ 1

Contaminants	
Metals	
Barium	Magnesium
Beryllium	Manganese
Cadmium	Nickel
Cobalt	Potassium
Copper	Sodium
Iron	Vanadium
Lead	Zinc
PAHs	
2-Methylnaphthalene	Chrysene
Acenaphthene	Dibenzo(a,h)anthracene
Anthracene	Fluoranthene
Benzo(a)anthracene	Fluorene
Benzo(a)pyrene	Indeno(1,2,3-cd)pyrene
Benzo(b)fluoranthene	Naphthalene
Benzo(g,h,i)perylene	Phenanthrene
Benzo(k)fluoranthene	Pyrene
SVOCs	
1,2-Dichlorobenzene	Dibenzofuran
1,4-Dichlorobenzene	
Pesticides	
4,4′-DDT	Total DDx
VOCs	
1,1-Dichloroethene	cis-1,2-Dichloroethene
1,2,4-Trimethylbenzene	Ethylbenzene
1,3,5-Trimethylbenzene	Isopropylbenzene
Benzene	Toluene

[110] Fifty-four TZW COPCs with HQs ≥ 1 were identified for benthic fish. Because petroleum compounds are not CERCLA contaminants, gasoline-range hydrocarbons were not included in the final count of contaminants posing potentially unacceptable risk; they are included here because they may nonetheless contribute to risk.

Table 7-44. TZW COPCs with HQ ≥ 1

Contaminants	
Carbon disulfide	Trichloroethene
Chlorobenzene	m,p-Xylene
Chloroethane	o-Xylene
Chloroform	Total xylenes
TPH	
Gasoline-range hydrocarbons[a b]	
Other Contaminants	
Cyanide	Perchlorate

[a] The COPC gasoline-range hydrocarbons was evaluated as five components (aliphatic hydrocarbons C4-C6, aliphatic hydrocarbons C6-C8, aliphatic hydrocarbons C8-C10, aliphatic hydrocarbons C10-C12, and aromatic hydrocarbons C8-C10). Gasoline-range hydrocarbons was identified as a COPC if any one of the five gasoline components exceeded its TRV.

[b] Because petroleum compounds are not CERCLA contaminants, gasoline-range hydrocarbons were not included in the final count of contaminants posing potentially unacceptable risk; they are included here because they may nonetheless contribute to risk.

CERCLA – Comprehensive Environmental Response, Compensation, and Liability Act
COPC – contaminant of potential concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
PAH – polycyclic aromatic hydrocarbon
SVOC – semivolatile organic compound
total DDx – sum of all six DDT isomers (2,4′-DDD; 4,4′-DDD; 2,4′-DDE; 4,4′-DDE; 2,4′-DDT; and 4,4′-DDT)
TPH – total petroleum hydrocarbons
TZW – transition zone water
VOC – volatile organic compound

7.5 ASSESSMENT OF BENTHIC FISH HEALTH AND PAH EXPOSURE

Per EPA's Problem Formulation (Attachment 2), an additional semi-quantitative LOE was evaluated for the exposure of benthic fish to PAHs. This assessment involved comparing Study Area sediment PAH concentrations to literature-derived PAH concentrations associated with the occurrence of skin or liver lesions. A comparison of fish health observations recorded in the field for Round 3 juvenile white sturgeon collected from the Study Area to fish health observation data reported in the greater Columbia River region is also presented. Qualitative fish health observation data were not available for other fish species (e.g., carp, sculpin, smallmouth bass).

Benthic fish are most likely exposed to PAHs through direct contact with bottom sediments and sediments that are suspended in water, and through dietary uptake (Johnson et al. 2002). Fish exposed to PAH-contaminated sediments through direct contact have been shown to have increased incidence of skin and liver lesions as well as other deformities (Myers et al. 1994; Pinkney et al. 2000). In addition, reduced lifespan in fish has been linked to cancerous lesions (Johnson et al. 2002; Baumann et al. 1987; Pinkney et al. 2000; Myers et al. 1994). The prevalence of both hepatic (i.e., liver) and epidermal (i.e., skin) lesions can be used as a criterion for identifying contaminated sites (Pinkney et al. 2004a).

The details of this assessment are presented in four subsections:

- Section 7.5.1 presents a review of the toxicological literature in which PAH concentrations in sediment have been detected in areas where lesion incidence has been observed in benthic fish.
- Section 7.5.2 presents a comparison of the Study Area sediment PAH concentrations to literature-based thresholds associated with lesion occurrence.
- Section 7.5.3 presents a summary of Study Area fish health field observations and compares these qualitative data to fish health observations made for Columbia River region fish.
- Section 7.5.4 presents the conclusions for this LOE.

The flowchart in Figure 7-15 shows the organization of the fish health and PAH exposure LOE.

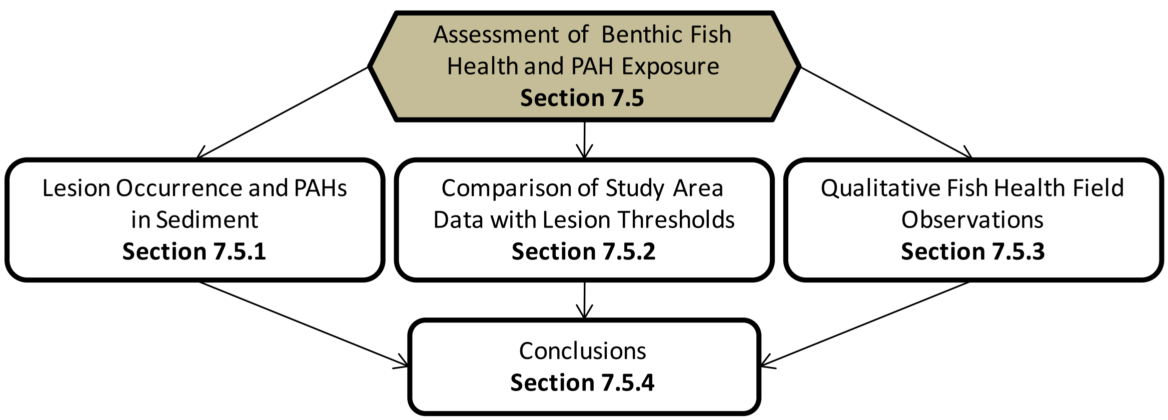

Figure 7-15. Overview of Assessment of Benthic Fish Health and PAH Exposure Section Organization

7.5.1 Lesion Occurrence and PAHs in Sediment

This section discusses the toxicological studies that examined PAH exposure and an increase in the incidence of lesions. Section 7.5.1.1 summarizes studies of external lesions and other deformities; Section 7.5.1.2 summarizes studies of hepatic lesions. The implications of lesion prevalence for population-level effects are discussed in Section 7.5.1.3. Section 7.5.1.4 discusses other factors that may contribute to lesions, and Section 7.5.1.5 summarizes the toxicological data relating sediment PAHs to lesions.

7.5.1.1 Toxicological Studies on External Lesions and Other Deformities

External lesions observed in fish exposed to PAHs in sediment include epidermal papillomas, mucoid plaques, and lip papillomas, as well as epidermal ulcerations/abrasions and fin lesions (Baumann et al. 1996; Mezin and Hale 2000). In one toxicological study (Mezin and Hale 2000), one group of mummichogs was exposed to PAH-contaminated sediment from the Elizabeth River Superfund site, and another group was exposed to uncontaminated sediment for 13 days. Ninety-four percent of the lesions observed on test fish were in the PAH-exposed group. Of these, 74% of the lesions were fin erosions; others manifested as epidermal ulcerations and abrasions, primarily in the anal region (Mezin and Hale 2000). In some cases, lesions were so severe that internal organs were exposed. Although specific sediment PAH concentrations were not reported in this study, PAH concentrations in Elizabeth River sediments have been reported at levels up to 170,000 µg/kg (Vogelbein and Unger 2003).

In another toxicological field study, Pinkney et al. (2004a) surveyed brown bullhead from the Anacostia River, which had total PAH concentrations in sediment that ranged from 15,200 to 30,900 µg/kg.[111] Tumor prevalence in Anacostia fish was compared with tumor prevalence in fish from an uncontaminated control site with an average total PAH[112] concentration in sediment of 190 µg/kg. Skin lesion prevalence in fish from the Anacostia site ranged from 13 to 23% in large brown bullhead versus 0% in fish from the reference site. Pinkney et al. (2004a, b) linked the prevalence of skin tumors to PAH biomarkers in bullhead from three PAH-contaminated Chesapeake Bay rivers. A significantly higher prevalence of skin tumors was observed in fish from the contaminated river sites than in fish from the reference site. The average total PAH concentration in sediment at the reference site was 187 µg/kg; total PAH concentrations in sediment from the Chesapeake Bay rivers were ranged from 6,480 to 6,750 µg/kg.

7.5.1.2 Toxicological Studies on the Prevalence of Hepatic Lesions

The link between PAH exposure and hepatic lesions is well-documented. A statistical study of eight investigations that explored the link between PAH contaminants in sediment and hepatic and kidney lesions in fish reported that overall hepatic lesion prevalence ranged from 4 to 16%, with the highest prevalence occurring at Eagle Harbor, Washington, in the Puget Sound (Landahl et al. 1990). The PAH concentration in sediment at Eagle Harbor was 540,000 µg/kg dw, higher than at any other location in the other studies. When six of the eight studies were considered as a whole, Landahl et al. (1990) found consistent relationships between PAH sediment concentrations and the development of hepatic lesions in four of the five hepatic lesion categories considered in the study.

When PAHs are metabolized in the livers of fish, metabolites are produced. Some of these metabolites are carcinogenic, mutagenic, or cytotoxic and are thought to be linked to hepatic lesion development (Johnson 2000; as cited in Malins et al. 2006; Myers et

[111] Sediment PCBs and chlordane concentrations, which were also elevated within the study area, were not reported.

[112] Total PAHs are the sum of the following individual analytes: acenaphthylene, acenaphthene, fluorene, phenanthrene, anthracene, 2-methylnaphthalene, fluoranthene, pyrene, benzo(a)anthracene, chrysene, total benzofluoranthenes, benzo(a)pyrene, indeno(1,2,3-cd)pyrene, dibenzo(a,h)anthracene, and benzo(g,h,i)perylene.

al. 1994; Pinkney et al. 2004a). Several studies that established a relationship between elevated sediment PAH concentrations and liver lesions were used to conduct a hockey stick regression[113] to determine sediment PAH effects concentration thresholds for several lesion types (Johnson et al. 2002). Lesion classes included neoplasms (i.e., tumors) and a category of "any lesions." The sediment total PAH concentration threshold for effects (and confidence limits) was calculated as 2,800 µg/kg (11 to 5,500 µg/kg)[114] for neoplasms and 620 µg/kg (300 to 1,000 µg/kg) for the "any lesions" category (Johnson et al. 2002).

Stern et al. (2003) conducted a separate hockey stick regression analysis of PAH concentrations in Puget Sound sediment using the "any lesion" category as defined in Johnson et al. (2002) and lesion data from that study as well as other lesion data. In this analysis, concentrations in sediment were characterized as the total PAH spatially weighted average concentration (SWAC)[115] over an assumed English sole foraging radius of 1 km. The resulting sediment total PAH effects concentration threshold for the "any lesion" category (and confidence limits) was 2,731 µg/kg (1,410 to 3,772 µg/kg).

7.5.1.3 Implications of Lesion Prevalence for Population-Level Effects

Although several studies have linked sediment PAH exposure to lesion prevalence, few have linked lesion prevalence to adverse effects at the population level. Several brown bullhead studies conducted on the Black River in Ohio reported liver histopathology data that shows a link between sediment PAH concentrations, liver lesions, and population age structure (Baumann 2000). In the early 1980s, PAH concentrations in Black River sediments were as high as several hundred parts per million (Baumann 2000, citing Baumann et al. 1982), largely the result of discharge from a steel and coke plant. Under these conditions, brown bullhead in the Study Area showed a high prevalence of liver lesions, with less than 20% having completely normal livers. A truncated age structure, whereby few individuals in the population survived beyond 4 years of age, was also observed. Baumann et al. (1987) reported that liver tumor prevalence increased significantly with fish age and attributed the truncated age structure to a reduced lifespan resulting from cancerous lesions.

Surveys conducted subsequent to the plant's closure in 1982 showed that tumor frequency in 3-year-old and older fish declined by approximately 50%, and the incidence of cancer was reduced to approximately 25% of earlier levels (Baumann 2000, citing Baumann and Harshbarger 1995). In addition, the age structure of the fish population shifted, with more 5-year-old fish captured and 6-year-old fish showing up in the surveys for the first time. After remedial dredging and a recovery period,

[113] A hockey stick regression is a dose-response type model that assumes a threshold must be reached before initiation of a response.

[114] Total PAHs were the sum of 10 LPAHs and 8 HPAHs.

[115] Based on the sum of 10 HPAHs.

sediment sampling in the Black River in 1997 and 1998 showed that nearly all PAH concentrations in sediment were down to 15,000 µg/kg or less. PAH concentrations in fish tissue were also significantly lower than earlier levels. In 1998, cancer prevalence in 5-year-old and older fish was 7%, and the age structure of the population more closely resembled that of populations in pristine sites, with over 60% of the fish population surveyed more than 5 years old and over 35% more than 6 years old (Baumann 2000). This analysis suggests that the population-level effects threshold for total PAHs may be greater than 15,000 µg/kg.

In a study of English sole, Johnson and Landahl (1994) examined the relationship between lesion prevalence and population-level effects by comparing estimated annual mortality rates at both highly contaminated (e.g., Eagle Harbor) and uncontaminated sites throughout Puget Sound. English sole mortality rates from contaminated sites associated with high liver lesion prevalence were not found to be significantly greater than mortality rates for English sole from Puget Sound as a whole. The investigators also examined the English sole population structure and found no evidence of increased age-related mortality in fish with lesions or in populations associated with areas of high levels of PAHs and PCBs. The authors concluded that fish populations that have high incidence of lesions do not necessarily have increased mortality. Other factors that affect English sole populations, such as fishing pressure, predation, and fluctuations in food supply, may mask population-level effects associated with chemical contamination and lesion incidence. Thus, this study (Johnson and Landahl 1994) did not identify a link between lesion prevalence and population structure in areas with widely varying ranges of PAH concentrations in sediment.

7.5.1.4 Other Factors Contributing to Lesion Prevalence

Several factors other than sediment PAH concentrations (e.g., age, sex, non-chemical stressors) have been shown to be significant risk factors for lesion prevalence in benthic fish (Baumann et al. 1987; Myers et al. 1994; Pinkney et al. 2000, 2004a). Age has been identified as an important risk factor for lesion development in brown bullhead, with the incidence of skin tumors increasing 2.5 times per year and the incidence of liver hepatocarcinoma increasing 3.5 times per year as fish age (Pinkney et al. 2000). Additional studies support the finding that age plays a role in lesion development in benthic fish (Baumann et al. 1987; Myers et al. 1994). The odds of developing hepatocarcinoma in brown bullhead have also been linked to sex, with a higher incidence (by a factor of 4.5) of these lesions observed in females (Pinkney et al. 2000). The accumulation of bile PAH metabolites has been identified as a risk factor for lesions, as well; for every 100 mg/kg increase in metabolite concentration, tumor prevalence was observed to increase by a factor of 1.1 to 1.8 (Pinkney et al. 2000). Additional factors including viruses, crowding, temperature change, and other biotic and abiotic factors may contribute to epidermal lesion development as environmental stressors suppress fish immune systems (Baumann et al. 1996).

Contaminants other than PAHs have been identified as risk factors for lesion development in benthic fish (Myers et al. 1994). PCB and DDT concentrations in sediment and liver tissues were found to be significant risk factors for neoplasms and pre-neoplastic lesions by Myers et al. (1994) and others (Stern et al. 2003, citing O'Niell et al. 1999). The role of PCBs in initiating neoplasms in fish is not well-understood, and it is not clear if PCBs alone can induce lesions in wild fish. Although Myers et al. (1994) reported PAH exposure as the most frequently identified risk factor, PCBs, DDTs, and other contaminants were thought to be toxicologically relevant risk factors in the etiology of hepatic lesions. Because PAHs, PCBs, DDTs, and other contaminants typically coexist in contaminated sediments, it is difficult to quantify their relative contributions to the development of liver lesions in benthic fish (Myers et al. 1994).

7.5.1.5 Summary of Toxicological Studies

Several studies have proposed hepatic lesion sediment effects thresholds for PAH ranging over an order of magnitude (230 to 4,000 µg/kg) (Horness et al. 1998; Johnson et al. 2002; Stern et al. 2003). Johnson et al. (2002) suggested a 1,000 µg/kg PAH threshold as being both protective of the majority of fish species and practical for making management decisions based on the “any lesions” category. Johnson et al. (2002) also reported a threshold of 2,800 µg/kg for neoplasms (benign or cancerous). Stern et al. (2003) reported an effects threshold of 2,700 µg/kg for the “any lesions” category and stated that a hockey stick regression using methods that account for the size of English sole’s foraging range would likely result in an even higher threshold for neoplasms. Pre- and post-dredge data from Eagle Harbor indicate that English sole hepatic lesion incidence was approximately equal to background levels at a SWAC in sediment of 4,000 µg/kg (Stern et al. 2003). Pre- and post-dredge data from the Black River indicate that brown bullhead lesion incidence and population structure were approximately equal to background levels at an area-weighted average sediment concentration of 15,000 µg/kg (Baumann 2000).

7.5.2 Comparison of Study Area Data with Lesion Thresholds

Study Area PAH concentrations in sediment were compared with the PAH concentrations associated with lesion occurrence in field studies. Total PAH[116] concentrations in surface sediment at the Study Area ranged from 6.3 to 7,300,000 µg/kg (n = 1,406). The site-wide UCL concentration of total PAH in sediment is 69,800 µg/kg. The UCL concentration is above the effects thresholds reported by Johnson et al. (2002) and Stern et al. (2003), ranging from 230 to 4,000 µg/kg. However, these toxicological threshold concentrations reported in the literature have not been linked to population-level effects.

The link between lesions and effects on fish populations have been demonstrated for brown bullhead from the Black River. However, the effects of specific classes of lesions (either cancerous or benign) and potentially pre-neoplastic lesions at the population

[116] Total PAHs are the sum of concentrations for the following chemicals: acenaphthylene, acenaphthene, anthracene, benzo(a)anthracene, total benzofluoranthenes, benzo(a)pyrene, benzo(g,h,i)perylene, chrysene, dibenzo(a,h)anthracene, fluoranthene, fluorene, indeno(1,2,3,-c,d)pyrene, naphthalene, phenanthrene, and pyrene.

level are uncertain, and the appropriate estimation method for sediment thresholds for this or other endpoints is still under refinement. Total PAH concentrations in Study Area sediments are greater than post-remediation concentrations reported in the Black River (15,000 µg/kg) that have been associated with recovered population structure in brown bullhead; however, because pre-remediation concentrations in the Black River greatly exceeded those in the Study Area, it is not known if population effects may occur in the Study Area.

A high degree of uncertainty is associated with literature-based PAH thresholds for fish lesion incidence for three reasons: 1) uncertainties are associated with the determination of the appropriate spatial scale of sediment PAH exposure for specific benthic fish (e.g., whether sediment PAH concentrations should be averaged across the area where fish are assumed to forage and be exposed to PAHs); 2) there are potential confounding effects of co-occurring contaminants (Myers et al. 1994); and 3) the sediment PAH effects threshold for occurrence of lesions in fish and associated reductions in survival, growth, or reproduction (and effects on the population); has not been demonstrated in field studies. Furthermore, the PAHs in sediment from areas associated with certain industrial activities are less bioavailable and therefore cause lower frequencies of liver lesions and DNA damage in sole (Johnson et al. 2009). Because the relative bioavailability of PAHs in Study Area sediments and those in the sediments of other studies is not known, the comparison of Study Area sediment concentrations to literature-based PAH thresholds is uncertain. Because of the combined uncertainties of this LOE, fish exposure to PAHs in sediment was not further evaluated quantitatively.

7.5.3 Qualitative Fish Health Field Observations

While the link between exposure to PAHs in Study Area sediment and population-level effects on benthic fish is not conclusive (Section 7.5.2), qualitative data on overall fish health and incidence of abnormalities were recorded from selected benthic fish collected in the Study Area.

A total of 165[117] juvenile (pre-breeding) sturgeon collected from the Study Area during Round 3 were examined visually for gross external abnormalities based on the data collection procedures outlined by USFWS(2007) and the USGS Biomonitoring of Environmental Status and Trends (BEST) protocol (Schmitt and Dethloff 2000). The USGS BEST protocol for examining fish provides generic identification of observable conditions, but it does not include specific diagnoses of fish health (USGS 2002). A formal diagnosis can only be determined through histopathology and other laboratory expertise.

Following the BEST protocol (Schmitt and Dethloff 2000), gross external abnormality data were collected as part of the Round 3 juvenile white sturgeon sampling event for

[117] Fish health observations were made for 150 juvenile sturgeon that were caught and released and for 15 legal-size-range (42 to 60 inches) juvenile sturgeon that were kept for tissue analysis.

the purpose of providing general information on sturgeon health. Fifty-five percent of juvenile sturgeon examined during the Round 3 sampling had external anomalies of the body, head, eyes, opercles, or gills. Consistent with EPA's Problem Formulation (Attachment 2), this percentage of anomalies does not include fin or other damage (such as recent body lesions, evidence of hook damage, fin damage, and hemorrhagic barbels) likely to be caused by catching, processing, or holding the fish.

These Study Area field observations were compared with Columbia River Basin data collected as part of the USGS BEST program. The Columbia River Basin BEST data were based on observations of external and internal lesions of seven fish species (i.e., carp, bass, largescale sucker, northern pikeminnow, longnose sucker, walleye, and rainbow trout) collected from 16 sampling locations in the Columbia River Basin from September 1997 to April 1998 (Hinck et al. 2004). Two of the sampling locations were located in the Willamette River: one sampling location at RM 10 (in the Study Area), and one sampling location at RM 30 (above the Study Area). Of all the fish collected in the Columbia River Basin study, between 25 and 46% were found to have external non-fin-related gross lesions (abnormalities) on the body, eyes, or opercles.[118]

The incidence of non-fin abnormality occurrence in Study Area sturgeon (55%) was slightly greater than the incidence of fish abnormalities observed in Columbia River Basin fish (25 to 46%). However, as noted in Hinck et al. (2004), the nature and magnitude of abnormality required for a specimen to be considered as having a recordable abnormality is variable depending on the study and researchers within a study; therefore, comparisons between studies should be viewed as highly uncertain. In addition, while certain lesions were identified as an incidental effect of fish holding and handling, lesion occurrence may also be the result of normal wear as a fish ages (Hinck et al. 2004). It may not be appropriate to compare the fish health observations compiled for juvenile white sturgeon to observations made for various other fish species. The average age of sturgeon collected from the Study Area during Round 3 was 13 years old (age ranged from 7 to 26 years),[119] and these sturgeon may be older than the fish collected from the Columbia River Basin with lower ages of sexual maturity. The home range of juvenile white sturgeon adds additional uncertainty to the evaluation of how Study Area contaminants may affect the occurrence of sturgeon lesions because juvenile white sturgeon are known to have a large home range (e.g., one PIT-tagged sturgeon collected from the Study Area was 72 miles from its initial tagging location), and exposure to contaminants and other factors outside of the Study Area may affect overall sturgeon health and body condition.

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[118] Fin-related lesions were not included in this estimate because most fin lesions were thought to have occurred as a result of fish collection and handling.

[119] Age analysis of juvenile sturgeon was determined by Ruth Farr and Michele Weaver at ODFW using pectoral fin ray samples following ODFW protocols (Beamesderfer et al. 1998).

In conclusion, while these qualitative data may suggest that benthic fish in the Study Area have been exposed to PAH-contaminated sediment, they are not conclusive with respect to population-level effects due to this exposure. Health observations for Study Area juvenile white sturgeon are not directly comparable to those for eight other fish species collected from the Columbia River Basin. The incidence of abnormalities in fish is nearly impossible to attribute to a single factor and is likely to result from confounding factors, including species, age, disease, organic matter, temperature, nutrition, season, and geographic location in addition to contaminants and catch methods (Adams et al. 1996). Because of the highly qualitative nature of the field health observations and the uncertainties associated with their interpretation, a conclusive link cannot be established between the field observations made on Study Area fish and overall status of the populations.

7.5.4 Conclusions

The evaluation of benthic fish health, through the incidence of lesions and abnormalities from excessive exposure to PAHs, is largely inconclusive and involves several uncertainties. The sediment concentrations established in the literature for lesion occurrence are highly variable and span over an order of magnitude. The link between lesion occurrence and sediment exposure is confounded by co-occurring contaminants (not just PAHs) at study sites as well as difficulties in establishing a conclusive link between population-level effects of survival, growth, or reproduction and the occurrence of lesions in fish. In addition, effect-level PAH concentrations in sediments have not been demonstrated in field studies. Fish condition data compiled for juvenile white sturgeon indicate that the incidence of abnormalities in the Study Area is somewhat higher than that reported for other fish species in the region; however the differences may be attributed to such factors as the difference of species, age of fish collected, fish handling methods, and the field researchers' qualitative assessments of abnormalities.

In summary, although the presence of abnormalities in benthic fish may indicate exposure to PAH-contaminated sediment, the data are not sufficient to conclude that benthic fish are experiencing population-level effects due toPAH exposure.

7.6 RISK CONCLUSIONS

This section presents the overall conclusions of the fish risk characterization. The fish risk assessment follows the conventional practice in ecological risk assessment of using organism-level TRVs defined in this manner to evaluate the potential for effects on populations. No explicit fish population modeling was included in this BERA. Rather, the BERA assesses whether COPCs occur at concentrations that have been shown to affect the survival, growth, or reproduction of aquatic organisms in the laboratory. If so, then the COPC is identified as posing potential risk to fish populations, triggering a semi-quantitative risk characterization that considers the spatial extent and magnitude of organism-level TRV exceedances, and the quality and relevance of the organism-level TRV as a predictor of population- or community-level effects. Background concentrations were considered, as appropriate, to examine whether calculated risks could be due to a localized source of contamination. Background concentrations were not, however, "subtracted out" or otherwise used to discount ecological risks. As per EPA ERAGs (EPA 1997), the risk conclusions identify the COPCs representing the majority of risk to fish assessment endpoints.

Through the BERA's risk characterization, several COPCs have, been identified as having HQs ≥ 1 by at least one LOE. These COPCs are identified and discussed below. All COPC-assessment endpoint pairs with an HQ ≥ 1 by at least one LOE were evaluated further. Consideration of uncertainties in the exposure and effects data, spatial extent of HQs ≥ 1, magnitude of HQs, level of effect represented by the TRV, and WOE (e.g., consistency across multiple lines of evidence) all were considered in characterizing ecological risk. For example, a COPC with a limited spatial distribution of HQs ≥ 1, low HQs, and a TRV based on a low level of effect (e.g., a small percent decrease in growth) is less likely to pose a risk to Study Area populations than a COPC with a broader distribution of HQs ≥ 1, high HQs, and a TRV based on high effect levels (e.g., a large increase in mortality).

Section 7.6.1 summarizes the fish COPCs with HQs ≥ 1 for each LOE. Section 7.6.2 further evaluates those COPC-assessment endpoint pairs. Section 7.6.3 presents the risk conclusions for all fish COPCs over all LOEs. In Section 11, the fish risk conclusions are combined with the risk conclusions for other ecological receptor groups to provide a holistic view of ecological risks, highlighting the COPCs posing the primary potential ecological risk across all assessment endpoints.

7.6.1 COPCs with HQs ≥ 1

Table 7-45 presents a summary of the COPCs, by LOE and receptor, resulting in HQs ≥ 1 at the final risk characterization step. Fifty-nine COPCs were identified as posing potentially unacceptable risk to at least one fish receptor based on the tissue, dietary, surface water, and TZW LOEs.[120] Of these, 44 COPCs have HQs ≥ 1 only for the TZW LOE. The spatial extent, magnitude, and potential ecological significance of TRV exceedances, and the concordance among LOEs for receptor-COPC pairs posing potentially unacceptable risk are discussed further below to determine risk conclusions.

Risk to fish from several COIs could not be evaluated because no screening-level TRVs were available or because DLs exceed the TRV: 17 tissue-residue COIs (Table 7-13), 11 dietary COIs (Table 7-16), 5 surface water COIs (Table 7-40) and 9 TZW COIs (Table 7-43).

[120] Fifty-four TZW COPCs with HQs ≥ 1 were identified for benthic fish. Because petroleum compounds are not CERCLA contaminants, gasoline-range hydrocarbons were not included in the final count of contaminants posing potentially unacceptable risk; they are included here because they may nonethelesss contribute to risk.

Table 7-45. Summary of Fish COPCs for Each LOE

COPC	Line of Evidence Resulting in HQ ≥ 1 [a] (Maximum HQ)					Receptor[b]
	Tissue-Residue	Diet	Surface Water	TZW	Fish Condition	
Metals						
Antimony	HQ ≥ 1 (5.4)	NE[c]	Not a COPC	Not a COI	Not a COI	Smallmouth bass
Barium	Not a COI	Not a COI	Not a COI	HQ ≥ 1 (1,100)	Not a COI	Sculpin, Pacific lamprey ammocoetes
Beryllium	Not a COI	Not a COI	Not a COI	HQ ≥ 1 (2)	Not a COI	Sculpin, Pacific lamprey ammocoetes
Cadmium	Not a COI	HQ ≥ 1 (4.2)	Not a COPC	HQ ≥ 1 (5.8)	Not a COI	Sculpin, juvenile Chinook salmon; Pacific lamprey ammocoetes
Cobalt	Not a COI	Not a COI	Not a COI	HQ ≥ 1 (3.6)	Not a COI	Sculpin, Pacific lamprey ammocoetes
Copper	HQ ≥ 1 (2.3)	HQ ≥ 1 (5.3)	Not a COPC	HQ ≥ 1 (1.3)	Not a COI	Sculpin, juvenile Chinook salmon, northern pikeminnow, Pacific lamprey ammocoetes, largescale sucker, juvenile white sturgeon, peamouth
Iron	Not a COI	Not a COI	Not a COI	HQ ≥ 1 (250)	Not a COI	Sculpin, Pacific lamprey ammocoetes
Lead	HQ ≥ 1 (280)	Not a COPC	Not a COPC	HQ ≥ 1 (3)	Not a COI	Sculpin, Pacific lamprey ammocoetes, Peamouth, smallmouth bass
Magnesium	Not a COI	Not a COI	Not a COI	HQ ≥ 1 (7)	Not a COI	Sculpin, Pacific lamprey ammocoetes
Manganese	NE[c]	NE[c]	Not a COI	HQ ≥ 1 (550)	Not a COI	Sculpin, Pacific lamprey ammocoetes
Mercury	Not a COPC	HQ ≥ 1 (4.5)	Not a COPC	Not a COI	Not a COI	Sculpin
Nickel	Not a COPC	NE[c]	Not a COPC	HQ ≥ 1 (1.6)	Not a COI	Sculpin, Pacific lamprey ammocoetes
Potassium	Not a COI	Not a COI	Not a COI	HQ ≥ 1 (3.7)	Not a COI	Sculpin, Pacific lamprey ammocoetes
Sodium	Not a COI	Not a COI	Not a COI	HQ ≥ 1 (55)	Not a COI	Sculpin, Pacific lamprey ammocoetes
Vanadium	Not a COI	Not a COI	Not a COI	HQ ≥ 1 (19)	Not a COI	Sculpin, Pacific lamprey ammocoetes
Zinc	HQ<1	Not a COPC	HQ ≥ 1 (1.2)	HQ ≥ 1 (14)	Not a COI	Sculpin, juvenile Chinook salmon, Pacific lamprey ammocoetes, smallmouth bass, northern pikeminnow

Table 7-45. Summary of Fish COPCs for Each LOE

COPC	Line of Evidence Resulting in HQ ≥ 1 [a] (Maximum HQ)					Receptor[b]
	Tissue-Residue	Diet	Surface Water	TZW	Fish Condition	
Butyltins						
Monobutyltin	NE[c]	NE[d]	HQ ≥ 1 (1.2)	Not a COI	Not a COI	Sculpin, smallmouth bass, northern pikeminnow
TBT	Not a COPC	HQ ≥ 1 (1.0[e])	Not a COPC	Not a COI	Not a COI	Sculpin
PAHs						
2-Methylnaphthalene	Not a COI	NE[f]	Not a COPC	HQ ≥ 1 (40)	Not a COI	Sculpin, Pacific lamprey ammocoetes
Acenaphthene	Not a COI	NE[g]	Not a COPC	HQ ≥ 1 (17)	Not a COI	Sculpin, Pacific lamprey ammocoetes
Anthracene	Not a COI	NE[g]	Not a COPC	HQ ≥ 1 (87)	Not a COI	Sculpin, Pacific lamprey ammocoetes
Benzo(a)anthracene	Not a COI	NE[g]	HQ ≥ 1 (10)	HQ ≥ 1 (1,200)	Not a COI	Sculpin, Pacific lamprey ammocoetes, smallmouth bass, northern pikeminnow
Benzo(a)pyrene	Not a COI	HQ<1	HQ ≥ 1 (14)	HQ ≥ 1 (2,700)	Not a COI	Sculpin, Pacific lamprey ammocoetes, smallmouth bass, northern pikeminnow, juvenile white sturgeon
Benzo(b)fluoranthene	Not a COI	NE[g]	Not a COPC	HQ ≥ 1 (49)	Not a COI	Sculpin, Pacific lamprey ammocoetes
Benzo(g,h,i)perylene	Not a COI	NE[g]	Not a COPC	HQ ≥ 1 (66)	Not a COI	Sculpin, Pacific lamprey ammocoetes
Benzo(k)fluoranthene	Not a COI	NE[g]	Not a COPC	HQ ≥ 1 (14)	Not a COI	Sculpin, Pacific lamprey ammocoetes
Chrysene	Not a COI	NE[g]	Not a COPC	HQ ≥ 1 (17)	Not a COI	Sculpin, Pacific lamprey ammocoetes
Dibenzo(a,h)anthracene	Not a COI	NE[g]	Not a COPC	HQ ≥ 1 (13)	Not a COI	Sculpin, Pacific lamprey ammocoetes
Fluoranthene	Not a COI	NE[g]	Not a COPC	HQ ≥ 1 (17)	Not a COI	Sculpin, Pacific lamprey ammocoetes
Fluorene	Not a COI	NE[g]	Not a COPC	HQ ≥ 1 (28)	Not a COI	Sculpin, Pacific lamprey ammocoetes
Ideno(1,2,3-cd)pyrene	Not a COI	NE[g]	Not a COPC	HQ ≥ 1 (61)	Not a COI	Sculpin, Pacific lamprey ammocoetes
Naphthalene	Not a COI	NE[g]	HQ ≥ 1 (50)	HQ ≥ 1 (1,100)	Not a COI	Sculpin, Pacific lamprey ammocoetes, smallmouth bass, northern pikeminnow.

Table 7-45. Summary of Fish COPCs for Each LOE

COPC	Line of Evidence Resulting in HQ ≥ 1 [a] (Maximum HQ)					Receptor[b]
	Tissue-Residue	Diet	Surface Water	TZW	Fish Condition	
Phenanthrene	Not a COI	NE[g]	Not a COPC	HQ ≥ 1 (57)	Not a COI	Sculpin, Pacific lamprey ammocoetes
Pyrene	Not a COI	NE[g]	Not a COI	HQ ≥ 1 (15)	Not a COI	Sculpin, Pacific lamprey ammocoetes
Total PAHs	Not a COI	HQ<1	NE[h]	NE[h]	Inconclusive	Benthic fish
Phthalates						
BEHP	NE[i]	Not a COI	HQ ≥ 1 (2.3)	Not a COI	Not a COI	Sculpin, smallmouth bass, northern pikeminnow
SVOCs						
1,2-Dichlorobenzene	Not a COI	Not a COI	Not a COI	HQ ≥ 1 (46)	Not a COI	Sculpin, Pacific lamprey ammocoetes
1,4-Dichlorobenzene	Not a COI	Not a COI	Not a COPC	HQ ≥ 1 (16)	Not a COI	Sculpin, Pacific lamprey ammocoetes
Dibenzofuran	Not a COI	Not a COI	Not a COPC	HQ ≥ 1 (2.2)	Not a COI	Sculpin, Pacific lamprey ammocoetes
PCBs						
Total PCBs	HQ ≥ 1 (9.4[j])	Not a COI	HQ<1[k]	Not a COI	Not a COI	Largescale sucker, sculpin, smallmouth bass, northern pikeminnow
Pesticides						
4,4′-DDT[l]	NE[m]	Not a COI	HQ < 1[n]	HQ ≥ 1 (160)	Not a COI	Sculpin, Pacific lamprey ammocoetes
Total DDx[l]	HQ ≥ 1 (1.9[j])	Not a COI	HQ ≥ 1[n] (1.8)	HQ ≥ 1 (280)	Not a COI	Sculpin, Pacific lamprey ammocoetes
VOCs						
1,1-Dichloroethene	Not a COI	Not a COI	Not a COPC	HQ ≥ 1 (1.6)	Not a COI	Sculpin, Pacific lamprey ammocoetes
cis-1,2-Dichloroethene	Not a COI	Not a COI	Not a COPC	HQ ≥ 1 (110)	Not a COI	Sculpin, Pacific lamprey ammocoetes
1,2,4-Trimethylbenzene	Not a COI	Not a COI	Not a COPC	HQ ≥ 1 (9.6)	Not a COI	Sculpin, Pacific lamprey ammocoetes
1,3,5-Trimethylbenzene	Not a COI	Not a COI	Not a COPC	HQ ≥ 1 (3)	Not a COI	Sculpin, Pacific lamprey ammocoetes

Table 7-45. Summary of Fish COPCs for Each LOE

	Line of Evidence Resulting in HQ ≥ 1 [a] (Maximum HQ)					
COPC	**Tissue-Residue**	**Diet**	**Surface Water**	**TZW**	**Fish Condition**	**Receptor[b]**
Benzene	Not a COI	Not a COI	Not a COPC	HQ ≥ 1 (30)	Not a COI	Sculpin, Pacific lamprey ammocoetes
Carbon disulfide	Not a COI	Not a COI	Not a COI	HQ ≥ 1 (870)	Not a COI	Sculpin, Pacific lamprey ammocoetes
Chlorobenzene	Not a COI	Not a COI	Not a COI	HQ ≥ 1 (190)	Not a COI	Sculpin, Pacific lamprey ammocoetes
Chloroethane	Not a COI	Not a COI	Not a COI	HQ ≥ 1 (3.4)	Not a COI	Sculpin, Pacific lamprey ammocoetes
Chloroform	Not a COI	Not a COI	Not a COI	HQ ≥ 1 (21)	Not a COI	Sculpin, Pacific lamprey ammocoetes
Ethylbenzene	Not a COI	Not a COI	HQ<1	HQ ≥ 1 (57)	Not a COI	Sculpin, Pacific lamprey ammocoetes
Isopropylbenzene	Not a COI	Not a COI	Not a COI	HQ ≥ 1 (2)	Not a COI	Sculpin, Pacific lamprey ammocoetes
Toluene	Not a COI	Not a COI	Not a COPC	HQ ≥ 1 (18)	Not a COI	Sculpin, Pacific lamprey ammocoetes
Trichloroethene	Not a COI	Not a COI	HQ ≥ 1 (4.1)	HQ ≥ 1 (1,900)	Not a COI	Sculpin, Pacific lamprey ammocoetes
m,p-Xylene	Not a COI	Not a COI	Not a COPC	HQ ≥ 1 (4.4)	Not a COI	Sculpin, Pacific lamprey ammocoetes
o-Xylene	Not a COI	Not a COI	Not a COPC	HQ ≥ 1 (12)	Not a COI	Sculpin, Pacific lamprey ammocoetes
Total xylenes	Not a COI	Not a COI	Not a COPC	HQ ≥ 1 (34)	Not a COI	Sculpin, Pacific lamprey ammocoetes
TPH [o]						
Gasoline-range aliphatic hydrocarbons C4-C6	Not a COI	Not a COI	Not a COI	HQ ≥ 1 (7.3)	Not a COI	Sculpin, Pacific lamprey ammocoetes
Gasoline-range aliphatic hydrocarbons C6-C8	Not a COI	Not a COI	Not a COI	HQ ≥ 1 (4.3)	Not a COI	Sculpin, Pacific lamprey ammocoetes
Gasoline-range aliphatic hydrocarbons C10-C12	Not a COI	Not a COI	Not a COI	HQ ≥ 1 (540)	Not a COI	Sculpin, Pacific lamprey ammocoetes
Gasoline-range aromatic hydrocarbons C8-C10	Not a COI	Not a COI	Not a COI	HQ ≥ 1 (2.7)	Not a COI	Sculpin, Pacific lamprey ammocoetes

Table 7-45. Summary of Fish COPCs for Each LOE

	Line of Evidence Resulting in HQ ≥ 1 [a] (Maximum HQ)					
COPC	**Tissue-Residue**	**Diet**	**Surface Water**	**TZW**	**Fish Condition**	**Receptor[b]**
Other Contaminants						
Cyanide	Not a COI	Not a COI	Not a COI	HQ ≥ 1 (4,400)	Not a COI	Sculpin, Pacific lamprey ammocoetes
Perchlorate	Not a COI	Not a COI	Not a COI	HQ ≥ 1 (19)	Not a COI	Sculpin, Pacific lamprey ammocoetes

[a] COPCs for which HQ were ≥ 1 based on the final step in the risk characterization including a relevant exposure scale (Table 7-1) and, for applicable receptors using the dietary LOE, a multiple prey species diet.
[b] These are the receptors for which a COPC has an HQ ≥ 1 for one or more LOEs.
[c] Not assessed because no screening-level TRV was available.
[d] Monobutyltin was not evaluated using the dietary LOEs beyond the SLERA because no baseline TRV was available.
[e] No individual samples results in an HQ ≥ 1.0 but individual sediment and prey samples in close proximity result in an HQ of 1.0.
[f] Not assessed because individual PAH was not a component of the total PAHs TRV and no other screening-level TRV was available.
[g] Individual PAHs other than benzo(a)pyrene were only evaluated as a component of total PAHs for the dietary LOE.
[h] Individual PAHs (not total PAHs) were evaluated for the surface water and TZW LOEs.
[i] BEHP could not be evaluated for the tissue-residue LOE because no acceptable BERA LOAEL TRV was available. A comparison of Study Area to upstream tissue contaminant concentrations is presented in Section 7.1.5.2.
[j] Maximum HQ presented is based on predicted concentrations.
[k] HQ presented is based on the alternative total PCBs TRV for protection of directly exposed aquatic organisms. Total PCBs surface water HQs are ≥ 1 in two samples (maximum HQ = 1.2) when calculated using the AWQC TRV (based on protection of mink via ingestion of contaminated prey). The alternative TRV is considered more appropriate for evaluating direct exposure of organisms.
[l] 2,4′-DDD, 2,4′-DDT, and 4,4′-DDD screened in as COPCs for the surface water and TZW LOEs and 4,4′-DDE screened in as a COPC for the TZW LOE but were not evaluated individually for the surface water or TZW LOEs; they were evaluated as a component of total DDx. 4,4′-DDT was evaluated both individually and as a component of total DDx because the TRV for total DDx is based on 4,4′-DDT.
[m] 4,4′-DDD and 4,4′-DDT screened in as COPCs but were not evaluated individually for the tissue residue LOE; they were evaluated as a component of total DDx.
[n] HQ presented is based on the alternative 4,4′-DDT TRV for protection of directly exposed aquatic organisms. 4,4′-DDT and total DDx HQs are ≥ 1 for all fish receptors (maximum HQ = 20) when calculated based on the AWQC TRV (for protection of brown pelican via ingestion of contaminated prey). The alternative TRV is considered more appropriate for evaluating direct exposure of aquatic organisms.
[o] Because petroleum compounds are not CERCLA contaminants, gasoline-range hydrocarbons were not included in the final count of contaminants posing potentially unacceptable risk; they are included here because they may nonetheless contribute to risk, as discussed in the risk characterization.

AWQC – ambient water quality criteria
BEHP – bis(2-ethylhexyl) phthalate
BERA – baseline ecological risk assessment
CERCLA – Comprehensive Environmental Response, Compensation, and Liability Act
COI – contaminant of interest
COPC – contaminant of potential concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
HQ – hazard quotient
LOAEL – no-observed-adverse-effect level
LOE – line of evidence
NE – not evaluated (COPC not evaluated using this LOE)
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
SVOC – semivolatile organic compound
TBT – tributyltin
total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT and 4,4′-DDT)
TPH – total petroleum hydrocarbons
TZW – transition zone water
VOC – volatile organic compound

7.6.2 Further Evaluation of COPCs and Assessment Endpoints with HQs ≥ 1

COPCs were divided into four groups to further evaluate COPCs and assessment endpoints with HQs ≥ 1: organics and organometals, inorganic metals, VOCs, and PAHs. Up to four LOEs (tissue-residue, dietary, surface water, and TZW) were used to derive HQs for organics and organometals and inorganic metals. Two LOEs (surface water and TZW) were evaluated for VOCs and three LOEs (dietary, surface water, and TZW) were evaluated for PAHs. Fish condition and direct contact with sediment were evaluated as a separate semi-quantitative LOE for PAHs but this LOE was inconclusive (Section 7.5.4).

7.6.3 Evaluation of Fish COPCs

EPA's Problem Formulation (Attachment 2) describes a WOE approach as "…a method to help identify and rank which LOEs for each receptor provide the most scientifically reliable indication of the status of each assessment endpoint from exposure to COPCs at the site and, hence, which might be most useful for making risk management decisions." When multiple LOEs did not agree, an evaluation of each LOE and the associated uncertainties was conducted, and a WOE evaluation was used to compare findings across LOEs. When only one LOE was used to evaluate a COPC or when HQs based on multiple LOEs were consistent, no WOE evaluation was necessary. Further evaluation of COPCs and assessment endpoints with HQs ≥ 1 took into account magnitude of HQs, uncertainties about exposures and effects, toxicological effects associated with the TRV, and, particularly for population-level assessment endpoints, the spatial extent of HQs ≥ 1.

Of the 72 COPCs identified by the SLERA and refined screening process for the fish receptors, 59 contaminants were identified as posing potentially unacceptable risk to at least one fish receptor. Upon consideration of the spatial extent, magnitude, WOE, and ecological significance of risks, it was concluded that COPCs with HQs ≥ 1 identified in the BERA using four primary LOEs (i.e., tissue-residue, dietary, surface water, and TZW LOEs) pose potentially unacceptable risk but pose negligible to low risk of ecologically significant adverse effects on Study Area fish populations. Some COPCs, however pose risk to individual fish (including Pacific lamprey for which the assessment endpoints are for protection at the level of the organism) within localized areas of the Study Area. These localized risks occurprimarily via the TZW LOE.

Fifty three[121] COPCs measured in TZW have baseline HQs ≥ 1. TZW exposure in benthic fish, including burrowing fish (Pacific lamprey ammocoetes) and in fish that feed on benthic organsims (sculpin) is much lower than that reflected by TZW EPCs (probably by a factor of ≥ 10) because, while lamprey live in shallow tubes and sculpin sometimes burrow into gravel to forage, they do not respire water below the sediment surface and are not directly exposed to TZW, especially at the depths sampled

[121] Fifty-four TZW COPCs with HQs ≥ 1 were identified for benthic fish. Because petroleum compounds are not CERCLA contaminants, gasoline-range hydrocarbons were not included in the final count of contaminants posing potentially unacceptable risk; they are included here because they may nonetheless contribute to risk. Petroleum hydrocarbons are also excluded from the counts of fish TZW COPCs with HQs >10 and with HQs ≤ 10.

(0 to 38 cm below the sediment surface). Of the 53 TZW COPCs with HQ ≥ 1, 15 have HQs ≤ 10 and are thus likely to pose negligible risk. The remaining 38 COPCs have HQs > 10. Three of these 38 COPCs with HQs >10 are naturally occuring metals (barium, iron, and manganese), and there is substantial uncertainty as to whether their source is anthropogenic.

TZW HQs > 10 were also calculated for vanadium, zinc, cyanide, and perchlorate. Although the average vanadium HQ at the Siltronic area is <10, HQs > 10 were calculated for some samples (Figure 6-22). The same is true for zinc at the ARCO area (Figure 6-22) and for perchlorate at the Arkema Chlorate Plant area (Figure 6-28). For cyanide at the Gasco and Siltronic areas, both individual and average HQs are >10 (Figure 6-28).

For 15 individual PAHs, TZW HQs > 10 occurred only at Gasco or Siltronic locations: 2-methylnaphthalene, acenaphthene, anthracene, benzo(a)anthracene, benzo(b)fluoranthene, benzo(g,h,i)perylene, benzo(k)fluoranthene, chrysene, dibenzo(a,h)anthracene, fluoranthene, fluorene, indeno(1,2,3-cd)pyrene, naphthalene, phenanthrene,and pyrene. Benzo(a)pyrene HQs > 10 were found not only at Gasco and Siltronic but also at the ARCO and Mobil Oil areas (see Section 6.6.5 for average HQs by area). Gasoline-range aliphatic hydrocarbons (C10-C12) also have HQs > 10 at ARCO, Mobil Oil, Gasco, and Siltronic areas and may therefore contribute to risk.

TZW HQs > 10 were calculated for 12 VOCs or SVOCs. The locations of these exceedances are Gasco or Sitronic areas for benzene, carbon disulfide, cis-1,2,-dichloroethene, ethylbenzene, o-xylene, toluene, total xylenes, and trichloroethene; the Rhone Poulenc area for 1,2-dichlorobenzene and 1,4-dichlorobenzene; and the Arkema Acid Plant area for chlorobenzene and chloroform. Of these SVOCs and VOCs, the only contaminants with an average HQ > 10 at any location are carbon disulfide (at the Gasco area), chlorobenzene (at the Arkema Acid Plant area), and trichloroethene (at the Siltronic area).

Because TZW exceedances are highly localized, none of the TZW COPCs is likely to pose risk to Study Area sculpin populations. However, TZW COPCs listed above whose concentrations in localized areas are more than 10 times as highas the TRV (i.e., HQ > 10) could adversely affect individual lamprey in those locations. The magnitude of risk to individual lamprey from these COPCs is unknown; however, because the TRVs were derived to be protective of the most sensitive species and may overpredict risk to Pacific lamprey, whose sensitivity to toxicity through various modes of action is equal to or less than that of most aquatic species (see Section 7.4.5.3).
HQs ≥ 1, and therefore potentially unacceptable risk, occur for total DDx in the surface water, tissue-residue, and TZW LOEs; however, ecologically significant risk to fish populations is not expected. Of 170 surface water samples from throughout the Study Area, the only total DDx concentration that exceeded the direct exposure surface water TRV is based on an N-qualified result, indicating probable analytical interference from a different chemical. Based on the tissue-residue LOE, sculpin is the only fish with tissue

concentrations greater than the total DDx 10th percentile LOAEL. All tissue exceedances occurred within a single localized area (approximately RM 7.1 to RM 7.4), with a maximum HQ of 1.9.

The only total DDx TZW exceedances also occurred in this vicinity (RM 6.8 and RM 7.4), with a maximum HQ of 280. The high TZW HQs are based on unfiltered samples, which may overestimate the true TZW concentration. However, the co-occurence of the tissue-residue and TZW exceedances indicates a low risk to individual benthic fish such as sculpin or Pacific lamprey in this localized area. The low magnitude (considering fish respire primarily surface water with concentrations that are a fraction of those measured in TZW) and limited spatial extent of exceedances indicate that exposure is not likely to result in ecologically significant adverse effects on Study Area fish populations.

HQs ≥1 occur for total PCBs based on the surface water and tissue-residue LOEs; however, total PCBs pose low risk of causing ecologically significant adverse effects on Study Area fish populations. All surface water samples had total PCB concentrations that were less than the surface water TRV based on direct exposure. Some composite tissue-residue samples of largescale sucker, sculpin, smallmouth bass, and northern pikeminnow had total PCB concentrations exceeding the 10th percentile LOAEL TRV (with HQs up to 9.4), indicating potentially unacceptable risk. However, these exceedances are expected to overpredict risk to fish based on the inclusion of highly uncertain and conservative toxicity studies in the derivation of the LOAEL TRV. While total PCBs pose low risk to Study Area fish populations, total PCBs may result in adverse effects on individual sculpin at RM 11 and individual smallmouth bass within Swan Island Lagoon in the exposure area from RM 10.5 to 11.5. Sculpin and smallmouth bass total PCB tissue concentrations from these respective locations were above the 10th percentile of LOAELs excluding the highly uncertain and conservative studies, although no acceptable toxicity data specific to sculpin or smallmouth bass were available. The limited spatial extent of these exceedances, however, indicates that risk is not likely to be ecologically significant to the Study Area populations. Total PCBs concentrations in Pacific lamprey tissue were below tissue-residue effects thresholds, indicating total PCBs pose negligible risk to Pacific lamprey ammocoetes.

Risk conclusions based on magnitude of HQs, uncertainties about exposures and effects, toxicological effects associated with the TRV, the spatial extent of HQs ≥ 1, and WOE for fish COPCs with HQ ≥ 1 are summarized in Table 7-46. In Section 11, the COPCs for the fish assessment endpoints are evaluated alongside the COPCs for all other ecological assessment endpoints to reach overall ecological risk conclusions. Risks conclusions associated with the nine TZW COPCs that have HQs ≥ 1 for another LOE in addition to TZW (i.e., cadmium, copper, lead, zinc, benzo(a)anthracene, benzo(a)pyrene, naphthalene, total DDx, and trichloroethene) are discussed in Table 7-46. Risk conclusions associated with the remaining 44 COPCs that result in HQs ≥ 1 only for the TZW LOE are discussed above but are not presented in Table 7-46. Effects from exposure to multiple chemicals that share the same mode of toxic action were not factored into the effects assessment. The same is true of effects due to other environmental stressors in the Study Area that could create additive, synergistic, or antagonistic interactions. Generally, it can be assumed that toxicity is additive if the individual chemicals induce the same health effects by a similar mode of action (EPA 2000a). There is, however, substantial uncertainty when assessing risk from mixtures, due primarily to the lack of toxicological data, particularly with three or more chemicals(EPA 2000b). Additionally, the nature of the chemicals in the mixture, their reactions in the ambient environment, and their interactions with one another increase the uncertainty associated with predicted toxicity. For example, the toxicity of metal mixtures is confounded by the different degrees of bioavailability among the individual metals and by the potentially conflicting methods of data interpretation among various toxicological studies (EPA 2007e). In the environment, one chemical can alter the toxicity of another and behave within a particular medium in a variety of ways. Interactions among multiple chemical contaminants may cause changes in form, bioaccumulation properties, and persistence of the individual components. The uncertainty can be exacerbated for relatively unstable chemicals, and for metals having multiple valence states. Because the combined effects of complex chemical mixtures and other stressors in the environment have not been sufficiently studied, effects of this uncertainty on risk predictions are unknown.

Table 7-46. Summary of Fish COPCs with HQ ≥ 1 and Risk Conclusions Across LOEs

	Max HQ[a] by Line of Evidence					
COPC by Receptor	**Tissue Residue**	**Dietary Dose**	**Surface Water**	**TZW[b]**	**Conclusion**	**Rationale for Risk Conclusion**
Large-Home-Range Fish (site-wide)						
Largescale Sucker						
Copper	Not a COPC	1.1	Not a COPC	Not a COI	Negligible risk	All LOEs in reasonable agreement. Maximum dietary HQ not indicative of ecologically significant risk. Dietary risk likely overestimated: HQ ≥ 1 based on clam-only diet; worm-only diet maximum HQ < 1. The selected LOAEL not repeatable in subsequent tests with same species, and just above range of nutritional requirements found in the literature for some but not all fish species. Maximum HQ < 1 when calculated using next lowest literature-based LOAEL. Surface water – based on large AWQC dataset is strongest LOE.
Total PCBs	1.6	Not a COI[c]	Step 1 HQ < 1[d]	Not a COI	Negligible risk	All LOEs in reasonable agreement. Maximum tissue-residue HQ not indicative of ecologically significant risk. Low tissue-residue HQ. HQ ≥ 1 in only 2 of 6 samples. Uncertainty in tissue-residue TRV more likely to over- than underpredict risk due to inclusion of uncertain LOAELs in SSD. Surface water – based on large AWQC dataset is strongest LOE.
Juvenile White Sturgeon						
Copper	Not a COPC	1.2	Not a COPC	Not a COI	Negligible risk	All LOEs in reasonable agreement. Maximum dietary HQ not indicative of ecologically significant risk. Dietary risk likely overestimated: HQ≥ 1 based on clam-only diet and assumed sediment ingestion rate of 56% (vs. 8% scenario); stomach content and worm diet HQ < 1. The selected LOAEL not repeatable in subsequent tests with same species, and just above range of nutritional requirements found in the literature for some but not all fish species. Maximum HQ < 1 when calculated using next lowest literature-based LOAEL. Surface water – based on large AWQC dataset is strongest LOE.

Table 7-46. Summary of Fish COPCs with HQ ≥ 1 and Risk Conclusions Across LOEs

	Max HQ[a] by Line of Evidence					
COPC by Receptor	**Tissue Residue**	**Dietary Dose**	**Surface Water**	**TZW[b]**	**Conclusion**	**Rationale for Risk Conclusion**
Juvenile Chinook Salmon						
Cadmium	Not a COPC	3.5	Not a COPC	Not a COI	Negligible risk	All LOEs in reasonable agreement. Maximum dietary HQ not indicative of ecologically significant risk. Dietary risk likely overestimated: diet consists primarily of pelagic prey rather than benthic prey as assumed. Dietary TRV conservative. Surface water – based on large AWQC dataset is strongest LOE.
Copper	Not a COPC	2.5	Not a COPC	Not a COI	Negligible risk	All LOEs in reasonable agreement. Maximum dietary HQ not indicative of ecologically significant risk. Dietary risk likely overestimated: diet consists primarily of pelagic prey rather than benthic prey as assumed. The selected LOAEL not repeatable in subsequent tests with same species, and just above range of nutritional requirements found in the literature for some but not all fish species. Maximum HQ < 1 when calculated using next lowest literature-based LOAEL. Surface water - based on large AWQC dataset is strongest LOE.
Peamouth						
Copper	Not a COPC	1.0	Not a COPC	Not a COI	Negligible risk	All LOEs in reasonable agreement. Maximum dietary HQ not indicative of ecologically significant risk. Dietary risk likely overestimated: diet consists substantially of pelagic prey rather than solely benthic prey as assumed. The selected LOAEL not repeatable in subsequent tests with same species, and just above range of nutritional requirements found in the literature for some but not all fish species. Maximum HQ < 1 when calculated using next lowest literature-based LOAEL. Surface water - based on large AWQC dataset is strongest LOE.
Lead	2.7	Not a COPC	Not a COPC	Not a COI	Negligible risk	All LOEs in reasonable agreement. Maximum tissue-residue HQ not indicative of ecologically significant risk. Low tissue-residue HQ. Tissue-residue LOE weak because fish actively regulate inorganic metals. High uncertainty in tissue-residue TRV. Surface water – based on large AWQC dataset is strongest LOE.

Table 7-46. Summary of Fish COPCs with HQ ≥ 1 and Risk Conclusions Across LOEs

	Max HQ[a] by Line of Evidence					
COPC by Receptor	**Tissue Residue**	**Dietary Dose**	**Surface Water**	**TZW[b]**	**Conclusion**	**Rationale for Risk Conclusion**
Pacific Lamprey Ammocoetes						
Cadmium	Not a COPC	Not a COI[e]	Not a COPC	5.8	Negligible risk	All LOEs in reasonable agreement. Maximum TZW HQ not indicative of ecologically significant risk. TZW risk likely overestimated by factor of > 10 because receptor not directly exposed to TZW; HQ < 10. Surface water – based on large AWQC dataset is strongest LOE.
Copper	2.2	Not a COI[e]	Not a COPC	1.3	Negligible risk	All LOEs in reasonable agreement. Maximum tissue-residue and TZW HQs not indicative of ecologically significant risk. Tissue-residue LOE relatively weak because fish actively regulate inorganic metals and tissue TRV is close to the nutritional requirement for some but not all fish species. Lamprey less sensitive to copper than most sensitive species. TZW risk likely overestimated by factor of > 10 because receptor not directly exposed to TZW; HQ < 10. Surface water – based on large AWQC dataset is strongest LOE.
Lead	Not a COPC	Not a COI[e]	Not a COPC	3	Negligible risk	All LOEs in reasonable agreement. Maximum TZW HQ not indicative of ecologically significant risk. TZW risk likely overestimated by factor of > 10 because receptor not directly exposed to TZW; HQ < 10. Surface water – based on large AWQC dataset is strongest LOE.
Zinc	Not a COPC	Not a COI[e]	Step 2 HQ < 1	14	Localized risk to individual organisms	Maximum TZW HQ indicative of risk to individual lamprey. TZW risk likely overestimated by factor of > 10 because receptor not directly exposed to TZW; HQ < 10, except at ARCO (maximum HQ = 14). TZW risk of limited spatial extent (HQ > 10 at 1 of 10 areas: 1 of 12 samples at ARCO). Lamprey not among most sensitive species in toxicity tests for most contaminants tested. All LOEs indicate negligible risk in other areas. Surface water – based on large AWQC dataset is strongest LOE.

Table 7-46. Summary of Fish COPCs with HQ ≥ 1 and Risk Conclusions Across LOEs

	Max HQ[a] by Line of Evidence					
COPC by Receptor	**Tissue Residue**	**Dietary Dose**	**Surface Water**	**TZW[b]**	**Conclusion**	**Rationale for Risk Conclusion**
Benzo(a)anthracene	Not a COI[f]	Not a COI[e]	Step 2 HQ < 1	1,200	Localized risk to individual organisms	Maximum TZW HQ indicative of risk to individual lamprey in localized areas. TZW risk likely overestimated by factor of > 10 because receptor not directly exposed to TZW; HQ < 10, except at Siltronic (maximum HQ = 1,200) and Gasco (maximum HQ = 120). TZW risk of limited spatial extent (HQ ≥ 10 at 2 of 6 areas: 3 of 12 samples at Gasco, and 11 of 32 at Siltronic). All LOEs indicate negligible risk in other areas. Lamprey insensitive to PAH toxicity.
Benzo(a)pyrene	Not a COI[f]	Not a COI[e]	Step 2 HQ < 1	2,700	Localized risk to individual organisms	Maximum TZW HQ indicative of low risk to individual lamprey. TZW risk likely overestimated by factor of > 10 because receptor not directly exposed to TZW; HQ < 10, except at Arco (maximum HQ = 15), Mobil Oil (maximum HQ = 25), Siltronic (maximum HQ = 2,700) and Gasco (maximum HQ = 210). TZW risk of limited spatial extent (HQ ≥ 10 at 4 of 6 areas: 1 of 12 samples at ARCO, 5 of 21 at Mobil Oil, 3 of 12 at Gasco, and 11 of 32 at Siltronic). All LOEs indicate risk in other areas is negligible. Lamprey insensitive to PAH toxicity.
Naphthalene	Not a COI[f]	Not a COI[e]	Step 2 HQ < 1	1,100	Localized risk to individual organisms	Maximum TZW HQ indicative of low risk to individual lamprey. TZW risk likely overestimated by factor of > 10 because receptor not directly exposed to TZW; HQ < 10, except at Siltronic (1,100) and Gasco (260). TZW risk of limited spatial extent (HQ ≥ 10 at 2 of 6 areas: 6 of 12 samples at Gasco, and 12 of 60 at Siltronic). All LOEs indicate risk in other areas is negligible. Lamprey insensitive to PAH toxicity.
4,4′-DDT	NE[g]	Not a COI[e]	Step 2 HQ < 1	160[g]	Localized risk to individual organisms	Maximum TZW HQ indicative of low risk to individual lamprey. TZW risk likely overestimated by factor of > 10 because receptor not directly exposed to TZW. Exposure likely to be overestimated by data from unfiltered samples. TZW risk of limited spatial extent (TZW HQ ≥10 in 3 of 12 samples at Arkema acid plant. All LOEs indicate risk in other areas is negligible. Lamprey not among most sensitive species in toxicity tests for most contaminants tested.

Table 7-46. Summary of Fish COPCs with HQ ≥ 1 and Risk Conclusions Across LOEs

	Max HQ[a] by Line of Evidence					
COPC by Receptor	**Tissue Residue**	**Dietary Dose**	**Surface Water**	**TZW[b]**	**Conclusion**	**Rationale for Risk Conclusion**
Total DDx	Step 1 HQ < 1	Not a COI[e]	Step 2 HQ < 1[g]	280[g]	Localized risk to individual organisms	Maximum TZW HQ indicative of low risk to individual lamprey. TZW risk likely overestimated by factor of > 10 because receptor not directly exposed to TZW. Exposure likely to be overestimated by data from unfiltered samples. TZW risk of limited spatial extent (TZW HQ ≥ 10 in areas: 5 of 12 samples at Arkema acid plant, maximum HQ = 280; 2 of 2 samples at Rhône-Poulenc, maximum HQ = 19. All LOEs indicate risk in other areas is negligible. Lamprey not among most sensitive species in toxicity tests for most contaminants tested.
Small--Home-Range Fish (1 mile or sample-specific)						
Sculpin (sample-specific)						
Cadmium	Not a COPC	4.2	Not a COPC	5.8	Negligible risk	All LOEs in reasonable agreement. Maximum dietary and TZW HQs not indicative of ecologically significant risk. Dietary risk of limited spatial extent: HQ ≥ 1 in 9 of 111 prey and 1 of 1,348 sediment samples (RM 2.0, east; RM 4.2, east; Slip 1; Swan Island Lagoon; International Slip). Dietary TRV highly conservative: lower than the nine NOAELs and four LOAELs in other studies. TZW risk likely overestimated by factor of > 10 because receptor not directly exposed to TZW; HQ < 10. Surface water – based on large AWQC dataset is strongest LOE.

Table 7-46. Summary of Fish COPCs with HQ ≥ 1 and Risk Conclusions Across LOEs

	Max HQ[a] by Line of Evidence					
COPC by Receptor	**Tissue Residue**	**Dietary Dose**	**Surface Water**	**TZW[b]**	**Conclusion**	**Rationale for Risk Conclusion**
Copper	2.3	5.3	Not a COPC	1.3	Negligible risk	All LOEs in reasonable agreement. Maximum tissue-residue, dietary and TZW HQs not indicative of ecologically significant risk. Tissue-residue risk of limited spatial extent (HQ ≥ 1 in only 3 of 38 samples). Tissue-residue LOE relatively weak because fish actively regulate inorganic metals. Dietary risk likely overestimated: The selected LOAEL not repeatable in subsequent tests with same species and just above range of nutritional requirements found in the literature for some but not all fish species; maximum HQ < 1 when calculated using next lowest literature-based LOAEL. TZW risk likely overestimated by factor of > 10 because receptor not directly exposed to TZW; HQ < 10. Surface water – based on large AWQC dataset is strongest LOE.
Lead	Not a COPC	Not a COPC	Not a COPC	3	Negligible risk	All LOEs in reasonable agreement. Maximum TZW HQ not indicative of ecologically significant risk. TZW risk likely overestimated by factor of > 10 because receptor not directly exposed to TZW; HQ < 10. Surface water – based on large AWQC dataset is strongest LOE.
Mercury	Not a COPC	4.5	Not a COPC	Not a COI[h]	Negligible risk	All LOEs in reasonable agreement. Maximum dietary HQ not indicative of ecologically significant risk. Dietary risk of limited spatial extent (HQ ≥ 1 for only one sediment sample, RM 6.7). Because mercury biomagnifies, tissue residue is strongest LOE.
Zinc	Not a COPC	Not a COPC	1.1	14	Negligible risk	All LOEs in reasonable agreement. Maximum surface water and TZW HQs not indicative of ecologically significant risk. Low surface water and TZW HQs, and limited spatial extent of HQs ≥ 1. Surface water HQ ≥ 1 in 1 of 167 samples (November 2004 low-flow sampling event at RM 9.7, west). TZW risk likely overestimated by factor of > 10 because receptor not directly exposed to TZW; HQ < 10, except at ARCO (maximum HQ = 14). TZW risk of limited spatial extent (HQ > 10 at 1 of 10 areas: 1 of 12 samples at ARCO). Surface water – based on large AWQC dataset is strongest LOE.

Table 7-46. Summary of Fish COPCs with HQ ≥ 1 and Risk Conclusions Across LOEs

	Max HQ[a] by Line of Evidence					
COPC by Receptor	**Tissue Residue**	**Dietary Dose**	**Surface Water**	**TZW[b]**	**Conclusion**	**Rationale for Risk Conclusion**
Monobutyltin	NE[j]	Not a COI[c]	1.2	Not a COI[h]	Negligible risk	All LOEs in reasonable agreement. Maximum surface water HQ not indicative of ecologically significant risk. Surface water risk of limited spatial extent and likely overestimated: HQ ≥ 1 for 1 of 167 samples (winter 2007 high-flow sampling event at RM 11, west); TRV based on a fish-protective AWQC for TBT, which is more toxic than monobutyltin. TBT tissue residue noted to be reliable predictor of toxicity is strongest LOE.
TBT	Not a COPC	1.0[k]	Not a COPC	Not a COI[h]	Negligible risk	All LOEs in reasonable agreement. Maximum dietary HQ not indicative of ecologically significant risk. Dietary risk of limited spatial extent and likely overestimated: HQ ≥ 1 based on a single lab worm sample combined with sediment samples (at mouth of Swan Island Lagoon); HQ for other prey species < 1. Selected LOAEL (reduced reproduction in medaka) conservative because observed effects not dose-responsive. TBT tissue residue noted to be reliable predictor of toxicity is strongest LOE.
Benzo(a)anthracene	Not a COI[f]	Not a COPC	10	1,200	Negligible risk	All LOEs in reasonable agreement. Maximum surface water and TZW HQs not indicative of ecologically significant risk. Surface water and TZW risk of limited spatial extent: Surface water HQ ≥ 1 for only 2 of 245 samples (July 2005 low-flow sampling event at RM 6.1, winter 2007 high-flow at RM 6.3); TZW risk likely overestimated by factor of > 10 because receptor not directly exposed to TZW; maximum TZW HQ < 10, except at Siltronic (1,200) and Gasco (120). TZW risk of limited spatial extent (HQ ≥ 10 at 3 of 12 samples at Gasco, and 11 of 32 at Siltronic). TRV for TZW and surface water LOEs is uncertain: based on Tier II value extrapolated from a *Daphnia* acute LC50 and may over- or underestimate risk to fish.

Table 7-46. Summary of Fish COPCs with HQ ≥ 1 and Risk Conclusions Across LOEs

	Max HQ[a] by Line of Evidence					
COPC by Receptor	**Tissue Residue**	**Dietary Dose**	**Surface Water**	**TZW[b]**	**Conclusion**	**Rationale for Risk Conclusion**
Benzo(a)pyrene	Not a COI[f]	Not a COPC	14	2,700	Negligible risk	All LOEs in reasonable agreement. Maximum surface water and TZW HQs not indicative of ecologically significant risk. Surface water risk of limited spatial extent: surface water HQ ≥ 1 for only 3 of 122 near-bottom samples (November 2004 low-flow sampling event at RM 6.1, July 2005 low-flow and winter 2007 high-flow events at RM 6.3); TZW risk likely overestimated by factor of > 10 because receptor not directly exposed to TZW; maximum TZW HQ < 10, except at Arco (maximum HQ = 15), Mobil Oil (maximum HQ = 25), Siltronic (maximum HQ = 2,700) and Gasco (maximum HQ = 210). TZW risk of limited spatial extent (HQ ≥ 10 in 1 of 12 samples at ARCO, 5 of 21 at Mobil Oil, 3 of 12 at Gasco, and 11 of 32 at Siltronic).
Naphthalene	Not a COI[f]	Not a COPC	50	1,100	Negligible risk	All LOEs in reasonable agreement. Maximum surface water and TZW HQs not indicative of ecologically significant risk. Surface water risk of limited spatial extent: surface water HQ ≥ 1 for only 10 of 268 samples all from RM 6.4 to RM 6.5 during May 2005 non-LWG sampling event. TZW risk likely overestimated by factor of > 10 because receptor not directly exposed to TZW; maximum TZW HQ < 10, except at Siltronic (1,100) and Gasco (260). TZW risk of limited spatial extent (HQ ≥ 10 at 2 of 6 areas: 6 of 12 samples at Gasco, and 12 of 60 at Siltronic).
BEHP	2.9[l]	Not a COI[c]	2.3	Not a COI[h]	Negligible risk	All LOEs in reasonable agreement. Maximum tissue-residue and surface water HQs not indicative of ecologically significant risk. Tissue risk likely overestimated and of limited spatial extent: TRV based on unbounded NOAEL as no LOAEL identified; HQ ≥ 1 for only 1 of 38 samples (mouth of Swan Island Lagoon). Surface water risk of limited spatial extent: surface water HQ ≥ 1 in only 2 of 190 samples (November 2006 storm runoff at RM 3.9, winter 2007 high-flow sampling at RM 6.7).

Table 7-46. Summary of Fish COPCs with HQ ≥ 1 and Risk Conclusions Across LOEs

COPC by Receptor	Max HQ[a] by Line of Evidence: Tissue Residue	Dietary Dose	Surface Water	TZW[b]	Conclusion	Rationale for Risk Conclusion
Total PCBs	9.4 (111[m])	Not a COI[c]	Step 1 HQ < 1[d]	Not a COI[h]	Low risk	Tissue-residue LOE indicates low risk of ecologically significant adverse effects; surface water LOE indicates negligible risk. Tissue-residue risk based on empirical data is of limited spatial extent: tissue-residue HQ ≥ 1 in only 4 of 38 samples (RM 2.3 to RM 2.4, east; Willamette Cove; RM 11.3, east; additional "predicted" locations). Spatial extent of tissue-residue HQ ≥ 1 based on predicted concentrations wider than that based on empirical concentrations: areas with predicted HQ ≥ 1 generally near or at locations with empirically based HQ < 1; empirical data assumed to be more representative than predicted data. Uncertainty in tissue-residue TRV more likely to over- than underpredict risk due to inclusion of uncertain LOAELs in SSD.
4,4′-DDT	NE[g]	Not a COI[e]	Step 2 HQ < 1	160[g]	Negligible risk	Maximum TZW HQ not indicative of ecologically significant risk. TZW risk likely overestimated by factor of > 10 because receptor not directly exposed to TZW. Exposure may be overestimated by data from unfiltered samples. TZW risk of limited spatial extent (TZW HQ ≥ 1 at 1 of 2 areas: 3 of 12 samples at Arkema acid plant.
Total DDx	1.9 (21[m])	Not a COI[c]	1.8[g]	280[g]	Negligible risk	All LOEs in reasonable agreement. Maximum tissue-residue, surface water, and TZW HQs not indicative of ecologically significant risk. Tissue-residue HQ ≥ 1 relatively low and present over a limited spatial extent (1of 38 empirical tissue samples, RM 7.3); HQ ≥ 1 based on predicted data co-located with empirical HQ ≥ 1; empirical data assumed to be more representative than predicted data. Surface water HQ ≥1 in only 1 of 170 samples (RM 2.0, N-qualified). Qualifier indicates interference from non-target analyte. TZW risk likely overestimated by factor of > 10 because receptor not directly exposed to TZW. Exposure may be overestimated by data from unfiltered samples. TZW risk of limited spatial extent (HQ ≥ 10 at 2 of 2 areas: 5 of 12 samples at Arkema acid plant, maximum HQ = 280; 2 of 2 samples at Rhône-Poulenc, maximum HQ = 19).

Table 7-46. Summary of Fish COPCs with HQ ≥ 1 and Risk Conclusions Across LOEs

	Max HQ[a] by Line of Evidence					
COPC by Receptor	**Tissue Residue**	**Dietary Dose**	**Surface Water**	**TZW[b]**	**Conclusion**	**Rationale for Risk Conclusion**
Ethylbenzene	Not a COI[f]	Not a COI[c]	Step 2 HQ < 1	57	Negligible risk	All LOEs in reasonable agreement. Maximum TZW HQ not indicative of ecologically significant risk. TZW risk likely overestimated by factor of > 10 because receptor not directly exposed to TZW. TZW risk of limited spatial extent (HQ ≥ 10 at 2 of 10 areas sampled: 1 of 8 samples at Gasco and 7 of 54 at Siltronic).
Trichloroethene	Not a COI[f]	Not a COI[c]	4.1	1,900	Negligible risk	All LOEs in reasonable agreement. Maximum surface water and TZW HQs not indicative of ecologically significant risk. Surface water risk of limited spatial extent: HQ ≥ 1 in only one sample (May 2005 non-LWG sampling event); sampling restricted to the one area thought to have highest potential for trichloroethene contamination (0.15-mile stretch between RM 6.4 and RM 6.5, west bank). TZW risk likely overestimated by factor of > 10 because receptor not directly exposed to TZW; TZW risk of limited spatial extent (HQ ≥ 10 at 1 of 10 areas sampled: 1 of 54 samples at Siltronic).
Smallmouth Bass (1 mile)						
Antimony	5.4	NE[n]	Not a COPC	Not a COI	Negligible risk	All LOEs in reasonable agreement. Max tissue-residue HQ not indicative of ecologically significant risk. Tissue-residue risk of limited spatial extent: HQ ≥ 1 in only 1 of 32 samples (RM 9.5 to 10.5 east bank), an outlier that also yielded the only lead HQ ≥ 1. Tissue-residue LOE weak: fish actively regulate inorganic metals; the selected LOAEL is highly uncertain due to low number of toxicity studies available. Surface water LOE TRV based on moderately sized Tier II dataset is stronger LOE.

Table 7-46. Summary of Fish COPCs with HQ ≥ 1 and Risk Conclusions Across LOEs

	Max HQ[a] by Line of Evidence					
COPC by Receptor	**Tissue Residue**	**Dietary Dose**	**Surface Water**	**TZW[b]**	**Conclusion**	**Rationale for Risk Conclusion**
Lead	280	Not a COPC	Not a COPC	Not a COI	Negligible risk	All LOEs in reasonable agreement. Max tissue-residue HQ not indicative of ecologically significant risk. Tissue-residue risk is of limited spatial extent: HQ ≥ 1 in only 1 of 32 samples (RM 9.5 to 10.5 east bank); sample concentration much higher (>2 to 5 orders of magnitude) than the other smallmouth bass concentration available in same exposure area (RM 9.5 to 10.5 west bank; HQ = 1.7) and elsewhere in Study Area; data compatible with discrete source (e.g., lead sinker). Tissue-residue LOE weak: fish actively regulate inorganic metals; the selected LOAEL is highly uncertain due to low number of toxicity studies available. Surface water – based on large AWQC dataset is strongest LOE.
Zinc	Not a COPC	Not a COPC	1.2	Not a COI	Negligible risk	All LOEs in reasonable agreement. Max surface water HQ not indicative of ecologically significant risk. Low surface water HQ and limited extent of risk: HQ ≥ 1 in only 1 of 167 samples (November 2004 low-flow sampling event at RM 9.7, west). Surface water – based on large AWQC dataset is strongest LOE.
Monobutyltin	NE[j]	NE[j]	1.2	Not a COI	Negligible risk	Surface water risk of limited spatial extent and likely overestimated: HQ ≥ 1 for 1 of 167 samples (winter 2007 high-flow sampling event; RM 11, west); TRV based on a fish-protective AWQC for TBT, which is more toxic than monobutyltin. TBT tissue residue noted to be reliable predictor of toxicity is strongest LOE.
Benzo(a)anthracene	Not a COI[f]	Not a COPC	4.7	Not a COI	Negligible risk	All LOEs in reasonable agreement. Max surface water HQ not indicative of ecologically significant risk. Surface water risk of limited spatial extent: HQ ≥ 1 for only 2 of 245 samples (July 2005 low-flow sampling event at RM 6.1 and winter 2007 high-flow event at RM 6.3). TBT tissue residue noted to be reliable predictor of toxicity is strongest LOE.
Benzo(a)pyrene	Not a COI[f]	Not a COPC	7.6	Not a COI	Negligible risk	All LOEs in reasonable agreement. Max surface water HQ not indicative of ecologically significant risk. Surface water risk of limited spatial extent: HQ ≥ 1 for only 3 of 122 near-bottom samples (November 2004 low-flow event at RM 6.1, July 2005 low-flow and winter 2007 high-flow events at RM 6.3).

Table 7-46. Summary of Fish COPCs with HQ ≥ 1 and Risk Conclusions Across LOEs

	Max HQ[a] by Line of Evidence					
COPC by Receptor	**Tissue Residue**	**Dietary Dose**	**Surface Water**	**TZW[b]**	**Conclusion**	**Rationale for Risk Conclusion**
Naphthalene	Not a COI[f]	Not a COPC	2.8	Not a COI	Negligible risk	All LOEs in reasonable agreement. Max surface water HQ not indicative of ecologically significant risk. Surface water risk of limited spatial extent: HQ ≥ 1 for only 10 of 268 samples all from RM 6.4 to RM 6.5 during May 2005 non-LWG sampling event.
BEHP	9.1	Not a COI[c]	2.3	Not a COI	Negligible risk	All LOEs in reasonable agreement. Max tissue-residue and surface water HQs not indicative of ecologically significant risk. Tissue risk is likely overestimated and of limited spatial extent: TRV is based on NOAEL as no LOAEL identified; HQ≥1 for only 2 of 32 samples (both at RM 3.5, east). Surface water risk of limited spatial extent: HQ ≥ 1 for only 2 of 190 samples (November 2006 storm runoff at RM 3.9, winter 2007 high-flow sampling at RM 6.7).
Total PCBs	7.1	Not a COI[c]	Step 1 HQ < 1[d]	Not a COI	Low risk	Tissue-residue LOE indicates low risk of ecologically significant effect; surface water LOE indicates negligible risk. Tissue-residue HQ ≥ 1 in 9 of 32 samples from 5 of 11 exposure areas (RM 1.5 to 2.5; RM 3.5 to 4.5; RM 6.5 to 7.5; Swan Island Lagoon; RM 10.5 to 11.5) indicating exceedances over a moderate spatial extent. Uncertainty in tissue-residue TRV more likely to over- than underpredict risk due to inclusion of uncertain LOAELs in SSD.
Northern Pikeminnow(1 mile)						
Copper	Not a COPC	1.5	Not a COPC	Not a COI	Negligible risk	All LOEs in reasonable agreement. Max dietary HQ not indicative of ecologically significant risk. Dietary risk likely overestimated: The selected LOAEL TRV not repeatable in subsequent tests with same species and is just above range of nutritional requirements found in the literature for some but not all fish species; maximum HQ < 1 when calculated using next lowest literature-based LOAEL. Surface water - based on large AWQC dataset is strongest LOE.

Table 7-46. Summary of Fish COPCs with HQ ≥ 1 and Risk Conclusions Across LOEs

	Max HQ[a] by Line of Evidence					
COPC by Receptor	**Tissue Residue**	**Dietary Dose**	**Surface Water**	**TZW[b]**	**Conclusion**	**Rationale for Risk Conclusion**
Zinc	Not a COPC	Not a COPC	1.2	Not a COI	Negligible risk	All LOEs in reasonable agreement. Maximum surface water HQ not indicative of ecologically significant risk. Low surface water HQ and limited spatial extent of risk: HQ ≥ 1 in only 1 of 167 samples (November 2004 low-flow sampling event at RM 9.7, west). Surface water – based on large AWQC dataset is strongest LOE.
Monobutyltin	Not a COPC[j]	NE[n]	1.2	Not a COI	Negligible risk	All LOEs in reasonable agreement. Maximum surface water HQ not indicative of ecologically significant risk. Surface water risk of limited spatial extent and likely overestimated: HQ ≥ 1 for 1 of 167 samples (winter 2007 high-flow sampling event at RM 11, west); TRV based on a fish-protective AWQC for TBT, which is more toxic than monobutyltin. TBT tissue residue noted to be reliable predictor of toxicity is strongest LOE.
Benzo(a)anthracene	Not a COI[f]	Not a COPC	4.7	Not a COI	Negligible risk	All LOEs in reasonable agreement. Maximum surface water HQ not indicative of ecologically significant risk. Surface water risk of limited spatial extent: HQ ≥ 1 for only 2 of 245 samples (July 2005 low-flow sampling event at RM 6.1 and winter 2007 high-flow event at RM 6.3).
Benzo(a)pyrene	Not a COI[f]	Not a COPC	7.6	Not a COI	Negligible risk	All LOEs in reasonable agreement. Maximum surface water HQ not indicative of ecologically significant risk. Surface water risk of limited spatial extent: HQ ≥ 1 for only 3 of 122 near-bottom samples (November 2004 low-flow sampling event at RM 6.1, and July 2005 low-flow and winter 2007 high-flow events at RM 6.3).
Naphthalene	Not a COI[f]	Not a COPC	2.8	Not a COI	Negligible risk	All LOEs in reasonable agreement. Maximum surface water HQ not indicative of ecologically significant risk. Surface water risk of limited spatial extent: HQ ≥ 1 for only 10 of 268 samples all from RM 6.4 to RM 6.5 during May 2005 non-LWG sampling event.

Table 7-46. Summary of Fish COPCs with HQ ≥ 1 and Risk Conclusions Across LOEs

	Max HQ[a] by Line of Evidence					
COPC by Receptor	Tissue Residue	Dietary Dose	Surface Water	TZW[b]	Conclusion	Rationale for Risk Conclusion
BEHP	Not a COPC	Not a COI[c]	2.3	Not a COI	Negligible risk	All LOEs in reasonable agreement. Maximum surface water HQ not indicative of ecologically significant risk. Surface water risk of limited spatial extent: HQ ≥ 1 in only 2 of 190 samples (November 2006 storm runoff at RM 3.9 and winter 2007 high-flow sampling event at RM 6.7).
Total PCBs	2.0	Not a COI[c]	Step 1 HQ < 1[d]	Not a COI	Negligible risk	All LOEs in reasonable agreement. Maximum tissue-residue HQ not indicative of ecologically significant risk. Low tissue-residue HQ. Tissue-residue risk of limited spatial extent: HQ ≥ 1 in 2 of 6 samples from 3 of 11 exposure areas (RM 6.5 to RM 7.5, RM 7.5 to RM 8.5, and RM 8.5 to RM 9.5). Uncertainty in tissue-residue TRV more likely to over- than underpredict risk due to inclusion of uncertain LOAELs in SSD.

Note: This table attempts to summarize the BERA's fish risk estimates and risk descriptions, the two major components of the risk characterization. Balancing and interpreting the different types of data evaluated in the BERA can be a major task requiring professional judgment. It can be difficult to prepare a concise summary of conclusions without losing important context, yet a concise summary is needed to help the risk manager judge the likelihood and ecological significance of the estimated risks (EPA 1997)

All the COPCs listed in this table have an HQ ≥ 1 in at least one LOE for at least one ecological receptor, and by definition pose potentially unacceptable risk. The likelihood and ecological significance of the potentially unacceptable risk may vary, though, from very low to very high. Therefore, the risk description may range from negligible to significant. For each receptor-COPC pair with a maximum HQ ≥ 1, this table provides maximum HQ by LOE, a synoptic risk description, and a very brief rationale for the risk description. This distillation of the body of knowledge presented in the BERA should not be taken out of context.

[a] The supporting tables in the individual risk characterization sections for each LOE provide the frequency of TRV exceedances and range of HQs over all exposure areas.

[b] The TZW LOE was evaluated only for sculpin and Pacific lamprey ammocoetes; all other fish receptors are expected to have negligible exposure to TZW.

[c] Dietary LOE was evaluated only for mercury, inorganic metals, and PAHs.

[d] HQ presented is based on the alternative total PCBs TRV for protection of directly exposed aquatic organisms. Total PCBs surface water HQs are ≥ 1 in two samples (maximum HQ = 1.2) when calculated using the AWQC TRV (based on protection of mink via ingestion of contaminated prey). The alternative TRV is considered more appropriate for evaluating direct exposure of organisms.

[e] The dietary LOE was not evaluated for Pacific lamprey ammocoetes.

[f] PAHs were not a COI for the tissue-residue LOE because fish metabolize PAHs.

[g] HQ presented is based on the alternative 4,4′-DDT TRV for protection of directly exposed aquatic organisms. 4,4′-DDT and total DDx surface water HQs are ≥ 1 for all fish receptors (maximum HQ = 20) when calculated based on the AWQC TRV (for protection of brown pelican via ingestion of contaminated prey). 4,4′-DDT and total DDx TZW HQs are ≥ 1 at two locations (Arkema acid plant area and Rhône-Poulenc (maximum HQ = 3,100) when calculated based on the AWQC TRV. The alternative TRV is considered more appropriate for evaluating direct exposure of aquatic organisms. DDT-related contaminants were evaluated as total DDx for the tissue-residue LOE, DDT metabolites such as 4,4′-DDT were not evaluated independently.

[h] Not evaluated because contaminant was not analyzed for in this medium.

[i] Ethylbenzene and trichloroethene were evaluated in surface water from only one location and were not evaluated in Step 2 for large-home-range receptors.

[j] Monobutyltin could not be evaluated because no LOAEL TRV was available from the literature. A LOAEL TRV was available for TBT. Because TBT is the most toxic butyltin (EPA 1991), risks from monobutyltin is assumed to be lower than those of TBT. TBT screens out in Step 1.

[k] No individual samples results in an HQ ≥ 1.0 but individual sediment and prey samples in close proximity result in an HQ of 1.0.

[l] HQ based on a NOAEL TRV because no LOAEL TRV was identified.

[m] Maximum HQ based on predicted tissue concentrations.

[n] COI was not evaluated because no screening-level TRV is available.

AWQC – ambient water quality criteria
BEHP – bis(2-ethylhexyl) phthalate
BERA – baseline ecological risk assessment
COI – contaminant of interest
COPC – contaminant of potential concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
LOE – line of evidence
LWG – Lower Willamette Group
NE – not evaluated
NOAEL – no-observed-adverse-effect level
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
RM – river mile
SSD – species sensitivity distribution
TBT – tributyltin
total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT and 4,4′-DDT)
TRV – toxicity reference value
TZW – transition zone water

8.0 WILDLIFE RISK ASSESSMENT

This section presents the draft final BERA for wildlife (birds and mammals) in the Study Area. Dietary exposure[122] is the main pathway by which wildlife receptors are exposed to sediment contaminants[123] (see Figure 3-2). To address the different ways wildlife may be exposed to sediment contaminants through their diets, six receptors representing four general feeding guilds were evaluated:

- Sediment-probing invertivorous birds – spotted sandpiper
- Omnivorous birds – hooded merganser
- Piscivorous birds[124] – bald eagle, osprey
- Aquatic-dependent carnivores – mink, river otter

Risks to wildlife receptors were evaluated using two LOEs:

- The dietary LOE, in which COPCs ingested via the diet are compared with dietary TRVs
- The egg-residue LOE, in which COPC concentrations in egg tissue residues for piscivorous birds are compared with egg-residue TRVs

The dietary LOE was used for all six wildlife receptors. Ingestion of both prey tissue and sediment (incidental) were accounted for in dietary-dose estimates. The tissue-residue (egg) LOE was used as a second LOE for the evaluation of risks to bald eagle and osprey.

Risk characterization was conducted using HQs, which were calculated by dividing medium-specific EPCs (i.e., egg tissue, prey tissue, and incidentally ingested sediment) by their respective effects thresholds. Among the factors to consider when estimating dietary exposure levelsare feeding rates, foraging areas, prey home ranges, and diets. Based on these data, dietary-dose TRVs (in mg/kg bw/day) were converted to receptor-specific threshold tissue and sediment concentrations (i.e., TTCs and TSCs, respectively; in mg/kg) to facilitate direct comparison to media concentrations. For all receptors, except bald eagle, the TRVs are expressed as LOAELs, which represent the threshold of exposure where effects have been observed in relatively sensitive species. Per EPA (2008j), as documented in EPA's Problem Formulation (Attachment 2), receptors that are designated in the Problem Formulation as special status species (e.g., the federally protected bald eagle) were evaluated at an organism level by comparing EPCs to NOAEL TRVs. NOAELs represent the highest experimental exposure level at which no adverse effects were observed. TRVs were selected from published studies and approved by EPA for use in the BERA.

[122] Incidental ingestion of sediment during feeding is considered a minor component of many wildlife receptor diets.

[123] Ingestion of water and dermal contact with air and water are generally classified as complete but insignificant pathways in EPA's CSM.

[124] Belted kingfisher is evaluated in the uncertainty assessment of the dietary-dose risk characterization text (Section 8.1.5.2.2).

The TRVs provide a basis for evaluating whether exposure concentrations are at or above a level that causes a significant increase in adverse effects on survival, growth, or reproduction of organisms in experimentally exposed populations relative to control populations. This approach follows the conventional practice in ecological risk assessment of using organism-level TRVs defined in this manner to evaluate the potential for effects on populations. However, organism-to-population extrapolation is a source of uncertainty. Levels of growth, mortality, and reproduction predicted from toxicity tests in the laboratory may differ from levels that pose risks to populations in the environment because of factors such as immigration, emmigration, natural fecundity rates, and mortality rates. For example, an increased mortality rate due to toxicity could have negligible effect on a population whose reproductive rate is greater than the carrying capacity of the available habitat, whereas the same increased mortality rate in a population with a naturally low reproductive rate could compromise the population's longevity in the local environment. All wildlife COPCs identified through the SLERA and refined screening process were evaluated. Risk characterization was a winnowing process that allowed proportionally more effort on the receptor-COPC combinations with the potential for unacceptable risk, incorporating principles (screening and iterative refinement) of ecological risk assessment (EPA 1997).

The dietary LOE process involved three steps progressing from more conservative to more realistic estimates of exposure. For each step, if estimates indicated a potentially unacceptable risk, then the exposure assumptions were developed in greater detail taking into account receptor-specific exposure areas and multi-species diets. This process progressed as follows:

- **Step 1 –** HQs were calculated on a sample-by-sample basis for each individual prey component. This calculation was performed in accordance with the methods described in the EPA's Problem Formulation (Attachment 2). If individual samples within a sediment or tissue component of a wildlife receptor's diet had an HQ $\geq$ 1, then the exposure assumptions were developed in greater detail in Step 2. This analysis was used to identify the sampling locations within the Study Area that contribute a greater proportion of exposure.
- **Step 2 –** Because individual composite samples do not represent "a conservative estimate of the average chemical concentration in an environmental medium… for each exposure unit within the site" (from EPA guidance on calculating EPCs (EPA 2002a)), HQs based on individual composite samples of prey or sediment are not appropriate estimates of risk to receptor populations in the Study Area. HQs were therefore calculated over relevant exposure scales based on species-specific foraging range assumptions. HQs were calculated separately for each prey species that the receptor might consume.

- **Step 3 –** If, and only if, the evaluation of any of the single prey species or sediment component in Step 2 yielded an HQ ≥ 1, the exposure was assessed for an appropriate multi-species diet within an appropriate foraging area. Resulting HQs reflect the diversity of prey items the receptor is assumed to ingest. Risk conclusions were based on the final step (i.e., Step 3 for the dietary LOE).[125]

Section 8.1 presents the wildlife risk evaluation process for the dietary LOE, a summary of the TRVs and exposure assumptions that were used, the risk evaluation results, and a discussion of uncertainties specific to the dietary LOE. Section 8.2 presents comparable information for the piscivorous bird tissue-residue (egg) LOE. Section 8.3 presents the overall conclusions about wildlife risk, including potentially unacceptable risks and a synoptic analysis of uncertainty. Section 8.3 establishes the ecological significance of risks associated with each COPC, considering the spatial extent and magnitude of HQs ≥ 1, key uncertainties in the exposure assessment and effects characterization, and a qualitative WOE analysis (where applicable).

8.1 DIETARY ASSESSMENT

Dietary dose was one of two methods used to evaluate risks from exposure to site-related chemicals for all bird and mammal receptors. As an additional LOE for evaluating risks to piscivorous birds (osprey and bald eagle), specific COPCs were evaluated by comparing concentrations measured in bird egg tissue to literature-derived bird egg TRVs (Section 8.2).

Receptor-specific dietary COPCs were identified in the SLERA and refined screen using SL dietary TRVs (Attachment 5). These COPCs were evaluated by comparing diet-based toxicity thresholds to the chemical concentrations in prey tissue and in sediments that are incidentally ingested while feeding. Toxicity thresholds were expressed as concentrations in tissue and sediment, back-calculated from dietary-dose thresholds using receptor-specific exposure assumptions.

For each receptor, COPCs with HQs ≥ 1 at the end of the three-step risk evaluation process were retained as chemicals posing potentially unacceptable risk. For these chemicals, the magnitude of HQs, the spatial distribution and frequency of HQs ≥ 1, the results of multiple LOEs (when applicable), and the associated exposure and effects assumptions were evaluated to arrive at risk conclusions.

The details of this dietary risk assessment for wildlife are presented as follows:

- Section 8.1.1 describes methods used to assess dietary risks to wildlife.
- Section 8.1.2 identifies COPCs evaluated in the dietary risk analysis.

[125] As agreed to between EPA and LWG on October 15, 2010, meeting (LWG 2010).

- Section 8.1.3 presents an overview of the assumptions used to derive exposure concentrations, including derivation of UCLs. Exposure data in this assessment are represented by COPC concentrations in composite samples of prey tissue and sediment. The rationale for exposure assumptions is presented in Attachment 16. All dietary exposure data (i.e., tissue and sediment concentrations) and calculated UCLs are presented in Attachment 4.
- Section 8.1.4 presents a summary of the effects data. Effects data in this assessment are represented by EPA-recommended NOAEL and LOAEL TRVs. Details and uncertainties associated with the selected TRVs for wildlife dietary COPCs are presented in Attachment 16. The comprehensive literature search process is presented in Attachment 14.
- Section 8.1.5 presents the risk characterization results, receptor-COPC pairs, and associated uncertainties. These COPCs are further assessed in the wildlife risk conclusions section (Section 8.3). The risk characterization results of the individual sample and dietary component analysis are presented in Attachment 17.

Figure 8-1 presents a flowchart of the wildlife dietary assessment section.

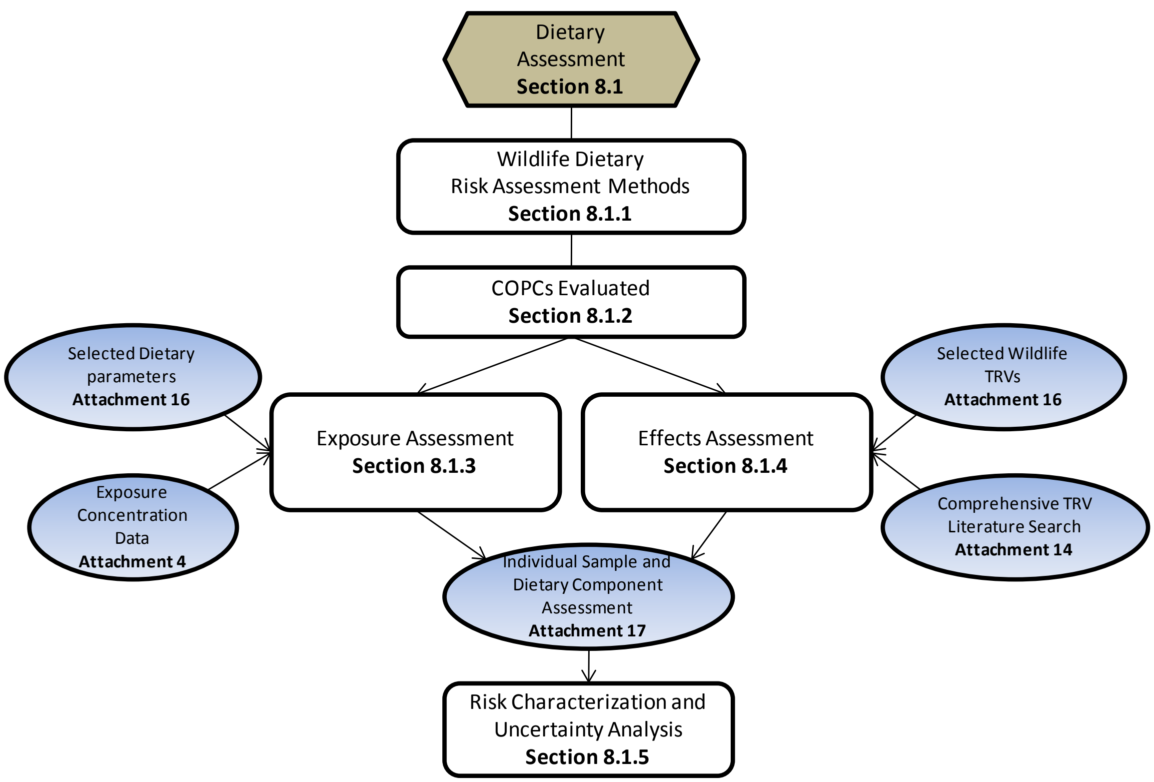


Figure 8-1. Overview of Wildlife Dietary Assessment Section Organization

8.1.1 Wildlife Dietary Risk Assessment Methods

Dietary HQs were calculated by comparing COPC concentrations in prey items and in sediment to receptor-specific toxicity thresholds. Back-calculated from dietary-dose thresholds using receptor-specific exposure assumptions, these thresholds are expressed as TTCs (expressed as mg/kg ww) in prey and TSCs (expressed as mg/kg dw) in sediment. The TTC and TSC derivation methods, and HQ calculation methods used to evaluate dietary risks for wildlife receptors are the same as those used to evaluate fish receptors, as described in Section 7.2.1.

As noted above, analysis of dietary risks to wildlife occurred in three steps, progressing from more conservative to more realistic estimates of exposure and risk:

- **Step 1 –** The derivation of HQs on a sample-by-sample basis for each composite sample of individual prey species and of sediment.
- **Step 2 –** The derivation of HQs over a relevant exposure area for individual prey species and sediment. For sandpiper, HQs were also calculated for individual beaches.
- **Step 3 –** The derivation of HQs over a relevant exposure area accounting for the ingestion of multiple prey species. This step was not conducted for sandpiper because of uncertainty in representative benthic invertebrate prey that sandpiper may ingest.

HQs in the first two steps were calculated according to EPA (2008j) direction, as outlined in EPA's Problem Formulation (Attachment 2). Results from the first two steps were used to narrow the list of COPCs for evaluation in the third step. For Steps 1 and 2, total HQs were calculated as the sum of the TSC HQ and the maximum TTC HQ. For Step 3, a proportion of the total diet for a given receptor was assigned to each of its prey items. Prey portions were used to derive total HQs using Equation 7-6 in Section 7.2.1.

Dietary risk conclusions are based on Step 3.[126] Results of the dietary LOE were further evaluated along with those from the bird egg LOE in light of the magnitude, spatial distribution, and frequency of HQs; the underlying uncertainties of exposure and effects data; and agreement of HQs for both LOEs (where applicable), as discussed in the risk conclusions for wildlife (Section 8.3).

8.1.2 COPCs Evaluated

This section presents the COPCs identified for evaluation for each wildlife receptor. It also briefly describes how TEQs were calculated for PCBs, dioxins, and furans. Finally, it documents the short list of COIs that could not be evaluated because of a lack of information about their toxicity to wildlife.

[126] As agreed to between EPA and LWG in the October 15, 2010, meeting (LWG 2010).

Receptor-COPC pairs were identified in the SLERA and refined screen (Table 8-1; see Attachment 5 for detailed methods and results). All but one of these receptor-COPC pairs were evaluated in a three-step wildlife risk evaluation process. Aluminum for birds[127] could not be evaluated because no baseline (LOAEL) TRV could be derived from the literature. The wildlife risk evaluation for aluminum is thus only an SL assessment. Aluminum is revisited in the risk characterization, where information about background concentrations is introduced to provide perspective on potential risk.

Table 8-1. Wildlife Dietary COPCs

	Birds				Mammals	
COPC	**Spotted Sandpiper**	**Hooded Merganser**	**Bald Eagle**	**Osprey**	**Mink**	**River Otter**
Metals						
Aluminum	X	X			X	X
Antimony					X	X
Arsenic	X					
Cadmium	X					
Chromium	X					
Copper	X	X			X	
Lead	X	X	X	X	X	X
Mercury	X	X	X	X	X	X
Selenium	X				X	
Thallium	X					
Zinc	X					
PAHs						
Benzo(a)pyrene	X	X		X		
Total HPAHs[a]					X	X
Total PAHs[a]	X					
Phthalates						
BEHP	X	X	X	X		
Dibutyl phthalate	X	X				
PCBs						
Total PCBs	X	X	X	X	X	X

[127] Aluminum was retained as a COPC because screening levels were exceeded in the spotted sandpiper and hooded merganser diets.

Table 8-1. Wildlife Dietary COPCs

COPC	Birds				Mammals	
	Spotted Sandpiper	Hooded Merganser	Bald Eagle	Osprey	Mink	River Otter
PCB TEQ[b]	X	X	X	X	X	X
Dioxins/Furans						
Total dioxin/furan TEQ[b]	X	X	X	X	X	X
Total TEQ[b]	X	X	X	X	X	X
Pesticides						
Aldrin	X					
4,4′-DDE			X	X		
Sum DDE	X	X	X			
Total DDx	X	X		X	X	X

[a] LPAH and HPAH were identified as COPCs for birds in the SLERA (Attachment 5) but (as per EPA direction) were evaluated as total PAHs (LWG 2010).

[b] Per EPA (Attachment 2), TEQ was evaluated as PCB TEQ, total dioxin/furan TEQ, and total TEQ.

BEHP – bis(2-ethylhexyl) phthalate
COPC – contaminant of potential concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
EPA – US Environmental Protection Agency
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
TEQ – toxic equivalent
total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT, and 4,4′-DDT)

TEQs for dioxins, furans, and PCBs were among the COPCs for birds and mammals. A TEQ is the TEF-weighted sum of constituent concentrations. TEQs were calculated consistent with EPA guidance (EPA 2008i). TEQ concentrations for birds and mammals were calculated using TEFs presented in Van den Berg et al. (1998; 2006). A constituent’s TEF is based on its affinity for binding to the aryl hydrocarbon (Ah) receptor relative to the Ah-binding affinity of 2,3,7,8-TCDD. The TEFs are derived from *in vivo* and *in vitro* studies. These data have been used to derive TEFs for PCB congeners that show structural similarity to dioxins and furans, bind to the Ah receptor, and elicit dioxin- or furan-specific biochemical and toxic responses. A key uncertainty in the TEQ approach is related to the derivation of consensus TEF values. Limitations in the underlying data used to derive TEFs, such as the relevance of the endpoints in the studies and the lack of information on interspecies variability, contribute to the uncertainty.

TEF Uncertainty

Among PCB congeners, the four most potent Ah receptor agonists in birds are the non-ortho PCB congeners 77, 81, 126, and 169. The variability in bird TEFs is high for PCB congeners that have been tested on multiple species (Van den Berg et al. 1998). For PCB 77, the five studies conducted resulted in a TEF range of < 0.0003 to 0.15 for ethoxyresorufin-O-deethylase (EROD) induction or *in ovo* effects in the various bird species tested. For PCB 81, two identified studies tested several species for EROD induction, with TEFs ranging widely from 0.001 to 0.5. For PCBs 126 and 169, data are available from only one study (*in ovo* with chickens). The relevance of TEFs derived by EROD induction or *in ovo* studies to risk assessment based on dietary exposure is also uncertain. It is not known if the uncertainties in the bird TEFs overestimate or underestimate risk.

The TEFs for mammals were derived from a large number of studies, with priority given to *in vivo* toxicity data over *in vitro* data. Despite the numerous biological variables such as species, strain, sex, and age included in these studies, the TEF values for a given congener generally fall within a range of about an order of magnitude for mammals(Sanderson and Van den Berg 1999). It is not known if the uncertainties in these TEFs overestimate or underestimate risk.

Nineteen dietary COIs could not be screened or otherwise evaluated because toxicological data are not available for birds and/or mammals (Table 8-2). The absence of bird and mammal data did not preclude these COIs from evaluation for other receptors (i.e., benthic invertebrates, fish, amphibians, plants,). Of these 19 COIs, 8 lacked avian toxicological data; these COIs were evaluated for mammals only (Table 8-2). In the absence of toxicological data, the dietary risks to birds and mammals from these chemicals are unknown.

Table 8-2. COIs Not Evaluated for Birds and/or Mammals

Metals	
Antimony[a]	Silver
Manganese	
PAHs	
1-Methylnaphthalene[a]	Dibenzothiophene[a]
2-Methylnaphthalene[a]	Perylene
Benzo(e)pyrene	Alkylated PAHs
SVOCs	
Benzoic acid	Dibenzofuran
Benzyl alcohol[a]	Hexachloroethane[a]
Carbazole	n-Nitrosodiphenylamine
Phenols	
2-Methylphenol	4-Methylphenol[a]
4-Chloro-3-methylphenol	Phenol[a]

[a] No bird dietary screening-level threshold was available; however, a mammal dietary threshold was available.

COI – contaminant of interest
SVOC –semivolatile organic compound
PAH –polycyclic aromatic hydrocarbon

In addition, dibutyl phthalate was not retained as a COPC for osprey because no detected prey tissue or sediment concentration exceeded the corresponding threshold concentration. Dibutyl phthalate was retained, however, as a COPC for spotted sandpiper and hooded merganser because maximum detected prey tissue concentrations (invertebrate tissue) exceeded receptor-specific TTCs. The dietary risk to osprey from dibutyl phthalate cannot be completely ruled out because at least one DL in a carp sample exceeded the TRV. However, carp are likely to constitute a small portion of the osprey diet (see Section 8.1.3.2.2). In all other samples of osprey prey, concentrations (where analyte was detected) or DLs (where analyte was not detected) were less than the TTC.

8.1.3 Exposure Assessment

This section presents the methods and assumptions used to estimate wildlife exposures to COPCs via prey tissue and surface sediment. Ultimately the prey tissue and sediment EPCs are combined (using an assumed fraction of sediment in the diet) to provide a single dietary exposure assessment for each receptor-COPC pair for each exposure area within the Study Area.

8.1.3.1 Exposure Point Concentrations

This section presents the methods used to derive EPCs in prey tissue and sediment for the dietary-dose approach. Prey tissue EPCs were evaluated to assess dietary risks. Sediment EPCs were evaluated to address potential exposure via incidentally ingested sediment. EPCs are represented by measured concentrations detected in composite samples collected from the Study Area or following laboratory bioaccumulation testing.[128]

As described in Section 8.0, the dietary LOE involved threes steps in which the data used to represent EPCs varied:

- **Step 1 –** EPCs were first represented by concentrations in composite tissue samples collected from individual prey species and by concentrations in composite samples of sediment from throughout the Study Area; this information was used to evaluate dietary risks on a sample-by-sample basis. COPC concentration data for all individual samples are presented in Attachment 4.
- **Step 2 –** For those receptor-COPC pairs whose sum of the maximum prey and sediment HQs was ≥ 1 in Step 1, prey tissue and sediment EPCs were then calculated as the UCL within the receptor-specific exposure areas (Table 8-3). The prey tissue and sediment EPCs are presented in Attachment 4.

The rationale for these exposure area assumptions is presented in Attachment 16. Maps 8-1 through 8-3 show the exposure areas for wildlife receptors. Where available data were insufficient for calculation of a UCL, the maximum concentration was used to represent the EPC.

[128] Chemical concentrations of neutral organic COPCs in tissues from bioaccumulation testing were adjusted for steady-state. See Attachment 3 for details.

UCL prey tissue and sediment EPCs were calculated using ProUCL Version 4.0 software (EPA 2007f). EPA's ProUCL software tests the goodness of fit for a given dataset and then computes the appropriate 95th UCL (as described in Section 7.1.3.1). Tissue data used to calculate UCLs were collected under EPA-approved QAPPs for the purpose of exposure calculation in this BERA (as described in Section 4). Under EPA guidance (EPA 2007f), both composite and discrete samples are appropriate for calculation of UCLs using ProUCL software. In the case where an insufficient number of detected data values was available (n < 6), the maximum concentrationwas used to represent the EPC.[129] EPCs based on tissue and sediment UCLs (or maximum concentrations) were used to calculate HQs using Equations 7-4 and 7-5.

Uncertainty is associated with the use of maximum concentrations to represent prey EPCs. The use of maximum values for small datasets carries a relatively high degree of uncertainty and provides relatively low confidence in estimating risk. Given the available data, the use of the maximum concentration as the EPC is considered the most appropriate estimator of the average, but it is less certain than a UCL based on a larger dataset. Maximum concentrations might over- or underestimate the the true population mean.

- **Step 3 –** To estimate dietary risks from ingestion of multiple prey species, a proportion of the total diet for a given receptor was assigned to each of its prey items. Prey portions were selected on the basis of diets reported in regional literature studies and are presented in Section 8.1.3.2.2. Prey portions were used to calculate HQs according to Equation 7-6. Risk conclusions are based on the exposure assumptions of Step 3.

Table 8-3. Summary of Receptor-Specific Exposure Area Assumptions

	Exposure Areas[a]			
Receptor	**Description**	**Location**	**Map**	**EPC Basis**
Spotted sandpiper	Shorebird beaches within 2-mile increments[b]	RM 1.9 to RM 3.9, RM 4.0 to RM 6.0, RM 7.0 to RM 9.0, RM 9.0 and above	Map 8-1	EPCs calculated within each 2-mile-increment exposure area for each prey species and beach sediment are based on UCL[b] COPC concentrations.
Hooded merganser, bald eagle, osprey, mink	1-mile increments	RM 1.5 to 2.5, RM 2.5 to 3.5, RM 3.5 to 4.5, RM 4.5 to 5.5, RM 5.5 to 6.5, RM 6.5 to 7.5, RM 7.5 to 8.5, Swan Island Lagoon, RM 8.5 to 9.5, RM 9.5 to 10.5, RM 10.5 to RM 11	Map 8-2	EPCs calculated within each 1-mile-increment exposure area for each prey species and sediment are based on UCL[c] COPC concentrations; EPCs were calculated as Study Area UCL[c] concentrations for prey with foraging ranges larger than a 1-mile area (i.e., carp, juvenile Chinook salmon, largescale sucker, and

[129] When the maximum concentration was a non-detected value, the full DL was used to represent the EPC.

Table 8-3. Summary of Receptor-Specific Exposure Area Assumptions

Receptor	Exposure Areas[a] Description	Exposure Areas[a] Location	Map	EPC Basis
				peamouth).
River otter	3-mile increments	RM 1.5 to 4.5, RM 4.5 to 7.5, RM 7.5 to 10.5, RM 10.5 and above	Map 8-3	EPCs calculated within each 3-mile-increment exposure area for each prey species and sediment are based on UCL[c] COPC concentrations; EPCs were calculated as Study Area UCL[c] concentrations for prey with foraging ranges larger than a 3-mile area (i.e., carp and largescale sucker).

[a] The rationale for selected exposure areas is presented in Attachment 16.
[b] Spotted sandpiper were also evaluated on a beach-by-beach basis, per EPA (2008j).
[c] When available data were insufficient for calculation of a UCL, the maximum concentration was used to represent the EPC.

COPC – contaminant of potential concern
EPA – US Environmental Protection Agency
EPC – exposure point concentration
RM – river mile
UCL – upper confidence limit on the mean

8.1.3.2 Exposure Parameters and Dietary Prey Assumptions

The following subsection presents the exposure parameters used for wildlife TTCs and TSCs. Dietary prey assumptions used to derive tissue EPCs are also presented.

8.1.3.2.1 Dietary-Dose Exposure Parameters

Body weights, FIRs, and sediment ingestion rates vary among bird and mammal receptors, as listed in Table 8-4. Details and the rationale for the receptor-specific exposure parameters selected as well as the associated uncertainties are provided in Attachment 16.

Table 8-4. Exposure Parameters Used for Wildlife Dietary Risk Calculations

Receptor	BW (kg)	BW Source	FIR (kg ww/day)	FIR Source	SI (%)[a]	% Moisture in Prey	SIR (kg dw/day)[b]
Spotted sandpiper	0.047	Maxson and Oring (1980)[c]	0.055	Nagy (2001)	18[d]	NA	0.0015
Hooded merganser	0.54	Dunning (1993)	0.20	Nagy (2001)	2[e]	NA	0.0011
Bald eagle	4.5	Wiemeyer (1991)[c]	0.54	Stalmaster and Gessaman (1984)[c]	2[e]	74%[f]	0.0028[g]
Osprey	1.9	Poole (1983)[c]	0.40	Poole (1983)[c]	2[e]	74%[f]	0.0021[g]
Mink	0.97	Hornshaw et al. (1983)[c]	0.16	Bleavins and Aulerich (1981)[c]	9.4[d, h]	74%[i]	0.0038[g]

Table 8-4. Exposure Parameters Used for Wildlife Dietary Risk Calculations

Receptor	BW (kg)	BW Source	FIR (kg ww/day)	FIR Source	SI (%)[a]	% Moisture in Prey	SIR (kg dw/day)[b]
River otter	7.7	USGS (2004)	0.76	Nagy (2001)	2[e]	NA	0.0047[g]

[a] Percent of incidental sediment ingestion.
[b] SIR = FIR × SI. The SIR was calculated as a percent of the FIR on a dw basis.
[c] As cited in EPA (1993).
[d] Based on Beyer et al. (1994).
[e] Based on best professional judgment.
[f] Average percent moisture in fish tissue collected from the Study Area.
[g] The SIR was calculated as a percent of the FIR on a dw basis. FIR (dw) = FIR (ww) × (1 - moisture content of diet).
[h] Sediment ingestion percent for mink based on raccoon. No data available for mink.
[i] Average percent moisture in fish and crayfish tissue collected from the Study Area.

BW – body weight
dw – dry weight
EPA – US Environmental Protection Agency
FIR – food ingestion rate
SI – sediment ingestion
SIR – sediment ingestion rate
NA – not applicable
USGS – US Geological Survey
ww – wet weight

Uncertainties Associated with Site Use and Incidental Sediment Ingestion

All wildlife receptors were assumed to forage solely within the site, per EPA's Problem Formulation (Attachment 2). Some receptors (particularly hooded merganser and bald eagle) probably forage in nearby aquatic and terrestrial environments and therefore use the site less than 100% of the time. Osprey also probably forage outside of the Study Area (fish prey are available from other water bodies in the area), and non-breeding osprey winter outside of the Study Area. This uncertainty is further evaluated in Section 8.1.5.1 as part of the risk characterization of hooded merganser, bald eagle, and osprey.

Incidental sediment ingestion rates (2% of the diet) for piscivorous birds (i.e., osprey and bald eagle) may overestimate actual sediment ingestion. However, because the contribution of incidental sediment on exposure concentrations is small, these incidental sediment assumptions do not affect risk estimates.

Dietary doses for all wildlife receptors are based primarily on female exposure parameters. Body weights and FIRs are based primarily on the literature presented in EPA's *Wildlife Exposure Factors Handbook* (EPA 1993). When species-specific data were not available from EPA (1993), FIRs were based on the allometric equations presented in Nagy (2001). Sediment ingestion rates were derived based on fraction of sediment in diets as reported by Beyer et al. (1994) or on best professional judgment when no data were available, using the following equation.

$$SIR = (FIR \times F_{solids}) \times SI \qquad \textbf{Equation 8-1}$$

Where:

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

To determine sediment ingestion rates on a dry-weight basis, FIRs based on dry weight were used as reported in the literature or were converted to dry weight based on the average percent moisture across relevant prey (Table 8-4).

8.1.3.2.2 Dietary Prey Assumptions

Fish and wildlife prey organisms in the BERA dataset include species collected in the Study Area and invertebrates that underwent laboratory bioaccumulation testing (i.e., clams and worms), as shown in Table 8-5. The species listed in the table were used to derive prey tissue EPCs. The diets selected for each receptor are based on the literature. Details regarding the selection of prey species are presented in Attachment 16, including the underlying assumptions and associated uncertainties.

Table 8-5. Receptor-Specific Prey Species Used to Derive Risk Estimates Based on Single Prey Consumption

	Birds				Mammals	
Prey Species	**Spotted Sandpiper**	**Hooded Merganser**	**Bald Eagle**	**Osprey**	**Mink**	**River Otter**
Invertebrates						
Crayfish		X			X	X
Clam	X[a]	X[a]				X[a]
Worm	X[b]					
Mussel					X	X
Fish						
Largescale sucker			X	X	X	X
Carp			X	X	X	X
Juvenile Chinook salmon					X	
Peamouth		X	X		X	
Sculpin		X			X	X
Smallmouth bass		X[c]		X	X	X
Northern pikeminnow			X	X	X	
Black crappie					X	X
Brown bullhead				X	X	

[a] Concentrations were evaluated for both laboratory- and field-collected clam tissue. Chemical concentrations of neutral organic COPCs in laboratory clam tissues were adjusted to steady-state concentrations.

[b] Worm data are based on laboratory tissue. Chemical concentrations of neutral organic COPCs in laboratory worm tissues were adjusted to steady-state concentrations.

[c] Per EPA (2008j), smallmouth bass were also used as a representative prey item for hooded mergansers. However, the size range of smallmouth bass collected from the Study Area (8.6 to 18 inches in length) is much greater than the prey size of fish consumed by mergansers; Use of this prey item to represent hooded merganser dietary concentrations introduces uncertainty.

COPC – contaminant of potential concern

EPA – US Environmental Protection Agency

Each prey species was evaluated individually in the first two steps of the wildlife risk evaluation process (i.e., it was assumed that the receptor diet consisted solely of that species).

Uncertainties Associated with Using Laboratory Bioaccumulation Testing Data to Represent Prey Chemical Concentrations

Uncertainty is associated with the use of lab worm and lab clam tissue concentrations to represent prey in the spotted sandpiper diet. Tissues were analyzed following 28-day laboratory bioaccumulation testing with field-collected sediment from the Study Area. Field and steady-state conditions may not be represented by tissue chemical concentrations determined in laboratory testing conditions because of the physical manipulation of sediments and possible changes in the chemical form that affect bioavailability and uptake.

COPCs with high K_{OW} values might not reach steady-state concentrations in tissues within the 28-day duration of the tests, but clam and worm tissue concentrations of neutral organic COPCs were adjusted to better estimate theoretical steady-state concentrations using the process in the Inland Testing Manual (EPA and USACE 1998). Attachment 3 presents the methods used to derive steady-state concentrations. The equations (based on McFarland (1995)) and assumptions (i.e., K_{OW} values) used to predict the steady-state adjusted concentrations are uncertain in that they do not reflect laboratory test conditions, a sediment matrix, or chemical mixtures. Adjusted clam and worm tissue chemical concentrations might over- or underestimate concentrations expected in Study Area field-collected clams and worms. Tissues in field-collected clams are more representative of field conditions than are tissues in laboratory-exposed clams.

For receptor-COPC pairs retained through the third step in the process, the proportional contributions of individual prey species were varied to better represent multi-species diets presented in the literature. Table 8-6 presents the portion assigned to each prey species to derive HQs. Details on the rationale for the selected prey portions are presented in Attachment 16. If no data were available for a given prey species, a surrogate prey species was used (e.g., if no data were available for largescale sucker, a species of similar trophic level-such as carp-was used to represent sucker concentrations). The effect on HQs of varying prey portions was evaluated as part of the uncertainty analysis (Section 8.1.5.2.1).

Table 8-6. Receptor-Specific Prey Species and Portions Used to Derive Risk Estimates Based on Multiple Prey Consumption

	Prey Consumption Portion					
	Birds				Mammals	
Prey Species	**Spotted Sandpiper[a]**	**Hooded Merganser**	**Bald Eagle**	**Osprey**	**Mink**	**River Otter**
Invertebrates						
Crayfish		0.05			0.20[b]	0.10
Clam	1.0[c]	0.25[d]				0.02[d]
Worm	1.0					
Fish						
Largescale sucker			0.45	0.83	0.20[e]	0.04[e]

Table 8-6. Receptor-Specific Prey Species and Portions Used to Derive Risk Estimates Based on Multiple Prey Consumption

	Prey Consumption Portion					
	Birds				Mammals	
Prey Species	Spotted Sandpiper[a]	Hooded Merganser	Bald Eagle	Osprey	Mink	River Otter
Carp			0.45	0.06	0.20	0.40
Peamouth		0.05	0.05			
Sculpin		0.65			0.20[f]	0.40
Smallmouth bass				0.02	0.20	0.04
Northern pikeminnow			0.05[g]	0.07[h]		
Black crappie						
Brown bullhead				0.02[i]		

a Two scenarios were evaluated for spotted sandpiper: one based on the ingestion of clams and one based on the ingestion of worms.
b Sculpin were used as a surrogate species when no crayfish data were available for the mink diet.
c HQs were calculated using laboratory clam tissues only at locations where field-collected tissue concentrations were not available. Field clams are more representative of concentrations in bivalves from the Study Area.
d HQs were calculated using only field clam tissue. Field clams are more representative of concentrations in bivalves from the Study Area.
e Carp was used as a surrogate species when no largescale sucker data were available for the mink and river otter diet.
f Smallmouth bass was used as a surrogate species when no sculpin data were available for the mink diet.
g Peamouth was used as a surrogate species when no northern pikeminnow data were available for the bald eagle diet.
h Smallmouth bass was used as a surrogate species when no northern pikeminnow data were available for the osprey diet.
i Smallmouth bass was used as a surrogate species when no brown bullhead data were available for the osprey diet.
HQ – hazard quotient

8.1.3.3 Data Used to Derive Spotted Sandpiper Exposure Concentrations

The exposure assessment for spotted sandpiper differs from that for the other wildlife receptors because the sandpiper forages only on beaches. Exposure parameters for spotted sandpiper are therefore discussed separately.

All sediment and tissue data from the Study Area were used to develop EPCs and evaluate risks to all wildlife receptors except sandpiper. In Steps 2 and 3 of the risk evaluation process, the sediment and tissue data used to evaluate spotted sandpiper EPCs were aggregated over individual shorebird beaches. Twenty-eight individual shorebird beaches were identified during the reconnaissance survey conducted as part of Round 2 sampling described in the July 16, 2004, memorandum from LWG to EPA (Saban and Andersen 2004). These beach areas represent potential exposure areas for shorebirds. Map 8-1 presents the shorebird beach locations (labeled B1 through B28).

Table 8-7 indicates the beach sediment transect samples that were collected and used to estimate spotted sandpiper EPCs at individual beaches. Prey tissue samples within or adjacent to beaches were also used to estimate spotted sandpiper EPCs at individual beaches. When possible, data from field-collected clams were used in preference to data from laboratory-exposed clams because the former are more representative of exposure concentrations in the Study Area. At certain beaches (i.e., B10, B11, B19, B21, B24, B27, and B28), field-collected clams were not analyzed for all COPCs; in these cases data from laboratory clams were used to represent clam EPCs. Chemical concentrations of neutral organic COPCs were steady-state adjusted in laboratory clam and worm tissues.

Table 8-7. Sediment and Tissue Data Used to Derive Risk Estimates for Spotted Sandpiper at Individual Beach Locations

Beach Area[a]	Transect Surface Sediment Samples	Clam Tissue Samples	Laboratory Worm Tissue Samples
B1	B001, B003, B005	CA02W	None[b]
B2	B002	FC001	LW001
B3	B004	FC002	LW002
B4	B006	None[b]	None[b]
B5	B007, 03B031	FC003, CA03W	LW003
B6	03B030, B008	FC004, FC005	LW004, LW005
B7	03B033	CA04W	None[b]
B8	04B024	None[b]	None[b]
B9	B010	FC008	LW008
B10	B011, B009	FC009[c]	LW009
B11	B012	FC011[c]	LW011
B12	04B023	FC012	FC012
B13	05B018	FC013	LW013
B14	B015	None[b]	None[b]
B15	06B022	FC016, 06R002	LW016
B16	B050, 07B024	FC017	LW017
B17	B018, 07B022	FC018, FC020, 07R003, 07R006	LW018, LW020
B18	B017	FC019	LW019
B19[a]	B019	FC021[c]	LW021
B20	B021, B022-1	FC024	LW024

Table 8-7. Sediment and Tissue Data Used to Derive Risk Estimates for Spotted Sandpiper at Individual Beach Locations

Beach Area[a]	Transect Surface Sediment Samples	Clam Tissue Samples	Laboratory Worm Tissue Samples
B21	07B023	FC022[c]	LW022
B22	B024	FC028	LW028
B23	B023	FC027-1	LW027-1, LW027-2
B24	B020, 09B024, 09B028, 08B032	FC026[c]	LW026, LW029
B25	09B026	FC031	LW031
B26	B025-1	FC030	LW030
B27	B026	FC032[c]	LW032
B28	09B027	FC033[c]	LW033

[a] Sandpipers were observed at these shorebird beach areas during the shorebird reconnaissance survey conducted in June 2004 (Saban and Andersen 2004).

[b] Tissue concentrations were calculated using the mechanistic model or using BSARs for those COPCs for which a relationship between sediment and tissue concentrations could be established.

[c] Clam concentrations are represented by laboratory clams when COPC was not analyzed in field clams.

BSAR – biota-sediment accumulation regression

COPC – contaminant of potential concern

At beaches where no clams or worms were collected (i.e., B1, B4, B7, B8, and B14), worm and clam tissue concentrations were estimated using either a mechanistic bioaccumulation model or site-specific BSARs. These predictive tools were selected to provide methodological consistency between BERA tissue-residue predictions and risk-based PRGs for the FS. The models are presented in the draft bioaccumulation modeling report for the Portland Harbor RI/FS (2009b). BSARs were developed for only those COPCs whose concentrations in co-located prey tissue (i.e., worm or field clam) and sediment demonstrate a predictable relationship.

The mechanistic model was available for predicting total PCB, pesticide, and dioxin and furan concentrations. The mechanistic model was not used for other COPCs because it is appropriate only for hydrophobic organic chemicals (Arnot and Gobas 2004). Site-specific BSARs were selected only for shorebird tissue COPCs that met appropriate regression analysis assumptions, had a statistically significant positive slope ($p \leq 0.05$), had an $r^2 \geq 0.30$, and were not modeled mechanistically. Details of the BSAR analysis and the mechanistic bioaccumulation model are presented separately (2009b).

The models used to estimate worm and clam tissue concentrations for shorebird COPCs are shown in Table 8-8. Seven COPCs were modeled mechanistically (i.e., total PCBs, PCB TEQ, total dioxin/furan TEQ, total TEQ, aldrin, sum DDE, and total DDx). Of shorebird COPCs that were not modeled mechanistically, only benzo(a)pyrene in clams, lead in clams, and lead in laboratory worms met the BSAR acceptability criteria noted above (Table 8-8). Modeled data are presented in Attachment 4. For benzo(a)pyrene in laboratory-exposed worms and the other two COPCs (i.e., copper and dibutyl phthalate) for which tissue concentrations could not be predicted, the the BSAR evaluation indicated no relationship between co-located sediment and tissue concentrations. This lack of relationship suggests that the organisms are bioregulating their tissue residues (e.g., for copper, an essential metal), that the exposure source is not limited to local sediments, or both. In the absence of either an empirical relationship between co-located sediment and tissue concentrations, or a mechanistic basis for relating the two, no BSAR can be developed. Therefore, no BSARs were developed for these COPCs, and no predicted shorebird prey tissue concentrations were calculated.

Table 8-8. Shorebird COPCs and Selected Models Used to Predict Prey Tissue Concentrations

	Field Clam		Lab Worm	
COPC	Tissue Concentration Predicted?	Selected Model	Tissue Concentration Predicted?	Selected Model
Metals				
Copper	No[a]	NA	No[a]	NA
Lead	Yes	BSAR	Yes	BSAR
PAHs				
Benzo(a)pyrene	Yes	BSAR	No[a]	NA
Phthalates				
Dibutyl phthalate	No[b]	NA	No[b]	NA
PCBs				
Total PCBs	Yes	Mechanistic model	Yes	Mechanistic model
PCB TEQ	Yes	Mechanistic model	Yes	Mechanistic model
Dioxins/Furans				
Dioxin/furan TEQ	Yes	Mechanistic model	Yes	Mechanistic model
Total TEQ	Yes	Mechanistic model	Yes	Mechanistic model
Pesticides				
Aldrin	Yes	Mechanistic model	Yes	Mechanistic model
Sum DDE	Yes	Mechanistic model	Yes	Mechanistic model
Total DDx	Yes	Mechanistic model	Yes	Mechanistic model

a Site-specific BSARs were not selected because these COPCs did not meet the appropriate BSAR analysis assumptions (Windward 2009b), did not have a statistically significant positive slope ($p < 0.05$), or had an $r^2 < 0.30$.

b No appropriate BSAR model could be developed because too few pairs of sediment and tissue detected concentrations were available (n = 5).

BSAR—biota-sediment accumulation regression
COPC – contaminant of potential concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
NA – not applicable
PCB – polychlorinated biphenyl
TEQ – toxic equivalent
total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT, and 4,4′-DDT)

8.1.4 Effects Assessment

This section presents the selected TRVs used to characterize effects for wildlife receptor-COPC pairs and the uncertainties associated with the values selected. Dietary-dose TRVs used in this assessment are expressed as mg/kg bw/day and are based on LOAELs and NOAELs derived from the toxicological literature. Dietary-dose TRVs were used to derive receptor-specific TTCs and TSCs following the methods described in Section 7.2.1.

A NOAEL and a LOAEL were selected for each COPC. TRVs for PCBs and dioxin/furans were selected for both total PCBs and TEQs (as total dioxin/furan TEQ, PCB TEQ, and total TEQ). The effects data presented in this section are assessed in combination with exposure data (presented in Section 8.1.3) in the risk characterization (Section 8.1.5).

Per EPA’s Problem Formulation (Attachment 2), LOAELs were used to assess effects at the population level for all receptors evaluated, except bald eagle. As directed in EPA’s Problem Formulation (Attachment 2), the assessment endpoint for wildlife receptors that are threatened, endangered, or of particular cultural significance is at the organism rather than population level. This variation applied only to bald eagles, and NOAELs were used to assess COPC effects on bald eagles.

Uncertainty is associated with the use of LOAELs to assess effects to populations, as LOAELs are based on organism-level effects. The endpoints used to derive the LOAEL for each COPC are discussed below to examine the ecological significance of TRV exceedances. See Section 8.11 for further discussion of how LOAELs were used in risk characterization.

The following subsections present the selected dietary-dose TRVs and the TTCs and TSCs that were back-calculated from the selected TRVs.

8.1.4.1 Selected Dietary TRVs

Bird and mammal TRVs for the BERA (EPA 2008f) were adopted for assessing risks to wildlife. Wildlife TRVs selected by EPA are based on the following hierarchy:

1. When available, NOAEL and LOAEL TRVs are based on EPA’s Eco-SSL documents.
2. If no Eco-SSLs are available, NOAEL and LOAEL TRVs are based on calculated dose values derived from the toxicological literature.

3. When TRVs are derived from the toxicological literature, EPA relied on LWG's comprehensive TRV review presented in Attachment 14 or on the TRVs reported in Sample et al. (1996). Attachment 16 presents the details, sources, and uncertainties associated with the selected TRVs. Attachment 14 includes a parallel effort conducted by the LWG to derive TRVs from available toxicological studies.

Generally, the EPA TRVs used in this BERA are similar to (i.e., within the same order of magnitude as) the TRVs recommended by LWG. This similarity arises because, in general, the same types of toxicological studies were considered in all three sources (i.e., Eco-SSLs, Sample et al. (1996), and LWG's review); differences arose because of variations in the assumptions (i.e., body weight, ingestion rate) and criteria used to select a TRV.

Derivation of TRVs from Eco-SSL Documents

EPA recommended that bird and mammal dietary NOAEL and LOAELTRVs be derived from data presented in Eco-SSL documents. However, per EPA (2008f), thresholds reported in the Eco-SSL documents are not appropriate as LOAEL TRVs: Eco-SSLs are based on NOAELs and are applicable only in SL assessments to ensure that all chemicals potentially contributing to risk are identified early in the process. Therefore for this BERA, the NOAELs used to derive Eco-SSLs were adopted for use as NOAEL TRVs. EPA derived corresponding LOAELs from the same datasets or studies on which the Eco-SSLs were based using the following approach:

- If the Eco-SSL was based on a NOAEL that was derived as a geometric mean of reported NOAELs in selected toxicological studies, then a LOAEL was derived as the geometric mean of LOAELs from the dataset presented in the Eco-SSL document.
- If the Eco-SSL was based on a NOAEL selected from a single study (e.g., the lowest appropriate study), then a LOAEL was selected from the same study, as presented in the Eco-SSL document.

Uncertainty is associated with the use of TRVs derived from toxicological data presented in the Eco-SSLs because the latter are based on studies that include multiple exposure pathways (i.e., dietary, gavage, drinking water). TRVs based on non-dietary exposure pathways such as gavage and drinking water do not represent the dietary pathway being evaluated. For wildlife receptors in the present assessment, drinking water ingestion involves a method of uptake and absorption different from that of the dietary pathway. Furthermore, drinking water is considered a minor pathway for wildlife receptors, and the bioavailability of chemicals from water may be different from that of food.

The TRVs for birds and mammals are presented in Tables 8-9 and 8-10, respectively, with their associated uncertainties. An uncertainty that applies to most of the TRVs arises when data from the test species are extrapolated to the receptor of concern. For the most part, receptor-specific toxicity data were not identified for any of the COPCs (the exceptions are mink TRVs for mercury, total PCBs, and 2,3,7,8-TCDD). When an Eco-SSL-based LOAEL or a lowest LOAEL is used as a TRV, the wildlife receptors in the Study Area are assumed to be as sensitive as the most sensitive species tested. This assumption reflects a conservative approach (except when based on receptor-specific effects data). In cases where few toxicological data points are available, the selected TRVs are highly uncertain and might either over- or underpredict risk because the range of species sensitivities is unknown.

Table 8-9. Bird Dietary-Dose TRVs

	TRV (mg/kg bw/day)			
COPC	**NOAEL**	**LOAEL**	**Source**	**Key Uncertainties**
Aluminum	157	NA	Carriere et al. (1986)	No LOAELs were identified; selected NOAEL TRV is unbounded and based on aluminum lactate, ionic form of aluminum, which is not directly comparable to the form present in the environment. According to ATSDR (2008), existing bioavailability data indicate that aluminum lactate has a higher bioavailability than the aluminum compounds typically found in the diet and drinking water.
Arsenic	2.24[a]	4.5[b]	Eco-SSL (EPA 2005c)	Lowest toxicity value in acceptable toxicological study reviewed (Attachment 14) is an order of magnitude higher than TRVs based on Eco-SSLs.
Cadmium	1.47[a]	6.34[b]	Eco-SSL (EPA 2005d)	The Eco-SSL is consistent with the lowest acceptable LOAEL reported in the literature (Attachment 14).
Chromium	2.66[a]	15.6[b]	Eco-SSL (EPA 2005e)	The Eco-SSL is consistent with the lowest acceptable LOAEL reported in the literature (Attachment 14).
Copper	4.05[a]	12.1[c]	Eco-SSL (EPA 2007a)	The selected LOAEL is lower than all bounded NOAELs and LOAELs reported in the literature. Literature-based LOAELs range from 29 to 66 mg/kg bw/day, and bounded NOAELs range from 16 to 47 mg/kg bw/day (Attachment 14).
Lead	1.63[a]	3.26[c]	Eco-SSL (EPA 2005f)	Lowest toxicity value in acceptable toxicological study reviewed (20 mg/kg bw/day) (Attachment 14) is an order of magnitude higher than TRVs based on Eco-SSLs.
Mercury	0.0064	0.064	Heinz (1975, 1979)	LOAEL is for several reproduction and chick behavior endpoints in mallard. NOAEL was extrapolated from the LOAEL using a factor of 10. Adverse effects on mallard at this LOAEL are uncertain. Heinz (1974, 1976) reported no effects on reproduction in mallards fed the same dose as the selected LOAEL, and a LOAEL 6.5 times as high. However, adverse effects on great egret growth and reproduction were reported at LOAELs similar to the selected LOAEL (Bouton et al. 1999; Spalding et al. 2000).
Selenium	0.29[a]	0.579[c]	Eco-SSL (EPA 2007c)	
Thallium	0.48	24	Hudson et al. (1984)	LOAEL is for 50% mortality in pheasants; only three toxicological studies (reported in two papers) were available. NOAEL was extrapolated from the LOAEL using a factor of 50.

Table 8-9. Bird Dietary-Dose TRVs

COPC	TRV (mg/kg bw/day) NOAEL	TRV (mg/kg bw/day) LOAEL	Source	Key Uncertainties
Zinc	66.1[a]	171[b]	Eco-SSL (EPA 2007d)	
TBT	6.8	16.9	Schlatterer et al. (1993)	LOAEL is for reduced reproduction in Japanese quail; only two toxicological studies were available.
Benzo(a)pyrene	0.28	1.4	Hough et al. (1993)	LOAEL is for reduced reproduction in Japanese quail; only two toxicological studies were available. TRV is based on weekly intramuscular injection. NOAEL was extrapolated from the LOAEL using a factor of 5.
Total PAHs[d]	40	NA	Patton and Dieter (1980)	No LOAELs were identified; selected NOAEL is based on growth of mallards exposed to petroleum hydrocarbon mixture that contained some individual PAHs not included as part of total PAHs. Only one toxicity study was available.
BEHP	1.1	11	Peakall (1974)	LOAEL was extrapolated from the NOAEL for effects on eggshell thickness in ringed dove using a factor of 10 based on SREL (1999), as cited by EPA (2008f). Lowest toxicity value in acceptable toxicological study reviewed (Attachment 14) is an order of magnitude higher than selected LOAEL (extrapolated from NOAEL).Only three toxicity studies were available.
Dibutyl phthalate	0.11	1.1	Peakall (1974)	No toxicological studies were available; NOAEL was extrapolated from the BEHP LOAEL using a factor of 10, as described above.
Total PCBs	0.29	0.58	Britton and Huston (1973)	TRVs are based on reduced egg hatchability in chickens, which have high sensitivity to PCBs compared with other species tested. A lower LOAEL for eggshell thinning in American kestrel was reported; however, the 5% magnitude of effect is unlikely to be ecologically significant. The LOAEL for mallards (15 mg/kg dw/day) is approximately 30 times as great as the selected LOAEL (Attachment 14), indicating the selected TRV may overestimate effects on ducks, such as hooded merganser.
PCB TEQ	1.4×10^{-5}	1.4×10^{-4}	Nosek et al. (1992)	Only two toxicological studies were identified; TRVs are based on several survival and reproduction endpoints in ring-necked pheasant following weekly intraperitoneal injection exposure to 2,3,7,8-TCDD.

Table 8-9. Bird Dietary-Dose TRVs

	TRV (mg/kg bw/day)			
COPC	**NOAEL**	**LOAEL**	**Source**	**Key Uncertainties**
Total dioxin/furan TEQ	1.4×10^{-5}	1.4×10^{-4}	Nosek et al. (1992)	Only two toxicological studies were identified; TRVs are based on several survival and reproduction endpoints in ring-necked pheasant following weekly intraperitoneal injection exposure to 2,3,7,8-TCDD.
Total TEQ	1.4×10^{-5}	1.4×10^{-4}	Nosek et al. (1992)	Only two toxicological studies were identified; TRVs are based on several survival and reproduction endpoints in ring-necked pheasant following weekly intraperitoneal injection exposure to 2,3,7,8-TCDD.
Total DDx	0.227[a]	2.27[c]	Eco-SSL (EPA 2007b)	The selected LOAEL is consistent with the lowest acceptable literature-based LOAEL (1.8 mg/kg bw/day), where eggshell thinning was statistically different from the control group with a difference of about 6% (1974). However, reproductive effects on field populations of birds are not documented for eggshell thinning of < 15 to 20% (Attachment 14).
Sum DDE	0.032	0.32	Mendenhall et al. (1983)	LOAEL is for eggshell breakage and reduced hatchling survival in barn owls. Per EPA (2008f), NOAEL was extrapolated from the LOAEL using a factor of 10. Literature-based NOAEL for eggshell thinning in American kestrel (Attachment 14) is 4 times as high as NOAEL extrapolated from the LOAEL.
Aldrin	0.008	0.04	DeWitt (1956)	LOAEL is for 97.5% reduction in quail survival; only two toxicological studies were available. NOAEL was extrapolated from the LOAEL using a factor of 5.

[a] NOAEL is based on the chemical-specific Eco-SSL.

[b] LOAELs are based on a geometric mean derived using the same data from which Eco-SSL was calculated.

[c] LOAEL was derived from on the same study as the NOAEL, which was used as the basis for the Eco-SSL.

[d] The TRV for total PAHs was used to evaluate bird exposure to PAHs rather than evaluating HPAHs and LPAHs separately.

BEHP – bis(2-ethylhexyl) phthalate
bw – body weight
COPC – contaminant of potential concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
Eco-SSL – ecological soil screening level
EPA – US Environmental Protection Agency
LOAEL – lowest-observed-adverse-effect level
NA – not available (no TRV selected)
NOAEL – no-observed-adverse-effect level
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
TBT – tributyltin
TCDD – tetrachlorodibenzo-*p*-dioxin
TEQ – toxic equivalent
total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT, and 4,4′-DDT)
TRV – toxicity reference value

Table 8-10. Mammal Dietary-Dose TRVs

COPC	TRV (mg/kg bw/day) NOAEL	TRV (mg/kg bw/day) LOAEL	Source	Key Uncertainties
Aluminum	34.4	75.8	Ondreicka et al. (1966), Golub et al. (1987)	TRVs are for reduced growth of mice. TRVs are based on ionic form of aluminum (aluminum lactate), which is not directly comparable to the form present in the environment.
Antimony	0.059[a]	0.59[b]	Eco-SSL (EPA 2005b)	TRVs are based on drinking water exposure; no toxicological studies reviewed (Attachment 14) reported dietary toxicity.
Copper	5.6[a]	9.34[b]	Eco-SSL (EPA 2007a)	Mink-specific toxicological studies reviewed (Attachment 14) reported dietary toxicity level slightly higher than TRVs based on Eco-SSLs.
Lead	4.7[a]	8.9[b]	Eco-SSL (EPA 2005f)	TRVs are based on drinking water exposure; dietary toxicological study reviewed (Attachment 14) reported toxicity level at an order of magnitude higher than TRVs based on Eco-SSLs.
Mercury	0.02	0.07	Dansereau et al. (1999)	TRVs are based on reduced reproduction in mink. Mink were fed field-collected fish (making up 40% of the prepared diet) that may have contained other, uncharacterized chemicals. A lower LOAEL for rat was not selected because relevance of rat to Study Area receptors is uncertain.
Selenium	0.143[a]	0.215[b]	Eco-SSL (EPA 2007c)	
Benzo(a)pyrene	2.0[c]	10	MacKenzie and Angevine (1981)	Only three toxicological studies were identified.
HPAHs	0.615[a]	3.07[b]	Eco-SSL (EPA 2007f)	TRVs are based on exposure only to benzo(a)pyrene; only three toxicological studies were identified.
Total PCBs	0.0074[c]	0.037	Restum et al. (1998)	TRVs are based on several mink reproduction endpoints. Mink were fed field-collected fish that contained detectable concentrations of other chemicals (e.g., dioxins/furans, DDTs, and other organochlorine pesticides). NOAEL was extrapolated from the LOAEL using a factor of 5.
PCB TEQ	2.2×10^{-7}	2.2×10^{-6}	Tillitt et al. (1996)	TRVs are based on several mink reproduction endpoints; similar LOAEL reported in additional mink studies; NOAEL was extrapolated from the LOAEL using a factor of 10.

Table 8-10. Mammal Dietary-Dose TRVs

COPC	TRV (mg/kg bw/day) NOAEL	TRV (mg/kg bw/day) LOAEL	Source	Key Uncertainties
Total dioxin/furan TEQ	2.2×10^{-7}	2.2×10^{-6}	Tillitt et al. (1996)	TRVs are based on several mink reproduction endpoints; similar LOAEL reported in additional mink studies; NOAEL was extrapolated from the LOAEL using a factor of 10.
Total TEQ	2.2×10^{-7}	2.2×10^{-6}	Tillitt et al. (1996)	TRVs are based on several mink reproduction endpoints; similar LOAEL reported in additional mink studies; NOAEL was extrapolated from the LOAEL using a factor of 10.
Total DDx	0.147[a]	0.735[b]	Eco-SSL (EPA 2007b)	Lowest toxicity value in acceptable toxicological study reviewed (Attachment 14) is twice as high as TRVs based on Eco-SSLs.

[a] NOAEL is based on the chemical-specific Eco-SSL.

[b] LOAEL was derived from on the same study as the NOAEL, which was used as the basis for the Eco-SSL.

COPC – contaminant of potential concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
Eco-SSL – ecological soil screening level
EPA – US Environmental Protection Agency
HPAH – high-molecular-weight polycyclic aromatic hydrocarbon
LOAEL – lowest-observed-adverse-effect level
NOAEL – no-observed-adverse-effect level
PCB – polychlorinated biphenyl
TEQ – toxic equivalent
total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT, and 4,4′-DDT)
TRV – toxicity reference value

8.1.4.2 Back-Calculated TTCs and TSCs

Once dietary TRVs were selected, receptor-specific TTCs and TSCs were back-calculated using receptor-specific parameters (i.e., body weight, prey ingestion rate, incidental sediment ingestion rate) following the methods described in Section 7.2.1. Tables 8-11 through 8-14 present the receptor-specific TTCs and TSCs derived for all bird and mammal receptor-COPC pairs.

Table 8-11. Calculated TTCs for Bird Receptor-COPC Pairs

COPC	Unit (ww)	TTC (Prey Tissue)							
		Spotted Sandpiper		Hooded Merganser		Bald Eagle		Osprey	
		NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL
Metals									
Aluminum	mg/kg	134	ND [a]	424	ND [a]	NA	NA	NA	NA
Arsenic	mg/kg	1.91	3.85	NA	NA	NA	NA	NA	NA
Cadmium	mg/kg	1.26	5.42	NA	NA	NA	NA	NA	NA
Chromium	mg/kg	2.27	13.3	NA	NA	NA	NA	NA	NA
Copper	mg/kg	3.46	10.3	10.9	32.7	NA	NA	NA	NA
Lead	mg/kg	1.39	2.79	4.4	8.8	13.6	27.2	7.74	15.5
Mercury	mg/kg	0.00547	0.0547	0.0173	0.173	0.0533	0.533	0.0304	0.304
Selenium	mg/kg	0.248	0.495	NA	NA	NA	NA	NA	NA
Thallium	mg/kg	0.41	20.3	NA	NA	NA	NA	NA	NA
Zinc	mg/kg	56.5	146	NA	NA	NA	NA	NA	NA
PAHs									
Benzo(a)pyrene	µg/kg	239	1,200	756	3,780	NA	NA	1,330	6,650
Total PAHs	µg/kg	34,200	NA	NA	NA	NA	NA	NA	NA
Phthalates									
BEHP	µg/kg	940	9,400	2,970	29,700	9,170	91,700	5,230	52,300
Dibutyl phthalate	µg/kg	94	940	297	2,970	NA	NA	NA	NA
PCBs									
Total PCBs	µg/kg	248	496	783	1570	2420	4,830	1,380	2,760

Table 8-11. Calculated TTCs for Bird Receptor-COPC Pairs

COPC	Unit (ww)	TTC (Prey Tissue)							
		Spotted Sandpiper		Hooded Merganser		Bald Eagle		Osprey	
		NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL
PCB TEQ	ng/kg	12	120	37.8	378	117	1,170	66.5	665
Dioxins/Furans									
Total dioxin/furan TEQ	ng/kg	12	120	37.8	378	117	1,170	66.5	665
Total TEQ	ng/kg	12	120	37.8	378	117	1,170	66.5	665
Pesticides									
Aldrin	µg/kg	6.84	34.2	NA	NA	NA	NA	NA	NA
Sum DDE	µg/kg	27.3	273	86.4	864	267	2670	152	1,520
Total DDx	µg/kg	194	1,940	613	6,130	NA	NA	NA	NA

[a] No data; no toxicological data were available for the derivation of an aluminum LOAEL TTC.

BEHP – bis(2-ethylhexyl) phthalate
COPC – contaminant of potential concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
LOAEL – lowest-observed-adverse-effect level
NA – not applicable (not a receptor-COPC pair)
ND – no data
NOAEL – no-observed-adverse-effect level
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
TEQ – toxic equivalent
total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT, and 4,4′-DDT)
TTC – threshold tissue concentration
ww – wet weight

Table 8-12. Calculated TSC for Bird Receptor-COPC Pairs

COPC	Unit (dw)	TSC							
		Spotted Sandpiper		Hooded Merganser		Bald Eagle		Osprey	
		NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL
Metals									
Aluminum	mg/kg	4,920	ND [a]	77,100	ND [a]	NA	NA	NA	NA
Arsenic	mg/kg	70.2	141	NA	NA	NA	NA	NA	NA
Cadmium	mg/kg	46.1	199	NA	NA	NA	NA	NA	NA
Chromium	mg/kg	83.3	489	NA	NA	NA	NA	NA	NA
Copper	mg/kg	127	379	1,990	5,940	NA	NA	NA	NA
Lead	mg/kg	51.1	102	800	1,600	2,620	5,240	1,470	2,950
Mercury	mg/kg	0.201	2.01	3.14	31.4	10.3	103	5.79	57.9
Selenium	mg/kg	9.09	18.1	NA	NA	NA	NA	NA	NA
Thallium	mg/kg	15	743	NA	NA	NA	NA	NA	NA
Zinc	mg/kg	2,070	5360	NA	NA	NA	NA	NA	NA
PAHs									
Benzo(a)pyrene	µg/kg	8,770	43,900	137,000	687,000	NA	NA	253,000	1,270,000
Total PAHs	µg/kg	1,250,000	NA	NA	NA	NA	NA	NA	NA
Phthalates									
BEHP	µg/kg	34,500	345,000	540,000	5,400,000	1,770,000	17,700,000	995,000	9,950,000
Dibutyl phthalate	µg/kg	3,450	34,500	54,000	540,000	NA	NA	NA	NA
PCBs									

Table 8-12. Calculated TSC for Bird Receptor-COPC Pairs

COPC	Unit (dw)	TSC Spotted Sandpiper NOAEL	TSC Spotted Sandpiper LOAEL	TSC Hooded Merganser NOAEL	TSC Hooded Merganser LOAEL	TSC Bald Eagle NOAEL	TSC Bald Eagle LOAEL	TSC Osprey NOAEL	TSC Osprey LOAEL
Total PCBs	µg/kg	9,090	18,200	142,000	285,000	466,000	932,000	262,000	525,000
PCB TEQ	ng/kg	439	4,390	6,870	68,700	22,500	225,000	12,700	127,000
Dioxins/Furans									
Total dioxin/furan TEQ	ng/kg	439	4,390	6,870	68,700	22,500	225,000	12,700	127,000
Total TEQ	ng/kg	439	4,390	6,870	68,700	22,500	225,000	12,700	127,000
Pesticides									
Aldrin	µg/kg	251	1,250	NA	NA	NA	NA	NA	NA
Sum DDE	µg/kg	1,000	10,000	15,700	157,000	51,400	514,000	29,000	290,000
Total DDx	µg/kg	7,110	71,100	111,000	1,110,000	NA	NA	NA	NA

[a] No data; no toxicological data were available for the derivation of an aluminum LOAEL TTC.

BEHP – bis(2-ethylhexyl) phthalate
COPC – contaminant of potential concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
dw – dry weight
LOAEL – lowest-observed-adverse-effect level
NA – not applicable (not a receptor-COPC pair)
ND – no data
NOAEL – no-observed-adverse-effect level
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
TEQ – toxic equivalent
total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT, and 4,4′-DDT)
TSC – threshold sediment concentration

Table 8-13. Calculated TTCs for Mammal Receptor-COPC Pairs

		TTC (Prey Tissue)			
		Mink		River Otter	
COPC	Unit (ww)	NOAEL	LOAEL	NOAEL	LOAEL
Metals					
Aluminum	mg/kg	206	461	344	770
Antimony	mg/kg	0.358	3.58	0.598	5.98
Copper	mg/kg	34	56.6	NA	NA
Lead	mg/kg	28.5	54	47.6	90.2
Mercury	mg/kg	0.121	0.424	0.203	0.709
Selenium	mg/kg	0.867	1.30	1.45	2.18
PAHs					
Total HPAHs	µg/kg	3,730	18,600	6,230	31,100
PCBs					
Total PCBs	µg/kg	44.9	224	75	375
PCB TEQ	ng/kg	1.33	13.3	2.23	22.3
Dioxins/Furans					
Total dioxin/furan TEQ	ng/kg	1.33	13.3	2.23	22.3
Total TEQ	ng/kg	1.33	13.3	2.23	22.3
Pesticides					
Total DDx	µg/kg	891	4,460	1,490	7,450

COPC – contaminant of potential concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
HPAH – high-molecular-weight polycyclic aromatic hydrocarbon
LOAEL – lowest-observed-adverse-effect level
NA – not applicable (not a receptor-COPC pair)
NOAEL – no-observed-adverse-effect level
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
TEQ – toxic equivalent
total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT, and 4,4′-DDT)
TTC – threshold tissue concentration
ww – wet weight

Table 8-14. Calculated TSCs for Mammal Receptor-COPC Pairs

		TSC			
		Mink		River Otter	
COPC	Unit (dw)	NOAEL	LOAEL	NOAEL	LOAEL
Metals					
Aluminum	mg/kg	8,680	19,400	55,700	125,000
Antimony	mg/kg	15.1	151	96.7	967
Copper	mg/kg	1,430	2,380	NA	NA
Lead	mg/kg	1,200	2,270	7,700	14,600
Mercury	mg/kg	5.11	17.9	32.8	115
Selenium	mg/kg	36.5	54.9	234	352
PAHs					
Total HPAHs	µg/kg	157,000	784,000	1,010,000	5,030,000
PCBs					
Total PCBs	µg/kg	1,890	9,440	12,100	60,600
PCB TEQ	ng/kg	56.2	562	360	3,600
Dioxins/Furans					
Total dioxin/furan TEQ	ng/kg	56.2	562	360	3,600
Total TEQ	ng/kg	56.2	562	360	3,600
Pesticides					
Total DDx	µg/kg	37,500	188,000	241,000	1,200,000

COPC – contaminant of potential concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
dw – dry weight
HPAH – high-molecular-weight polycyclic aromatic hydrocarbon
LOAEL – lowest-observed-adverse-effect level
NA – not applicable (not a receptor-COPC pair)
NOAEL – no-observed-adverse-effect level
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
TEQ – toxic equivalent
total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT, and 4,4′-DDT)
TSC – threshold sediment concentration

8.1.5 Risk Characterization and Uncertainty Analysis

The following section presents the risk characterization results for birds and mammals for the dietary LOE. An HQ approach was used to quantify risk estimates (Equation 6-1) following the three-step risk process described in Section 8.0. Section 8.1.5 presents the risk characterization results and uncertainty evaluation for each wildlife receptor. Section 8.1.5.4 summarizes the risk characterization for all bird and mammal receptors.

As part of an uncertainty evaluation, risks to piscivorous birds were characterized in a dietary-dose assessment conducted for belted kingfisher. Results are reported in Section 8.1.5.2.2.

8.1.5.1 Risk Characterization Results and Uncertainty Evaluation

The HQ results from the first two steps (presented in Attachment 17) were used to narrow the list of COPCs for evaluation in the third step. Table 8-15 lists the receptor-COPC pairs retained for further evaluation after Step 2 (i.e., HQs ≥ 1 based on individual prey). COPCs resulting in HQs < 1 in Step 2 were not included in Step 3. The following subsections present the dietary HQs for each bird and mammal receptor based on multiple prey within relevant exposure areas.

Table 8-15. Maximum HQs for Dietary COPCs Based on Individual Prey Species

COPC	Spotted Sandpiper	Hooded Merganser	Bald Eagle	Osprey	Mink	River Otter
Metals						
Aluminum	NC	NC	Not a COPC	Not a COPC	**1.8**	NA
Antimony	Not a COPC	Not a COPC	Not a COPC	Not a COPC	**1.6**	Not a COPC
Copper	**1.3**	0.67	Not a COPC	Not a COPC	0.48	Not a COPC
Lead	0.72	**130**	0.88	**71**	**20**	**12**
Mercury	NA	0.96	**9.4**	**1.6**	**1.3**	NA
PAHs						
Benzo(a)pyrene	**1.6**	NA	Not a COPC	NA	Not a COPC	Not a COPC
Phthalates						
BEHP	NA	**2.9**	NA	**1.7**	Not a COPC	Not a COPC
Dibutyl phthalate	**1.4**	NA	Not a COPC	Not a COPC	Not a COPC	Not a COPC
PCBs						
Total PCBs	**12**	**5.6**	**7.9**	**6.9**	**85**	**51**
PCB TEQ	**11**	NA	0.58	NA	**3.4**	**1.9**
Dioxins/Furans						
Total dioxin/furan TEQ	**17**	0.66	0.25	NA	**4.2**	**6.2**
Total TEQ	**20**	0.82	0.62	NA	**13**	**6.8**
Pesticides						
Aldrin	**1.7**	Not a COPC	Not a COPC	Not a COPC	Not a COPC	Not a COPC
Sum DDE	**1.3**	NA	**2.2**	NA	Not a COPC	Not a COPC
Total DDx	**1.4**	NA	Not a COPC	NA	NA	NA

BEHP – bis(2-ethylhexyl) phthalate
COPC – contaminant of potential concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
HQ – hazard quotient
NA – not applicable (Step 1 sum of maximum sample-by-sample tissue and sediment HQs < 1 [Attachment 17]).
Bold identifies HQs ≥ 1.
NC – not calculated (no LOAEL was available)
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
TEQ – toxic equivalent
total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT, and 4,4′-DDT)

8.1.5.1.1 Spotted Sandpiper

Twenty COPCs were identified for spotted sandpiper in the SLERA and refined screen. HQs could not be calculated for aluminum because no LOAEL was available. Eleven COPCs have HQs ≥ 1 in Step 1 based on individual prey components: copper, lead, benzo(a)pyrene, dibutyl phthalate, total PCBs, PCB TEQ, total dioxin/furan TEQ, total TEQ, aldrin, sum DDE, and total DDx (Attachment 17). All COPCs with HQs ≥ 1 were further evaluated on an individual beach basis (Attachment 17).

The spotted sandpiper home range was assumed, based on the literature, to be 2 miles, and the diet was represented by clams,[130] lab worms, and sediment (incidental ingestion only). Individual prey portions were not assigned to the representative prey species because both clam and worms are meant to be representative of benthic invertebrate prey that sandpiper may ingest. Two sets of HQs were calculated, one for ingestion of only clams and incidental sediment, and the other for ingestion of only lab worms and incidental sediment. HQs across each 2-mile exposure area based on UCLs are presented in Table 8-16.

Table 8-16. Spotted Sandpiper LOAEL HQs Within 2-Mile Beach Exposure Areas

Exposure Area (Beach Areas Within 2 Miles)	Approximate RM	Total HQ Based on Clam Diet	Total HQ Based on Lab Worm Diet
Copper			
B1-B6	RM 1.9 – RM 3.9	**1.1**	0.34[a]
B7-B13	RM 4.0 – RM 6.0	**1.3**	0.67[a]
B14-B24	RM 7.0 – RM 9.0	**1.1**	0.41
B25-28	RM 9.0 – RM 10.0	0.99[a]	0.31[a]
Lead			
B1-B6	RM 1.9 – RM 3.9	0.30	0.40
B7-B13	RM 4.0 – RM 6.0	0.44	0.59

[130] Clam data were represented by field clams; when no field clam data were available, lab clams were used.

Table 8-16. Spotted Sandpiper LOAEL HQs Within 2-Mile Beach Exposure Areas

Exposure Area (Beach Areas Within 2 Miles)	Approximate RM	Total HQ: Based on Clam Diet	Total HQ: Based on Lab Worm Diet
B14-B24	RM 7.0 – RM 9.0	0.62	0.72
B25-28	RM 9.0 – RM 10.0	0.34	0.41
Benzo(a)pyrene			
B1-B6	RM 1.9 – RM 3.9.0	0.013	0.067[a]
B7-B13	RM 4.0 – RM 6.0	0.044	**1.6**[a]
B14-B24	RM 7.0 – RM 9.0	0.059	0.13
B25-28	RM 9.0 – RM 10.0	0.0073[a]	0.034[a]
Dibutyl phthalate			
B1-B6	RM 1.9 – RM 3.9	0.0089	0.0089
B7-B13	RM 4.0 – RM 6.0	0.024	0.15
B14-B24	RM 7.0 – RM 9.0	**1.4**[a]	0.51[a]
B25-28	RM 9.0 – RM 10.0	0.085	0.19
Total PCBs			
B1-B6	RM 1.9 – RM 3.9	0.55	**11**
B7-B13	RM 4.0 – RM 6.0	0.24	**1.7**[a]
B14-B24	RM 7.0 – RM 9.0	**2.2**	**12**
B25-28	RM 9.0 – RM 10.0	0.95[a]	**7.7**[a]
Total Dioxin/Furan TEQ			
B1-B6	RM 1.9 – RM 3.9	0.018	0.13
B7-B13	RM 4.0 – RM 6.0	0.061	0.96[a]
B14-B24	RM 7.0 – RM 9.0	0.20	**17**
B25-28	RM 9.0 – RM 10.0	0.029[a]	0.12[a]
PCB TEQ			
B1-B6	RM 1.9 – RM 3.9	0.38	**10**[a]
B7-B13	RM 4.0 – RM 6.0	0.073[b]	0.58[a, b]
B14-B24	RM 7.0 – RM 9.0	0.56	**11**
B25-28	RM 9.0 – RM 10.0	0.15[a]	0.88[a]
Total TEQ			
B1-B6	RM 1.9 – RM 3.9	0.40	**11**[a]

Table 8-16. Spotted Sandpiper LOAEL HQs Within 2-Mile Beach Exposure Areas

Exposure Area (Beach Areas Within 2 Miles)	Approximate RM	Total HQ Based on Clam Diet	Total HQ Based on Lab Worm Diet
B7-B13	RM 4.0 – RM 6.0	0.12[b]	**1.3**[a]
B14-B24	RM 7.0 – RM 9.0	0.40	**20**[b]
B25-28	RM 9.0 – RM 10.0	0.18[a]	**1.0**[a]
Aldrin			
B1-B6	RM 1.9 – RM 3.9	0.0080	0.037[a]
B7-B13	RM 4.0 – RM 6.0	0.0088	0.056[a]
B14-B24	RM 7.0 – RM 9.0	0.039	**1.7**
B25-28	RM 9.0 – RM 10.0	0.011[a]	0.044[a]
Sum DDE			
B1-B6	RM 1.9 – RM 3.9	0.054	0.20
B7-B13	RM 4.0 – RM 6.0	0.073	0.44[a]
B14-B24	RM 7.0 – RM 9.0	0.17	**1.3**
B25-28	RM 9.0 – RM 10.0	0.047[a]	0.15[a]
Total DDx			
B1-B6	RM 1.9 – RM 3.9	0.016	0.044
B7-B13	RM 4.0 – RM 6.0	0.017	0.088
B14-B24	RM 7.0 – RM 9.0	0.099	**1.4**
B25-28	RM 9.0 – RM 10.0	0.010[a]	0.037[a]

[a] EPC based on maximum detected value.

[b] PCB TEQ and total TEQ were not available in sediment at exposure area B14-B24. Total HQ is based only on prey tissue.

DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
EPC – exposure point concentration
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
PCB – polychlorinated biphenyl
RM – river mile
TEQ – toxic equivalent
total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT, and 4,4′-DDT)

Bold identifies HQs ≥ 1.

Ten COPCs have HQs ≥ 1 within a 2-mile exposure area: copper, benzo(a)pyrene, dibutyl phthalate, total PCBs, total dioxin/furan TEQ, PCB TEQ, total TEQ, aldrin, sum DDE, and total DDx. Risk estimates from total dioxin/furan TEQ and PCB TEQ are components of risk from total TEQ because the total TEQ includes dioxin-like effects from both PCBs and dioxins/furans. Similarly, risk from sum DDE is a component of risk from total DDx.

Several uncertainties are associated with the assumptions used to derive the risk estimates, as described below. Uncertainty related to exposure assumptions is discussed first, followed by that related to effects assumptions.

- **FIR –** The assumed spotted sandpiper food ingestion rate (120% body weight per day) was estimated from a point estimate for common sandpiper, which was calculated from metabolic rates (2001). Use of the allometric equation for all birds (EPA 1993) results in a food ingestion rate of 110% body weight per day, 8% lower than that assumed in the exposure analysis. This difference in food ingestion rates for spotted sandpiper would have negligible influence on HQs. The three copper HQs of 1.1 to 1.3 would decrease to 1.0.
- **SIR –** The assumed sediment ingestion rate is 18% of the diet, on a dry-weight basis. This sediment ingestion rate is based on the mean of four other sandpiper species reported in Beyer et al. (1994), whose rates ranged from 7.3 to 30%. If it is assumed that the spotted sandpiper sediment ingestion rate could fall anywhere within this range, the sediment HQ component of the total HQ could be 59% lower or 67% higher than that listed in Table 8-15. For almost all chemical and exposure area scenarios this difference is insignificant, as sediment ingestion HQs for spotted sandpiper are usually less than 0.1, and often substantially less. Neither a higher or lower sediment ingestion rate would change any of the risk conclusions.
- **Use of clams and worms as prey species –** The spotted sandpiper diet was modeled based on the available benthic prey data for clams and worms; however, sandpipers are more likely to feed on amphipods and terrestrial and aquatic insects. Depending on the tissue concentrations in amphipods and insects relative to those of clams and worms, risk may be over- or underestimated. In addition, the use of lab worm and clam chemical concentrations adjusted for steady-state concentrations may not be representative of benthic invertebrate tissue concentrations in the field and may contribute to over- or underestimates of risks.
- **Prey portions –** Different but plausible assumptions about prey portions would not change HQs from ≥ 1 to < 1 or vice versa for any COPC (see Section 8.1.5.2.1).
- **Use of predicted rather than measured concentrations in prey species –** A mechanistic model or a BSAR was used to predict prey species concentrations where no data were available for certain beaches (e.g., B1, B4, B7, B8, and B14). The risk may have been over- or underestimated. A mechanistic model was used to predict concentrations in clam and worm tissue for total PCBs, PCB TEQ, total dioxin/furan TEQ, total TEQ, aldrin, sum DDE, and total DDx. A BSAR was used to predict clam concentrations for benzo(a)pyrene. For copper (clam and worm), and benzo(a)pyrene (worm only), no significant sediment–tissue

relationship existed, precluding development of BSARs. As a result, risks from copper could not be evaluated for beaches B1, B4, B7, B8, and B14; risks from benzo(a)pyrene at these same beaches could be evaluated for consumption of clams but not worms. Because dibutyltin data (clam and worm) were insufficient for development of BSARs, risks could not be evaluated for beaches B1, B4, B7, B8, and B14. The absence of a relationship between sediment and tissue concentrations means that there is not a relationship between dietary risk (should it occur) and sediment concentrations.

Additional uncertainties are associated with the effects data used to derive the risk estimate for each COPC. The TRVs (Table 8-9) represent an estimate of the effects thresholds for birds in general. No sandpiper-specific toxicity data were identified for any of the COPCs. The conservative assumption that sandpipers are as sensitive as the most sensitive bird species adds uncertainty and, except where noted below, may overestimate risks. Uncertainties associated with the effects assumptions are discussed below for each COPC whose HQ $\geq$ 1.

- **Copper –** The selected LOAEL (12.1 mg/kg bw/day) is based on an Eco-SSL (EPA 2007a) and is lower than all bounded NOAELs and LOAELs reported in the literature, suggesting that the TRV underestimates the effects threshold for birds and overestimates risk to the spotted sandpiper.
- **Lead –** The selected LOAEL (3.26 mg/kg bw/day) is based on an Eco-SSL (EPA 2005f) and is an order of magnitude lower than the lowest acceptable literature-based LOAEL (20 mg/kg bw/day) (Attachment 14), suggesting that this TRV underestimates the effects threshold for birds and overestimates risk to the spotted sandpiper.
- **Benzo(a)pyrene –** The degree of uncertainty is high because only two toxicological studies were identified and the TRV is based on the ecologically unrealistic exposure mechanism of intramuscular injection. The uncertainty about the exposure mechanism may result in an under- or overestimate of the effects threshold for birds and an over- or underestimate of risk to spotted sandpiper.
- **Dibutyl phthalate –** The degree of uncertainty is high because no toxicological data were identified. The use of the BEHP TRV as a surrogate may over- or underestimate risk. However, all LOAELs available in the acceptable toxicological literature are higher than the LOAEL TRV, which was extrapolated from a NOAEL per EPA (2008h), suggesting further that dibutyl phthalate risk estimates based on the BEHP LOAEL TRV are overestimated.
- **Total PCBs –** The TRV for total PCBs may overestimate risk to wild birds because it is based on a LOAEL for chickens, which have been shown to be highly sensitive to PCBs (see Attachment 14).
- **Total dioxin/furan TEQ, PCB TEQ, total TEQ (TRV is based on 2,3,7,8-TCDD) –** The degree of uncertainty is high because only two toxicological studies were identified and the selected TRV is based on the

ecologically unrealistic exposure mechanism of intramuscular injection. These uncertainties may result in an under- or overestimate of the effects threshold for birds and an over- or underestimate of risk to spotted sandpiper.

- **Aldrin –** The degree of uncertainty is high because only two toxicological studies were identified and the TRV is based on 97% reduction in survival. The uncertainties may result in an under- or overestimate of the effects threshold for birds and an over- or underestimate of risk to spotted sandpiper.
- **Sum DDE –** The degree of uncertainty is relatively low because the TRV is based on the lowest reported LOAEL from sufficient toxicity literature.
- **Total DDx –** The selected total DDx LOAEL (2.27 mg/kg bw/day) is based on Eco-SSL (EPA 2007b) and is consistent with the lowest acceptable literature-based LOAEL (1.8 mg/kg bw/day), where eggshell thinning was statistically different from the control group with a difference of about 6% (1974). However, the selected LOAEL may overpredict risk to populations, because reproductive effects in field populations of birds are not documented for eggshell thinning of < 15 to 20% (Attachment 14).

8.1.5.1.2 Hooded Merganser

Fourteen COPCs were identified for hooded merganser in the SLERA and refined screen. HQs could not be calculated for aluminum because no LOAEL was available. Three COPCs have HQs ≥ 1 based on individual prey components in Step 2: lead, BEHP, and total PCBs (Table 8-15). Individual prey items resulting in HQs ≥ 1 in Step 2 based on 1-mile exposure areas are clam for total PCBs; peamouth for lead; sculpin for total PCBs; and smallmouth bass for lead, BEHP, and total PCBs (Attachment 17). In Step 3, HQs were calculated for these COPCs across multiple prey species with incidental sediment ingestion using LOAEL-based TTCs and TSCs (Table 8-17).

Table 8-17. Hooded Merganser LOAEL HQs Within 1-Mile Exposure Areas

	Total HQ		
Exposure Area	**Lead[a]**	**BEHP[b]**	**Total PCBs[a]**
RM 1.5 to RM 2.5	0.097	0.0051	**1.5**
RM 2.5 to RM 3.5	0.013	0.0030	0.10
RM 3.5 to RM 4.5	0.054	0.011	0.26
RM 4.5 to RM 5.5	0.14	0.22	0.11
RM 5.5 to RM 6.5	0.039	0.0034	0.10
RM 6.5 to RM 7.5	0.21	0.0048	**1.8**
RM 7.5 to RM 8.5	0.032	0.0022	0.14
RM 8.5 to RM 9.5	0.071	0.0033	0.23
Swan Island Lagoon	0.047	0.67	0.33

Table 8-17. Hooded Merganser LOAEL HQs Within 1-Mile Exposure Areas

	Total HQ		
Exposure Area	Lead[a]	BEHP[b]	Total PCBs[a]
RM 9.5 to RM 10.5	0.034	0.0034	0.42
RM 10.5 to RM 11.8	0.033	0.0012	**3.8**

[a] Total HQ was calculated using the following prey portions: 65% sculpin, 5% peamouth, 25% field clams, and 5% crayfish.

[b] Total HQ was calculated using the following prey portions: 70% sculpin, 25% field clams, and 5% crayfish. Peamouth tissue was not analyzed for BEHP.

BEHP – bis(2-ethylhexyl) phthalate
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
PCB – polychlorinated biphenyl
RM – river mile
Bold identifies HQs ≥ 1.

Total PCBs has HQs ≥ 1 in Step 3 based on a multiple-prey diet in three exposure areas (i.e., RM 1.5 to RM 2.5, RM 6.5 to RM 7.5, and RM 10.5 to RM 11.8). HQs based on multiple prey for all exposure areas are < 1 for lead and BEHP.

Several uncertainties are associated with the assumptions used to derive the risk estimates, as described below. Uncertainty related to exposure assumptions is discussed first, followed by that related to effects assumptions.

- **FIR –** No additional food ingestion rate data beyond those used for the exposure analysis were identified for mergansers.
- **Prey portions –** Different but plausible assumptions about prey portions would not change HQs from ≥ 1 to < 1 or vice versa for any COPC (see Section 8.1.5.2.1). Additionally, the magnitude of total PCB HQs does not substantially change when calculated using prey portions matched to the relative abundance of prey in the Study Area, rather than baseline prey portions derived from the literature.
- **Size of smallmouth bass assumed as prey –** Smallmouth bass included in the samples used to model exposure (8.6 to 18 inches in length) are much larger than the size of fish consumed by mergansers (< 2 inches in length). Because fish tissue concentrations of PCBs tend to increase with body size (EPA 2009e), inclusion of smallmouth bass data probably overestimates PCB risks to merganser from ingestion of small fish in the Study Area.
- **Foraging range –** The assumption that hooded mergansers forage year-round within a 1-mile area is conservative. No published data from the literature are available to develop an alternative foraging area; however, because mergansers may forage in ponds and other aquatic environments outside of the Study Area, the assumed SUF of 1 may cause risk to be overestimated. A SUF of 0.75 would

reduce the HQs by 25%; however, HQs would remain ≥ 1 (ranging from 1.1 to 2.9)[131] in the three exposure areas.

Additional uncertainty is associated with assumptions about effects on hooded merganser. No hooded merganser-specific toxicity data were identified for any of the COPCs. The conservative assumption that mergansers are as sensitive as the most sensitive bird species adds uncertainty and, except where noted, may overestimate risk. Uncertainties associated with the effects assumptions for lead and total PCBs for hooded merganser are as discussed above for spotted sandpiper. There is high uncertainty associated with BEHP because only three toxicological studies were identified, and the LOAEL TRV was extrapolated from a NOAEL.

8.1.5.1.3 Bald Eagle

Eight COPCs were identified for bald eagle in the SLERA and refined screen for the dietary-dose assessment. Three COPCs have dietary-dose HQs ≥ 1 based on individual prey components in Step 2: mercury, total PCBs, and sum DDE (Table 8-15). Individual prey items resulting in HQs ≥ 1 over 1-mile exposure areas in Step 2 are carp for total PCBs, largescale sucker for mercury, northern pikeminnow for mercury and sum DDE, and peamouth for mercury (Attachment 17). In Step 3, dietary-dose HQs were calculated for these COPCs across multiple prey species and incidental sediment ingestion using NOAEL-based TTCs and TSCs, as shown in Table 8-18.

Table 8-18. Bald Eagle NOAEL HQs Within 1-Mile Exposure Areas

	Total HQ[a]		
Exposure Area	**Mercury**	**Total PCBs**	**Sum DDE**
RM 1.5 to RM 2.5	**1.2**	**3.8**	0.63
RM 2.5 to RM 3.5	**1.3**	**3.9**	0.64
RM 3.5 to RM 4.5	**1.2**	**3.8**	0.63
RM 4.5 to RM 5.5	**1.5**	**3.8**	0.65
RM 5.5 to RM 6.5	**1.5**	**3.8**	0.65
RM 6.5 to RM 7.5	**1.7**	**3.9**	0.71
RM 7.5 to RM 8.5	**1.5**	**3.9**	0.62
RM 8.5 to RM 9.5	**1.5**	**3.9**	0.63
Swan Island Lagoon	**1.3**	**3.9**	0.61
RM 9.5 to RM 10.5	**1.2**	**3.8**	0.63
RM 10.5 to RM 11.8	**1.2**	**3.8**	0.63

[131] The HQ of 1.5 multiplied by 75% = 1.1; the HQ of 3.8 multiplied by 75% = 2.9.

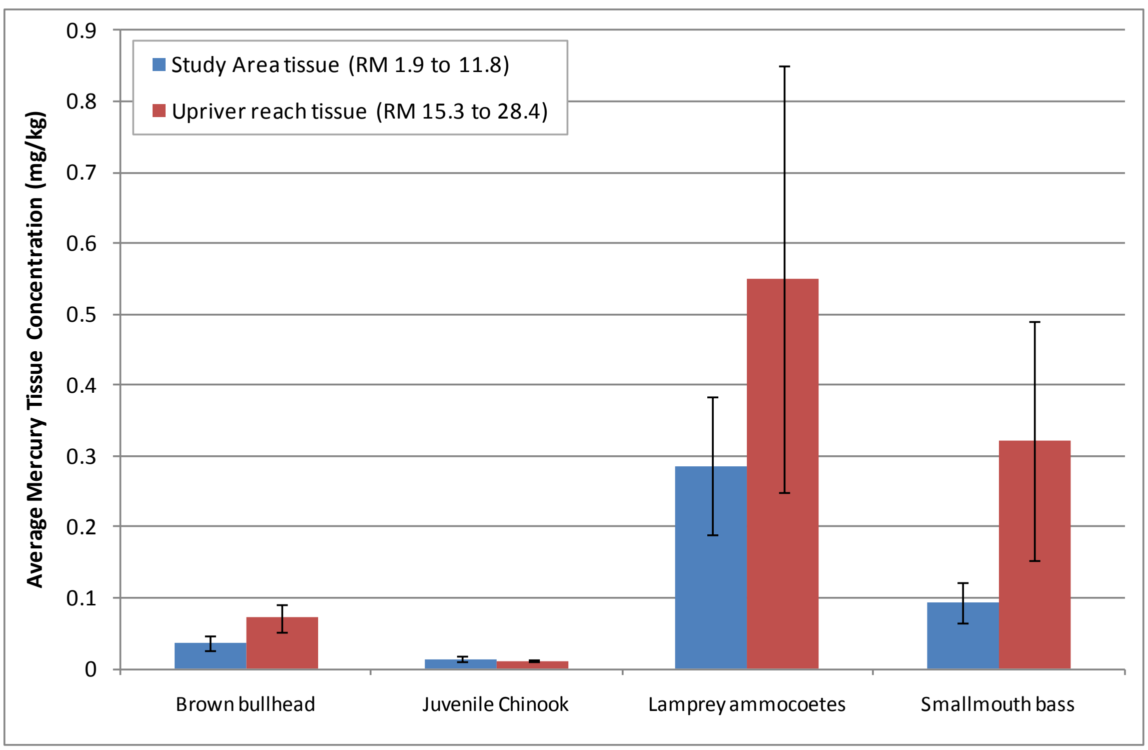
[a] Total HQ was calculated using the following prey portions**:** 45% carp, 45% largescale sucker, 5% peamouth, and 5% northern pikeminnow. When no northern pikeminnow data were available, peamouth were assigned a prey portion of 10%.

DDE – dichlorodiphenyldichloroethylene
HQ – hazard quotient
NOAEL – no-observed-adverse-effect level
PCB – polychlorinated biphenyl
RM – river mile

Bold identifies HQs ≥ 1.

In all exposure areas, mercury and total PCBs have HQs ≥ 1 for a diet of multiple prey species. HQs for all exposure areas are < 1 for sum DDE.

Mercury concentrations are of concern throughout the Willamette River Basin (Oregonian 2006). Average mercury concentrations in brown bullhead, lamprey ammocoetes, and smallmouth bass were greater in tissues collected from the upriver reach (RM 15.3 to RM 28.4) than in those collected from the Study Area (Figure 8-2). Upriver reach data were not available for fish species assigned as bald eagle prey (i.e., northern pikeminnow, largescale sucker, carp, and peamouth). The number of whole-body tissue composite samples, the number of fish in each composite, and the weights and lengths of fish species collected in the Study Area and at upstream locations are noted Section 7.1.5.2.2. Except for smallmouth bass, upriver and Study Area fish were of similar sizes; the smallmouth bass collected upriver were larger than those from the Study Area. Since mercury is highly bioaccumulative, fish tissue burdens typically increase with fish age and size (EPA 2000a).



Note: Error bars represent standard deviation of the mean.

Figure 8-2. Comparision of Study Area and Upriver Reach Mercury Tissue Concentrations in Fish

Willamette River-wide average mercury concentrations in tissue of carp, largescale sucker, northern pikeminnow, and smallmouth bass (as reported by Hope (2003)) are approximately 2 to 6 times as high as those in the Study Area. For example, northern pikeminnow mercury tissue concentrations throughout the Willamette River averaged 0.60 mg/kg ww while those in the Study Area averaged 0.28 mg/kg ww (Table 8-19). The variability of concentrations, as reflected in the coefficient of variation for mercury in northern pikeminnow tissue, was similar: 0.52 mg/kg ww for the Willamette basin and 0.54 mg/kg ww for the Study Area.[132] Although Study Area concentrations exceed bald eagle TTCs, the same is true for concentrations in fish from other portions of the Willamette River.

Table 8-19. Mean Fish Tissue Mercury Concentrations

	Fish from Study Area		Fish from Entire Willamette Basin[a]	
Species	**n**	**Mercury Concentration (mg/kg ww) (standard deviation)**	**n**	**Mercury Concentration (mg/kg ww) (standard deviation)**
Carp	15	0.045 (0.0088)	64	0.28 (0.18)
Largescale sucker	6	0.068 (0.016)	135	0.22 (0.16)
Northern pikeminnow	6	0.28 (0.15)	95	0.60 (0.31)
Smallmouth bass	32	0.093 (0.028)	10	0.28 (0.19)

[a] As reported by Hope (2003).
n – number of samples
ww – wet weight

Several uncertainties are associated with the assumptions used to derive the risk estimates, as described below. Uncertainty related to exposure assumptions is discussed first, followed by that related to effects assumptions.

- **FIR –** The FIR assumed in the exposure analysis (12% body weight per day) is at the upper end of the range reported by EPA (1993) (i.e., 6.5 to 14% body weight per day). Alternative but nonetheless plausible FIRs could yield HQs for all COPCs that range from approximately 45% lower to 17% greater than those calculated above. Such lower ingestion rates could cause HQs for mercury and total PCBs to fall below 1; however, the FIR used in the BERA is appropriately conservative given the range reported by EPA (1993).
- **Use of only fish as prey species –** Bald eagles were assumed to ingest 100% fish. However, eagles are also known to feed on aquatic and terrestrial birds and mammals, carrion, and garbage. Therefore, exposure estimates may underestimate risk to bald eagles if they have preyed on aquatic birds and mammals that have bioaccumulated contaminants through the Study Area aquatic food web. Or, more likely, exposure estimates may overestimate risk to bald eagles whose diet

[132] The coefficient of variation is the standard deviation divided by the mean.

includes terrestrial prey, prey species that have not accumulated aquatic contaminants from the Study Area, or both.

- **Prey portions –** As discussed below in Section 8.1.5.2.1, varying the prey portions in the diet is unlikely to lower HQs from ≥ 1 to < 1 for any COPC or vice versa. Mercury HQs could be slightly underestimated and total PCBs HQs could be overestimated.
- **Use of surrogate prey data –** Northern pikeminnow tissue data were not available for any COPC in four exposure areas (RM 1.5 to RM 2.5, RM 3.5 to RM 4.5, RM 9.5 to RM 10.5, and RM 10.5 to RM 11.8) and northern pikeminnow tissue BEHP data were not available for any exposure area. Peamouth COPC data were used as a surrogate in these exposure areas. In exposure areas where data were available for both species, concentrations of mercury and total PCBs were on average somewhat lower in peamouth. Because northern pikeminnow constitutes only 5% of the bald eagle diet, this uncertainty is not likely to affect risk conclusions.
- **Foraging range –** The 1-mile foraging range assumed for bald eagle per EPA's Problem Formulation (Attachment 2) is smaller than the home ranges of lower Columbia River breeding bald eagles, reported as 3.5 miles by Garrett et al. (1993). However, given the fact that the majority of the bald eagle diet is modeled using prey with large home ranges (i.e., carp and largescale sucker), an increase in the exposure area is unlikely to change the risk estimates for bald eagle.
- **Site use –** The assumption that bald eagles forage year-round only within the Study Area (i.e., a SUF of 1) is conservative. Because bald eagles forage in the Study Area for only part of the year and in the terrestrial environment as well as the Study Area, a SUF of 0.5 is probably more realistic. Although use of this lower value reduces the HQs by 50%, HQs for total PCBs would remain ≥ 1; mercury HQs would fall to < 1.

Additional uncertainty is associated with assumptions about effects on bald eagles. No bald eagle-specific toxicity data were identified for any of the COPCs. The conservative assumption that eagles are as sensitive as the most sensitive bird species adds uncertainty and, except where noted, may overestimate risks.

Uncertainties associated with the effects assumptions for total PCBs for the bald eagle are as discussed above for spotted sandpiper. Uncertainties associated with mercury and sum DDE are as follows.

- **Mercury** – The NOAEL TRV was extrapolated from a mallard LOAEL using a factor of 10. In a study other than that reporting the selected LOAEL, effects on mallard were not observed at higher concentrations.
- **Sum DDE** – The NOAEL was extrapolated from a LOAEL using a factor of 10; as discussed in Section 8.1.4.1, literature-based NOAELs suggest this extrapolation may overestimate risks.

8.1.5.1.4 Osprey

Ten COPCs were identified for osprey in the SLERA and refined screen for the dietary-dose assessment. Four COPCs had dietary-dose HQs ≥ 1 based on individual prey components in Step 2: lead, mercury, BEHP, and total PCBs (Table 8-15). Individual prey items resulting in HQs ≥ 1 within 1-mile exposure areas are carp for total PCBs; northern pikeminnow for mercury; and smallmouth bass for lead, BEHP, and total PCBs (Attachment 17). Dietary-dose HQs were calculated for these COPCs across multiple prey species and incidental sediment ingestion using LOAEL-based TTCs and TSCs, as shown in Table 8-20:

Table 8-20. Osprey LOAEL HQs Within 1-Mile Exposure Areas

	Total HQ[a]			
Exposure Area	**Lead**	**Mercury**	**BEHP**	**Total PCBs**
RM 1.5 to RM 2.5	0.023	0.25	0.047	0.92
RM 2.5 to RM 3.5	0.018	0.29	0.049	0.89
RM 3.5 to RM 4.5	0.031	0.28	0.20	0.91
RM 4.5 to RM 5.5	0.045	0.32	0.048	0.88
RM 5.5 to RM 6.5	0.028	0.32	0.049	0.88
RM 6.5 to RM 7.5	0.099	0.38	0.048	0.94
RM 7.5 to RM 8.5	0.023	0.34	0.048	0.91
RM 8.5 to RM 9.5	0.040	0.34	0.048	0.91
Swan Island Lagoon	0.027	0.28	0.051	0.93
RM 9.5 to RM 10.5	**7.8**	0.28	0.053	0.89
RM 10.5 to RM 11.8	0.024	0.28	0.053	**1.1**

[a] Total HQ was calculated using the following prey portions**:** 83% largescale sucker, 7% northern pikeminnow, 6% carp, 2% smallmouth bass, and 2% brown bullhead. When no northern pikeminnow data were available, smallmouth bass were assigned a prey portion of 9%, and when no brown bullhead data were available, smallmouth bass were assigned an additional prey portion of 2%.

BEHP – bis(2-ethylhexyl) phthalate
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
PCB – polychlorinated biphenyl
RM – river mile

Bold identifies HQs ≥ 1.

HQs based on multiple prey in Step 3 were ≥ 1 for lead and total PCBs, each in only one exposure area. HQs based on multiple prey are < 1 for mercury and BEHP in all exposure areas. The lead HQ is ≥ 1 for osprey only in the exposure area between RM 9.5 and RM 10.5. The HQ was calculated under the assumption that smallmouth bass constitute 11% of the osprey diet. All other osprey prey data result in HQs < 1. The maximum lead concentration from smallmouth bass collected at this exposure area (RM 9.5 to RM 10.5) was 1,100 mg/kg ww, which is over 100 times that for lead in the other smallmouth bass from this exposure area (6.8 mg/kg ww, collected from the east bank between RM 9.5 and RM 10.5) and 100 to 100,000 times that detected in smallmouth bass from all other areas (0.0048 to 1.8 mg/kg ww). The single composite sample is an outlier for both antimony and lead, suggesting that a fish in the sample might have swallowed a fishing sinker. Antimony can be mixed with lead as a hardener for lead-based products (ATSDR 1992). For example, one fish tackle supplier notes that fishing sinkers contain 94% lead and 6% antimony for hardness and color (Blue Ocean Tackle 2011) Without this sample, the lead HQ would be well below 1 and similar to all the other exposure areas.

Several uncertainties are associated with the assumptions used to derive the risk estimates, as described below. Uncertainty related to exposure assumptions is discussed first, followed by that related to effects assumptions.

- **FIR –** The assumed osprey FIR of 21% body weight per day appears to be slightly conservative. This value is a point estimate based on a population in Massachusetts ((Poole 1983), as cited in EPA (1993)). When the allometric equation for all birds and the range of female osprey body weights reported by EPA (1993) is used instead, the resulting FIRs are 18 to 19% body weight per day, a value 10 to 14% lower than the default rate. Slightly lower FIRs would have negligible influence on the risk conclusions. The total PCBs HQ of 1.1 at RM 10.5 to RM 11.8 would fall to slightly less than 1 if the 18% FIR was assumed.
- **Prey portions –** Osprey prey assumptions are based on region-specific studies (Attachment 16). As discussed below in Section 8.1.5.2.1, varying the prey portions in the diet within the range of plausible assumptions consistent with region-specific studies would affect the risk conclusions. Specifically, if smallmouth bass were assumed to constitute a smaller fraction of the osprey diet, the lead HQ would drop from ≥ 1 to < 1. The same would be true for the total PCBs HQ if carp and smallmouth bass were excluded as components of the osprey diet.
- **Site use –** The assumption that osprey forage year-round only within the Study Area (i.e., a SUF of 1) is conservative. Osprey probably forage outside of the Study Area (fish prey are available from other water bodies in the area); in addition, osprey are migratory and winter outside of the Study Area in the southern U.S., Mexico, and Central America.
- **Use of surrogate prey data for northern pikeminnow –** Northern pikeminnow tissue data were not available for any COPC in four exposure areas (RM 1.5 to RM 2.5, RM 3.5 to RM 4.5, RM 9.5 to RM 10.5, and RM 10.5 to RM 11.8). Northern pikeminnow tissue BEHP data were not available for any exposure area. Smallmouth bass COPC data were used as a surrogate in these exposure areas. In exposure areas where data were available for both species, concentrations of lead were on average somewhat higher in smallmouth bass, concentrations of mercury were somewhat lower in smallmouth bass, and concentrations of total PCBs were similar between species. Northern pikeminnow were assumed to constitute only

7% of the osprey diet; except for lead as discussed above (under prey portions), the uncertainty associated with the use of surrogate data is not likely to affect risk conclusions.

- **Use of surrogate prey data for brown bullhead –** Brown bullhead data were not available for any COPC in three exposure areas (RM 1.5 to RM 2.5, RM 9.5 to RM 10.5, and RM 10.5 to RM 11.8); smallmouth bass data were used as a surrogate in these exposure areas. In exposure areas where data were available for both species, concentrations of all COPCs were on average higher in smallmouth bass than brown bullhead. Brown bullhead were assumed to constitute only 2% of the osprey diet; except for lead as discussed above (under prey portions), the uncertainty associated with use of the surrogate data is not likely to affect risk conclusions.

Additional uncertainty is associated with assumptions about effects on osprey. No osprey-specific toxicity data were identified for any COPC. The conservative assumption that ospreys are as sensitive as the most sensitive bird species adds uncertainty and, except where noted, may overestimate risks.

Uncertainties associated with the effects assumptions for lead and total PCBs are as discussed above for spotted sandpiper. Uncertainties associated with mercury and BEHP are as follows:

- **Mercury –** Uncertainty in the effects threshold for birds arises from the absence of observed effect in the most sensitive species (mallard) at concentrations higher than the selected TRV in a study other than that reporting the LOAEL used; however, a similarly low LOAEL was reported for a different species in a separate study.
- **BEHP –** Uncertainty in the effects threshold for birds is due to the availability of only three acceptable studies. Uncertainty in the selected LOAEL arises because it was extrapolated from a NOAEL.

A further consideration in interpreting the HQs presented above is field data that provide a direct measure of local osprey populations. As discussed in Section 2.2.3.1.4, nesting success and population growth of osprey throughout the Willamette River system, including the Study Area, have increased from 1993 to 2001 (Henny et al. 2009). These data indicate that the osprey nesting population in the LWR (including the Study Area) has increased in recent years and that the productivity is above that necessary to maintain a stable population.

8.1.5.1.5 Mink

Twelve COPCs were identified for mink in the SLERA and refined screen. Eight COPCs have HQs ≥ 1 based on individual prey components in Step 2: aluminum, antimony, lead, mercury, total PCBs, PCB TEQ, total dioxin/furan TEQ, and total TEQ (Table 8-15). Individual prey items resulting in HQs ≥ 1 within 1-mile exposure areas in Step 2 are black crappie for total PCBs; brown bullhead for total PCBs; carp for total PCBs;

crayfish for total PCBs, total dioxin/furan TEQ, and total TEQ; largescale sucker for total PCBs; peamouth for total PCBs; northern pikeminnow for mercury and total PCBs; sculpin for total PCBs, PCB TEQ, total dioxin/furan TEQ, and total TEQ; smallmouth bass for antimony, lead, total PCBs, PCB TEQ, total dioxin/furan TEQ, and total TEQ; and sediment for aluminum and total TEQ (Attachment 17). In Step 3, HQs were calculated for these COPCs across multiple prey and incidental sediment ingestion using LOAEL-based TTCs and TSCs, as shown in Table 8-21.

Table 8-21. Mink LOAEL HQs Within 1-Mile Exposure Areas

	Total HQ							
Exposure Area	**Aluminum[a]**	**Antimony[a]**	**Lead[a]**	**Mercury[a]**	**Total PCBs[a]**	**PCB TEQ[b]**	**Dioxin/Furan TEQ[b]**	**Total TEQ[b]**
RM 1.5 to RM 2.5	**1.6**	0.0036	0.018	0.14	**23**	**2.1**	0.19	**2.3**
RM 2.5 to RM 3.5	**1.4**	0.011	0.015	0.15	**19**	**1.7**	0.22	**1.9**
RM 3.5 to RM 4.5	**1.5**	0.018	0.030	0.16	**20**	**1.7**	0.23	**1.9**
RM 4.5 to RM 5.5	**1.5**	0.0077	0.055	0.16	**19**	**1.4**	0.21	**1.6**
RM 5.5 to RM 6.5	**1.3**	0.0051	0.025	0.17	**19**	**1.4**	0.24	**1.6**
RM 6.5 to RM 7.5	**1.5**	0.014	0.12	0.28	**22**	**1.5**	**2.0**	**12[c]**
RM 7.5 to RM 8.5	**1.6**	0.0079	0.018	0.19	**19**	**1.4**	0.27	**1.7**
RM 8.5 to RM 9.5	**1.6**	0.026	0.041	0.20	**20**	**1.7**	0.24	**1.8**
Swan Island Lagoon	**1.5**	0.0063	0.025	0.15	**23**	**1.6**	0.27	**1.8**
RM 9.5 to RM 10.5	**1.4**	0.33	**4.0**	0.18	**20**	**1.5**	0.25	**1.8**
RM 10.5 to RM 11.8	**1.5**	0.020	0.019	0.16	**33**	**2.4**	0.22	**2.6**

[a] Total HQ was calculated using the following prey portions**:** 20% carp, 20% sculpin, 20% largescale sucker, 20% smallmouth bass, and 20% crayfish.

[b] Total HQ was calculated using the following prey portions: 40% carp, 20% sculpin, 20% smallmouth bass, and 20% crayfish. Largescale sucker were not analyzed for TEQs. When no sculpin data were available, smallmouth bass were assigned a prey portion of 40%. When no crayfish data were available, sculpin were assigned a prey portion of 40%.

[c] Total TEQ HQ includes a sediment HQ of 8.6 based on a sediment UCL greater than the maximum concentration.

HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
PCB – polychlorinated biphenyl
RM – river mile
TEQ – toxic equivalent
UCL – upper confidence limit on the mean

Bold identifies HQs ≥ 1.

When calculated for a multiple prey diet, aluminum, lead, total PCBs, PCB TEQ, total dioxin/furan TEQ, total TEQ have HQs ≥ 1 in at least one exposure area. Risk estimates from total dioxin/furan TEQ and PCB TEQ are components of risk from total TEQ because the total TEQ includes dioxin-like effects from both PCBs and dioxins/furans. HQs based on multiple prey are < 1 for antimony and mercury in all exposure areas.

The aluminum HQ is ≥ 1 because the TSC was exceeded in all exposure areas; no prey items resulted in HQs ≥ 1. The lead HQ for mink is ≥ 1 only in the exposure area between RM 9.5 and RM 10.5. Although the HQ was calculated under the assumption that smallmouth bass comprise 20% of the mink diet, smallmouth bass contributed essentially 100% to the risk estimate (i.e., HQ) because no other prey data were available from this exposure area. The maximum lead concentration from smallmouth bass collected at this exposure area (RM 9.5 to RM 10.5) is 1,100 mg/kg ww, which is over 100 times as great as the other smallmouth bass concentration available from this exposure area (6.8 mg/kg ww, collected from the east bank between RM 9.5 and RM 10.5) and 2 to 5 orders of magnitude greater than lead concentrations detected in smallmouth bass from all other areas (0.0048 to 1.8 mg/kg ww). The single composite sample is an outlier for both antimony and lead, suggesting that a fish in the sample might have swallowed a fishing sinker. Antimony can be mixed with lead as a hardener for lead-based products (ATSDR 1992). For example, one fish tackle supplier notes that fishing sinkers contain 94% lead and 6% antimony for hardness and color (Blue Ocean Tackle 2011). Without this sample the HQ would be well below 1 and comparable with the HQs for the other Study Area reaches.

Several uncertainties are associated with the exposure and effects assumptions used to derive the risk estimates, as described below. Uncertainty related to exposure assumptions is discussed first, followed by that related to effects assumptions.

- **FIR –** The assumed mink FIR of 16% of body wieght per day appears to be conservative, falling at the upper end of the range reported by EPA (1993). HQs could be as much as 25% lower using the lowest ingestion rate reported by EPA (12% body weight per day). However, even at the lower ingestion rate, all mink HQs for total PCBs and total TEQ would still be ≥ 1.
- **Prey portions –** There is some uncertainty associated with the prey portions assigned to each of the mink prey species. Prey portions were assigned on the basis of those presented in the literature and on the relative abundance of potential prey items in the Study Area (Attachment 16). As described below in Section 8.1.5.2.1, risk results would change if the prey portions were changed. Specifically, if smallmouth bass were assumed to consitute a larger but still plausible fraction of the mink diet, the antimony HQ would rise from < 1 to ≥ 1 in the exposure area RM 9.5 to RM 10.5. However, the sample that would cause this change is the same one discussed above that drives risk from lead; all other samples result in antimony HQs < 1. Additionally, total PCBs HQs, although decreased by a factor of 2 to 3, would remain ≥ 1 when calculated from prey portions that match the relative abundance of prey in the Study Area, rather than on baseline prey portions derived from the literature. With the above exceptions, varying the portions of mink's potential aquatic prey has little impact on the magnitude or extent of HQs ≥ 1.
- **Use of only fish as prey species –** Mink were assumed to eat fish and crayfish only. In the wild, mink diets vary greatly by season and availability to include birds, mammals, and amphibians; in some cases, these other taxa are the most important food source (Eagle and Whitman 1987). Not particularly agile in water, mink tend to catch a higher proportion of small and slow-moving fish than large

or swift ones such as salmonids (Melquist et al. 1981; Dunstone and Birks 1987). Some of the mink COPCs with HQs ≥ 1 (i.e., PCBs and dioxins/furans) tend to bioaccumulate in the tissues of higher-trophic-level organisms. It is therefore possible that exposure estimates may underestimate risk to mink if the dietary proportion of salmon or piscivorous birds is high (although the contribution of PCBs in adult salmon tissue to mink risk would not be attributable to Study Area sediment because their PCB burdens are attributable to exposure through the marine food web, rather than uptake during their relatively brief freshwater migration (e.g., O'Neill et al. 1998)). The exposure estimates are more likely to overestimate risks to mink because much of its common prey consists of smaller fish and herbivorous waterbirds and mammals.

- **Site use –** The assumption that mink forage only within the Study Area (i.e., a SUF of 1) is conservative. Although mink forage primarily on the land along waterways, they may also exploit adjacent uplands in pursuit of terrestrial prey and use other water bodies near the Study Area. Furthermore, mink prefer riparian cover within their foraging habitat and may not use industrial areas that offer no riparian cover (Allen 1986). Therefore, the SUF is most likely < 1.
- **Use of surrogate prey data for crayfish –** In two exposure areas (RM 4.5 to RM 5.5 and RM 7.5 to RM 8.5), sculpin data were used as a surrogate for crayfish PCB TEQ, total dioxin/furan TEQ, and total TEQ concentrations. In exposure areas where both species were present, sculpin TEQ concentrations were similar to or higher than those in crayfish. This comparison suggests that sculpin is a conservative surrogate for crayfish, yielding similar or higher risk estimates than those generated by a mixed species diet.
- **Use of surrogate prey data for sculpin –** In one exposure area (RM 2.5 to RM 3.5), smallmouth bass data were used as a surrogate for sculpin PCB TEQ, total dioxin/furan TEQ, and total TEQ tissue data. For exposure areas where both species were present, smallmouth bass TEQ concentrations were similar to or higher than those of sculpin. This comparison suggests that smallmouth bass is a conservative surrogate for sculpin, yielding similar or higher risk estimates than would be obtained if sculpin data were available.
- **Use of surrogate data for largescale sucker –** Largescale sucker PCB TEQ, total dioxin/furan TEQ, and total TEQ data were not available for any exposure area; carp data were used as a surrogate. Total PCB concentrations in carp were more than 10 times those in largescale sucker, indicating that the PCB TEQ and PCB fraction of the total TEQ could be overestimated. Because largescale sucker were assumed to constitute 20% of the mink diet, using carp data as a surrogate for largescale sucker overestimates PCB TEQ and total TEQ risks. However, using only 10% of the carp tissue concentration as a surrogate for largescale sucker would have only a small effect on HQs and would not change any HQs from ≥ 1 to < 1.

- **Use of TEFs to derive TEQ EPCs –** Uncertainty associated with the PCB TEQ, total dioxin/furan TEQ, and total TEQ is related to the derivation of mammal TEFs. The TEFs for mammals are based on *in vivo* toxicity (when data are available). TEF values for a given congener generally fall within a range of about an order of magnitude for mammals (Sanderson and Van den Berg 1999). The uncertainties in these TEFs may overestimate or underestimate risk.

Mink and River Otter Habitat in the Lower Willamette River

Mink and otter are considered aquatic-dependent mammals and live much of their lives in close proximity to water, utilizing similar habitats. Access to permanent water, reliable food sources, and dense riparian vegetation are key features of their ideal habitat. Mink prey on both terrestrial and aquatic animals. Although otter share a similar diet, prey are more likely to be fish, crustaceans, amphibians, and reptiles. The home range of mink is significantly smaller than that of otter—on the order of acres versus miles; however, the home range of any individual animal is a function of habitat quality and prey density. As strong swimmers, otter are able to range farther and utilize discontinuous habitats in comparison to mink.

The upland environment along the LWR is primarily urban or industrial, with fragmented areas of riparian forest, wetlands, and associated upland forests. Historical development along the shoreline and filling of channels and wetlands has left only small strips or isolated pockets of riparian wildlife habitat, with the exception of areas such as Harborton Wetlands, Oaks Bottom, Forest Park, and Powers Marine Park.

Within the Study Area, isolated wildlife habitat areas do exist but linkages to the larger landscape are limited. Significant habitat that may be used by otter and mink in the Study Area includes the South Rivergate corridor, the Harborton forest and wetlands near the confluence of the river with the Multnomah Channel, Willamette Cove, the railroad corridor, and Swan Island beaches and lagoon (Adolfson et al. 2000). Small pocket beaches that might be used for foraging by otter are found throughout the Study Area. The habitat represented by these pocket beaches may be too fragmented to support self-sustaining mink populations.

Mink exposure may also be limited by the types of prey they are physically able to catch. Typically, mink foraging is restricted to invertebrates and small fish in nearshore environments. Therefore, mink may have limited exposure to contaminant levels in large, older fish such as large carp, or contaminants in prey that inhabit deepwater or offshore areas. This latter point may be important when assessing the risk reduction that might result from remediation of offshore or deepwater areas.

Additional uncertainty is associated with assumptions about effects on mink. Mink-specific toxicity data were identified for total PCBs, PCB TEQ, total dioxin/furan TEQ, and total TEQ; these TRVs are predictive of mink-specific adverse effects. Mink-specific toxicity data were not available, however, for antimony and lead, adding uncertainty to the use of the selected TRVs and perhaps to an overestimate of risk.

Other uncertainties associated with the effects assumptions are as follows:

- **Aluminum –** The TRV is highly uncertain because only one dietary LOAEL was identified, and it is based on exposure of mice to aluminum lactate, an ionic form of aluminum not present in the environment.
- **Antimony** – The TRV is highly uncertain because all effects data are based on drinking water exposure.
- **Lead –** As for antimony, the TRV is based on drinking water exposure. The effects threshold in mammals from dietary exposure appears to be an order of magnitude higher, indicating that risk may be overestimated.

- **Mercury –** The TRV is based on exposure to mink and is thus directly pertinent. Uncertainty arises, however, because the TRV studies exposed mink to field-collected fish; adverse effects in mink could be attributable in part to other chemicals in the fish.
- **Total PCBs, PCB TEQ,** total **dioxin/furan TEQ, and total TEQ –** Low uncertainty is associated with the LOAELs used to derive mink HQs for total PCBs and TEQs. The selected total PCB and TEQ TRVs represent the lowest LOAELs based on chronic mink reproductive studies in which mink were fed field-collected carp from the Great Lakes region (Restum et al. 1998; Tillitt et al. 1996). Other chemicals were present in these field-collected carp (e.g., dioxins/furans, PCBs, DDE, DDD, chlordane); however, LOAELs based on these studies are consistent with the effect threshold based on mink reproduction where adult mink were fed a laboratory mixture of PCBs in food for 18 months (2001).

8.1.5.1.6 River Otter

Nine COPCs were identified for river otter in the SLERA and refined screen. Five COPCs have HQs ≥ 1 based on individual prey components in Step 2: lead, total PCBs, PCB TEQ, total dioxin/furan TEQ, and total TEQ (which is the sum of the PCB TEQ and total dioxin/furan TEQ) (Table 8-15). Individual prey items resulting in HQs ≥ 1 within 3-mile exposure areas in Step 2 are carp for total PCBs, PCB TEQ, and total TEQ; field clam for total PCBs; crayfish for total PCBs; laboratory clam for total dioxin/furan TEQ and total TEQ; largescale sucker for total PCBs; sculpin for total PCBs, PCB TEQ, and total TEQ; and smallmouth bass for lead, total PCBs, PCB TEQ, and total TEQ (Attachment 17). In Step 3 HQs were calculated for these COPCs across multiple prey and incidental sediment ingestion using LOAEL-based TTCs and TSCs, as shown in Table 8-22.

Table 8-22. River Otter LOAEL HQs Within 3-Mile Exposure Areas

	Total HQ				
Exposure Area	**Lead[a]**	**Total PCBs[a]**	**PCB TEQ[b]**	**Total Dioxin/Furan TEQ[b]**	**Total TEQ[b]**
RM 1.5 to RM 4.5	0.0067	**25**	**1.3**	0.13	**1.5**
RM 4.5 to RM 7.5	0.013	**22**	0.96	0.96	**2.3**
RM 7.5 to RM 10.5	0.49	**21**	0.95	0.14	**1.1**
Above RM 10.5	0.0051	**31**	**1.5**	0.13	**1.6**

[a] Total HQ was calculated using the following prey portions: 40% carp, 40% sculpin, 10% crayfish, 4% smallmouth bass, 4% largescale sucker, and 2% clams.

[b] Total HQ was calculated using the following prey portions: 44% carp, 40% sculpin, 10% crayfish, 4% smallmouth bass, and 2% clams. Largescale sucker were not analyzed for dioxins/furans and PCB congeners.

HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
PCB – polychlorinated biphenyl
RM – river mile
TEQ – toxic equivalent
Bold identifies HQs ≥ 1.

Total PCBs, PCB TEQ, and total TEQ have HQs ≥ 1 in at least one exposure area based on a multiple prey diet. HQs based on multiple prey for all exposure areas are < 1 for lead and total dioxin/furan TEQ (however, total TEQ includes dioxins and furans).

Several uncertainties are associated with the exposure and effects assumptions used to derive the risk estimates, as described below. Uncertainty related to exposure assumptions is discussed first, followed by that related to effects assumptions.

- **FIR –** The assumed FIR of 10% of body weight per day is slightly lower than that assumed by Sample and Suter (1999). The use of Sample and Suter's (1999) value (11% body weight per day) would have only a marginal influence on HQs (for example, an HQ of 10 would become 11).
- **Prey portions –** As discussed below in Section 8.1.5.2.1, varying the prey portions in the diet would affect the risk conclusions. Specifically, if smallmouth bass were assumed to constitute a larger fraction of the river otter diet, the lead HQ would rise from < 1 to ≥ 1 in the RM 9.5 to RM 10.5 exposure area. However, the sample that would cause this change is the same one discussed above for mink that drives risk from lead. All other samples result in lead HQs < 1. Total dioxin/furan TEQ risks could also result in HQs ≥ 1 in RM 4.5 to RM 7.5 if river otter were assumed to eat a larger but nonetheless plausible fraction of crayfish or sculpin. Dioxins and furans are already accounted for as contributing to potentially unacceptable risk because total TEQ HQ is ≥ 1 in RM 4.5 to RM 7.5. The magnitude of total PCBs HQs decreases by a factor of 2 but remains ≥ 1, when calculated assuming prey portions based on the relative abundance of prey in the Study Area rather than on baseline prey portions derived from the literature. In general, however, varying prey portions has little impact on the magnitude of HQs.
- **Use of TEFs to derive TEQ EPCs –** Uncertainty associated with the PCB TEQ, total dioxin/furan TEQ, and total TEQ is related to the derivation of mammal TEFs as discussed above for mink. The uncertainties in these TEFs may overestimate or underestimate risk.
- **Use of surrogate prey data –** Largescale sucker PCB TEQ, total dioxin/furan TEQ, and total TEQ data were not available for any exposure area; carp data were used as a surrogate. Total PCB concentrations were more than 10 times as high in carp as in largescale sucker indicating that the PCB TEQ and PCB fraction of the total TEQ could be overestimated. However, largescale sucker were assumed to constitute only 4% of the mink diet, and assuming 10 times lower largescale sucker concentrations has only a small effect on HQs (e.g., RM 1.5 to RM 2.5 HQ is reduced from 2.3 to 1.6) and does not change any HQs from ≥ 1 to < 1.

Additional uncertainty is associated with assumptions about effects on river otter. No river otter-specific toxicity data were identified for any COPC. The conservative assumption that river otters are as sensitive as the most sensitive mammal species adds uncertainty to the use of the selected TRVs and, except where noted, may overestimate risks.

The general uncertainties associated with the effects assumptions for lead, total PCBs, PCB TEQ, total dioxin/furan TEQ, and total TEQ are the same as those discussed above for mink.

8.1.5.2 Evaluation of Additional Uncertainties

Uncertainties associated with exposure assumptions, effect thresholds (TRVs), and risk characterization methods are identified in previous subsections. This subsection presents evaluations for two additional uncertainties identified as part of EPA's Problem Formulation (Attachment 2): the selected prey portions and risk estimates for belted kingfisher.

8.1.5.2.1 Evaluation of Varying Prey Portions

Selected prey portions (Table 8-6) are based on dietary information presented in the literature. In EPA's Problem Formulation (Attachment 2), EPA requested that prey portions be varied probabilistically from 0 to 100%. An evaluation was conducted to determine how varying the prey portions in the diet would change the risk conclusions (i.e., whether or not an HQ would change from ≥ 1 to < 1, or vice versa for all receptor-COPC pairs in which multiple prey species were evaluated in the diet.[133]

In Step 2 of the risk characterization, HQs were calculated assuming single prey species constituted 100% of the diet (Table 8-15). The prey portion uncertainty evaluation identified the range of possible HQs when the contribution of individual prey species to the diet varied from 0 to 100%. The range of HQs was used to determine whether HQs for any COPC would change from ≥ 1 to < 1 or vice versa under different but plausible assumptions about prey portions (default portions are presented in Table 8-6). The range of plausible prey fractions were determined based on the dietary information in Attachment 16. Table 8-23 presents the results of this evaluation.

[133] The spotted sandpiper diet was not evaluated because sandpiper prey species (i.e., worms and clams) were evaluated individually, and no multi-species diet was evaluated for this receptor.

Table 8-23. Summary of Dietary Prey Portion Uncertainty Evaluation

COPC by Receptor	HQ	Exposure Area	Selected Prey Portions Used in Risk Characterization	Prey Portion Uncertainty Evaluation	Does Uncertainty Evaluation Change COPC Status?
Hooded Merganser					
Total PCBs	1.5 – 3.8	RM 1.5 – RM 2.5; RM 6.5 – RM 7.5; RM 10.5 – RM 11.8	65% sculpin, 5% peamouth, 25% clams, 5% crayfish	HQs could range from 0.002 to 5.6 based on 100% ingestion of crayfish and sculpin, respectively.	No; would have an HQ < 1 if diet was primarily crayfish and peamouth; however, this is not a reasonable possibility (the merganser diet probably includes sculpin).
Lead	0.013 – 0.21	All exposure areas	65% sculpin, 5% peamouth, 25% clams, 5% crayfish	More than 83% of diet would have to be represented by peamouth to push HQs to ≥ 1 in all exposure areas; peamouth prey portion of > 83% is not supported by the literature.	No, because of low probability that peamouth is > 83% of diet.
BEHP	0.0012 – 0.67	All exposure areas	70% sculpin, 25% clams, 5% crayfish	HQs are < 1 regardless of prey portion of selected prey species.	No; would have an HQ < 1 regardless of prey portions.
Bald Eagle					
Mercury	1.2 – 1.7	All exposure areas	45% largescale sucker, 45% carp, 5% northern pikeminnow, 5% peamouth	HQs could range from 0.93 to 9.4 based on 100% ingestion of carp and northern pikeminnow, respectively.	No; would have an HQ < 1 only if the eagle diet consisted solely of carp; this is not a reasonable possibility.
Total PCBs	3.8 – 3.9	All exposure areas	45% largescale sucker, 45% carp, 5% northern pikeminnow, 5% peamouth	HQs could range from 0.12 to 7.9 based on 100% ingestion of peamouth and carp, respectively.	No; would have an HQ < 1 only if carp was absent from the diet; this is not a reasonable possibility.

Table 8-23. Summary of Dietary Prey Portion Uncertainty Evaluation

COPC by Receptor	HQ	Exposure Area	Selected Prey Portions Used in Risk Characterization	Prey Portion Uncertainty Evaluation	Does Uncertainty Evaluation Change COPC Status?
Sum DDE	0.61 – 0.71	All exposure areas	45% largescale sucker, 45% carp, 5% northern pikeminnow, 5% peamouth	More than 83% of prey would have to be represented by northern pikeminnow for HQs to be ≥ 1 at three exposure areas (between RM 4.5 and RM 7.5); literature on prey species does not support a high prey portion (> 45%) of pikeminnow by bald eagles.	No, because of low probability that northern pikeminnow is > 45% of diet.
Osprey					
Lead	7.8	RM 9.5 – RM 10.5	83% largescale sucker, 6% carp, 7% northern pikeminnow, 2% smallmouth bass, 2% brown bullhead	More than 1% of diet would have to be smallmouth bass for the HQ to be ≥ 1	Yes; HQ < 1 if smallmouth bass < 1% of the diet, which is a reasonable possibility.
Mercury	0.25 – 0.38	All exposure areas	83% largescale sucker, 6% carp, 7% northern pikeminnow, 2% smallmouth bass, 2% brown bullhead	HQs could range from 0.15 to 1.6 based on 100% ingestion of brown bullhead and northern pikeminnow, respectively; More than 63% of diet would have to be northern pikeminnow for the HQ to be ≥ 1.	No, because of low probability that northern pikeminnow is > 63% of diet.
BEHP	0.047 – 0.2	All exposure areas	83% largescale sucker, 6% carp, 7% northern pikeminnow, 2% smallmouth bass, 2% brown bullhead	More than 59% of the diet would have to be represented by smallmouth bass for the HQ to be ≥ 1 at RM 3.5 to RM 4.5.	No, because of low probability that smallmouth bass is > 59% of diet.

Table 8-23. Summary of Dietary Prey Portion Uncertainty Evaluation

COPC by Receptor	HQ	Exposure Area	Selected Prey Portions Used in Risk Characterization	Prey Portion Uncertainty Evaluation	Does Uncertainty Evaluation Change COPC Status?
Total PCBs	1.1	RM 10.5 – RM 11.8 Swan Island Lagoon	83% largescale sucker, 6% carp, 7% northern pikeminnow, 2% smallmouth bass, 2% brown bullhead	HQs could range from 0.047 to 6.9 based on 100% ingestion of brown bullhead and carp, respectively. Less than 56% of diet would have to be represented by smallmouth bass and < 10% of diet represented by carp for the HQ to be < 1 site-wide.	Yes; would have an HQ < 1 under the reasonable possibility that the osprey diet includes < 10% carp and < 56% smallmouth bass.
Mink					
Aluminum	1.3 – 1.6	All exposure areas	20% carp, 20% sculpin, 20% largescale sucker, 20% smallmouth bass, 20% crayfish	HQ > 1 regardless of prey portions of selected prey species	No; would not have an HQ ≥ 1 regardless of prey portions.
Antimony	0.0036 – 0.33	All exposure areas	20% carp, 20% sculpin, 20% largescale sucker, 20% smallmouth bass, 20% crayfish	More than 62% of the diet would have to be represented by smallmouth bass for the HQ to be ≥ 1 at RM 9.5 to RM 10.5; 100% ingestion of smallmouth bass results in an HQ of 1.6 only at RM 9.5 to RM 10.5.	Yes; would have an HQ ≥ 1 if > 62% of diet was composed of smallmouth bass, which is a reasonable possibility.
Lead	4.0	RM 9.5 – RM 10.5	20% carp, 20% sculpin, 20% largescale sucker, 20% smallmouth bass, 20% crayfish	More than 5% of diet would have to be represented by smallmouth bass for the HQ to be ≥ 1.	No; would have an HQ < 1 only if smallmouth bass represented <5% of the diet; this is not a reasonable possibility given the opportunistic nature of mink and abundance of smallmouth bass in the Study Area.
Mercury	0.14 – 0.28	All exposure areas	20% carp, 20% sculpin, 20% largescale sucker, 20% smallmouth bass, 20% crayfish	HQs are < 1 regardless of prey portion of selected prey species.	No; would not have an HQ ≥ 1 regardless of prey portions.

Table 8-23. Summary of Dietary Prey Portion Uncertainty Evaluation

COPC by Receptor	HQ	Exposure Area	Selected Prey Portions Used in Risk Characterization	Prey Portion Uncertainty Evaluation	Does Uncertainty Evaluation Change COPC Status?
Total PCBs	19 – 33	All exposure areas	20% carp, 20% sculpin, 20% largescale sucker, 20% smallmouth bass, 20% crayfish	HQs could range from 0.014 to 85 based on 100% ingestion of crayfish and carp, respectively.	No; would have an HQ ≥ 1 in at least one exposure area regardless of prey portions. HQs would be < 1 in only 3 of the 11 exposure areas (RM 2.5 to RM 4.5, RM 4.5 to RM 5.5, RM 5.5 to RM 6.5) if diet consisted only of crayfish or sculpin; however, ingestion of a single species is not a reasonable possibility given the opportunistic nature of mink, and HQs would still be > 1 in other exposure areas.
PCB TEQ	1.4 – 2.4	All exposure areas	40% carp, 20% sculpin, 20% smallmouth bass, 20% crayfish	HQs could range from 0.028 to 3.4 based on 100% ingestion of crayfish and carp, respectively.	No; would have an HQ < 1 if diet consisted primarily of crayfish; this is not a reasonable possibility given the opportunistic nature of mink and abundance of alternative prey in the Study Area.
Total dioxin/furan TEQ	2.0	RM 6.5 – RM 7.5	40% carp, 20% sculpin, 20% smallmouth bass, 20% crayfish	HQs could range from 0.035 to 4.2 based on 100% ingestion of crayfish and smallmouth bass, respectively.	No; would have an HQ < 1 if diet consisted only of carp;; this is not a reasonable possibility given the opportunistic nature of mink and abundance of alternative prey in the Study Area.
Total TEQ	1.6 – 12	All exposure areas	40% carp, 20% sculpin, 20% smallmouth bass, 20% crayfish	HQs could range from 0.076 to 12.9 based on 100% ingestion of crayfish and smallmouth bass, respectively.	No; would have an HQ < 1 if diet consisted primarily of crayfish; this is not a reasonable possibility given the opportunistic nature of mink and abundance of alternative prey in the Study Area.
River Otter					
Lead	0.0051 – 0.49	All exposure areas	40% carp, 40% sculpin, 10% crayfish, 4% smallmouth bass, 4% largescale sucker, 2% clams	More than 8% of diet would have to be represented by smallmouth bass for HQ to be ≥ 1 at RM 9.5 to RM 10.5; 100% ingestion of smallmouth bass results in an HQ of 12 only at RM 9.5 to RM 10.5.	Yes; would have an HQ ≥ 1 if > 8% of diet was composed of smallmouth bass, which is a reasonable possibility.

Table 8-23. Summary of Dietary Prey Portion Uncertainty Evaluation

COPC by Receptor	HQ	Exposure Area	Selected Prey Portions Used in Risk Characterization	Prey Portion Uncertainty Evaluation	Does Uncertainty Evaluation Change COPC Status?
Total PCBs	21 – 31	All exposure areas	40% carp, 40% sculpin, 10% crayfish, 4% smallmouth bass, 4% largescale sucker, 2% clams	HQs could range from 0.15 to 51 based on 100% ingestion of crayfish and carp, respectively.	No; would have an HQ < 1 if diet consisted primarily of crayfish; this is not a reasonable possibility given the opportunistic nature of otter and abundance of alternative prey in the Study Area.
PCB TEQ	0.95 – 1.5	All exposure areas	44% carp, 40% sculpin, 10% crayfish, 4% smallmouth bass, 2% clams	HQs could range from 0.027 to 1.9 based on 100% ingestion of crayfish and carp, respectively.	No; would have an HQ < 1 in any exposure area only if diet consisted primarily of crayfish and clams; this is not a reasonable possibility.
Total dioxin/furan TEQ	0.13 – 0.96	All exposure areas	44% carp, 40% sculpin, 10% crayfish, 4% smallmouth bass, 2% clams	The diet would have to be represented by more than 77% crayfish, 83% sculpin, or 67% smallmouth bass for HQs to be ≥ 1 in one exposure area (RM 4.5 to 7.5); 100% ingestion of crayfish, sculpin, or smallmouth bass results in HQs ranging from 1.2 to 1.5 between RM4.5 and RM 7.5.	Yes; would have an HQ ≥ 1 if > 77% or 83% of diet could be composed of crayfish or sculpin, respectively, which is a reasonable possibility.
Total TEQ	1.1 – 2.3	All exposure areas	44% carp, 40% sculpin, 10% crayfish, 4% smallmouth bass, 2% clams	HQs could range from 0.077 to 3.1 based on 100% ingestion of crayfish and carp, respectively.	No; would have an HQ < 1 only if diet consisted primarily of crayfish and clams; this is not a reasonable possibility.

BEHP – bis(2-ethylhexyl) phthalate
COPC – contaminant of potential concern
DDE – dichlorodiphenyldichloroethylene
HQ – hazard quotient
PCB – polychlorinated biphenyl
RM – river mile
TEQ – toxic equivalent

Additional dietary sensitivity analysis was conducted for receptor-COPC pairs that appear to be the primary contributors to potentially unacceptable Study Area risks, or whose dietary uncertainty could influence risk conclusions (Table 8-24). The objective was to evaluate the uncertainty and variability in the composition of wildlife receptor diets using reasonable assumptions. The sensitivity analysis considered relative abundance of each receptor's likely aquatic prey species in the Study Area. The osprey population was excluded from this sensitivity evaluation because dietary information used in the risk calculations is site-specific and provides a realistic approximation of diet in the Study Area. The sandpiper was excluded also because more realistic approximations of diet could not be evaluated with the available tissue chemistry data.

Table 8-24. Chemical-Receptor Pairs Considered in the Sensitivity Analysis of Wildlife Prey Fractions

Chemical	Mink	River Otter	Bald Eagle	Hooded Merganser
Mercury			X	
Total PCBs	X	X	X	X
Total TEQ	X	X		

PCB – polychlorinated biphenyl
TEQ – toxic equivalent

To evaluate dietary sensitivity, dietary composition ranges were identified for each wildlife receptor. For the largely opportunistic receptors—mink and river otter—the site-specific fish abundance data were used to define these ranges because these receptors would consume fish species in quantities approximately proportional to their abundance. For more selective feeders that tend to ingest fish of a certain size range—hooded merganser and bald eagle—ranges for the dietary fraction of each prey item were assigned on the basis of professional judgment considering the relative abundane of potential prey and receptor-specific foraging data (Attachment 16). The fraction of invertebrates was also based on professional judgment because no data were available on their abundance in the Study Area relative to other potential prey. Because analyses for total TEQ constituents were not conducted in samples of largescale sucker, northern pikeminnow, or peamouth, surrogate species were used. Carp was considered a surrogate for largescale sucker, smallmouth bass was considered a surrogate for northern pikeminnow, and sculpin was considered a surrogate for peamouth. The prey items and dietary fractions used in this analysis are shown in Table 8-25. Also summarized in Table 8-25 are relative abundance data for fish in the Study Area (taken from Table 2-6 in Attachment 16).

Table 8-25. Prey Species and Dietary Fraction Ranges Considered in the Wildlife Receptor HQ Sensitivity Analysis

	Dietary Fraction by Receptor						
	Mink		River Otter		Bald Eagle	Hooded Merganser	
Prey	**Total PCBs**	**Total TEQ**	**Total PCBs**	**Total TEQ**	**Mercury and Total PCBs**	**Total PCBs**	**Relative Abundance of Fish in Study Area**
Black crappie	0 – 5%	0 – 5%	0 – 5%	0 – 5%	0%	0%	0 – 5%
Brown bullhead	0 – 5%	0 – 5%	0 – 5%	0 – 5%	0%	0%	0 – 5%
Carp	5 – 20%	40 – 70%	5 – 20%	40 – 70%	5 – 20%	0%	5 – 20%
Chinook, juvenile salmon	0 – 1%	0 – 1%	0 – 1%	0 – 1%	0%	0%	0 – 5%
Clam	0%	0%	0 – 2%	0 – 2%	0%	0 – 25%	0%
Crayfish	0 – 47%	0 – 47%	0 – 47%	0 – 47%	0%	0 – 5%	0%
Largescale sucker	35 – 50%	ND[a]	35 – 50%	ND[a]	35 – 50%	0%	35 – 50%
Northern pikeminnow	5 – 25%	ND[a]	5 – 25%	ND[a]	5 – 25%	0%	5 – 25%
Peamouth	0 – 30%	ND[a]	0 – 30%	ND[a]	0 – 30%	0 – 50%	0 – 30%
Sculpin	5 – 15%	5 – 45%	5 – 15%	5 – 45%	0%	0 – 50%	5 – 15%
Smallmouth bass	0 – 35%	5 – 60%	0 – 35%	5 – 60%	0%	0%	0 – 35%
Worm	0%	0%	0%	0%	0%	0%	0%

[a] Analyses for total TEQ constituents were not conducted in samples of largescale sucker, northern pikeminnow, or peamouth. Carp was considered a surrogate for largescale sucker, smallmouth bass was considered a surrogate for northern pikeminnow, and sculpin was considered a surrogate for peamouth.

HQ – hazard quotient

ND – no data

PCB – polychlorinated biphenyl

TEQ – toxic equivalent

Sensitivity of the HQ to assumptions concering the relative proportion of prey in the diet of wildlife receptors was then evaluated using probabilistic risk assessment methods. The software program @Risk (Palisade Corporation) was used to conduct Monte Carlo sampling from the ranges of each dietary prey item assuming a uniform distribution. A total of 1,000 Monte Carlo iterations was conducted for each receptor-chemical pair. Because, given the random sampling approach, the percentages of each prey species in a receptor's diet could add up to greater than or less than 100%, the predicted dietary fractions from each of the 1,000 Monte Carlo iterations were weighted so that the sum of all dietary fractions equaled 1.0 (i.e., 100%). After each dietary fraction was weighted, iterations that had resulted in dietary fractions outside of the dietary composition ranges defined in Table 8-25 were excluded. For example, if a Monte-Carlo iteration for the mink total TEQ analysis returned a normalized fraction of carp that was less than 40%, the results from that iteration of the model run would be excluded. Finally, for each of the remaining Monte Carlo-generated estimates of the dietary composition, HQs were calculated and plotted as box-and-whisker plots. The HQs based on the default dietary assumptions used in the risk assessment were also plotted for comparison.

The results of the sensitivity analysis based on dietary composition are discussed by individual receptor.

Mink

The mink sensitivity analysis addressed only total PCBs and total TEQ. For total PCBs, the 95th percentile (and often the maximum) of the 1,000 Monte Carlo-generated HQs are less than the HQs based on the default diet assumed in risk assessment (Figure 8-3). This result indicates that conservatism is inherent in the default dietary assumptions of the mink risk assessment. This result occurs in large part because those species with relatively high total PCBs concentrations were assumed to constitute a larger fraction of the diet than is suggested by their relative abundance in the Study Area. In particular, the total PCBs concentration in carp (EPC = 19 mg/kg ww) is much higher than that in other mink prey (EPCs range from < 1 to 8.8 mg/kg ww; Attachment 4) and carp was assumed to constitute 20% of the default diet. This fraction is at the upper end of the estimated range of carp's contribution (5 to 20%) to the mink diet and is, therefore, conservative.

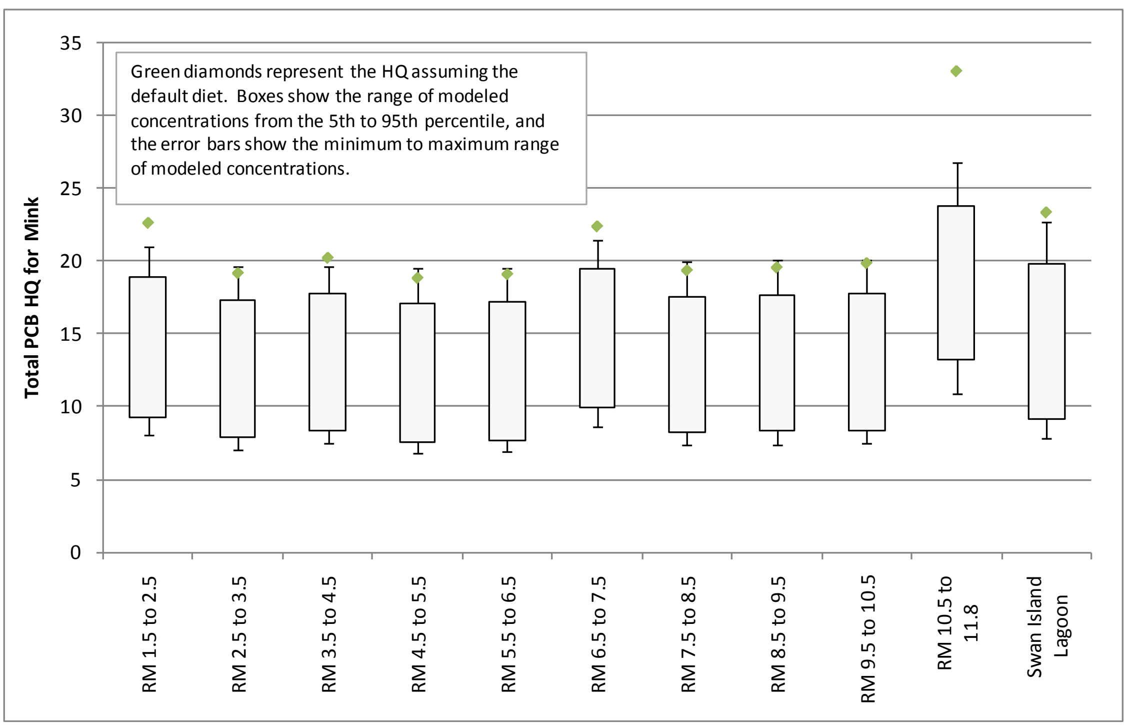


Figure 8-3. Dietary Composition Sensitivity Analysis for Mink and Total PCBs

Similarly, at some river reaches, the total PCBs concentration is relatively high in sculpin (EPC = 8.8 mg/kg ww). Sculpin were assumed to represent 20% of the mink default diet, while the dietary range defined for the Monte-Carlo simulation was 5 to 15%. As for carp, the baseline dietary fraction assumed for sculpin in the risk assessment is conservative.

For total TEQ, the mink HQs based on the default diet are equal to or slightly greater than the 5th percentile HQs from the sensitivity analysis (Figure 8-4). Accordingly, the mink total TEQ HQs assuming the default diet are not as conservative as those for total PCBs; however, the variability in the total TEQ HQs as a function of the dietary composition assumptions is also not as great, with the ratios of the maximum HQ to minimum HQ ranging from 1.1 to 1.7 depending on the river reach. Some of the difference between the results for total TEQ and total PCBs analyses is likely due to the lower number of species for which total TEQ data are available. The limited total TEQ data were assumed to apply to prey species for which data were unavailable and are thus overrepresented in the mink diet (i.e., the surrogate species represent a larger fraction of the mink diet than would be indicated by their relative abundance in the Study Area fish community).

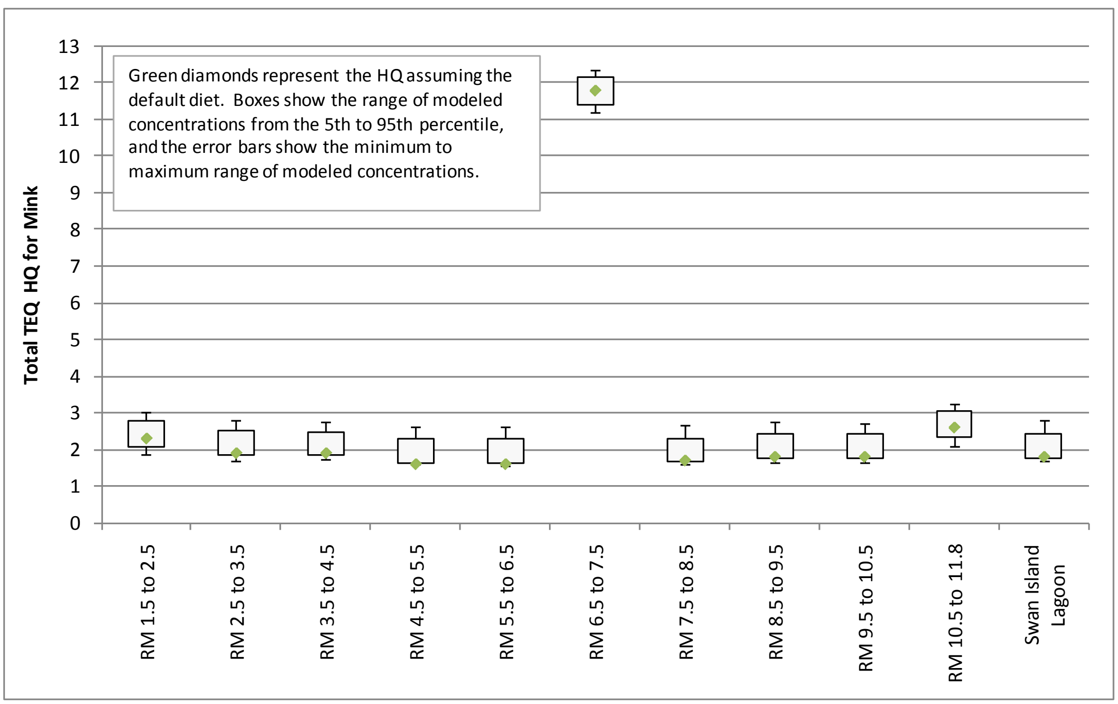


Figure 8-4. Dietary Composition Sensitivity Analysis for Mink and Total TEQ

When considering variability in dietary composition overall, the baseline HQs for mink exposed to total PCBs are conservative. The baseline HQs for total TEQ are less conservative, but given the narrow range of HQs output from the Monte Carlo analysis, do not over- or underpredict risk.

River Otter

Results of the river otter sensitivity analysis are similar to those observed for mink. The HQs based on the default diet for total PCBs appear to be conservative when considering a range of reasonable dietary fractions (the default HQs are approximately double the maximum HQs from the dietary fraction sensitivity analysis; Figure 8-5). The total TEQ HQs assuming the default diet are less conservative than those for total PCBs, but still fall within the range of the 5th to 95th percentile from the dietary fraction sensitivity analysis (Figure 8-6). As for mink, some of the difference between the results for total TEQ and total PCBs analyses is likely due to the availability of total TEQ data for fewer species. The limited available total TEQ data were assumed to apply to prey species for which data were unavailable and are thus overrepresented in the otter diet.

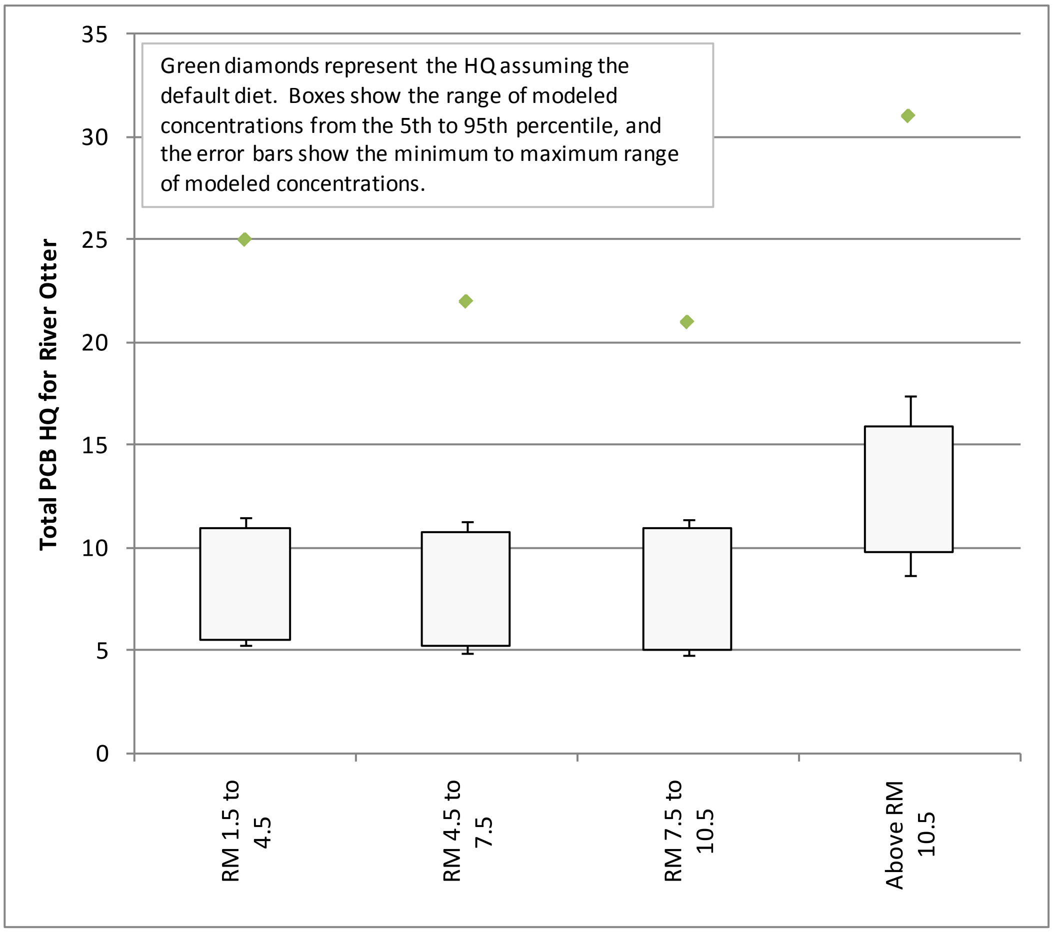


Figure 8-5. Dietary Composition Sensitivity Analysis for River Otter and Total PCBs

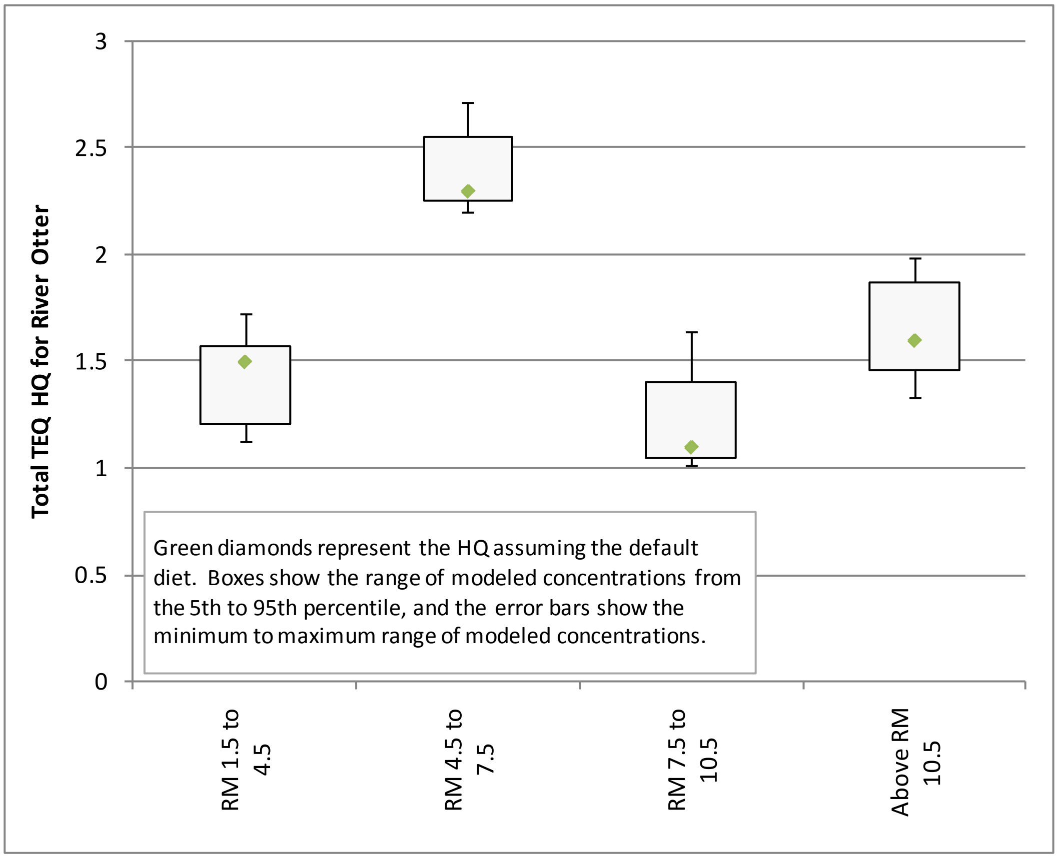


Figure 8-6. Dietary Composition Sensitivity Analysis for River Otter and Total TEQ

Bald Eagle

The bald eagle sensitivity analysis evaluated the influence of the dietary composition assumptions on the HQs for mercury and total PCBs. For mercury, the HQs based on the default diet are equal to or slightly less than the minimum HQs from the dietary composition sensitivity analysis (Figure 8-7). It was assumed in the default diet, and in this sensitivity analysis, that bald eagles feed on carp, largescale sucker, northern pikeminnow, and peamouth. The range of dietary fractions assumed for the sensitivity analysis included a larger proportion of fish with higher mercury concentrations than was assumed for the default diet. For example, northern pikeminnow mercury concentrations (EPCs range from 0.17 to 0.4 mg/kg ww) were higher than for other fish (EPCs range from 0.02 to 0.2 mg/kg ww). In the default diet, pikeminnow represent 5% of the bald eagle diet. In the sensitivity analysis, pikeminnow varied from 5 to 25% of the diet. Consequently, pikeminnow-based HQs were weighted more heavily in the sensitivity analysis than in the default analysis.

Overall, the results of the sensitivity analysis do not have an important influence on the risk conclusions for bald eagles exposed to mercury. All HQs based on the default diet are ≥ 1 for all river reaches. The mercury HQs based on the senstivity analysis under different dietary compositions are slightly higher than the default values; the 95th percentile HQs are all less than 2 times the HQs based on the default diet (Figure 8-7).

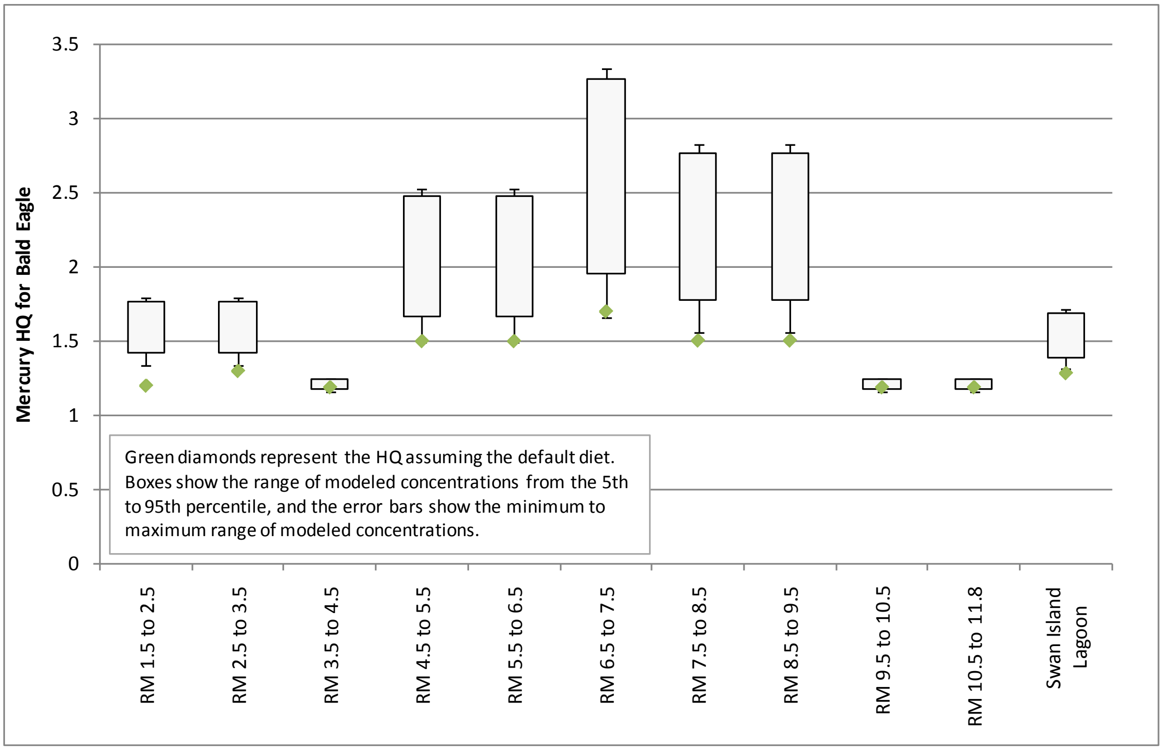


Figure 8-7. Dietary Composition Sensitivity Analysis for Bald Eagle and Mercury

For total PCBs, the HQs based on the default diet are conservatively high, being approximately double the maximum HQs from the sensitivity analysis (Figure 8-8). The default diet HQs are more conservative than those from the sensitivity analysis because carp represent a smaller fraction of the diet in the sensitivity analysis (5 to 20%) than in the default diet (45%). As discussed above for mink, carp have higher total PCBs concentrations than other fish, with the result that lesser fractions of carp result in lower HQs.

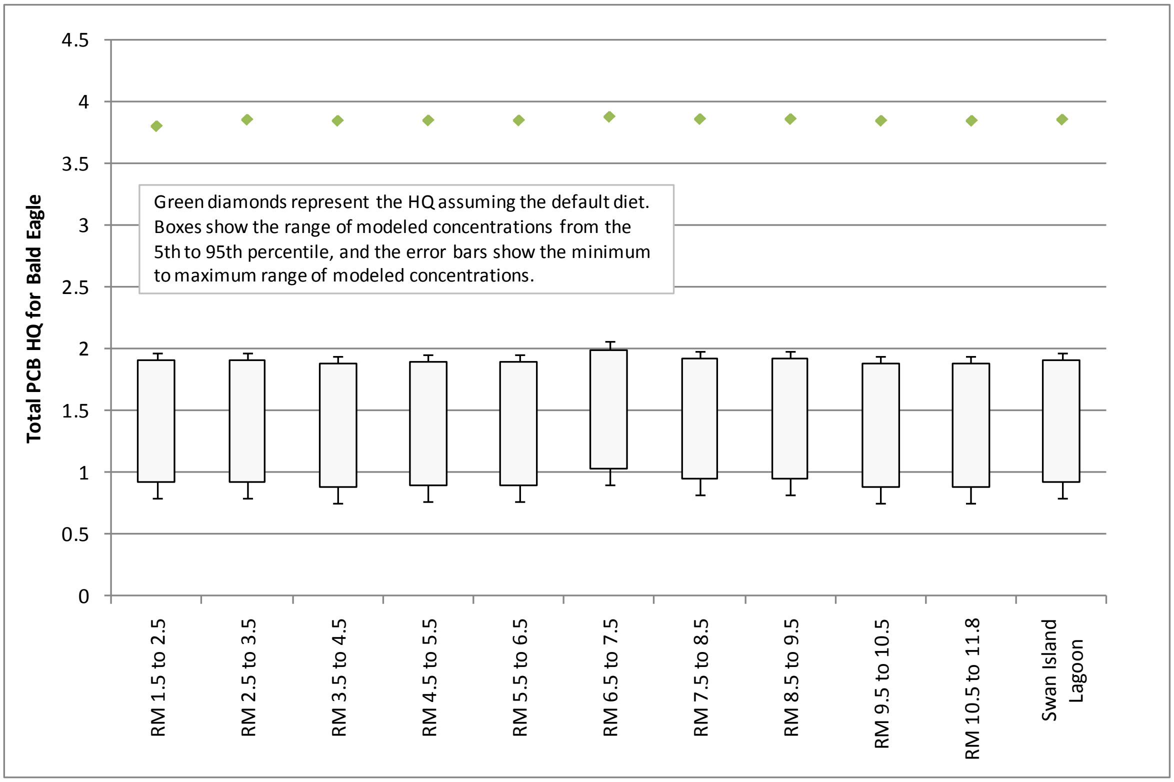


Figure 8-8. Dietary Composition Sensitivity Analysis for Bald Eagle and Total PCBs

Hooded Merganser

The hooded merganser sensitivity analysis evaluated the influence of the dietary composition assumptions on the HQs for total PCBs. The baseline HQs for total PCBs are typically within the range of HQs from the sensitivity analysis, and sometimes greater than the maximum HQ from the sensitivity analysis (Figure 8-9). The dietary composition sensitivity analysis indicates that risks may be overestimated in RM 1.5 to RM 2.5, RM 6.5 to RM 7.5, and RM 10.5 to RM 11.8.

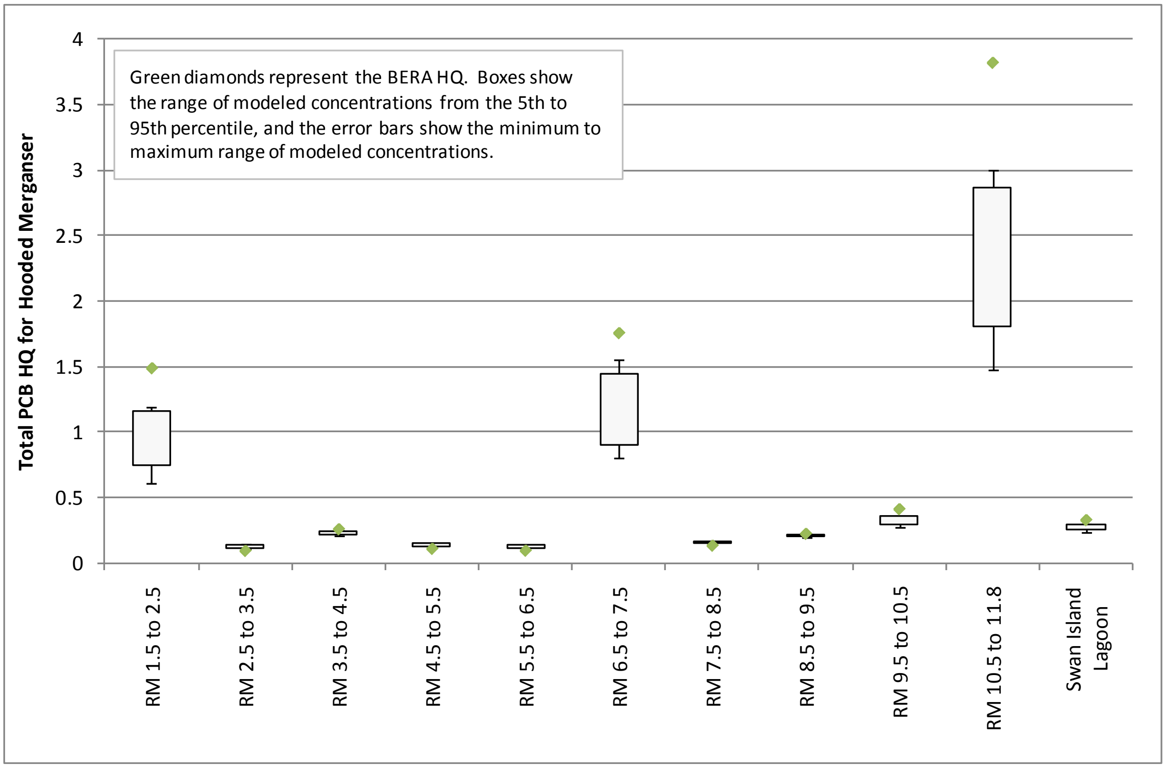


Figure 8-9. Dietary Composition Sensitivity Analysis for Hooded Merganser and Total PCBs

8.1.5.2.2 Evaluation of Belted Kingfisher

Per EPA (2008j), the belted kingfisher was evaluated as part of the wildlife dietary uncertainty assessment to represent small piscivorous birds in the Study Area. The results of the risk characterization for belted kingfisher were compared with the results of the risk characterization for selected bird and mammal receptors to ensure that the selected receptors were protective of belted kingfisher. With the same methods used to derive dietary COPCs for other bird receptors in the SLERA and refined screen (Attachment 5), 13 dietary COPCs were identified for the belted kingfisher (Table 8-26).

Table 8-26. Belted Kingfisher COPCs

COPCs	
Metals	
Aluminum	Lead
Copper	Mercury
PAHs	
Benzo(a)pyrene	
Phthalates	
BEHP	Dibutyl phthalate

Table 8-26. Belted Kingfisher COPCs

COPCs	
PCBs and Dioxins/Furans	
Total PCBs	Dioxin TEQ
PCB TEQ	Total TEQ
Pesticides	
Sum DDE	Total DDx

BEHP – bis(2-ethylhexyl) phthalate
COPC – contaminant of potential concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
TEQ – toxic equivalent
total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT, and 4,4′-DDT)

Exposure Assumptions

The exposure assumptions used to derive EPCs for belted kingfisher are presented in Table 8-27. These assumptions are based on EPA (2008j), except for the sediment ingestion rate (SIR), which was corrected to reflect an incidental sediment ingestion rate of 2%. Details and the rationale for the selected receptor-specific exposure parameters and uncertainties are presented in Attachment 16.

Table 8-27. Belted Kingfisher Exposure Parameters

Parameter	Value	Notes
BW	0.148 kg	Based on EPA (1993)
FIR	0.080 kg ww/day	Based on Nagy (1987)
SIR	0.00033 kg ww/day	Based on assumed 2% incidental sediment ingestion of the dry diet
SUF	1	Based on Puchy and Marshall (1993)
Exposure scale	1 mile	Based on home and foraging data reported in multiple sources (Brooks and Davis 1987; as cited in EPA 1993; Csuti et al. 2001; Cornwell 1963)

BW – body weight
EPA – US Environmental Protection Agency
FIR – food ingestion rate
SIR – sediment ingestion rate
SUF – site use factor
ww – wet weight

Belted kingfishers generally feed within 1 mile of their nesting sites but may have a foraging range up to 5 miles. A 1-mile exposure scale was assumed for this analysis.

Four species were used to represent belted kingfisher prey: juvenile Chinook salmon, peamouth, sculpin, and clam (as a surrogate for mussels). The selected diet for belted kingfisher was based on information from the literature. In the third step of the risk characterization, sculpin and clam were assigned prey portions of 0.9 and 0.1, respectively, to estimate belted kingfisher HQs based on multiple prey. Details and the rationale for the assumptions for selected prey species and associated uncertainties are presented in Attachment 16.

Effects Assumptions

The dietary TRVs presented for birds in Section 8.1.4.1 were used to derive receptor-specific TTCs and TSCs for the belted kingfisher. As described in Section 8.1.4.2, TTCs and TSCs were calculated using receptor-specific parameters (presented in Attachment 16). TTCs and TSCs for all belted kingfisher COPCs are listed in Table 8-28.

Table 8-28. Calculated TTCs and TSCs for Belted Kingfisher COPCs

		TTC (ww)		TSC (dw)	
COPC	**Unit**	**NOAEL**	**LOAEL**	**NOAEL**	**LOAEL**
Metals					
Aluminum	mg/kg	294	NA	71,400	NA
Copper	mg/kg	7.59	22.7	1,840	5,500
Lead	mg/kg	3.06	6.11	741	1,480
Mercury	mg/kg	0.012	0.12	2.91	29.1
PAHs					
Benzo(a)pyrene	µg/kg	525	2,630	127,000	636,000
Phthalates					
BEHP	µg/kg	2,060	20,600	500,000	5,000,000
Dibutyl phthalate	µg/kg	206	2,060	50,000	500,000
PCBs					
Total PCBs	µg/kg	544	1,090	132,000	264,000
PCB TEQ	ng/kg	26.3	263	6,360	63,600
Dioxins/Furans					
Dioxin/furan TEQ	ng/kg	26.3	263	6,360	63,600
Total TEQ	ng/kg	26.3	263	6,360	63,600
Pesticides					
Sum DDE	µg/kg	60	600	14,500	145,000

Table 8-28. Calculated TTCs and TSCs for Belted Kingfisher COPCs

COPC	Unit	TTC (ww)		TSC (dw)	
		NOAEL	LOAEL	NOAEL	LOAEL
Total DDx	µg/kg	426	4260	103,000	1,030,000

BEHP – bis(2-ethylhexyl) phthalate
COPC – contaminant of potential concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
dw – dry weight
LOAEL – lowest-observed-adverse-effect level
NA – not applicable (not a receptor-COPC pair)
NOAEL – no-observed-adverse-effect level
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
TEQ – toxic equivalent
total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT, and 4,4′-DDT)
TSC – threshold sediment concentration
TTC – threshold tissue concentration
ww – wet weight

Risk Characterization

This section presents the risk characterization process and results for the belted kingfisher. The same process outlined in Section 8.1.5.1 to characterize risks to other wildlife receptors was used to characterize risks to belted kingfisher.

Thirteen COPCs were identified for belted kingfisher in the SLERA and refined screen. HQs could not be calculated for aluminum because no LOAEL was available. Six COPCs have HQs ≥ 1 based on individual prey components (Attachment 17): lead, mercury, BEHP, total PCBs, total TEQ, and sum DDE. HQs were calculated for these COPCs across multiple prey and incidental sediment ingestion using LOAEL-based TTCs and TSCs, as shown in Table 8-29.

Table 8-29. Belted Kingfisher LOAEL HQs Within 1-Mile Exposure Areas

Exposure Area	Total HQ					
	Lead	Mercury	BEHP	Total PCBs	Total TEQ	Sum DDE
RM 1.5 to RM 2.5	0.051	0.53	0.0089	**2.8**	0.44	0.054
RM 2.5 to RM 3.5	0.014	0.36	0.0045	0.15	0.053	0.028
RM 3.5 to RM 4.5	0.065	0.33	0.019	0.37	0.31	0.040
RM 4.5 to RM 5.5	0.21	0.40	0.41	0.17	0.031	0.052
RM 5.5 to RM 6.5	0.044	0.49	0.0045	0.15	0.055	0.056
RM 6.5 to RM 7.5	0.27	1.0	0.0066	**2.4**	0.67	1.0
RM 7.5 to RM 8.5	0.036	0.85	0.0027	0.21	0.049	0.26
RM 8.5 to RM 9.5	0.086	0.65	0.0052	0.35	0.13	0.062
Swan Island Lagoon	0.057	0.51	**1.3**	0.56	0.042	0.041

Table 8-29. Belted Kingfisher LOAEL HQs Within 1-Mile Exposure Areas

Exposure Area	Total HQ					
	Lead	Mercury	BEHP	Total PCBs	Total TEQ	Sum DDE
RM 9.5 to RM 10.5	0.042	0.63	0.0055	0.70	0.083	0.045
RM 10.5 to RM 11.8	0.038	0.43	0.0017	**7.3**	0.12	0.025

[a] Total HQ was calculated using the following prey portions: 90% sculpin and 10% field clams.

BEHP – bis(2-ethylhexyl) phthalate
DDE – dichlorodiphenyldichloroethylene
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
PCB – polychlorinated biphenyl
RM – river mile
TEQ – toxic equivalent

Bold identifies HQs ≥ 1.\

HQs based on multiple prey for all exposure areas are < 1 for lead, mercury, total TEQ, and sum DDE. However, HQs for BEHP and total PCBs are ≥ 1 in at least one exposure area.

A high degree of uncertainty is associated with the effects assumptions used to derive the risk estimates for BEHP. The LOAEL for BEHP (11 mg/kg bw/day) was extrapolated from the NOAEL using aUF of 10, per EPA (2008f). The extrapolation of a LOAEL from a NOAEL is unprecedented and furthermore, the extrapolated BEHP LOAEL is an order of magnitude lower than the LWG-recommended LOAEL derived directly from the literature (Attachment 14). The literature-based LOAEL of 329 mg/kg bw/day was calculated from Ishida et al. (1982) and was the only LOAEL reported in the three toxicological studies reviewed. At this LOAEL, egg production ceased in domestic chickens following 230 days of exposure (also during a critical life stage) (Ishida et al. 1982). Though there is uncertainty associated with the literature-derived LOAEL (because the literature dataset for BEHP toxicity to birds is limited to three studies using highly variable dose concentrations), the literature-based LOAEL is more appropriate for evaluating risks to birds than the extrapolated LOAEL. Using the literature-based BEHP LOAEL (329 mg/kg bw/day), the TTC and TSC for belted kingfisher are 617 mg/kg ww and 500,000 mg/kg dw, respectively. No individual samples exceeded these TTCs or TSCs for BEHP.

The characterization of BEHP based on the EPA directed LOAEL is too uncertain to draw conclusions about potentially unacceptable risk to kingfisher. The literature-based LOAEL was not exceeded. Total PCBs exceed the LOAEL for belted kingfisher in the same areas as those identified for hooded merganser and bald eagle. Therefore, the selected bird ecological receptors are protective of belted kingfisher.

8.1.5.3 COIs for Which Risks Cannot Be Quantified

COIs for which dietary risks to birds and mammals cannot be quantified based on the dietary LOE are listed in Table 8-30. These COIs represent chemicals for which no TRV is available as well as chemicals whose maximum DL exceeded a TRV but whose detected values did not.

Table 8-30. Wildlife Dietary COIs with No Available TRV or with DLs Exceeding Screening-Level TRVs

COI	Rationale for Absence of Quantitative Risk Evaluation
Metals	
Antimony[a]	Dietary risk to birds unknown; no dietary TRV available.
Manganese	Dietary risk to birds and mammals unknown; no dietary TRV available.
Silver	Dietary risk to birds and mammals unknown; no dietary TRV available.
PAHs	
1-Methylnaphthalene[a]	Dietary risk to birds unknown; no dietary TRV available.
2-Methylnaphthalene[a]	Dietary risk to birds unknown; no dietary TRV available.
Benzo(e)pyrene	Dietary risk to birds and mammals unknown; no dietary TRV available.
Dibenzothiophene[a]	Dietary risk to birds unknown; no dietary TRV available.
Perylene	Dietary risk to birds and mammals unknown; no dietary TRV available.
Alkylated PAHs	Dietary risk to birds and mammals unknown; no dietary TRV available.
SVOCs	
Benzoic acid	Dietary risk to birds and mammals unknown; no dietary TRV available.
Benzyl alcohol[a]	Dietary risk to birds unknown; no dietary TRV available.
Carbazole	Dietary risk to birds and mammals unknown; no dietary TRV available.
Dibenzofuran	Dietary risk to birds and mammals unknown; no dietary TRV available.
Hexachloroethane[a]	Dietary risk to birds unknown; no dietary TRV available.
n-Nitrosodiphenylamine	Dietary risk to birds and mammals unknown; no dietary TRV available.
2-Methylphenol	Dietary risk to birds and mammals unknown; no dietary TRV available.
4-Chloro-3-methylphenol	Dietary risk to birds and mammals unknown; no dietary TRV available.
4-Methylphenol[a]	Dietary risk to birds unknown; no dietary TRV available.
Phenol[a]	Dietary risk to birds unknown; no dietary TRV available.
Phthalates	
Dibutyl phthalate	Dietary risk to osprey unknown; 40% of non-detected carp tissue samples had DLs > osprey TTC, but no detected fish prey or sediment samples exceeded screening-level TRVs; dibutyl phthalate was retained as dietary COPC for spotted sandpiper and hooded merganser.

[a] No bird dietary screening-level threshold was available; however, a mammal dietary threshold was available.

COI – contaminant of interest
COPC – contaminant of potential concern
DL – detection limit
PAH –polycyclic aromatic hydrocarbon
SVOC –semivolatile organic compound
TRV – toxicity reference value
TTC – threshold tissue concentration

8.1.5.4 Summary of Bird and Mammal Diet LOE

Twelve bird COPCs have HQs ≥ 1 in Step 3 for at least one avian receptor: copper, lead, mercury, benzo(a)pyrene, dibutyl phthalate, total PCBs, PCB TEQ, total dioxin/furan TEQ, total TEQ, aldrin, DDE (as sum DDE or 4,4′-DDE), and total DDx. The mammal COPCs with HQs ≥ 1 in Step 3 for mink, river otter, or both are lead, total PCBs, and total TEQ, as well as PCB TEQ and total dioxin/furan TEQ, the two components of total TEQ. The HQs and uncertainties associated with the baseline dietary assumptions are described in Section 8.1.5.1 and 8.1.5.2.2. The magnitude, spatial distribution, and frequency of HQs ≥ 1; the underlying uncertainties of exposure and effects data; and agreement of HQs across LOEs (where applicable) are discussed in Section 8.3.3 to determine the risk conclusions for wildlife.

8.2 BIRD EGG TISSUE ASSESSMENT

The tissue residue LOE, wherein, concentrations measured in bird egg tissue were compared to literature-derived bird egg TRVs was one of two LOEs used to evaluate risks to piscivorous birds (osprey and bald eagle) from exposure to site-related chemicals. The dietary LOE was the other LOE used for evaluating risks to piscivorous birds (Section 8.1).

The following section presents the assessment based on chemical residues in osprey eggs. COPCs were identified in the SLERA and refined screen by comparing osprey egg data to screening-level egg tissue TRVs (Attachment 5). These COPCs were evaluated for both bald eagle and osprey by comparing egg tissue toxicity thresholds to the chemical concentrations in osprey eggs collected from the Study Area.

For each receptor, COPCs with HQs ≥ 1 pose potentially unacceptable risk. For COPCs with HQs ≥ 1, the magnitude of HQs, the spatial distribution and frequency of HQs ≥ 1, results of multiple LOEs (when applicable), and the associated exposure and effects assumptions are evaluated in Section 8.2.4 to arrive at risk conclusions for pisciviorous birds.

The details of this bird egg risk assessment are presented as follows:

- Section 8.2.1 summarizes the COPCs identified for bald eagle and osprey.
- Section 8.2.2 summarizes the exposure data, as represented by COPC concentrations in samples of individual bird eggs. All egg tissue concentrations are presented in Attachment 4.
- Section 8.2.3 summarizes the effects data, as represented by EPA-recommended NOAEL and LOAEL TRVs. Details and uncertainties associated with the selected TRVs for wildlife dietary COPCs are presented in Attachment 16. The comprehensive literature search process is presented in Attachment 14.
- Section 8.2.4 presents the risk characterization results, receptor-COPC pairs resulting in HQs ≥ 1, and associated uncertainties. These COPCs are further

assessed in the wildlife risk conclusions section (Section 8.3). The risk characterization results of the individual sample analysis are presented in Attachment 17.

Figure 8-10 presents a flowchart of the bird egg assessment section organization.

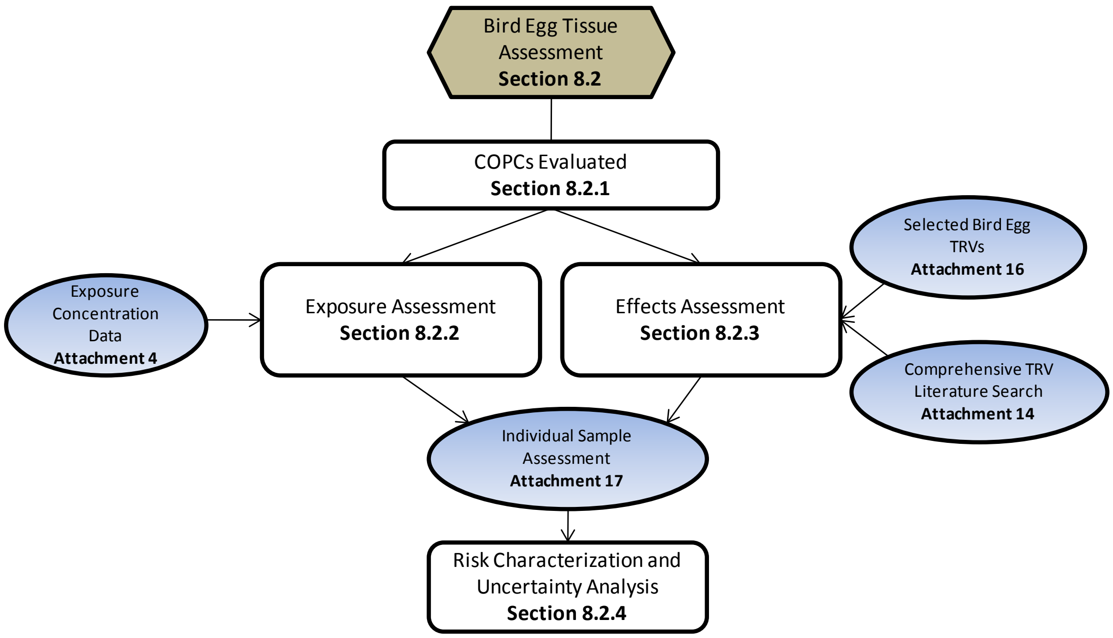


Figure 8-10. Overview of Bird Egg Assessment Section Organization

8.2.1 COPCs Evaluated

Receptor-COPC pairs were identified in the SLERA and refined screen (Attachment 5). The bird egg COPCs evaluated are listed in Table 8-31.

Table 8-31. Bird Egg COPCs

COPC	Bald Eagle	Osprey
PCBs		
Total PCBs	X	X
PCB TEQ[a]	X	X
Dioxins/Furans		
Total dioxin/furan TEQ[a]	X	X
Total TEQ[a]	X	X
Pesticides		
4,4′-DDE	X	X

[a] Per EPA (Attachment 2), TEQ was evaluated as PCB TEQ, total dioxin/furan TEQ, and total TEQ.

COPC – contaminant of potential concern
PCB – polychlorinated biphenyl
DDE – dichlorodiphenyldichloroethylene
TEQ – toxic equivalent
EPA – US Environmental Protection Agency

DDE was assessed as 4,4′-DDE in the evaluation of bird eggs. The best documented response to DDE is eggshell thinning in birds, which can result in embryo mortality and decreased hatchling survival (Heath et al. 1969; Lincer 1975). The leading hypothesis for DDE-induced thinning involves an inhibition of prostaglandin synthesis in the shell gland mucosa by 4,4′-DDE (but not by 2,4′-DDE, or DDD or DDT isomers) (EPA 2007b; Lundholm 1997).

TEQs for dioxins, furans, and PCBs are the remaining bird egg COPCs. Derivation of TEQs and associated uncertainties are discussed in Section 8.1.2.

8.2.2 Exposure Assessment

Bird egg tissue EPCs in this assessment are represented by chemical concentrations measured in samples of indiviudal osprey eggs collected from the Study Area. Although bird egg tissue data are available only for osprey, they were used in the assessment of both osprey and bald eagle. Osprey data are expected to be protective of bald eagles for several reasons. With foraging ranges of similar size (Table 8-3), the spatial extent of exposure from the Study Area is likely to be similar for both receptors. Osprey consume solely fish, whereas the bald eagle diet includes items not exposed to Study Area contaminants (Attachment 16). Additionally, relative to their body weight, the FIR for osprey is higher than that for bald eagle (Table 8-4), indicating that ospreys have a greater rate of contaminant intake. It is also important to note that osprey overwinter in Mexico and Central America, and they nest and lay eggs within a short time after returning to the lower Willamette (Henny et al. 2003), whereas bald eagle may be migratory or resident.Bird egg EPCs are represented by concentrations in the five available individual samples, as shown in Table 8-32. Each was presumed to be representative of a 1-mile exposure area (Table 8-3) despite what is known about the migratory behavior of osprey that nest in the Study Area and its likely implication for the source of egg tissue COPC residues.

Table 8-32. Osprey Egg EPCs Within 1-Mile Exposure Areas

Exposure Area[a]	Dioxin TEQ (pg/g ww)	PCB TEQ (pg/g ww)	Total TEQ (pg/g ww)	Total PCBs (µg/kg ww)	4,4′-DDE (µg/kg ww)
RM 2.5 to RM 3.5	14 T	74.3 T	88.3 T	3,660 JT	1,490
RM 5.5 to RM 6.5	20.1 T	93.3 T	113 T	8,550 JT	2,450
RM 6.5 to RM 7.5	26.3 T	44.5 T	70.8 T	3,800 JT	1,100
RM 8.5 to RM 9.5	21.1 T	101 T	122 T	19,700 JT	1,112
RM 10.5 to RM 11.8	58.6 T	9.64 T	68.2 T	580 JT	1,150

[a] Exposure areas for which egg data are not available are not listed.

DDE – dichlorodiphenyldichloroethylene
EPC – exposure point concentration
J – estimated concentration
PCB – polychlorinated biphenyl
RM – river mile
T – value calculated or selected from multiple results
TEQ – toxic equivalent
ww – wet weight

8.2.3 Effects Assessment

This section presents the selected TRVs used to characterize effects for bird egg COPCs and the uncertainties associated with these selected values. Bird egg TRVs are expressed as mg/kg ww in bird egg tissues and are based on LOAELs and NOAELs derived from the toxicological literature. A NOAEL and a LOAEL were selected for each COPC. TRVs for PCBs and dioxin/furans were selected for both total PCBs and TEQs (as dioxin/furan TEQ, PCB TEQ, and total TEQ). The effects data presented in this section are assessed in combination with exposure data (presented in Section 8.2.2) in the risk characterization (Section 8.2.4).

EPA's Problem Formulation (Attachment 2) requires the use of LOAELs and a population-level assessment for osprey but NOAELS and an organism-level assessment for bald eagle.

Per EPA (2008j), bird egg TRVs are based on field data for representative piscivorous bird species from the Willamette River region, when available. Field-based TRVs were derived from detected chemical concentrations in eggs associated with adverse effects (LOAEL) or no adverse effects (NOAEL). Attachment 16 summarizes the details, sources, and uncertainties associated with all of the selected TRVs. Attachment 14 presents details of the literature-based bird egg TRVs for all COPCs. The bird egg TRVs adopted for this BERA are listed in Table 8-33. Key uncertainties are also noted.

Table 8-33. Bird Egg Tissue Residue TRVs

	TRV (mg/kg ww)			
COPC	**NOAEL**	**LOAEL**	**Source**	**Key Uncertainty**
PCBs and Dioxins/Furans				
Total PCBs	3.0	4.5	NOAEL – Wiemeyer et al. (1993); LOAEL – Wiemeyer et al. (1984)	NOAEL and LOAEL are based on national bald eagle field data associated with productivity.
PCB TEQ	2.3×10^{-6}	3.198×10^{-5}	NOAEL – Henny et al. (2003); LOAEL – Anthony et al. (1993)	NOAEL and LOAEL are based on regional osprey and bald eagle field data associated with productivity and eggshell thinning.
Total dioxin/furan TEQ	2.3×10^{-6}	3.198×10^{-5}	NOAEL – Henny et al. (2003); LOAEL – Anthony et al. (1993)	
Total TEQ	2.3×10^{-6}	3.198×10^{-5}	NOAEL – Henny et al. (2003); LOAEL – Anthony et al. (1993)	

Table 8-33. Bird Egg Tissue Residue TRVs

COPC	TRV (mg/kg ww) NOAEL	TRV (mg/kg ww) LOAEL	Source	Key Uncertainty
Pesticides				
4,4′-DDE	1.3	3.5	NOAEL and LOAEL – Wiemeyer et al. (1984)	NOAEL and LOAEL are based on national bald eagle field data associated with productivity.

COPC – contaminant of potential concern
DDE – dichlorodiphenyldichloroethylene
LOAEL – lowest-observed-adverse-effect level
NOAEL – no-observed-adverse-effect level
PCB – polychlorinated biphenyl
TEQ – toxic equivalent
TRV – toxicity reference value

Uncertainty of Bird Egg TRVs Based on Field Data

Unlike the selected dietary TRVs, bird egg TRVs are based primarily on field-collected data. Per EPA (2008j), bird egg TRVs are based on thresholds reported in field studies, when data are available. The use of field-collected data allows for the selection of toxicological data based on Willamette-specific receptors (i.e., osprey and bald eagle) to ensure that these receptors are protected. However, uncertainties are associated with the use of TRVs based on field-collected data.

NOAELs based on field data were derived from egg residues in bird populations in which no effects were reported. Because these NOAELs reflect multiple stressors and complex chemical mixtures associated with field conditions, they are reliable in that lower concentrations are unlikely to cause adverse effects. However, because other uncharacterized chemicals and stressors may contribute to adverse effects, field-based NOAELs may not represent the upper range of the NOAEL for a given chemical. LOAELs based on field data were derived from egg tissue concentrations in which adverse population effects (e.g., productivity, eggshell thinning) were reported. Bird egg tissues in the field may contain other uncharacterized chemicals that could have contributed to the observed reproductive toxicity. Non-chemical stressors (such as habitat degradation) can also contribute to the adverse reproductive effects observed in the field.

Elliott and Harris (2001\2002) conducted a comprehensive review of available field and laboratory toxicity data for birds to select appropriate PCB, TEQ, and DDx effects thresholds for bald eagle. This review indicates that the EPA-selected TRVs for total PCBs and DDE are conservative but appropriate for the BERA. For total PCBs, Elliott and Harris (2001\2002) recommend a LOAEL other than the 4.5 mg/kg ww presented in (Wiemeyer et al. 1993), because of the intercorrelation of PCBs and DDE and the stronger influence of DDE on productivity. Instead, a LOAEL for PCBs of 20 mg/kg is recommended, based on an assessment of PCB effects on bald eagles in the Fox River/Green Bay system. This value is reasonably consistent with studies of other species that suggest higher thresholds for total PCBs compared to DDE (Elliott and Harris 2001\2002).

For DDE, Elliott and Harris (2001\2002) recommend 6 mg/kg ww as a threshold value at which productivity is affected. Various individual studies found threshold values ranging from 3.6 to 12 mg/kg ww, depending upon the period of egg collection. Using this wide range of studies from both published and unpublished data, Elliott and Harris (2001\2002) developed a linear relationship between productivity and DDE concentrations in eggs and determined that 6 mg/kg ww was the strongest estimate they could determine as a threshold of effects. Thus, the EPA-selected NOAEL and LOAEL TRVs of 1.3 and 3.5 mg/kg ww, respectively, are conservative effect concentrations representing data from only one study rather than the multiple lines of evidence approach used by Elliott and Harris (2001\2002).

The EPA-selected NOAEL TEQ effects threshold of 2.3 x 10^{-6} is approximately 2 orders of magnitude lower than the value of 3.03 x 10^{-4} mg/kg ww recommended by Elliott and Harris (2001\2002). The Elliott and Harris value was derived from a NOAEL of 2.10 x 10^{-4} ww, representing the 2,3,7,8-TCDD concentration in bird eggs from sites in the vicinity of a kraft pulp and paper mill in British Columbia at which no effects on hatchability were observed compared to a reference site. These data were further assessed in an unpublished report cited by Elliott and Harris (2001\2002), and the NOAEL was determined to be 3.03 x 10^{-4} mg/kg ww. These data indicate that the EPA-selected LOAEL for this BERA is highly conservative.

8.2.4 Risk Characterization and Uncertainty Analysis

The following section presents the bird egg risk characterization. A deterministic risk characterization was conducted using available osprey egg data. An HQ was used to quantify risk estimates (Equation 8-2). HQs were derived for bird egg tissue COPCs using the following equation:

$$HQ = \frac{EPC}{TRV}$$

Equation 8-2

Where:

HQ = hazard quotient

EPC = exposure point concentration

TRV = toxicity reference value

The EPC and TRV are represented by bird egg tissue-residue concentrations expressed as mg/kg ww.

Section 8.2.4.1 presents the risk characterization results and uncertainty evaluation for bald eagle, and Section 8.2.4.2 presents those for osprey. An evaluation of osprey egg data from the Willamette River outside of the Study Area is presented in Section 8.2.4.3. Results of the piscivorous bird egg LOE, along with results of the dietary LOE were evaluated together considering the relative strengths of each LOE and associated uncertainties, as presented in the wildlife risk conclusions section (Section 8.3).

8.2.4.1 Bald eagle

Five COPCs were identified in the SLERA and refined screen: total PCBs, PCB TEQ, total dioxin/furan TEQ, total TEQ, and 4,4′-DDE (Table 8-31). Bald eagle HQs were calculated for these COPCs by comparing osprey egg tissue concentrations (Attachment 4) to NOAELs (Table 8-34). All five COPCs have HQs ≥ 1 in at least one exposure area (Map 8-4).

Table 8-34. Bald Eagle Bird Egg NOAEL HQs Within 1-Mile Exposure Areas

	HQ				
Exposure Area[a]	**Total PCBs**	**PCB TEQ**	**Total Dioxin/Furan TEQ**	**Total TEQ**	**4,4′-DDE**
RM 2.5 to RM 3.5	**1.2**	**32**	**6.1**	**38**	**1.1**
RM 5.5 to RM 6.5	**2.9**	**41**	**8.7**	**49**	**1.9**
RM 6.5 to RM 7.5	**1.3**	**19**	**11**	**31**	0.85
RM 8.5 to RM 9.5	**6.6**	**44**	**9.2**	**53**	0.86
RM 10.5 to RM 11.8	0.19	**4.2**	**25**	**30**	0.88

[a] Exposure areas for which egg data are not available are not listed.

DDE – dichlorodiphenyldichloroethylene
HQ – hazard quotient
NOAEL – no-observed-adverse-effect level
PCB – polychlorinated biphenyl
RM – river mile
TEQ – toxic equivalent

Bold identifies HQs ≥ 1.

Several uncertainties are associated with the exposure and effects assumptions used to derive the risk estimates using the bird egg approach. The primary uncertainty related to exposure is the assumption that osprey egg data are representative of bald eagle egg data. Bald eagles forage in both aquatic and terrestrial environments, and they have lower ingestion rates relative to their body weight than do ospreys (12% body weight per day for bald eagle and and 21% body weight per day for osprey). However, osprey are migratory, whereas bald eagle may be migratory or resident to the Study Area. Given these differences, the use of osprey as a surrogate for bald eagle egg tissue-residue concentrations may over- or underestimate exposure and risk to bald eagle.

Uncertainty in the use of TEQ data arises from the inter-species variability of bird TEFs and the relevance of TEFs based on EROD induction to evaluate individual and population risks from dioxins/furans and dioxin-like PCBs. Use of a TEQ approach may contribute to over- or underestimates of risk by more than an order of magnitude from chemicals with a dioxin-like mode of action.

An additional source of uncertainty is the TRVs used to derive risk estimates, which are more conservative than values suggested in a comprehensive review of the effects of chorinated hydrocarbons on bald eagle populations (Elliott and Harris 2001\2002). As discussed in Section 8.2.3, Elliott and Harris (2001\2002) conducted a thorough review of available field and laboratory toxicity data for birds to select appropriate PCB, TEQ, and DDx effects thresholds for bald eagle. The effect threshold concentrations are 20 mg/kg ww for total PCBs and 6 mg/kg ww for DDE. An effect threshold was not available for TEQ, but the no-observed effect threshold was determined to be 3.03 x 10^{-4} mg/kg ww.

To evaluate the uncertainty associated with using the more conservative TRVs selected for the BERA calculations compared to the more comprehensive values from Elliott and Harris (2001\2002), bald eagle HQs were recalculated using the values from Elliott and Harris (Table 8-35). All HQs are < 1 based on these effects thresholds.

Table 8-35. Comparison of Bald Eagle Bird Egg NOAEL HQs with HQs Based on Recommended Effects Thresholds from Elliott and Harris

COPC	BERA NOAEL HQs	Elliott and Harris 2001/2002 HQ
PCBs		
Total PCBs	0.19 **– 6.6**	0.03 – 0.99
PCB TEQ	**4.2 – 44**	0.03 – 0.33
Dioxins/Furans		
Total dioxin/furan TEQ	**6.1 – 25**	0.05 **–** 0.99
Total TEQ	**30 – 53**	0.23 – 0.40
Pesticides		
4,4′-DDE	0.85 **– 1.9**	0.31 – 0.68

Source: Elliott and Harris (2001\2002)

BERA – baseline ecological risk assessment
COPC – contaminant of potential concern
DDE – dichlorodiphenyldichloroethylene
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
NOAEL – no-observed-adverse-effect level
PCB – polychlorinated biphenyl
TEQ – toxic equivalent

8.2.4.2 Osprey

Five COPCs were identified in the SLERA and refined screen: total PCBs, PCB TEQ, total dioxin/furan TEQ, total TEQ, and 4,4′-DDE (Table 8-31). Osprey HQs were calculated for these COPCs by comparing osprey egg concentrations to LOAELs (Table 8-36).

Table 8-36. Osprey Bird Egg LOAEL HQs Within 1-Mile Exposure Areas

	HQ				
Exposure Area[a]	**Total PCBs**	**PCB TEQ**	**Dioxin/Furan TEQ**	**Total TEQ**	**4,4′-DDE**
RM 2.5 to RM 3.5	0.81	**2.3**	0.44	**2.8**	0.43
RM 5.5 to RM 6.5	**1.9**	**2.9**	0.63	**3.5**	0.70
RM 6.5 to RM 7.5	0.84	**1.4**	0.82	**2.2**	0.31
RM 8.5 to RM 9.5	**4.4**	**3.2**	0.66	**3.8**	0.32
RM 10.5 to RM 11.8	0.13	0.30	**1.8**	**2.1**	0.33

[a] Exposure areas for which egg data are not available are not listed.

DDE – dichlorodiphenyldichloroethylene
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
PCB – polychlorinated biphenyl
RM – river mile
TEQ – toxic equivalent

Bold identifies HQs ≥ 1.

Total PCBs, PCB TEQ, total dioxin/furan TEQ, and total TEQ have HQs ≥ 1 in at least one exposure area. HQs are < 1 for 4,4′-DDE.

Several uncertainties are associated with the exposure and effects assumptions used to derive the risk estimates under the bird egg approach. The size of exposure areas for osprey is based on regional literature. The SUF was assumed to be 1 for osprey; however, the SUF is probably less than 1 because ospreys are likely to forage outside of the Study Area (fish prey are available from other water bodies in the area) and osprey winter outside of the Study Area (Henny et al. 2003). As such the default SUF probably overestimates Study Area exposure and risk. Uncertainty in the use of TEQ data arises from inter-species variability of bird TEFs and the relevance of TEFs based on EROD induction to evaluate individual and population risks from dioxins/furans and dioxin-like PCBs. Use of a TEQ approach may contribute to over- or underestimates of risk by more than an order of magnitude.

Further uncertainty with the bird egg approach arises because TRVs used to derive risk estimates are based on field LOAELs that attribute all obeserved toxicity to a single contaminant. Avian reproductive effects have been associated with PCBs (including dioxin-like PCBs), dioxins/furans, and DDTs, and it is not possible in field studies to isolate the effects due to any particular contaminant because birds in the field are exposed to mixtures of these three contaminant groups. Non-contaminant stressors may also contribute to declines in reproduction, further confounding the field LOAELs. Because the TRVs are based on field LOAELs they are biased to underestimate the effect threshold and overestimate risk.

Because the field-collected effects data are based on bald eagle studies, they may over- or underestimate risk to osprey. The regional data for osprey provide only unbounded NOAELs for PCBs and TEQs, leading to uncertainty in the osprey-specific effect threshold. No additional data are available on the relative sensitivity of osprey and bald eagle to total PCBs. For TEQs, the only osprey-specific TEQ LOAEL (136 ng/kg ww for reduced chick growth in osprey from the Wisconsin River [see Attachment 14]) is higher than the selected bald eagle LOAEL by a factor of approximately 4. However, the bald eagle LOAEL (32 ng/kg ww) is within the bounds of the available osprey-specific NOAEL and LOAEL (2.3 and 136 ng/kg ww, respectively). Furthermore, EPA (2003b) notes that the field-based total TEQ effects thresholds are believed to be lower than laboratory-based 2,3,7,8-TCDD effects thresholds for chickens (by far the most sensitive species tested) due to effects of non-dioxin-like co-contaminants in the field.

There is an additional source of uncertainty associated with the TRVs used to derive risk estimates based on a comparison to effects thresholds presented in a comprehensive review of the effects of chorinated hydrocarbons on bald eagle populations (Elliott and Harris 2001\2002). This review of bald eagles indicates that the selected TRVs for osprey, a species similar to bald eagle, may overestimate risk. As discussed in Section 8.2.3, Elliott and Harris (2001\2002) conducted a thorough review of available field and laboratory toxicity data for birds to select appropriate PCB, TEQ, and DDx effects thresholds for bald eagle. The effect threshold concentrations for total PCBs and DDE based on this review were 20 and 6 mg/kg ww, respectively, slightly higher than the selected LOAEL TRVs of 4.5 and 3.5 mg/kg ww, respectively. An effect threshold was not available for TEQ, but the no-observed effect threshold was determined to be 3.0×10^{-4} mg/kg ww, two orders of magnitude higher than the selected NOAEL TRV of 2.3×10^{-6} mg/kg ww. HQs calculated using the effects thresholds from Elliott and Harris compared with the BERA HQs are the same as those calcuated for bald eagle (Table 8-35), and all are < 1.

Field data on osprey reproductive success (i.e., productivity) in terms of number of fledged young per nest (young/successful nest) in the Study Area are limited; however, some data are available from the lower reach of the Willamette River (RM 0 to RM 26), which includes the Study Area. Nesting success of osprey was monitored along the Willamette River system between 1993 and 2001 (Henny et al. 2009). Nests were classified as occupied (adult pair present), active (eggs laid), and/or successful (fledged young observed). Between RM 0 and RM 26, the number of osprey nests increased from 1993 to 2001; one active nest was observed in 1993, and 10[134] active nests were observed in 2001. The productivity of osprey in 2001 in the section of the Willamette River from RM 0 to 26 was reported by Henny et al. (2009) as 1.75 young per all types of nest (occupied, active, successful). This rate of 1.75 is similar to the productivity of osprey that Henny et al. (2009) reported in upstream sections of the Willamette River (average 1.77 young/active nest in the Upper River and Santiam River sections combined) and well above the rates of 0.7 and 0.8 young/active nest that have been reported to be the minimum required rate to maintain stable bald eagle and osprey populations, respectively

[134] Three of the 10 nests in the LWR segment (from RM 0 to RM 26) were located in the Study Area, and one of the nests was located between RM 12 and RM 26. The locations of the other six nests within the LWR segment were not reported (Henny et al. 2009).

(Wiemeyer et al. 1984; Henny et al. 2009). Buck and Kaiser (2011) report that that the productivity of osprey in the Study Area in 2008 was also above the rate necessary to maintain a stable population. Additionally, thickness of osprey eggshells (a measure of potential reproductive impairment) collected in 2008 was not statistically different between the Study Area and the upstream reference area.

8.2.4.3 Evaluation of Non-Study Area Data

Concentrations of COPCs in bird eggs from outside of the Study Area were evaluated to determine the potential contribution of regional sources of contamination to Study Area risk. The following subsections present the HQs for all COPCs in each bird egg tissue sample.

Between May 2008 and September 2009, USGS/USFWS, in a joint effort with the Portland Harbor Natural Resource Trustees, collected osprey egg samples from five locations in the Study Area and from five locations in each of two reaches outside of the Study Area: the mid-Willamette River (RM 69 to RM 77) and Multnomah Channel (from Sauvie Island Bridge to the mouth of the Columbia River). Tissue samples of individual eggs from the five Study Area locations were analyzed for each bird egg COPC. Two mid-Willamette River and three Multnomah Channel individual egg tissue samples were analyzed for dioxin TEQ, PCB TEQ, total TEQ, and total PCB congeners. Analysis for 4,4′-DDE was conducted in all (15) tissue samples. All data are included in Attachment 4.

For DDE, the concentrations outside of the Study Area were compared with the bird egg LOAELs using analysis of variance (ANOVA). No statistical comparisons were made for total PCBs, PCB TEQ, or total TEQ concentrations because of the small sample size of bird egg data available from the mid-Willamette River reference area and Multnomah Channel (n = 2 and n = 3 samples for mid-Willamette River and Multnomah Channel, respectively). Both non-parametric (Kruskall Wallis) and parametric ANOVA tests were used to detect statistical differences based on log-transformed data. The Dunnett T3 *post hoc* multiple comparison test was also used to test for differences in log-transformed concentrations between pairs of areas.

Total PCB Concentrations

The total PCB congener LOAEL TRV was exceeded in two samples from the Study Area (HQs = 1.9 and 4.4). Total PCB concentrations for the three areas are shown by RM in Figure 8-11. The sample average concentration from the Study Area is greater than that for both the mid-Willamette River and Multnomah Channel. No statistical comparison was conducted because insufficient data were available from the mid-Willamette River and Multnomah Channel sites.

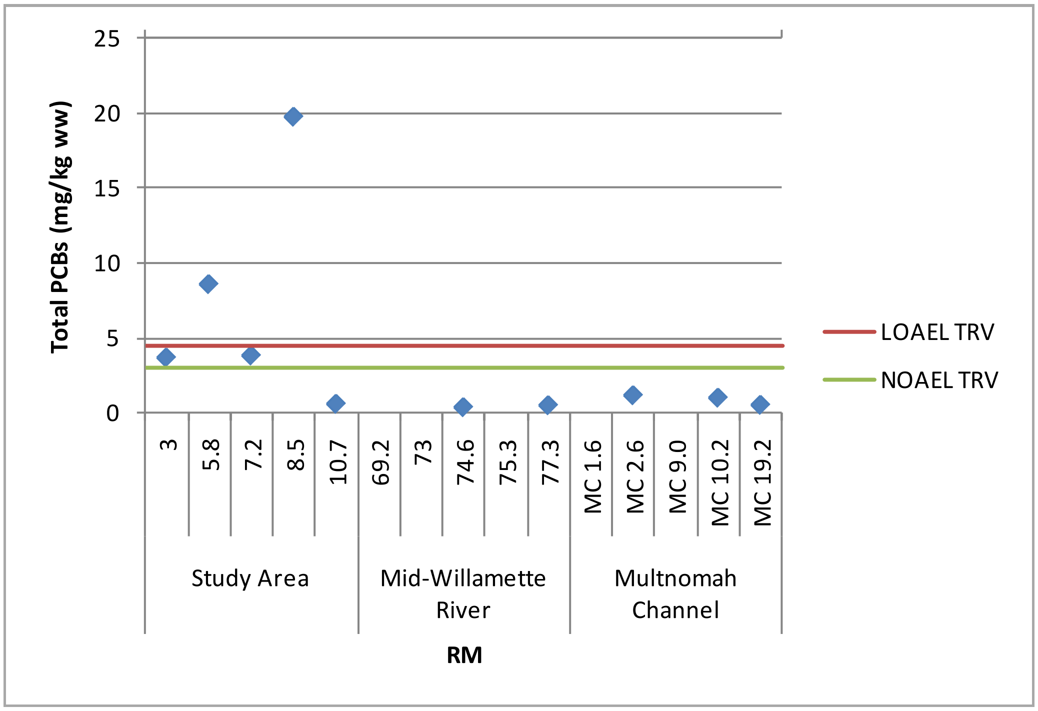


Figure 8-11. Total PCB Concentrations in Osprey Egg Tissue by RM from Study Area and Mid-Willamette River and Multnomah Channel Reference Areas

TEQ Concentrations

The bird egg TEQ HQs in the Study Area and mid-Willamette River, and Multnomah Channel are as follows:

- Dioxin TEQ – One of five Study Area bird egg samples exceeded the LOAEL TRV (HQ = 1.8) and one of two mid-Willamette River bird egg samples exceeded the LOAEL TRV (HQ = 1.4). None of the three bird egg samples from the Multnomah Channel exceed this LOAEL TRV.
- PCB TEQ – Four of the five Study Area bird egg samples exceeded the PCB TEQ LOAEL (HQs = 1.4 to 3.2) and one of the three Multnomah Channel samples exceeds the PCB TEQ LOAEL (HQ = 1.1). Neither mid-Willamette River sample exceeded this TRV.
- Total TEQ – Each of the five bird egg samples collected within the Study Area exceeded the total TEQ LOAEL (HQs = 2.1 to 3.8), as did one of the two mid-Willamette River samples and two of three Multnohah Channel samples samples (HQ = 1.7 and HQs = 1.2 and 1.4, respectively).

Figures 8-12 though 8-14 present TEQ concentrations by RM from the Study Area mid-Willamette River, and Multnomah Channel. Mid-Willamette River PCB TEQ concentrations were lower than Multnomah Channel and Study Area concentrations, and four of the five Study Area samples had higher concentrations than the maximum concentration from Multnomah Channel (Figure 8-12). No statistical comparison was conducted because only insufficient data were available from the mid-Willamette River and Multnomah Channel areas.

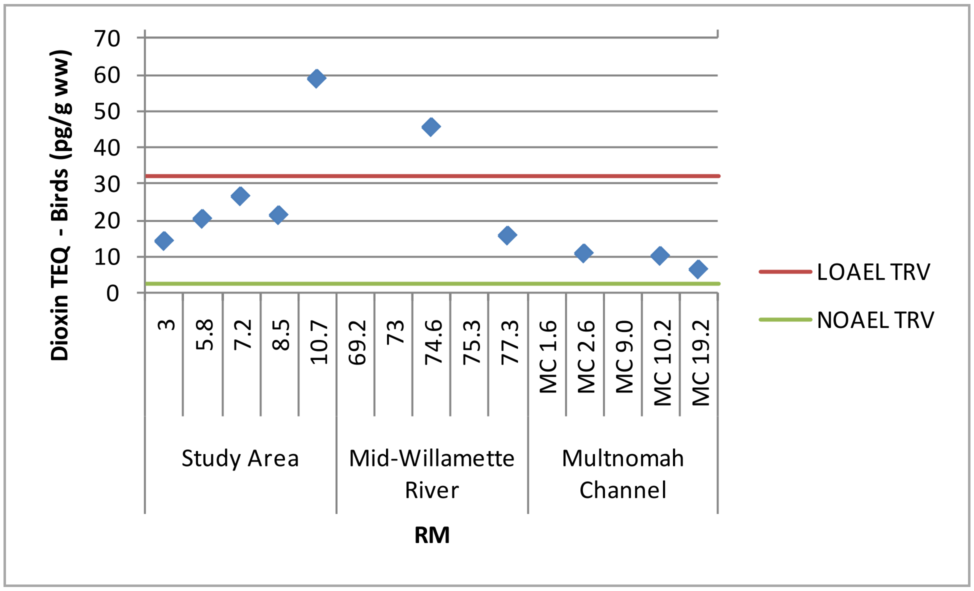


Figure 8-12. Dioxin TEQ Concentrations in Bird Egg Tissue by RM from Study Area and Mid-Willamette River and Multnomah Channel Reference Areas

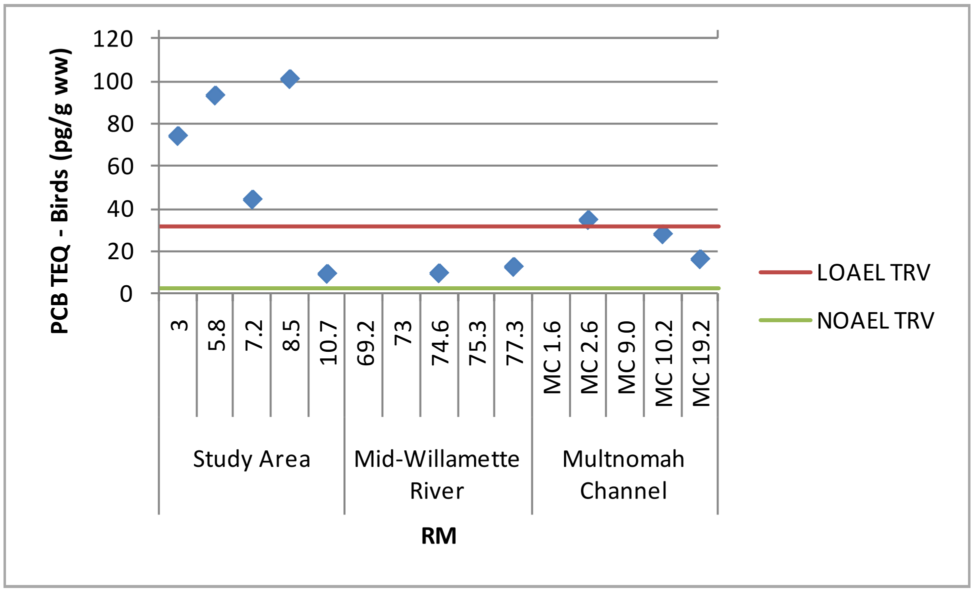


Figure 8-13. PCB TEQ Concentrations in Bird Egg Tissue by RM from Study Area and Mid-Willamette River and Multnomah Channel Reference Areas

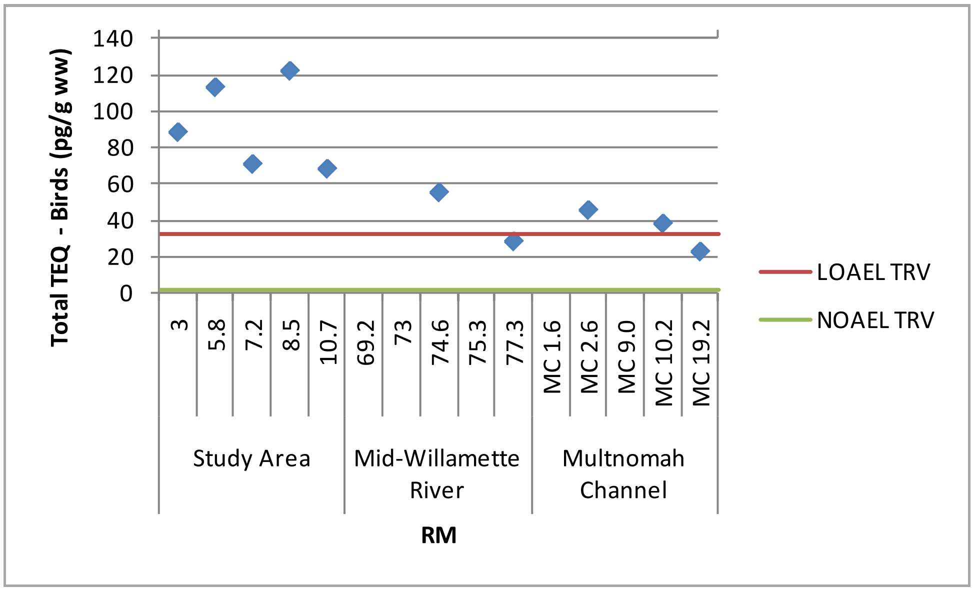


Figure 8-14. Total TEQ Concentrations in Bird Egg Tissue by RM from Study Area and Mid-Willamette River and Multnomah Channel Reference Areas

4,4'-DDE Concentrations

Plots of 4,4′-DDE concentrations for the three areas are shown in Figure 8-15. The t-test indicates that the average concentration in the Study Area was significantly higher than in the mid-Willamette River ($p = 0.021$); however, no samples from either of the three areas exceeded the 4,4′-DDE LOAEL. Concentrations of 4,4′-DDE in all of the Study Area eggs lie between the two highest concentrations in Multnomah Channel. The non-parametric ANOVA was more powerful than the parametric for 4,4′-DDE and detected a statistical difference among the three areas ($p = 0.035$). Based on the Dunnett T3 *post hoc* test on log-transformed concentrations, the means of Study Area and mid-Willamette concentrations are significantly different ($p = 0.04$), but Study Area and Multnomah Channel concentrations do not differ significantly ($p = 0.55$).

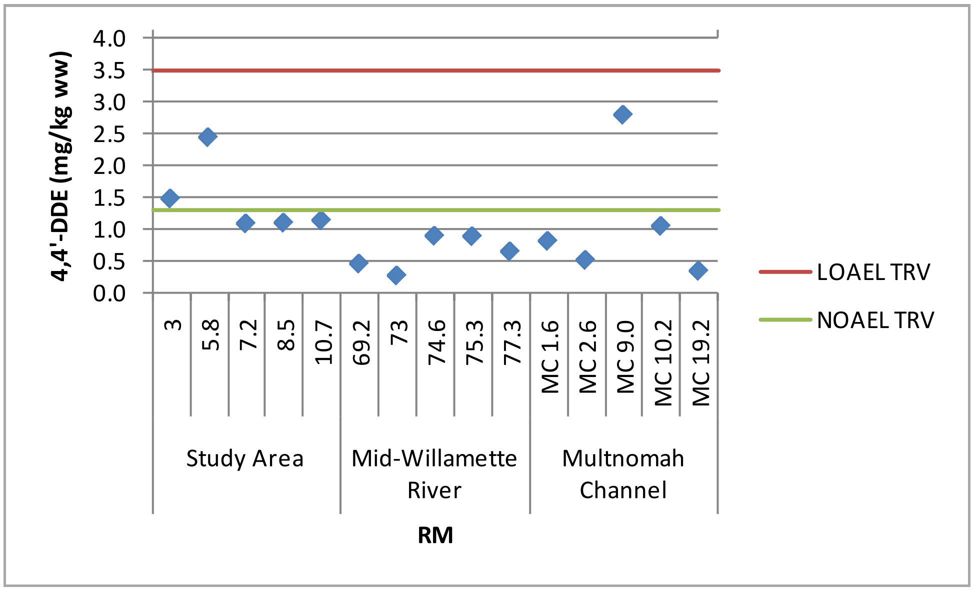


Figure 8-15. 4,4′-DDE Concentrations in Bird Egg Tissue by RM from Study Area and Mid-Willamette River and Multnomah Channel Reference Areas

8.3 RISK CONCLUSIONS

This section presents a summary of the overall conclusions of the wildlife risk assessment. Bird and mammal risks were identified using the dietary dose and (for bald eagles and osprey) bird egg LOEs. Risk conclusions were reached by evaluating the magnitude of HQs, the spatial distribution and frequency of HQs ≥ 1, and the uncertainty of exposure and effects assumptions. Background concentrations were considered, as appropriate, to put risk conclusions in context. Background concentrations were not, however, “subtracted out” or otherwise used to discount ecological risks. The outcome of the WOE analysis of the dietary dose and bird egg LOEs was used, in part, to determine conclusions for bald eagle and osprey. As per EPA ERAGs (EPA 1997), the risk conclusions identify receptor-COPC pairs that are estimated to be primary contributors to potentially unacceptable ecological risk to wildlife.

Several COPCs have been identified as posing potentially unacceptable risk for one or more wildlife receptors. However, a substantial amount of uncertainty is involved with using TRVs based on organism-level attributes to extrapolate to population-level risk (refer to green box in Section 3.2). Four additional types of information were therefore considered when assessing the risk of ecologically significant effect at the population level: the level of effect observed in the study from which the TRV was derived, the magnitude of the HQ, the spatial extent of the HQs ≥ 1, and uncertainties in the exposure and effects data. Field data available on the reproductive success of osprey in a portion of the Study Area were considered in drawing risk conclusions for osprey.

Potentially unacceptable risk is posed by COPCs whose HQs are ≥ 1 in the final step of the risk characterization process, which includes examination of relevant exposure scales and dietary assumptions. The foraging range assumptions and resulting foraging areas used for exposure analyses and risk characterization are presented for each wildlife receptor in Table 8-3. The uncertainties associated with risk estimates for individual COPCs, the spatial distribution of COPC exceedances, the magnitude of exceedance, and the level of effect represented by the TRV all play a role in assessing whether chemicals pose a population-level risk. For example, a COPC with a limited spatial distribution of HQs ≥ 1, low HQs, and a TRV based on a low level of effect (e.g., a small percent decrease in growth) is not likely to pose a significant risk to Study Area populations. Conversely, a COPC with a broad distribution of HQs ≥ 1, high HQs, and a TRV based on substantial effects (e.g., a large increase in mortality) is more likely to pose ecologically significant risk at the population level.

After consideration of these factors, PCBs were found to be the most significant contributor to potential wildlife risk. The mink population is the wildlife receptor most vulnerable to PCB exposure. The organism-level assessment probably overestimates the potential population-level risk to mink because the population has some capacity to compensate for individual kits lost to PCB toxicity, and because an analysis of the exposure uncertainties (presented below in Section 8.3.3.2.1) indicates that exposure probably has been overestimated.

This risk conclusions section summarizes the wildlife COPCs with HQs ≥ 1 (Section 8.3.1), presents a WOE evaluation for bald eagle and osprey (Section 8.3.2), and presents risk conclusions for all wildlife COPCs (Section 8.3.3). In Section 11.0, the wildlife conclusions are combined with those for other ecological receptor groups to provide a comprehensive view of ecological risks; COPCs and receptors associated with ecologically significant risks in the Study Area are highlighted.

8.3.1 Bird and Mammal COPCs with HQs ≥ 1

Table 8-37 tabulates the wildlife receptor-COPC pairs resulting in HQs ≥ 1 in the final step of risk characterization. These receptor-COPC pairs represent potentially unacceptable risk because the EPC for the final step in the risk characterization exceeded the selected TRVs at a relevant exposure scale. Total PCBs was the only COPC resulting in an HQ ≥ 1 for all six wildlife receptors. This phenomenon was not unexpected; PCBs frequently drive assessed risk at contaminated sediment sites, and PCBs have also been identified in the BHHRA as the predominant contributor to potential human health risk in the Study Area. For the hooded merganser, total PCBs was the only COPC resulting in an HQ ≥ 1; for river otter the only COPCs resulting in an HQ ≥ 1 were the PCB groups.

Table 8-37. Wildlife COPCs with Maximum HQ ≥ 1 from Final Step of Risk Characterization

	Receptor					
COPC	**Spotted Sandpiper**	**Hooded Merganser**	**Bald Eagle**	**Osprey**	**Mink**	**River Otter**
Metals						
Aluminum	NE[a]	NE[a]	Not a COPC	Not a COPC	**1.6**	Step 1 HQ < 1
Copper	**1.3**	Step 2 HQ < 1	Not a COPC	Not a COPC	Step 2 HQ < 1	Not a COPC
Lead	Step 3 HQ < 1	Step 3 HQ < 1	Step 2 HQ < 1	**7.8**	**4.0**	Step 3 HQ < 1
Mercury	Step 1 HQ < 1	Step 2 HQ < 1	**1.7**[b] HQ < 1[c]	Step 3 HQ < 1[b, c]	Step 3 HQ < 1	Step 1 HQ < 1
PAHs						
Benzo(a)pyrene	**1.6**	Step 1 HQ < 1	Not a COPC	Step 1 HQ < 1	Not a COPC	Not a COPC
Phthalates						
Dibutyl phthalate	**1.4**	Step 1 HQ < 1	Not a COPC	Not a COPC	Not a COPC	Not a COPC
PCBs						
Total PCBs	**12**	**3.8**	**3.9**[b],**6.6**[c]	**1.1**[b], **4.4**[c]	**33**	**31**
PCB TEQ	**11**	Step 2 HQ < 1	Step 2 HQ < 1[b], **44**[c]	Step 1 HQ < 1[b], **3.2**[c]	**2.4**	**1.5**
Dioxins/Furans						
Total dioxin/furan TEQ	**17**	Step 2 HQ < 1	Step 2 HQ < 1[b], **25**[c]	Step 1 HQ < 1[b], **1.8**[c]	**2.0**	Step 3 HQ < 1
Total TEQ	**20**	Step 2 HQ < 1	Step 2 HQ < 1[b], **53**[c]	Step 1 HQ < 1[b], **3.8**[c,d]	**12**	**2.3**
Pesticides						
Sum DDE[d]	**1.3**	Step 1 HQ < 1	Step 3 HQ < 1[b], **1.9**[c,d]	Step 1 HQ < 1[b, c]	NE[e]	NE[e]
Total DDx	**1.4**	Step 1 HQ < 1	NE[e]	NE[e]	Step 1 HQ < 1	Step 1 HQ < 1
Aldrin	**1.7**	Not a COPC	Not a COPC	Not a COPC	Not a COPC	Not a COPC

a Not evaluated because no LOAEL TRV was identified.

[b] Based on the dietary-dose LOE.
[c] Based on the bird egg LOE.
[d] Sum DDE assessed as 4,4′-DDE in the bird egg LOE.
[e] Not evaluated because DDT and its metabolites were assessed as total DDx for mammals. For birds, the assessment used sum DDE in the dietary LOE and 4,4′-DDE in the bird egg LOE.

COPC – contaminant of potential concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
LOE – line of evidence
NE – not evaluated
PCB – polychlorinated biphenyl
TEQ – toxic equivalent
total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT, and 4,4′-DDT)

Bold indicates maximum HQ ≥ 1 in Step 3 of risk characterization.

The receptor with the most HQs ≥ 1 is spotted sandpiper (10), followed by bald eagle and mink (six each), osprey (five), river otter (three), and hooded merganser (one). In total, 13 COPCs have final HQs ≥ 1 for at least one wildlife receptor. The DDx COPCs are redundant in the sense that different forms were used to assess ecological risk (i.e., 4,4′-DDE was used in the bird egg LOE, and sum DDE and total DDx were used in the dietary-dose LOE). Sum DDE, 4,4′-DDE, and total DDx are three metrics for assessing the same risk. PCB TEQ, total dioxin/furan TEQ, and total TEQ risks are likewise three metrics for assessing the same risk. In a practical sense, then, ten COPCs pose potentially uncacceptable risk to wildlife: aluminum, copper, lead, mercury, benzo(a)pyrene, dibutyl phthalate, total PCBs, total TEQ (represented by PCB TEQ, total dioxin/furan TEQ, and total TEQ), aldrin, and DDx compounds (represented by sum DDE, 4,4′-DDE, and total DDx). The spatial extent, magnitude, and potential ecological significance of TRV exceedances and the concordance among LOEs for receptor-COPC pairs posing potentially unacceptable risk are discussed in Section 8.3.3 to determine risk conclusions.

8.3.2 WOE Evaluation for Piscivorous Birds

For osprey and bald eagle, both dietary and piscivorous bird egg LOEs were used to evaluate risks from specific bioaccumulative chemicals[135] including mercury, total PCBs, dioxins, furans, dioxin-like PCB congeners (evaluated as PCB TEQ, total dioxin/furan TEQ, and total TEQ), and 4,4′-DDE.

Per EPA’s Problem Formulation (Attachment 2), a WOE approach is needed to integrate the results of each LOE, the ultimate goal of which is:

> …to develop a method to help identify and rank which LOEs for each receptor provide the most scientifically reliable indication of the status of each assessment endpoint from exposure to COPCs at the site and, hence, which might be most useful for making risk management decisions.

[135] As agreed upon by LWG and EPA for the Ecological PRE (Windward 2005a),

A WOE was necessary only if there were multiple LOEs evaluated for a given COPC and their results did not agree. When only one LOE was used to evaluate a COPC or when HQs based on multiple LOEs agreed, no WOE was necessary to arrive at risk conclusions.[136] When the results of multiple LOEs were consistent, risk conclusions were based on the concordance of LOEs, taking into account the magnitude of HQs, spatial extent of HQs ≥ 1, uncertainties of exposure and effects assumptions, and the likelihood of ecologically significant adverse effects based on the TRV endpoints. When multiple LOEs did not agree, an evaluation of each LOE and the associated uncertainties was necessary to arrive at risk conclusions. The dietary and piscivorous bird egg LOEs are in agreement in identifying total PCBs as posing potentially unacceptable risk to both bald eagle and osprey. In contrast, the conclusions for the two LOEs for mercury, total TEQ, and DDx compounds conflict for at least one of the two receptors evaluated (Table 8-37), and a WOE evaluation is therefore needed. The WOE is considered in the risk conclusions in Section 8.3.3.

8.3.3 Wildlife Risk Assessment Conclusions

Two LOEs were evaluated to determine the risk resulting from exposure: direct exposure as measured through ingestion of prey and sediment, and, for piscivorous birds, egg tissue residues. The following subsections present a summary of the overall wildlife risk conclusions (Section 8.3.3.1) and a detailed evaluation of the results for PCBs and mink (Section 8.3.3.2).

8.3.3.1 Overall Conclusions Across All COPCs

Several factors affect interpretation of the quantitative risk analysis:

- Results of multiple LOEs for osprey and bald eagle COPCs
- Magnitude of HQs
- Spatial extent of HQs ≥ 1
- Implications of TRV exceedances based on COPC-specific toxicological data
- Uncertainty of exposure and effects assumptions

Of the 22 COPCs identified by the SLERA and refined screening process for the wildlife receptors, the primary contributor to potentially unacceptable risk is PCBs. Calculated risk estimates indicate that both mink and river otter populations in the Study Area might be experiencing reduced reproductive success because of exposure to PCBs. The potential risk to the mink population is estimated to be greater than the potential risk to the river otter population because mink metabolic requirements are higher. Reproductive success in spotted sandpipers, bald eagles, and ospreys might also be reduced because of

[136] When only one LOE was used, risk conclusions were derived on the basis of the single LOE, taking into account the magnitude of HQs, spatial extent of HQs ≥1.0, uncertainties of exposure and effects assumptions, and the likelihood of ecologically significant adverse effects based on the TRV endpoint.

PCB exposure. Overall, a greater degree of uncertainty is associated with PCB risk estimates for birds than for mammals because of uncertainty about exposure, uncertainty in the effects data, and because wildlife studies confirm a hatching and fledging success and a growing population of osprey within the upper and lower Willamette River, including the Study Area (Henny et al. 2009).

Total TEQ exposure also poses the potential for reduced reproductive success in mink, river otter, sandpiper, bald eagle, and osprey. Total TEQ HQs are generally lower than those for total PCBs. PCBs are responsible for the majority of total TEQ exposure, but the total dioxin/furan TEQ also exceeds TRVs in some locations of the Study Area. As is the case for total PCBs, a greater degree of uncertainty is associated with total TEQ risk estimates for birds than for mammals because of uncertainties in the former's exposure and effects data. Because the COPC most likely to cause a population-level effect is PCBs and because the receptor most likely to experience that effect is mink, a detailed examination of the PCB-mink assessment was undertaken, as reported in Section 8.3.3.2.

The calculated osprey and mink HQs for lead are ≥ 1 in one 1-mile exposure area (RM 9.5 to RM 10.5). The lead exposure estimate for both receptors is driven by one extreme outlier, specifically a composite sample of smallmouth bass with a lead concentration of 1,100 mg/kg ww. This concentration is over 100 times the other smallmouth bass concentration available from the same RM (6.8 mg/kg ww) and 2 to 5 orders of magnitude greater than lead concentrations detected in all other Study Area smallmouth bass samples (0.0048 to 1.8 mg/kg ww).

In addition to lead, total PCBs, and both PCB and total dioxin/furan TEQs have HQs ≥ 1. At the same time, field data evaluating nesting success in the LWR (including the Study Area) indicate that osprey populations have increased in recent years. From these data, it appears that osprey populations in the LWR (including the Study Area) are not at risk because they do not appear to be exhibiting adverse effects at the population level.

Risk to bald eagle from DDE based on osprey egg data indicates that DDx compounds pose low to negligible risks of reduced reproductive success to individual bald eagles within limited portions of the Study Area. Uncertainties in exposure and effects data make bald eagle risk predictions uncertain. The only other receptor with a DDx HQ > 1 is the spotted sandpiper population. DDx compounds are not likely to pose risk to the spotted sandpiper population because the HQs are of low magnitude over a limited spatial extent and likely overestimate risk for the reasons summarized in Table 8-38.

The remaining COPCs resulting in HQs ≥ 1 (i.e., aluminum, copper, mercury, benzo(a)pyrene, and aldrin) were not found to pose ecologically significant risk to the wildlife receptors evaluated, given the low magnitude of HQ values and the limited spatial extent of the exceedances; these low risks were estimated using conservative assumptions (e.g., based on Eco-SSLs or extrapolated TRVs that are lower than literature-based dietary TRVs, 100% ingestion of the most contaminated prey [worms]). Risk conclusions regarding exposure of birds to benzo(a)pyrene could not be reached because of the high degree of uncertainty in the selected dietary dose TRVs, which are based on studies involving weekly intraperitoneal injection.

Wildlife COPCs, HQs, uncertainties associated with exposure and effects, and risk conclusions are summarized in Table 8-38.Results of the wildlife analysis are integrated with those of all other ecological receptor groups to arrive the overall ecological risk conclusions in Section 11.0.

Table 8-38. Summary of Wildlife COPCs with HQ ≥ 1.0 and Risk Conclusions Across LOEs

COPC by Receptor	Max HQ by Line of Evidence[a] Tissue Residue[b]	Dietary Dose	Conclusion	Rationale for Risk Conclusion
Mammals				
Mink (RM-specific, unless otherwise noted)				
Aluminum	NE	1.6	Negligible risk	Max HQ is not indicative of ecologically significant risk. All HQs are low (1.3 to 1.6) and risk is likely overestimated. Exceedances limited to sediment samples in all exposure areas, no prey concentrations exceed the effects threshold. Selected LOAEL highly uncertain because only one dietary LOAEL identified, with LOAEL based on exposure of mice to an ionic form of aluminum not present in the environment.
Lead	NE	4.0	Negligible risk	Max HQ not indicative of ecologically significant risk. Risk of limited spatial extent: HQ ≥ 1 for only one smallmouth bass sample (RM 9.5 to RM 10.5). Lead in this sample likely an outlier. Selected LOAEL derived from drinking water exposure and an order of magnitude lower than the only literature-based dietary LOAEL.
Total PCBs	NE	33	Significant risk	Magnitude and spatial extent of HQ ≥ 1 indicate potential reduction in reproductive success of a Study Area population. HQ ≥ 1 (19 to 35) in all exposure areas. Given less conservative exposure estimates, total PCBs HQs would remain ≥ 1. Unknown effect of associated reduction in fecundity on viability of a Study Area mink population.
Total TEQ[c]	NE	12	Significant risk	Magnitude and spatial extent of HQ ≥ 1 indicate potential reduction in reproductive success of a Study Area population. Uncertainty in mammal TEFs may result in over- or underestimate of exposure (within an order of magnitude). HQ ≥ 1 (1.6 to 12) in all exposure areas. Total TEQ risks primarily due to PCB TEQ rather than the total dioxin/furan TEQ component; risk largely redundant with that from total PCBs. Total dioxin/furan TEQ risk of limited spatial extent: HQ ≥ 1 in only 1 of 15 crayfish, 1 of 21 sculpin, and 2 of 32 smallmouth bass samples, with all HQs ≥ 1 in samples from RM 6.5 to RM 7.5.
River Otter (all exposure areas)				
Total PCBs	NE	31	Significant risk	Magnitude and spatial extent of HQ ≥ 1 indicate potential reduction in reproductive success of a Study Area population. HQ ≥ 1 (21 to 31) in all exposure areas. Given less conservative exposure estimates, total PCBs HQs would remain ≥ 1. River otter-specific effects data not available and selected TRV for mink—consistently the most sensitive species tested—may overestimate risk.

Table 8-38. Summary of Wildlife COPCs with HQ ≥ 1.0 and Risk Conclusions Across LOEs

COPC by Receptor	Max HQ by Line of Evidence[a] Tissue Residue[b]	Dietary Dose	Conclusion	Rationale for Risk Conclusion
Total TEQ[c]	NE	2.3	Low risk	Magnitude and spatial extent of HQ ≥ 1 indicate limited potential for reduction in reproductive success of a Study Area population. HQ ≥ 1 (1.1 to 2.3) in all exposure areas. Uncertainty in mammal TEFs may result in over- or underestimate of exposure (within an order of magnitude). Total TEQ risks primarily due to PCB TEQ rather than the total dioxin/furan TEQ component; risk largely redundant with that from total PCBs. Total dioxin/furan TEQ HQs < 1 in all exposure areas. River otter-specific effects data not available and selected TRV for mink—the most sensitive species identified—may overestimate risk.
Birds				
Spotted Sandpiper (RM-specific, unless otherwise noted)				
Copper	NE	1.3	Negligible risk	Max HQ not indicative of ecologically significant risk. Risk likely overestimated: HQ ≥ 1 based on clam-only diet; worm-only diet max HQ < 1. Exposure via worms and clams may over- or underestimate exposure via amphipods and terrestrial invertebrates. The selected Eco-SSL-based LOAEL may overestimate risk, I lower than all bounded NOAELs and LOAELs reported in the literature.
Benzo(a)pyrene	NE	1.6	Negligible risk	Max HQ not indicative of ecologically significant risk. Risk of limited spatial extent: HQ > 1 in only 1 of 27 lab worm samples; clam-only diet max HQ < 1. Exposure via worms and clams may over- or underestimate exposure via amphipods and terrestrial invertebrates. Use of BSAR to predict clam concentrations when no data available (i.e., beaches B1, B4, B7, B8, and B14) may over- or underpredict exposure. Selected literature-based LOAEL TRV highly uncertain: only one study identified, with TRV based on injection exposure.
Dibutyl Phthalate	NE	1.4	Negligible risk	Max HQ not indicative of ecologically significant risk. Risk of limited spatial extent: HQ > 1 only 1 of 28 clam samples; worm-only diet max HQ < 1. Exposure via worms and clams may over- or underestimate exposure via amphipods and terrestrial invertebrates. Selected LOAEL TRV likely overestimates risk: extrapolated from BEHP NOAEL using a UF of 10. Dibutyl phthalate concentration in background sediments higher than the average in Study Area (Attachment 11).

Table 8-38. Summary of Wildlife COPCs with HQ ≥ 1.0 and Risk Conclusions Across LOEs

COPC by Receptor	Max HQ by Line of Evidence[a] Tissue Residue[b]	Max HQ by Line of Evidence[a] Dietary Dose	Conclusion	Rationale for Risk Conclusion
Total PCBs	NE	12	Significant risk	Magnitude and spatial extent of HQ ≥ 1 indicate potential reduction in reproductive success of Study Area spotted sandpiper population. HQ ≥ 1 (1.7 to 12) in all exposure areas assuming worm only diet; clam-only diet HQ ≥ 1 only in RM 7.0 to RM 9.0 (HQ = 2.2). Exposure via worms and clams may over- or underestimate exposure via amphipods and terrestrial invertebrates. Use of mechanistic model to estimate clam and worm concentrations for some beaches and use of adjusted steady-state lab worm concentrations may over- or underestimate prey concentrations. Receptor-specific effects data not available, selected TRV for reproduction of chicken—consistently the most sensitive species tested—may overestimate risk.
Total TEQ[c]	NE	20	Low risk	Magnitude and spatial extent of HQ ≥ 1 indicate limited but uncertain potential for reduction in reproductive success of Study Area spotted sandpiper population. HQ ≥ 1 (10 at RM 1.9 to RM 3.9, and 11 at RM 7.0 to RM 9.0) in two of four exposure areas assuming worm only diet; clam-only diet HQs < 1. Uncertainty in bird TEFs may result in over- or underestimate of exposure (by more than an order of magnitude). Exposure via worms and clams may over- or underestimate exposure via amphipods and terrestrial invertebrates. Adjusted steady-state lab worm concentrations may over- or underestimate prey concentrations. Total TEQ risks primarily due to PCB TEQ rather than the total dioxin/furan TEQ component, risk largely redundant with that from total PCBs. Total dioxin/furan TEQ risk of limited spatial extent: HQ ≥ 1 in only 1of 27 clam and 2 of 27 worm samples, both from RM 7.0 to RM 9.0. Selected LOAEL highly uncertain: based on the lower of two identified literature-reported LOAELs. Study based on injection of ring-necked pheasants, an unrealistic exposure mechanism.
Sum DDE	NE	1.3	Negligible risk	Max HQ not indicative of ecologically significant risk. Risk of limited spatial extent and likely overestimated: HQ > 1 in only one of four exposure areas (RM 7.0 to RM 9.0) based on worm only diet; clam-only or mixed diet max HQ < 1. Worm or clam diet may over- or underestimate risk from amphipod and terrestrial invertebrate diet. Use of mechanistic model to estimate clam and worm tissue concentrations for some beaches may over- or underestimate exposure. Selected Eco-SSL-based LOAEL consistent with the lowest literature-based LOAEL where mallard eggshell thinning of about 6% was statistically different from control. However, reproductive effects in field populations of birds not documented for eggshell thinning of < 15 to 20%.
Total DDx	NE	1.4	Negligible risk	Rationale, exposure and effects uncertainties mirror those for sum DDE.

Table 8-38. Summary of Wildlife COPCs with HQ ≥ 1.0 and Risk Conclusions Across LOEs

COPC by Receptor	Max HQ by Line of Evidence[a]		Conclusion	Rationale for Risk Conclusion
	Tissue Residue[b]	Dietary Dose		
Aldrin	NE	1.7	Negligible risk	Max HQ not indicative of ecologically significant risk. Risk of limited spatial extent. HQ > 1 only in only 1 of 27 lab worm samples (from RM 7.0 to RM 9.0); HQ < 1 for all other clam and worm samples. Exposure via worms and clams may over- or underestimate exposure via amphipods and terrestrial invertebrates. Use of mechanistic model to estimate clam and worm tissue concentrations for some beaches may over- or underestimate exposure. Adjusted steady-state lab worm concentrations may over- or underestimate prey concentrations. Selected TRV highly uncertain: only two toxicological studies available.
Osprey (RM-specific)				
Lead	NE	7.8	Negligible risk	Max HQ not indicative of ecologically significant risk. Risk of limited spatial extent: HQ ≥ 1 for only one smallmouth bass sample (RM 9.5 to 10.5). Lead in this sample likely an outlier. Selected LOAEL (3.26 mg/kg bw/day) based on an Eco-SSL (EPA 2005f) and an order of magnitude lower than lowest acceptable literature-based LOAEL (20 mg/kg bw/day) (Attachment 14). Data from Henny et al. (2009) indicate that osprey populations from LWR (including the Study Area) have increased in recent years, with productivity above that necessary for maintaining a stable population.
Total PCBs	4.4 (egg)	1.1	Low risk	Results of tissue-residue and dietary LOEs somewhat contradictory. Magnitude and spatial extent of tissue-residue HQs ≥ 1 indicate limited but uncertain potential for reduction in reproductive success of Study Area osprey population. Tissue-residue risk of limited spatial extent: HQ ≥ 1 in only 2 of 5 exposure areas (RM 5.5 to RM 6.5 and RM 8.5 to RM 9.5). Max dietary HQs not indicative of ecologically significant risk. Dietary risk of limited spatial extent: HQ≥ 1 in only 2 of 15 carp samples and 3 of 32 smallmouth bass samples with the mixed diet HQ only slightly > 1 for RM 10.5 to RM 11.8. Max dietary HQ < 1 if no smallmouth bass or carp in osprey diet, a plausible assumption. Dietary-dose max HQ may over- or underestimate Study Area exposure because osprey may forage in nearby water bodies. Osprey-specific dietary effects data not available, selected TRV for reproduction of chicken - consistently the most sensitive species tested - may overestimate risk. Data from Henny et al. (2009) indicate that osprey populations from LWR (including the Study Area) have increased in recent years, with productivity above that necessary for maintaining a stable population.

Table 8-38. Summary of Wildlife COPCs with HQ ≥ 1.0 and Risk Conclusions Across LOEs

COPC by Receptor	Max HQ by Line of Evidence[a]		Conclusion	Rationale for Risk Conclusion
	Tissue Residue[b]	Dietary Dose		
Total TEQ[c]	3.8 (egg)	Step 1 HQ <1	Low risk	Results of tissue-residue and dietary LOEs somewhat contradictory. Magnitude and spatial extent of tissue-residue HQs ≥ 1 indicate limited but uncertain potential for reduction in reproductive success of the Study Area population. Uncertainty in bird TEFs may result in over- or underestimate of exposure (by more than an order of magnitude). Tissue-residue HQ ≥ 1 (2.1 to 3.8) in all exposure areas for which data available. Selected tissue-residue LOAEL based on bald eagle-specific effects data; the only osprey-specific LOAEL is 4 times as high as selected LOAEL (all HQs < 1 based on the osprey-specific LOAEL). These are field-based LOAELs, which may overestimate risk due to effects from co-contaminants. The bald eagle LOAEL between the osprey NOAEL and LOAEL so is within the bounds of uncertainty in osprey-specific effects threshold. Dietary HQ < 1 in all exposure areas. Selected LOAEL highly uncertain: based on lower of two identified literature-reported LOAELs. Study based on injection of ring-necked pheasants, an unrealistic exposure mechanism.
Bald Eagle (RM-specific, unless otherwise noted)				
Mercury	HQ <1 (egg)	1.7	Negligible risk	LOEs in reasonable agreement. Max dietary HQ not indicative of ecologically significant risk. HQ ≥ 1 (1.2 to 1.7) in all exposure areas. Dietary exposure likely overestimated: diet assumed to be fish only; HQ < 1 possible if diet includes terrestrial prey. Selected dietary NOAEL uncertain. Bald eagle-specific effects data not available. Selected NOAEL extrapolated from LOAEL using a UF of 10 and may over- or underestimate risk. All tissue-residue HQs < 1, with less uncertainty than dietary LOE. Tissue-residue exposure estimated based on osprey data, which may over- or underestimate exposure to bald eagle. Data not available for all exposure areas. The selected tissue-residue NOAEL based on bald eagle field studies. No bald eagle LOAEL identified, so effects threshold for bald eagle unknown; however, the NOAEL unlikely to contribute to underestimate of risk.

Table 8-38. Summary of Wildlife COPCs with HQ ≥ 1.0 and Risk Conclusions Across LOEs

COPC by Receptor	Max HQ by Line of Evidence[a]		Conclusion	Rationale for Risk Conclusion
	Tissue Residue[b]	Dietary Dose		
Total PCBs	6.6 (egg)	3.9	Low risk	Dietary and egg LOEs indicate potential for reduction in reproductive success of bald eagles in the Study Area. Egg HQs ≥ 1 (1.2 to 6.6) in 4 of 5 exposure areas. Egg exposure uncertainties mirror those discussed above for mercury. Selected egg NOAEL based on bald eagle-specific field studies and may overestimate but is not likely to underestimate risk; egg concentrations are below Elliott and Harris (2001\2002) recommended eagle-specific effects thresholds. Dietary HQ ≥ 1 (3.8 to 3.9) in all exposure areas. Exclusion of terrestrial prey likely overestimates Study Area-related exposure but not likely to result in HQ < 1. Receptor-specific effects data not available; selected TRV for reproduction of chicken - consistently the most sensitive species tested - may overestimate risk.
Total TEQ[c]	53 (egg)	Step 2 HQ <1	Low risk	Results of tissue-residue and dietary LOEs somewhat contradictory. Uncertainty in bird TEFs may result in over- or underestimate of exposure (by more than an order of magnitude). Magnitude and spatial extent of egg HQ ≥ 1 indicate potential for reduction in reproductive success of Study Area bald eagle. Egg HQ ≥ 1 (30 to 53) in all exposure areas for which data available. Egg exposure uncertainties mirror those discussed above for mercury. TEQ NOAEL based on osprey data, as no bald eagle-specific NOAEL identified; egg concentrations are below Elliott and Harris (2001\2002) recommended eagle-specific effects thresholds. Dietary HQ < 1 in all exposure areas. Dietary exposure uncertainties mirror those discussed above for mercury. Selected dietary LOAEL highly uncertain: effects uncertainty mirrors that discussed above for spotted sandpiper – total PCBs

Table 8-38. Summary of Wildlife COPCs with HQ ≥ 1.0 and Risk Conclusions Across LOEs

COPC by Receptor	Max HQ by Line of Evidence[a] Tissue Residue[b]	Dietary Dose	Conclusion	Rationale for Risk Conclusion
Sum DDE	1.9 (egg)[d]	Step 3 HQ <1	Low to negligible risk	Results of tissue-residue and dietary LOEs in reasonable agreement. Egg HQ ≥ 1 (1.1 at RM 2.5 to RM 3.5 and 1.9 at RM 5.5 to RM 6.5) in 2 of 5 exposure areas. Uncertainties in egg exposure data mirror those discussed above for eagle - mercury. Selected egg NOAEL based on bald eagle-specific field studies and, as NOAEL, may overestimate risk. Exposure concentrations do not exceed Elliott and Harris (2001\2002) recommended eagle-specific effects thresholds (max HQ = 0.68). Given uncertainties, reasonable to conclude that individual bald eagles face low to negligible risk of reduced reproductive success. Dietary LOE indicates negligible risk. Based on multi-species diet, dietary HQ < 1 in all exposure areas. HQ ≥ 1 for only 1 of 15 carp and 2 of 6 northern pikeminnow samples. Bald eagle-specific dietary effects data not available. NOAEL extrapolated from a LOAEL using factor of 10; literature-based NOAELs suggest extrapolation may overestimate risk to birds. Availability of bald eagle-specific effects data for the egg LOE results in a higher confidence in the egg LOE relative to the dietary LOE.
Hooded Merganser (RM-specific)				
Total PCBs	NE	3.8	Negligible risk	Max HQ not indicative of ecologically significant risk. Risk of limited spatial extent: HQ ≥ 1 for only 1 of 41 clam, 4 of 38 sculpin, and 4 of 32 smallmouth bass samples. Based on multi-species diet, HQ ≥ 1 (1.5 to 3.8) only in 3 of 12 exposure areas (RM 1.5 to RM 2.5; RM 6.5 to RM 7.5, and RM 10.5 to RM 11.8). Exposure may be overestimated because smallmouth bass used to model exposure larger than fish merganser eat and total PCB concentrations increase with fish size. Receptor-specific effects data not available. Selected TRV for reproduction of chicken - consistently the most sensitive species tested - likely overestimates risk. LOAEL for mallards is approximately 30 times as great as selected LOAEL.

Note: This table attempts to summarize the BERA's wildlife risk estimates and risk descriptions, the two major components of the risk characterization. Balancing and interpreting the different types of data evaluated in the BERA can be a major task requiring professional judgment. It can be difficult to prepare a concise summary of conclusions without losing important context, yet a concise summary is needed to help the risk manager judge the likelihood and ecological significance of the estimated risks (EPA 1997).

All the COPCs listed in this table have an HQ ≥ 1 in at least one LOE for at least one ecological receptor, and by definition pose potentially unacceptable risk. The likelihood and ecological significance of the potentially unacceptable risk may vary, though, from very low to very high. Therefore, the risk description may range from negligible to significant. For each receptor-COPC pair with a maximum HQ ≥ 1,this table provides maximum HQ by LOE, a synoptic risk description, and a very brief rationale for the risk description. This distillation of the body of knowledge presented in the BERA should not be taken out of context.

[a] Max HQ for each receptor in final step of risk characterization over all exposure areas (river miles).

[b] A tissue-residue approach was used only for osprey and bald eagle for the COPCs total PCBs, PCB TEQ, total dioxin/furan TEQ, total TEQ, and 4,4'-DDE.

[c] TEQ was also assessed as PCB TEQ and total dioxin/furan TEQ. Risks from all TEQ analyses are summarized with total TEQ risks.

[d] HQ is based on 4,4'-DDE.

BEHP – bis(2-ethylhexyl) phthalate
BERA – baseline ecological risk assessment
BSAR – biota-sediment accumulation regression
bw – body weight
COPC – contaminant of potential concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
Eco-SSL – ecological soil screening level
EPA – US Environmental Protection Agency
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
LOE – line of evidence
LWR – Lower Willamette River
NE – not evaluated
NOAEL – no-observed adverse effect level
PCB – polychlorinated biphenyl
RM – river mile
TEF – toxicity equivalency factor
TEQ – toxic equivalent
total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT, and 4,4′-DDT)
TRV – toxicity reference value
UF – uncertainty factor

8.3.3.2 Mink and PCB Evaluation

The primary potential risk to wildlife in the Study Area is associated with a single COPC—total PCBs. The receptor with the greatest potential risk is the mink population. The spatial extent and magnitude of risk to river otter from PCBs is similar to risk to mink, however, because no otter-specific toxicity data are available, risk to otter has greater associated uncertainty than that for mik. A detailed evaluation of the PCB assessment for mink is presented in this section, including an uncertainty evaluation and discussion of the exposure and effects assumptions. Although the Study Area has not been reported to support active populations of mink and river otter, it offers at least marginally suitable habitat, and both species have been collected nearby on Multnomah Channel and the Columbia River (including RM 90, RM 108, and RM 119.5 near Portland[137]) (Elliott et al. 1999; Henny et al. 1996).

8.3.3.2.1 Evaluation of Exposure Assumptions

The following subsection examines the exposure assumptions used in the PCB-mink evaluation. Results of site-specific exposure modeling combined with literature-reported effects data indicate a possibility of risk to the mink population in the Study Area.

Risk to mink from PCBs was estimated by comparing exposure, as characterized by a conservative estimate of the average daily chemical dose, with the daily chemical dose reported in the non site-specific toxicological literature to be associated with adverse effects on mink.[138] PCB exposure was estimated by applying model output of daily feeding rates for mink to concentrations of PCBs and dioxins and furans in fish and crayfish from 1-mile exposure areas within the Study Area. As opportunistic feeders, mink consume a range of prey, such as muskrats, fish, frogs, crayfish, small mammals, and birds found near water (Csuti et al. 2001). In a study of mink diets from the Columbia River upstream from the Willamette River,[139] it was reported that birds, mammals, and fish were consumed at similar frequencies; crayfish were consumed at a higher frequency (WDG 1980).[140] For this BERA, it was assumed that the mink diet

[137] The mouth of the LWR is located at RM 101 of the Columbia River.

[138] In Section 8.1, the dietary risk equation was mathematically rearranged so that LOAELs were expressed as threshold tissue and sediment chemical concentrations that would result in an HQ of 1.0. The threshold approach facilitates comparison of a single effects threshold to a range of exposure concentrations, whereas the dietary-dose approach used for calculations in this section facilitates comparison of a given exposure concentration to potential effects across a range of doses. Resulting HQs are the same, regardless of the approach.

[139] Data were collected between RM 107 and RM 546 of the Columbia River including two mink specimens collected between RM 107 and RM 146. The mouth of the Willamette River is located at RM 101 of the Columbia River.

[140] Data were based on the percent frequency of occurence in mink scats and do not necessarily reflect the relative biomass of prey; however, because biomass data are not available, percent frequency was assumed to correspond to the relative importance of each prey type.

consists solely of fish and crayfish from the Study Area, although the Washington Department of Game (WDG) (1980) reported that remains of fish were found in only 32% of mink feces and that remains of crayfish were found in only 47% of mink feces.[141] The literature also notes that mink diets vary seasonally, for example shifting towards fish in the winter and birds in the summer. The default dietary assumption, based on the relative abundance of fish and crayfish in the Study Area, is that the mink diet consists of equal portions by weight of five prey species: crayfish, largescale sucker, carp, sculpin, and smallmouth bass (i.e., each component is 20% of the diet).[142] Excluding terrestrial prey from the assumed diet imparts a substantial over-estimation bias to the exposure analysis.

Table 8-39 presents mink exposure to PCBs in the diet (expressed as a dose), the selected dose-based LOAEL, and resulting HQs. Calculated doses for all exposure areas are greater than the selected LOAEL.

Table 8-39. Mink PCB Exposure Doses and HQs for All Exposure Areas

Exposure Area	Exposure Dose (mg/kg bw/day)[a]	LOAEL (mg/kg bw/day)	HQ
RM 1.5 – RM 2.5	0.85	0.037	**23**
RM 2.5 – RM 3.5	0.70	0.037	**19**
RM 3.5 – RM 4.5	0.74	0.037	**20**
RM 4.5 – RM 5.5	0.70	0.037	**19**
RM 5.5 – RM 6.5	0.70	0.037	**19**
RM 6.5 – RM 7.5	0.81	0.037	**22**
RM 7.5 – RM 8.5	0.70	0.037	**19**
RM 8.5 – RM 9.5	0.74	0.037	**20**
Swan Island Lagoon	0.85	0.037	**23**
RM 9.5 – RM 10.5	0.74	0.037	**20**
RM 10.5 – RM 11.8	1.22	0.037	**33**

[a] Exposure dose estimate assuming 20% ingestion of each of the following prey: carp, sculpin, largescale sucker, smallmouth bass, and crayfish.

bw – body weight
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
PCB – polychlorinated biphenyl
RM – river mile

[141] Otter also consume non-aquatic prey, though to a lesser extent, with birds, mammals, and prey other than fish and crayfish constituting up to 15% or more of their diets (USEPA 1993).

[142] Other parameters used to model mink exposure included body weight, food ingestion rate, and sediment ingestion rate. These parameters were based on literature-reported values as described in Attachment 16.

To evaluate the potential for risk to mink in the upriver reach (RM 15.3 to RM 28.4), a mink exposure dose and HQ were calculated from tissue data available for this area. Tissue data from the upriver reach were available for juvenile Chinook salmon, smallmouth bass, brown bullhead, and Pacific lamprey ammocoetes. The mink diet was assumed to consist of equal portions by weight of juvenile Chinook salmon, smallmouth bass, and brown bullhead.[143] The resulting exposure dose is 0.021 mg/kg bw/day, with an HQ of 0.57 at a LOAEL of 0.037 mg/kg bw/day. Thus, negligible risk to mink is expected on the basis of the limited data from the upriver reach; however, tissue data are not available for most of the predominant mink prey species (i.e., carp, largescale sucker, crayfish, and sculpin). Depending on the relative concentrations in tissue of upriver prey and Study Area prey, upstream risk estimates may be over- or underestimated.

To test the sensitivity of risk estimates to mink dietary composition, PCB doses were recalculated under various assumptions. Specifically, PCB doses were calculated by assuming that mink are strictly monophagous, eating only a single prey species. Each of 10 potential prey species (i.e., crayfish and the nine fish species collected from the Study Area, each of which represents potential mink prey) served as the sole food item in this analysis. Results are shown for each 1-mile exposure area in Table 8-40 (mink dietary PCB doses) and Table 8-41 (resulting HQs). When assumed to be the sole element of the mink diet, each prey species except for juvenile Chinook salmon resulted in a dietary dose greater than the dose-based LOAEL in at least one 1-mile exposure area.

The dietary doses (Table 8-40) and HQs (Table 8-41) do not account for exposure to PCBs via incidental ingestion of sediment. From the foraging habits of raccoons (Beyer et al. 1994), incidental ingestion of sediment in mink was assumed to be about 9% of dietary intake on a mass basis. Because sediment represents a trivial fraction of overall exposure (i.e., sediment contributes < 1% of the exposure dose and HQ), the exposure doses and HQs are expected to be approximately the same whether incidental ingestion of sediment is included or excluded. Reducing the sediment ingestion rate would not affect risk estimates.

[143] Smallmouth bass, brown bullhead, and juvenile Chinook salmon EPCs were estimated as the maximum concentration for each species because limited species-specific data were available (n < 6).

Table 8-40. Mink Dietary Doses for Total PCBs Based on Consumption of a Single Prey Item

	Calculated Dietary Dose (mg/kg bw/day)									
Exposure Area	**Black Crappie**	**Brown Bullhead**	**Carp**	**Juvenile Chinook Salmon**	**Crayfish**	**Largescale Sucker**	**Northern Pikeminnow**	**Peamouth**	**Sculpin**	**Smallmouth Bass**
RM 1.5 to RM 2.5	ND	ND	NA	NA	0.01	NA	ND	NA	*0.56*	*0.23*
RM 2.5 to RM 3.5	*0.02*	*0.02*	NA	NA	0.001[a]	NA	*0.12*	NA	*0.03*	*0.13*
RM 3.5 to RM 4.5	*0.02*	*0.02*	NA	NA	0.05	NA	ND	NA	0.06	*0.24*
RM 4.5 to RM 5.5	*0.02*	*0.02*	NA	NA	*0.0004*[a]	NA	*0.07*	NA	*0.03*	*0.06*
RM 5.5 to RM 6.5	*0.02*	*0.02*	NA	NA	*0.01*	NA	*0.07*	NA	*0.03*	*0.11*
RM 6.5 to RM 7.5	*0.04*	*0.28*	NA	NA	*0.01*	NA	*0.31*	NA	*0.41*	*0.33*
RM 7.5 to RM 8.5	*0.04*	*0.28*	NA	NA	*0.01*	NA	*0.17*	NA	*0.04*	*0.15*
RM 8.5 to RM 9.5	*0.04*	*0.28*	NA	NA	*0.01*	NA	*0.17*	NA	*0.06*	*0.16*
Swan Island Lagoon	*0.04*	*0.28*	NA	NA	*0.01*	NA	*0.13*	NA	*0.10*	*0.81*
RM 9.5 to RM 10.5	ND	ND	NA	NA	*0.02*	NA	ND	NA	*0.13*	*0.13*
RM 10.5 to RM 11.8	ND	ND	NA	NA	*0.20*	NA	ND	NA	*1.4*	*1.07*
Site-wide[b]	NA	NA	3.1	0.032	NA	0.25	NA	*0.048*	NA	NA

[a] Dietary dose calculated from maximum concentration that is based on one-half DL (where one-half DL > maximum detected concentration or where COPC is not detected).
[b] Dietary dose calculated for large-home-ranging fish based on site-wide UCL concentrations.

bw – body weight
COPC – contaminant of potential concern
DL – detection limit
NA – not applicable
ND – no data available
PCB – polychlorinated biphenyl
RM – river mile
UCL – upper confidence limit on the mean

Italics identify dietary doses derived from maximum concentration; UCL concentration could not be derived (n detects < 6).

Table 8-41. Mink PCB HQs Based on Consumption of a Single Prey Item

	Prey Item (Minimum – Maximum Percent Occurrence)[a]									
Exposure Area	**Black Crappie (0 – 5%)**	**Brown Bullhead (0 – 5%)**	**Carp (5 – 20%)**	**Juvenile Chinook Salmon (0 – 1%)**	**Crayfish (0 – 47%[b])**	**Largescale Sucker (35 – 50%)**	**Northern Pikeminnow (5 – 25%)**	**Peamouth (0 – 30%)**	**Sculpin (5 – 15%)**	**Smallmouth Bass (0 – 35%)**
RM 1.5 to RM 2.5	ND	ND	NA	NA	*0.14*	NA	ND	NA	***15***	***6.3***
RM 2.5 to RM 3.5	*0.41*	*0.58*	NA	NA	*0.019*[c]	NA	***3.2***	NA	*0.76*	***3.5***
RM 3.5 to RM 4.5	*0.41*	*0.58*	NA	NA	**1.3**	NA	ND	NA	**1.7**	***6.5***
RM 4.5 to RM 5.5	*0.41*	*0.58*	NA	NA	*0.011*[c]	NA	***2.0***	NA	*0.89*	***1.7***
RM 5.5 to RM 6.5	*0.41*	*0.58*	NA	NA	*0.2*	NA	***2.0***	NA	*0.71*	***3.0***
RM 6.5 to RM 7.5	***1.1***	***7.6***	NA	NA	*0.25*	NA	***8.5***	NA	***11***	***9.0***
RM 7.5 to RM 8.5	***1.1***	***7.6***	NA	NA	*0.21*	NA	***4.5***	NA	*1.0*	***4.0***
RM 8.5 to RM 9.5	***1.1***	***7.6***	NA	NA	*0.3*	NA	***4.5***	NA	***1.6***	***4.3***
Swan Island Lagoon	***1.1***	***7.6***	NA	NA	*0.23*	NA	***3.5***	NA	***2.8***	***22***
RM 9.5 to RM 10.5	ND	ND	NA	NA	*0.49*	NA	ND	NA	***3.6***	***3.6***
RM 10.5 to RM 11.8	ND	ND	NA	NA	***5.3***	NA	ND	NA	***39***	***29***
Site-wide[d]	NA	NA	**85**	0.87	NA	**6.7**	NA	***1.3***	NA	**NA**

[a] Based on the relative abundance of fish caught in three fish community studies in the LWR (see Section 2.5.6 of Attachment 16).
[b] Crayfish range was estimated from data on the percent frequency of crayfish occurrence in mink scats from the Columbia River (see Section 2.5.6 of Attachment 16).
[c] HQs based on maximum concentration that is based on one-half DL (where one-half DL > maximum detected concentration or where COPC is not detected).
[d] HQs for large-home-ranging fish based on site-wide EPCs.

COPC – contaminant of potential concern
DL – detection limit
EPC – exposure point concentration
HQ – hazard quotient
LWR – Lower Willamette River
NA – not applicable
ND – no data available
PCB – polychlorinated biphenyl
RM – river mile

Italics identify HQ derived from maximum concentrations; UCL concentration could not be derived (n detects < 6).

Bold identifies HQs ≥ 1.

Dietary doses and HQs associated with any prey portion assumption can be determined from the information in Tables 8-40 and 8-41 by calculating the weighted sum of dietary fractions and doses (or HQs) across all prey (i.e., the dietary dose is simply the dose multiplied by the dietary fraction for each prey item summed over all prey items). Weighting by the default assumption of 20% each of carp, crayfish, largescale sucker, sculpin, and smallmouth bass results in the HQs presented in Table 8-39, again showing that HQs ≥ 1 are predicted for all 1-mile exposure areas.

Because mink are opportunistic hunters, eating more frequently the prey that are more abundant (Melquist et al. 1981; Racey and Euler 1983; Ward et al. 1986; Wise et al. 1981), the available fish community studies of the LWR were examined to determine the relative abundance of prey fish in the Study Area (Attachment 16). For each possible prey species, the minimum and maximum percent occurrence was used in the dietary sensitivity analysis presented in Section 8.1.5.2.1. The results of that senstivity analysis show that HQs for all exposure areas are always ≥ 1 regardless of the prey assumptions; however, the 95th percentile HQs, and usually the maximum HQs, were less than the HQs based on default assumptions (Figure 8-16).

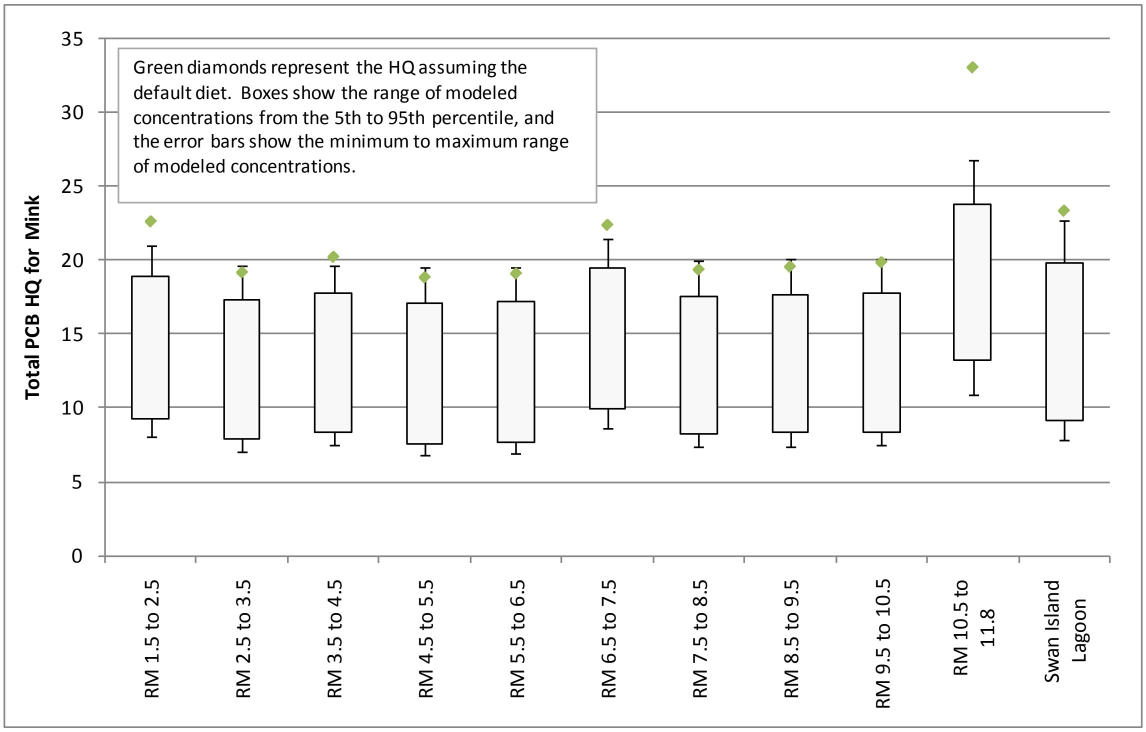


Figure 8-16. Range of Total PCB Mink HQs Based on Abundance of Mink Prey Species

Mink diets from the Columbia River and the relative abundance of fish species in the LWR (presented in Attachment 16) both suggest that carp and largescale sucker are likely to be important prey for mink in the Study Area. Because Study Area-wide tissue concentrations for these two species are high, even the smallest likely fractions in the mink diet (5% of diet for carp and 35% of diet for largescale sucker) would result in

HQs ≥ 1 throughout the Study Area (HQ in all exposure areas would be 6.6; 5% multiplied by the carp HQ of 85 plus 35% multiplied by the largescale sucker HQ of 6.7 = 6.6). In this scenario, carp and largescale sucker together constitute only 40% of the mink diet. Adding dietary contributions from other prey species further increases the PCB dose to mink (and HQ), regardless of how low the doses are. Thus, assuming that mink eat only crayfish (60% of diet) and fish (5% carp and 35% largescale sucker) and no birds or mammals, the minimum realistic dietary dose from the Study Area is about 0.20 mg/kg bw/day, which results in an HQ of 6.6.

A SUF of 1 was assumed for mink in the calculatation of HQs; however, the assumption that mink acquire all of their food from the Study Area is probably unrealistic. As discussed above, data from the Columbia River indicate that more than half of the mink diet consists of birds and mammals (WDG 1980). To the extent that the bird and mammal food items are not entirely dependent on the Study Area, consumption of such birds and mammals would reduce Study Area-related risk to mink. If it is assumed that mink consume only one-third of their diet from the Study Area and two-thirds of the diet from birds and mammals with no Study Area-related PCBs, the minimum realistic dietary dose would be one-third as high (0.067 mg/kg bw/day) and result in an HQ of 2.2.

Finally, mink exposure is a function of FIRs and body weights. HQs increase with FIR and decrease with body weight; however, because FIR is a function of body weight, risk estimates are not affected by changes in body weight. No FIRs other than 16% body weight per day were identified for female mink; FIRs for male mink ranged from 12 to 22% body weight per day (EPA 1993). These FIRs bracket those assumed for female mink, so different FIRs could result in higher or lower risk.

From this analysis, it is apparent that all reasonable mink prey portion exposure scenarios and all prey portion scenarios based on relative percent occurrence for the Study Area result in dietary doses that exceed the selected LOAEL. In addition, decreasing the selected incidental sediment ingestion rate or SUF still results in PCB LOAEL exceedances.

8.3.3.2.2 Evaluation of Selected LOAEL TRV

The following subsection evaluates the TRV study that was used to derive the LOAEL by which risk to mink from PCBs was assessed. The selected LOAEL of 0.037 mg/kg bw/day is the lowest adverse effect concentration reported in any of the 12 studies identified from the toxicological literature showing effects on mink following dietary exposure to PCBs (Attachment 14). In this study, Restum et al. (1998) showed that when mink were fed field-collected carp containing PCBs from the Great Lakes for multiple generations, mink birth rates (whelping) were reduced and the resulting offspring (kits) had lower body weights relative to mink fed uncontaminated diets. Adverse effects observed in this study may have been due, to some extent, to other contaminants present in the field-collected fish. However, similar results were also observed when mink were fed otherwise uncontaminated diets to which PCBs had been added in the laboratory (Brunström et al. 2001). Over all 12 studies reviewed, dietary doses associated with the LOAELs ranged from 0.037 mg/kg bw/day for reduced birth rate and kit body weight (Restum et al. 1998) to 2.6 mg/kg bw/day for reduced birth weight, reduced growth rate of kits, and reduced adult female survival (1986). Over all 12 studies, adverse effects on survival or growth of newborns through the first few weeks of life generally occurred at low PCB doses (0.037 to 2.6 mg/kg bw/day); adverse effects on adult mink were generally not observed until much higher dosages (0.32 to 2,000 mg/kg bw/day).

Contrary to other receptors and COPCs for which toxicological data specific to the receptor are generally lacking, literature-reported adverse effects data specific to mink and PCBs are fairly available. Adverse effects on the survival of newborn mink kits following dietary PCB exposure were reported in 10 studies (reported in 11 papers) (Tillitt et al. 1996; Heaton et al. 1995; Restum et al. 1998; Hornshaw et al. 1983; Jensen et al. 1977; Aulerich et al. 1985; Kihlstrom et al. 1992; Brunström et al. 2001; Aulerich and Ringer 1977; Bleavins et al. 1980; Wren et al. 1987). Five toxicological studies reported the relationship between PCB dietary doses administered and the magnitude of effects on kit growth and survival at 5 or 6 weeks after whelping (Heaton et al. 1995; Tillitt et al. 1996; Restum et al. 1998; Wren et al. 1987; Hornshaw et al. 1983). Adverse effects on kit growth or survival occurred at doses between the selected LOAEL (0.037 mg/kg bw/day) and about 0.7 mg/kg bw/day. At higher doses, 100% mortality of kits is consistently observed. The PCB dietary doses and associated magnitude of effects on kit growth and survival at 5 or 6 weeks after whelping reported in the toxicological literature (Wren et al. 1987; Restum et al. 1998; Tillitt et al. 1996; Heaton et al. 1995; Hornshaw et al. 1983) are presented in Figure 8-17. The dose-response line best fitting the body weight data indicates, for example, that a 50% reduction in kit body weight would occur at a dose of approximately 0.06 mg/kg bw/day. Calculated PCB doses for mink from the Study Area (Table 8-39; Figure 8-17) are higher than any of the doses associated with reductions in kit growth or survival, indicating that mink eating from the Study Area might not reproduce successfully and that the mink population might be adversely affected.

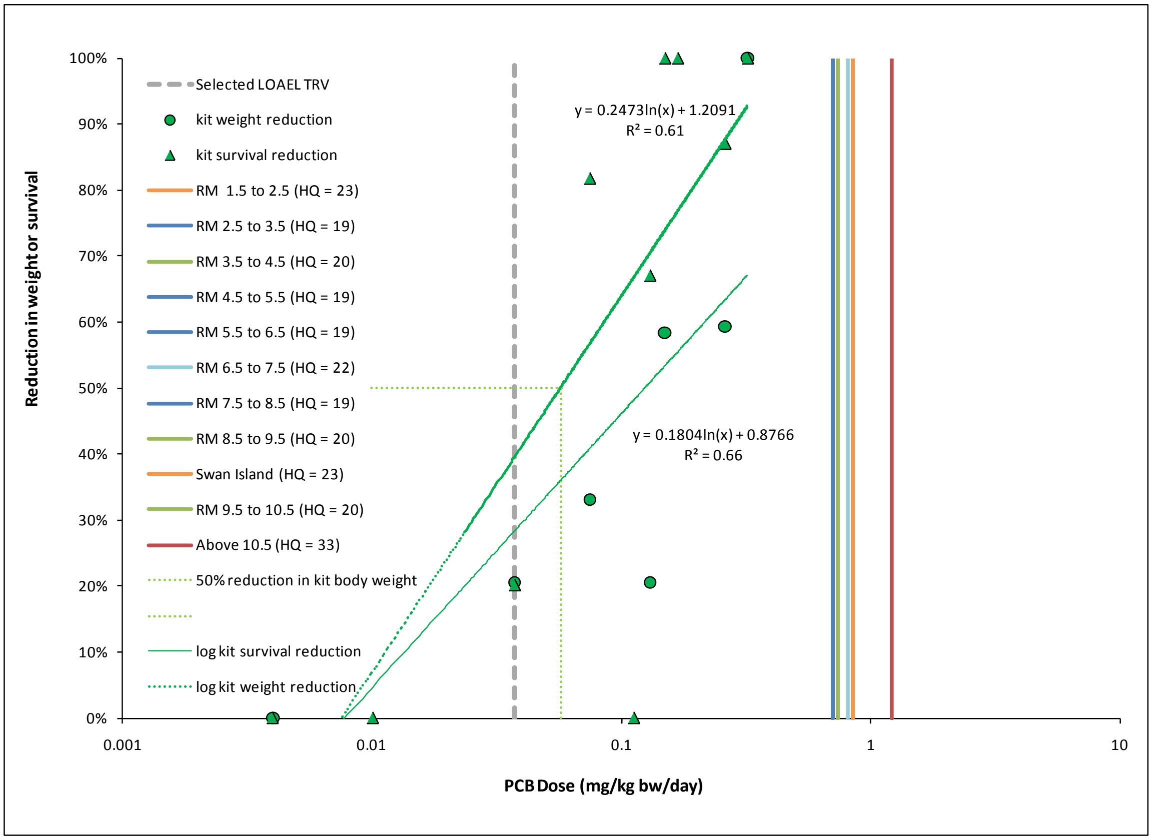


Figure 8-17. Toxicity of PCBs on Mink Kit Body Weight and Survival 5 or 6 Weeks After Whelping

8.3.3.2.3 Conclusions

Based on this analysis, it is likely that Study Area PCB concentrations are high enough to cause adverse effects on mink. The exceedance of the LOAEL TRV might or might not result in population-level risk to mink. The LOAEL is associated with adverse effects on mink reproduction. What is not clear, without conducting a population-level risk assessment, is whether the magnitude of the reproductive effect is sufficient to affect the overall health of a mink population in the Study Area.

This detailed evaluation of PCB risk to mink are considered in the overall ecological risk conclusions in Section 11.0.

9.0 AMPHIBIAN RISK ASSESSMENT

This section presents the draft BERA for amphibians in the Study Area. The toxicological thresholds available for the amphibian BERA are the same as those used for the SLERA and refined screening steps. The BERA differs from the SLERA and refined screen in that it incorporates more life history information into the exposure assessment and risk characterization, using the red-legged frog (*Rana aurora*) as a representative amphibian species. An adult northern red-legged frog (*Rana aurora aurora*) was the only amphibian species visually observed[144] during the 2002 Portland Harbor amphibian and aquatic plant reconnaissance survey (Integral et al. 2004a). The larval life stage is most likely to be affected; eggs hatch in March through April.

The ecological CSM (Figure 3-2) classifies ingestion of prey, direct contact with surface water, and direct contact with TZW as complete and significant exposure pathways for amphibians. The surface water assessment is presented in Section 9.1. The TZW evaluation is presented in Section 9.2. As required by EPA, TZW was screened against surface water screening values. TZW data are limited.

The complete and significant dietary exposure pathway for amphibians (Figure 3-2) is not evaluated in the BERA per EPA's Problem Formulation (Attachment 2). One reason is that dietary-based TRVs for quantitative assessment are not available. In addition, exposure to contaminated prey within the Study Area may be limited, given the red-legged frog's reliance on riparian habitat and terrestrial prey; adults of some species or subspecies of red-legged frog often live for months in dense riparian habitats, feeding on mice, other amphibians (most often tree frogs), and terrestrial insects (Hayes and Tennant 1985). Although not evaluated specifically, risk associated with the amphibian dietary exposure pathway is covered indirectly by the evaluation of other small-home-range receptors (e.g., sculpin), whose home range is much smaller than that of amphibians (up to 2 miles for frogs (USFWS 2002)).

9.1 SURFACE WATER ASSESSMENT

Surface water is the primary LOE for evaluating risk to amphibians. Surface water COPCs were identified in the SLERA and refined screen using water TRVs based on AWQC and other TRVs available in the literature (Attachment 5). Figure 9-1 describes the layout of the surface water assessment section.

- Section 9.1.1 presents the general approach.
- Section 9.1.2 lists the the COPCs evaluated.
- Section 9.1.3 describes how exposure concentrations were derived. All surface water chemical concentrations are presented in Attachment 4.

[144] The presence of the Pacific tree frog (*Hyla regilla*) was audibly confirmed based on frog calls during the 2002 amphibian and aquatic plant reconnaissance survey.

- Section 9.1.4 summarizes the effects data. Details on the development of the water TRVs are presented in Attachment 10.
- Section 9.1.5 presents the risk characterization results and associated uncertainties. These COPCs are further assessed in the amphibian risk conclusions section (Section 9.3).

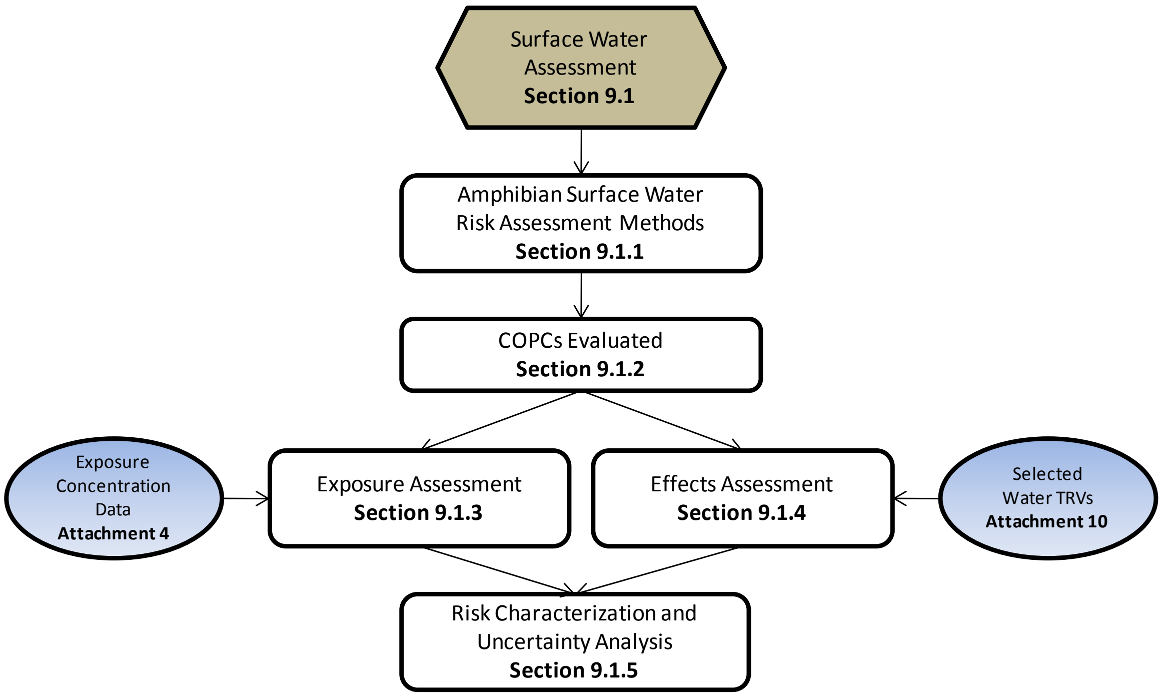


Figure 9-1. Overview of Amphibian Surface Water Section Organization

9.1.1 Amphibian Surface Water Risk Assessment Methods

Surface water HQs were calculated by comparing COPC concentrations in water samples to chronic water TRVs. These TRVs were developed from water quality criteria and literature-based TRVs, according to a hierarchy articulated in Attachment 10.

Baseline risk to amphibians was evaluated only for those areas of the Study Area that have potentially suitable amphibian habitat (Map 9-1). The comparison of surface water concentrations to water TRVs was conducted on an individual sample basis, as directed by EPA (Attachment 2).

HQs based on individual samples were derived for all COPCs using Equation 6-1 to quantify surface water risk estimates. The EPC and TRV are expressed as COPC concentrations in water. Site-wide HQs were also calculated using UCLs to characterize potential risks for the Study Area amphibian population as a whole. COPCs with HQ $\geq$ 1 for any individual surface water sample within an amphibian habitat area were identified as posing potentially unacceptable risk. The quantitative risk results (i.e., magnitude, spatial distribution, and frequency of HQs $\geq$ 1), the seasonal and sampling method patterns of HQs, and underlying uncertainties of exposure and effects data are presented in the risk characterization (Section 9.1.5). For all contaminants posing potentially unacceptable risk, the spatial distribution and magnitude of HQs and the associated exposure and effects assumptions were evaluated to arrive at risk conclusions for amphibians (Section 9.3).

9.1.2 COPCs Evaluated

Eleven of the 14 surface water COPCs identified in the SLERA and refined screen (Attachment 5) are evaluated in the BERA. Three individual DDT metabolites identified in the SLERA (2,4′-DDD, 2,4′-DDT, and 4,4′-DDD) were evaluated as part of total DDx and not individually; 4,4′-DDT was evaluated both individually and as total DDx because the TRV for DDx is based on 4,4′-DDT. All of the other 14 COPCs were evaluated in this assessment (Table 6-30).

Nineteen surface water COIs were not evaluated in the SLERA and refined screen because no toxicological data were available (Table 6-31). The risks to amphibians from these chemicals in surface water are unknown because of the absence of toxicological data. Four of these COIs (4-chloroaniline, aniline, 2,4-DB, and MCPP) were infrequently detected; when they were detected, it was in isolated areas and during different sampling events. Surface water thresholds are unavailable for individual dioxins and furans other than 2,3,7,8-TCDD. Dioxins and furans are evaluated for fish, birds, and mammals as a toxicity-weighted sum based on the toxicity of each congener relative to 2,3,7,8-TCDD, using TEFs. TEFs are not available for amphibians and were therefore not evaluated. The toxicity of 2,3,7,8-TCDD to other vertebrates (fish and wildlife) is higher than that of other congeners; the same was assumed for amphibians. Thus, evaluation of dioxins based solely on 2,3,7,8-TCDD toxicological data is expected to be protective of amphibians from all dioxin and furan congeners.

Aluminum was not identified as a COPC as per agreement with EPA because the AWQC were developed using toxicity data from acidic waters and are not applicable to the Study Area. Aluminum concentrations in background surface water and sediment were evaluated to identify local sources of aluminum contamination within the Study Area, if any (see Section 6.5.5.3). Like aluminum, zinc is naturally occurring in the environment, and background zinc concentrations were also evaluated.

One COI (2,4′-DDE) was not retained as a COPC in the refined screen because no detected concentration exceeded the corresponding TRV (although at least one DL exceeded the TRV). However, this COI was evaluated as a component of total DDx.

9.1.3 Exposure Assessment

This section presents the exposure concentrations used to evaluate risks to amphibians. An overview of all LWG- and non-LWG-collected surface water data (i.e., sampling events and rationale, and sample types) and general trends in COPC concentrations are presented in the benthic risk assessment (Section 6.5.3).

9.1.3.1 Exposure Areas

For amphibians, only surface water samples collected within amphibian/aquatic plant exposure areas (amphibian and aquatic plant exposure areas are equivalent) were used to represent EPCs (Table 9-1; Map 9-1). Amphibian/aquatic plant exposure areas were identified as the amphibian/aquatic plant habitat areas and quiescent areas. The latter are defined as areas in the Study Area potentially capable of sustaining aquatic plant growth because of slow-moving, shallow water. Amphibian habitat areas were identified based on professional judgment during a summer 2002 reconnaissance survey conducted within the Initial Study Area (RM 3.5 to RM 9.2). Locations with low-sloping beaches and riprapped or rocky banks were classified as amphibian habitat areas (see Section 2.2.4).

Seven surface water transect locations were not evaluated as potential exposure areas for amphibians because either they did not offer appropriate amphibian habitat or samples were collected from deep water in the middle of the channel: W005,W011,W023, W023E, W023M, W023W, and W025M. Near-bottom and near-surface samples from two single-point locations (W030, W037 and W038) were also excluded because both locations are near the channel of the LWR in highly industrialized areas lacking suitable amphibian habitat.

Table 9-1. Surface Water Sampling Locations Identified as Occurring in Amphibian and Aquatic Plant Exposure Areas

Sampling Location ID and Approximate Location	Reason for Inclusion		
	Amphibian Habitat[a]	Evidence of Amphibians[a]	Quiescent Area[b]
W001, RM 2	X		
W002, RM 2.2	X		
W003, RM 3	X	X	
W004, RM 3.7	X	X	X
W006, RM 4	X		X
W007, RM 4.4	X	X	X
W008, RM 4.6	X	X	X
W009, RM 5.6	X		X
W010, RM 5.7			X
W012, RM 6.3	X		X
W013-1, RM 6.7	X		X
W013-2, RM 6.7	X		X
W014, RM 6.7	X		X
W015, RM 6.9	X		X
W016-1, RM 7.2			X

Table 9-1. Surface Water Sampling Locations Identified as Occurring in Amphibian and Aquatic Plant Exposure Areas

Sampling Location ID and Approximate Location	Reason for Inclusion		
	Amphibian Habitat[a]	Evidence of Amphibians[a]	Quiescent Area[b]
W016-2, RM 7.2			X
W017, RM 7.5	X		X
W018, RM 8.3			X
W019, RM 8.6	X		X
W020, RM 9.1	X		X
W021, RM 8.7	X	X	X
W022, RM 9.7	X		
W025E, RM2	X		
W025W, RM 2	X		
W026, RM 2.1	X		
W027, RM 2.9	X		
W028, RM 3.6			X
W029, RM 4.4	X		X
W031, RM 6.1			X
W032, RM 6.7	X		X
W033, RM 7	X		X
W034, RM 7.5			X
W035, RM 8.5	X	X	X
W036, RM 8.6			X
GP082, RM 6.4	X		

[a] Evidence of amphibians includes egg masses, calls, or sightings, as documented in the 2002 amphibian/plant reconnaissance survey, presented in Appendix B of the Programmatic Work Plan (Integral et al. 2004a).

[b] Based on data presented in the Round 2 FSP for surface water (Integral 2004a).

FSP – field sampling plan

ID – identification

RM – river mile

9.1.3.2 Seasonal Relevance of Exposure to Amphibians

Amphibians such as the red-legged frog require a variety of riparian and aquatic habitats for breeding, dispersal, foraging, and refuge. Use of Study Area aquatic habitat is likely seasonal. The breeding season for amphibians that may occur within the Study Area (red-legged frog, Pacific tree frog, and bullfrog) typically ranges from late winter to early spring (Corkran and Thoms 1996). Red-legged frogs most frequently lay their eggs in March (USFWS 2002), typically on submerged or emergent vegetation in ponds, pools, or other quiescent water. Early life stages (eggs and tadpoles) are thus exposed to surface water contaminants. Red-legged frog tadpoles mature into terrestrial adults after 11 to 20 weeks (USFWS 2002); the estimated exposure period for tadpoles is from spring to summer. During wet weather, juvenile and adult red-legged frogs may disperse over upland habitat; distances of up to 2 miles have been documented (USFWS 2002).

The March 2005 surface water sampling event was selected by EPA to coincide with the early exposure period for amphibian egg masses. Data from this sampling event likely represent a sensitive exposure period for amphibians (i.e., during reproduction). Data from the July 2005 sampling event may also represent early life stage exposure, as several egg masses were observed during the June 2002 Portland Harbor amphibian and aquatic plant reconnaissance survey (Integral et al. 2004a), and amphibian tadpoles may also be present during this time.

Amphibians use aquatic habitat to a more limited extent during non-reproductive or non-early life stages. Surface water samples collected during other sampling events in winter and fall were used to evaluate risks to amphibians. However, these data are uncertain for use in estimating amphibian exposure concentrations and may not appropriately represent exposure.

Uncertainty Associated with Amphibian Use of the Study Area

In general, amphibian habitat is limited in the Study Area because riparian and quiescent backwater habitats have been lost to industrialization and channelization. The distribution and population abundance of amphibians in the Study Area is uncertain, and it is unknown whether amphibians are present in or use all habitat areas where surface water samples were collected. During the June 2002 Portland Harbor amphibian and aquatic plant reconnaissance survey (Integral et al. 2004a), frogs or egg masses were observed at the following locations: at the mouth of Multnomah Channel; at RM 2.3 (eastern bank); and inside the International Slip, Slip1, Slip 3, and Swan Island Lagoon (Map 9-1). Surface water samples were collected near or adjacent to five of these six locations.

Additional uncertainty is associated with the amphibian habitat areas. Amphibian habitat areas were identified only within the boundaries of the Initial Study Area (RM 3.5 to RM 9.2). Potential amphibian habitat areas outside of the Initial Study Area but within the Study Area (i.e., RM 1.9 to RM 3.5 and RM 9.2 to RM 11.8) were not identified and thus could not be evaluated. Risks to amphibians in these areas are unknown.

9.1.3.3 Surface Water COPC Concentrations in Amphibian Exposure Areas

All surface water data, including data from amphibian exposure areas (i.e., quiescent and amphibian habitat areas) are presented in Attachment 4. General trends in surface water COPC concentrations from samples within these exposure areas are described below.

Metals and Butyltins – Zinc and monobutyltin were analyzed only in peristaltic pump samples. Zinc (dissolved) was detected in about half of all samples at concentrations of 0.9 to 41.9 µg/L. Monobutyltin was infrequently detected (in 6% of samples) at concentrations of 0.002 to 0.02 µg/L. The highest concentrations for metals and butyltins were detected outside of the amphibian egg exposure period during the May 2005 sampling event; maximum concentrations for zinc and butyltins were detected during the November low-flow event.

SVOCs – The SVOCs (PAHs and BEHP) were analyzed in both peristaltic pump samples and XAD samples. These compounds were detected more frequently in the XAD samples (27 to 90%) than in the peristaltic pump samples (7 to 13%), which had higher DLs. However, detected concentrations of PAHs were substantially higher in the peristaltic pump samples (mean concentrations of 32 to 76,000 ng/L) than the XAD samples (mean concentrations of 1.3 to 9.6 ng/L). Detected concentrations of BEHP were also higher in the peristaltic pump samples than in the XAD samples (mean concentrations of 1,800 and 16 ng/L, respectively). The highest PAH concentrations (naphthalene at RM 6.4) were measured in peristaltic samples collected during a single May sampling event. The highest BEHP concentration was collected during a January high-flow event at Willamette Cove (RM 6.7).

PCBs – Total PCBs were detected in 100% of the XAD samples and in 13% of the peristaltic pump samples; XAD samples had lower DLs. As for SVOCs, detected concentrations of total PCBs were lower in the XAD samples (mean concentration of 0.886 ng/L) than in the peristaltic pump samples (mean concentration of 11 ng/L). The highest PCB concentration was detected during the November low-flow event.

VOCs – Ethylbenzene and trichloroethene concentrations were analyzed in only one surface water sample collected from amphibian exposure areas (RM 6.4, west bank, May 2005). Ethylbenzene was detected (3.46 µg/L) but trichloroethene was not. There is uncertainty in the exposure data for VOCs; one sample does not offer representative spatial data.

DDx – Total DDx and metabolites were detected in 96 to 100% of the XAD samples and in 0 to 28% of the peristaltic pump samples; XAD samples had lower DLs. The mean concentrations of total DDx in XAD and peristaltic pump samples were 0.86 and 3.4 ng/L, respectively. Nineteen percent of total DDx concentrations in the peristaltic pump samples were based on N-qualified results. N-qualification indicates “the presence of an analyte that has been ‘tentatively identified,’ and the associated numerical value represents its approximate concentration” (EPA 1999b). The qualification indicates that the analyst believed that the result was based on analytical interference from a chemical other than the target analyte. All N-qualified results are therefore biased high and may result in an overestimation of risk. The highest DDx concentrations were measured during a sampling event that captured the targeted amphibian egg exposure period (March 2005).

9.1.3.4 Surface Water EPCs

Surface water EPCs in this assessment are represented by detected concentrations in all individual surface water samples collected within quiescent areas or amphibian habitat areas. Surface water EPCs were also represented by site-wide UCL concentrations calculated using all surface water sampling locations collected within amphibian exposure areas. Site-wide amphibian habitat UCLs were calculated for all surface water COPCs, except for ethylbenzene and trichloroethene, which were analyzed in only one sample located within an amphibian habitat area (RM 6.4, west bank). COPC surface water concentration data for all individual samples and calculated site-wide amphibian habitat UCL concentrations are presented in Attachment 4. At locations where both XAD and peristaltic samples were collected and analyzed for organic COPCs, the results of the peristaltic samples (i.e., the low-resolution results) were removed from the dataset used to derive UCLs. These data were removed because COPC concentrations from XAD samples are based on high-resolution analyses with lower DLs and greater accuracy. Surface water EPCs based on a UCL were calculated using ProUCL Version 4.0 software (EPA 2007f). EPA's ProUCL software tests the goodness of fit for a given dataset and then computes the appropriate 95th UCL (as described in Section 7.1.3.1).

Surface water EPCs were compared to water TRVs to characterize risks to amphibians via exposure to surface water. Surface water COPC concentration data for all individual samples are presented in Attachment 4.

Uncertainty Associated with Surface Water Sampling Methods

Surface water sampling methods varied across sampling events, creating some uncertainty about data comparability. Surface water samples were collected both as single-point samples and as transect samples (vertical, horizontal, or both) using two types of sampling methods (XAD and the peristaltic method). Samples were collected over seven sampling events; and not all surface water locations were sampled during each event. Surface water transect samples are representative of a wider range of conditions because the technique collects more water over a greater area and longer period. Horizontal transects were sampled at only two locations within the Study Area that were within amphibian/aquatic plant habitat areas (at RM 2.0 and at the mouth of Multnomah Channel) and thus are limited spatially.

The evaluation of transect, single-point, XAD, and peristaltic samples provides a larger dataset for estimating amphibian surface water EPCs. The advantage of having more data at least partially offsets the disadvantage of adding unquantified uncertainty about data comparability across sampling methods.

9.1.4 Effects Assessment

Per agreement with EPA (2008f), chronic water TRVs were developed for all surface water COPCs based on the hierarchy detailed in Attachment 10. Section 6.5.4 and Table 6-32 in the benthic risk assessment present the water TRVs developed for all surface water COPCs. Use of the surface water effects thresholds to assess risk to amphibians is uncertain because amphibian data are rarely included in datasets used to derive AWQC or other criteria on which the TRVs are based. Therefore, use of these TRVs for evaluating risk to amphibians may over- or underestimate risks.

Because the selected AWQC for total PCBs (0.014 µg/L) and 4,4′-DDT (0.001 µg/L) are based on the protection of mammals and birds, respectively, risk estimates for aquatic receptors based on these TRVs are uncertain. Alternative TRVs specifically protective of aquatic receptors were developed in this BERA using methods consistent with those used for AWQC derivation. The alternative water TRV for total DDx was calculated as 0.011 µg/L. The alternative water TRV for total PCBs was calculated as 0.19 µg/L. Derivation of these alternative TRVs is described in Section 6.5.4 and Attachment 10. No aquatic toxicity data for amphibians were included in the datasets from which the alternative total PCB TRV (see Attachment 10) and alternative total DDx TRV (EPA 1980a) were derived. For evaluating direct exposure of aquatic organisms to water, the alternative TRVs are considered more appropriate than the AWQC-based TRVs.

Two sets of water HQs were derived for total PCBs, 4,4′-DDT, and total DDx: one using the selected AWQC-based TRVs and one using the alternative water TRVs. The alternative TRVs are more appropriate than the AWQC values for assessing risk to amphibians and therefore were used as the primary line of evidence to determine risk conclusions.

Surface Water TRV Uncertainties

Surface water TRVs are based on the hierarchy presented in Attachment 10 and are intended to be protective of the most sensitive aquatic organism; however, these TRVs may overestimate effects to amphibians in cases where amphibians are less sensitive than the organisms in the study setting the TRV (or underestimate effects where amphibians are more sensitive). The water TRVs for all COPCs, except BEHP, were established by the AWQC and Tier II sources and were based on or included toxicological data on invertebrate (primarily *Daphnia* spp.), fish, and bird species. BEHP was the only water TRV derived using toxicity data for amphibians (see Attachment 10). TRVs that are protective of other receptors may under- or overestimate risks to amphibians. The TRV uncertainties are discussed in more detail on a chemical-specific basis in Section 9.1.5.

The water TRV for total DDx is based on the 4,4′-DDT AWQC, which was derived from the effects data for brown pelican. Birds are known to be highly sensitive to DDx compounds, based on eggshell thinning studies, and, because of the bioaccumulative nature of DDx, may be exposed to different concentrations than amphibians. The water TRV that is protective of birds is of questionable relevance and probably overestimates risk to amphibians. Similarly, the total PCBs AWQC is based on the protection of mink via ingestion of contaminated prey. Alternative water TRVs were developed for total PCBs and DDx following the methods used to develop AWQC values and are more appropriate for evaluating risk to aquatic organisms directly exposed to surface water. Although both the selected and alterative water TRVs were used to derive water HQs, only the alternative TRVs were used to draw risk conclusions.

9.1.5 Risk Characterization and Uncertainty Analysis

This section presents the risk estimates for amphibians based on the surface water LOE. An HQ approach was used to quantify risk following the risk characterization process described in Section 9.1.1. HQs were calculated for all COPCs using Equation 6-1. The EPC and TRV are represented by surface water concentrations.

Section 9.1.5.1 presents the risk characterization results and uncertainty evaluation. Section 9.1.5.2 presents an evaluation of background concentrations. Section 9.1.5.3 presents the COIs for which risks to amphibians cannot be quantified. Section 9.1.5.4 summarizes surface water COPCs with HQ ≥ 1.

9.1.5.1 Risk Characterization Results and Uncertainty Evaluation

Table 9-2 presents a summary of the individual HQs calculated across all surface water samples collected within amphibian exposure areas for all surface water COPCs. Table 9-3 presents the HQs calculated using site-wide UCLs based on the surface water samples located within the amphibian habitat areas. COPCs with maximum HQ ≥ 1 are identified as contaminants posing potentially unacceptable risk. The spatial extent, magnitude, and potential ecological significance of TRV exceedances, and the concordance among LOEs (i.e., surface water and TZW) for contaminants posing potentially unacceptable risk are discussed in Section 9.3 to determine risk conclusions.

Table 9-2. Number of Surface Water Samples in Amphibian Exposure Areas with HQs ≥ 1

COPC	Number of Samples with HQs ≥ 1 (Max HQ)	Percentage of Samples with HQs ≥ 1
Metals		
Zinc (dissolved)	1 of 117 **(1.1)**	< 1%
Butyltins		
Monobutyltin ion	0 of 117	0%
PAHs		
Benzo(a)anthracene	2 of 158 **(10)**	1.3%
Benzo(a)pyrene	3 of 158 **(14)**	1.9%
Naphthalene	1 of 159 **(50)**	< 1%
Phthalates		
BEHP	1[a] of 129 **(1.2)**	< 1%
PCBs		
Total PCBs	0 of 111 (0.089)[b]	0%
Pesticides		
4,4′-DDT	0 of 121 (0.35)[c]	0%
Total DDx	1 of 121 **(1.8)**[d]	< 1%
VOCs		
Ethylbenzene	0 of 1	0%
Trichloroethene	0 of 1	0%

a An additional two samples had DLs that were greater than the TRV. The maximum HQ based on a DL is 1.4 for BEHP.

b Maximum HQ and the number/percentage of samples with HQs ≥ 1 presented in the table are based on the alternative TRV. Two of 111 samples had total PCB concentrations greater than the AWQC total PCB TRV of 0.014 µg/L, which is specific to protection of mink via consumption of contaminated prey (maximum HQ = 1.2).

c Maximum HQ and the number/percentage of samples with HQs ≥ 1 presented in the table are based on the alternative TRV. Thirteen of 121 and 26 of 121 samples had 4,4′-DDT and total DDx concentrations, respectively, greater than the AWQC-based 4,4′-DDT TRV of 0.001 µg/L, which is based on protection of birds (maximum HQs = 3.9 and 20, respectively). An additional three samples had DLs greater than the AWQC-based TRV. The maximum HQ based on a DL is 1.6 for both 4,4′-DDT and total DDx.

d The only sample resulting in an HQ ≥ 1 is based on N-qualified data, indicating that the elevated concentration was likely due to analytical interference from a different chemical.

AWQC – ambient water quality criteria
BEHP – bis(2-ethylhexyl) phthalate
HQ – hazard quotient
PAH – polycyclic aromatic hydrocarbon

COPC – contaminant of potential concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
DL – detection limit
Bold identifies HQs ≥ 1.
PCB – polychlorinated biphenyl
total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT, and 4,4′-DDT)
TRV – toxicity reference value
VOC – volatile organic compound

Table 9-3. Summary of Site-Wide Amphibian and Aquatic Plant Habitat Surface Water UCL HQs

COPC	Unit	UCL	Water TRV	HQ (unitless)
Metals				
Zinc	µg/L	2.9	36.5	0.079
Butyltins				
Monobutyltin ion	ng/L	3	72	0.042
PAHs				
Benzo(a)anthracene	ng/L	16	27	0.59
Benzo(a)pyrene	ng/L	13	14	0.93
Naphthalene	ng/L	38,000	12,000	**3.2**
Phthalates				
BEHP	ng/L	450	3,000	0.15
PCBs				
Total PCBs	ng/L	2.5	190 (14)[a]	0.013 (0.18)[a]
Pesticides				
4,4′-DDT	ng/L	0.53	11 (1)[a]	0.063 (0.53)[a]
Total DDx	ng/L	1.9	11 (1)[a]	0.15 (**1.9**)[a]

[a] Alternative TRVs and HQs (the AWQC-based TRV and HQ are shown in parentheses).

AWQC – ambient water quality criteria
BEHP – bis(2-ethylhexyl) phthalate
COPC – contaminant of potential concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
HQ – hazard quotient
Bold identifies HQs ≥ 1.
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT, and 4,4′-DDT)
TRV – toxicity reference value
UCL – upper confidence limit on the mean

The following COPCs have HQs ≥ 1 in at least one surface water sample collected within amphibian exposure areas: zinc, benzo(a)anthracene, benzo(a)pyrene, naphthalene, BEHP, and total DDx. Risk from total PCBs, 4,4′-DDT, and total DDx was evaluated based on alternative TRVs (as described in Section 9.1.4). When risks are calculated using the alternative PCB and DDT TRVs, all but one HQ is < 1 (one total DDx HQ ≥ 1). The AWQC TRVs for total PCBs and 4,4′-DDT are based on protection of mammals and birds, respectively, via ingestion of contaminated prey; the alternative TRVs are more realistic for aquatic organisms directly exposed to surface water. Map 9-2 presents the sampling locations for COPCs with HQ ≥ 1. Site-wide amphibian habitat HQs are ≥ 1 for one COPC (i.e., naphthalene).[145] No site-wide UCLs were derived for VOCs (ethylbenzene and trichloroethene) because of the limited spatial extent of the data.A discussion of COPCs with HQs ≥ 1, including an evaluation of the key uncertainties and their frequency and location, is presented below. Key uncertainties associated with the appropriateness of using water TRVs for assessing risks to amphibians are also discussed.

Metals –Zinc HQs are ≥ 1 in only one sample (HQ = 1.1, November 2004 sampling event) and the site-wide HQ is < 1. Uncertainty is associated with the zinc TRV (36.5 µg/L) because it is based on risk to fish and aquatic invertebrates. Amphibian-specific toxicity data for zinc in water indicate that amphibians may be more sensitive than fish; however, toxicity data are limited. In developmental toxicity tests, the most sensitive amphibians tested had zinc LC50 and LC10 (concentration that is lethal to 10% of an exposed population) values of 10 and 3 µg/L, respectively (Birge et al. 2000). Amphibians appear to be as sensitive or more sensitive to zinc than assumed when using the fish- and invertebrate-based AWQC TRVs. Only one of the 117 samples analyzed for zinc had a concentration greater than the LC50 of 10 µg/L for the most sensitive amphibian tested by Birge et al. (2000), and 12 samples had concentrations slightly greater than the LC10 of 3 µg/L (11 samples ranged from 3.2 to 6.6 µg/L and one sample had a concentration of 41.9 µg/L).

PAHs and BEHP – HQs ≥ 1 were calculated for benzo(a)anthracene, benzo(a)pyrene, naphthalene, and BEHP in fewer than 2% of samples; HQs < 1 were calculated for these COPCs in all samples collected during the sensitive amphibian exposure period in March 2005. The site-wide HQs for benzo(a)pyrene, benozo(a)anthracene, and BEHP are < 1 and the site-wide HQ for naphthalene is 3.2.

Uncertainty is associated with the TRVs for benzo(a)anthracene (0.027 µg/L), benzo(a)pyrene (0.014 µg/L), and naphthalene (12 µg/L) because they are based on risks to fish or aquatic invertebrates. Very limited amphibian-specific PAH toxicity data are available. Sublethal cellular effects have been observed in amphibians following exposure to PAHs; micronucleated erythrocytes were induced following exposure to 10 µg/L of benzo(a)pyrene, 3.12 to 750 µg/L of benzo(a)anthracene,[146] and 500 µg/L of naphthalene. Ultraviolet (UV) radiation (via sunlight) can affect the toxicity of PAHs to amphibians. Lower effects thresholds for benzo(a)pyrene have been reported in amphibians when they are also exposed to UV radiation; 100% mortality was reported in newt tadpoles following simultaneous exposure to UV-A radiation and 6.25 to 12.5 µg/L of benzo(a)pyrene (Sparling 2000). The TRV selected for the BERA (i.e., based on fish

[145] The site-wide amphibian HQ for total DDx was < 1 when based on the alternative TRV (and was ≥ 1 only when based on the AWQC TRV); therefore total DDx is not included in the count of COPCs with HQ≥ 1 for the site-wide analysis.

[146] Induction of micronucleated erythrocytes were observed at 3.12 and 6.25 µg/L of benzo(a)anthracene with UV-A exposure and at 750 µg/L of benzo(a)anthracene with no UV-A exposure.

and invertebrate Tier II values) appears to be at least as protective as effects values from Sparling (2000).

Total PCBs – Using the alternative total PCB TRV of 0.19 µg/L, all individual sample and site-wide HQs are < 1. Limited amphibian-specific PCB toxicological data are available in the published literature. PCB Aroclor 1254 amphibian thresholds of 63 µg/L (acute) and 6.3 µg/L (chronic) were derived from a 96-hour study (Zhou et al. 2004) in which 70% mortality was observed. In this study, UFs of 4 and 40 were applied to the water concentration associated with mortality to derive acute and chronic thresholds of 16 and 1.6 µg/L, respectively. PCB Aroclor 1254 LC50s for amphibians at 4 days post-hatch and as hatchlings ranged from 1.0 to 3.7 µg/L and 3.5 to 38.2 µg/L, respectively (Sparling 2000). The alternative PCB water TRV of 0.19 µg/L is based on risks to fish, and Sparling (2000) reported that rainbow trout and redear sunfish are more sensitive to PCBs than are amphibians. Because available amphibian thresholds are higher than fish thresholds, risks to amphibians from PCBs in water may be overestimated using the water TRV.

Total DDx – Total DDx concentration in only one sample (W001, RM 2.0) exceeded the TRV (HQ based on alternative TRV = 1.8). However, this result is based on N-qualified data collected via peristaltic pump. The N-qualification indicates that the elevated concentration was likely due to analytical interference from a different chemical. Total DDx HQs based on non-N-qualified data (n = 20) range from 0.003 to 0.9. The peristaltic samples (i.e., the low-resolution results) have lesser accuracy than the XAD samples.

The total DDx HQ used in this assessment is based on the alternative TRV of 0.011 µg/L. A high degree of uncertainty is associated with the total DDx AWQC-based TRV (0.001 µg/L), which was derived from bird effects data and cannot be meaningfully applied to amphibians.

The limited data available on the toxicity of DDT to amphibians indicate that amphibian thresholds are much higher than the selected and alternative TRVs. Amphibian sensitivity to total DDx varies with stage of development (Sparling 2000). High mortality (80 to 100%) was reported in frog and toad tadpoles at various stages of development following acute exposure to 50 to 500 µg/L; however, mortality was 15% or less following exposure to 5 µg/L (Cooke 1972). Sublethal effects in amphibians have included hyperactivity and abnormal snout development, which occurred at a DDT concentration of 20 µg/L (Cooke et al. 1970, as cited in Sparling 2000).

9.1.5.2 Evaluation of Background Concentrations

By agreement with EPA, aluminum was not identified as a water COPC because its AWQC was developed using toxicity data from acidic waters and is not applicable to the circumneutral waters of the Study Area. Study Area and background sediment and surface water concentrations were compared to determine if any risk to amphibians in the Study Area, if any, may be due to background levels of this crustal element.

Background aluminum concentrations were established as part of the RI (see Section 7.0 of the draft final RI (Integral et al. 2011)). Background and Study Area concentrations in sediment and surface water were compared (see Attachment 11 and Section 6.5.5.3). Aluminum concentrations in sediment and surface water for the Study Area are generally similar to or below the background UCL and UPL. Aluminum and other trace elements are major constituents of the mineral fraction of sediment but contribute to the analytical chemical results because of the acid extraction step during analysis. Because aluminum is not biologically available or not toxic at naturally occurring concentrations generally found in surface water, aluminum is not expected to pose unacceptable risk to amphibians.

Zinc is also a naturally occurring crustal element in the environment. A background water concentration could not be established because the number of data points was limited (see Attachment 11). The Study Area UCL water concentration of zinc (2.5 µg/L) is greater than the range of zinc concentrations detected in background[147] (1.4 to 2.2 µg/L). The Study Area UCL sediment zinc concentration (164 mg/kg dw) is greater than the background sediment UCL and UPL (79 and 110 mg/kg dw, respectively). These data indicate that zinc concentrations are elevated above background.

9.1.5.3 COIs for Which Risks Cannot Be Quantified

COIs for which risks to amphibians cannot be quantified based on surface water data are the same as those for benthic invertebrates listed in Table 6-35. No TRV is available for aluminum, 4-chloroaniline, aniline, 2,4-DB, MCPP, and individual dioxin and furan congeners other than 2,3,7,8-TCDD.

9.1.5.4 Summary of Surface Water LOE

Six surface water COPCs with with HQs ≥ 1 were identified for amphibians: zinc, benzo(a)anthracene, benzo(a)pyrene, naphthalene, BEHP and total DDx.

9.2 TZW ASSESSMENT

EPA's Problem Formulation (Attachment 2) calls for analysis of TZW[148] data relative to surface water effects thresholds for amphibians. TZW is evaluated in detail in Section 6.6 as an LOE for benthic invertebrates. Sections 7.4 and 10.2 evaluate these same data as LOEs for fish and plants, respectively.The TZW samples evaluated in this BERA were collected primarily during a 2005 sampling effort focusing on areas offshore of nine upland sites with known or likely pathways for discharge of upland contaminated groundwater to the Study Area. Sampling locations were selected at each of the nine study sites based on results of a groundwater discharge mapping field effort. The RI

[147] Zinc concentrations were detected in only 3 of 22 surface water samples included in the background dataset (see Section 7.0 of the draft RI).

[148] For the purpose of the BERA, TZW is the porewater associated with sediment matrix within the top 38 cm of the sediment column. TZW is composed of some percentage of both groundwater and surface water.

Appendix C2 presents the process used to select these sites per agreement with EPA, ODEQ, and LWG. The findings of the discharge mapping effort were considered in conjunction with relevant site data (e.g., hydrogeology, surface sediment texture delineation, distribution of COIs in upland groundwater and sediments) to identify zones of possible contaminated groundwater discharge. The TZW sampling locations selected for each site focused primarily on the zones of possible groundwater plume discharge, based on the GWPA discharge mapping effort. Additional sampling locations were specified to provide comparative data for TZW quality outside of the potential discharge zones (Integral et al. 2011).

Because the primary objective of RI groundwater pathway assessment was to evaluate whether transport pathways from upland contaminated groundwater plumes to the river were complete, TZW target analyte lists varied from site to site and were derived primarily based on the COIs in the upland contaminated groundwater plumes. Therefore, not all COIs in sediments were analyzed in TZW samples. As described in Sections 4.4.3.1 and 6.1.5.2 of the draft final RI (Integral et al. 2011), there also may be other groundwater plumes in the Study Area that may be discharging into river sediments where TZW samples have not been collected.

The remainder of this section is organized as follows:

- Section 9.2.1 describes the general approach used to assess risks to amphibians from TZW.
- Section 9.2.2 summarizes the TZW COPCs evaluated. Some COPCs were not evaluated because no toxicity thresholds are available.
- Section 9.2.3 explains how exposure concentrations were estimated and describes uncertainties in those estimates. All TZW chemical concentrations are presented in Attachment 4.
- Section 9.2.4 summarizes the effects data. Details on the development of the water TRVs are presented in Attachment 10.
- Section 9.2.5 presents the risk characterization results and associated uncertainties.

Figure 9-2 shows how the TZW evaluation is organized.

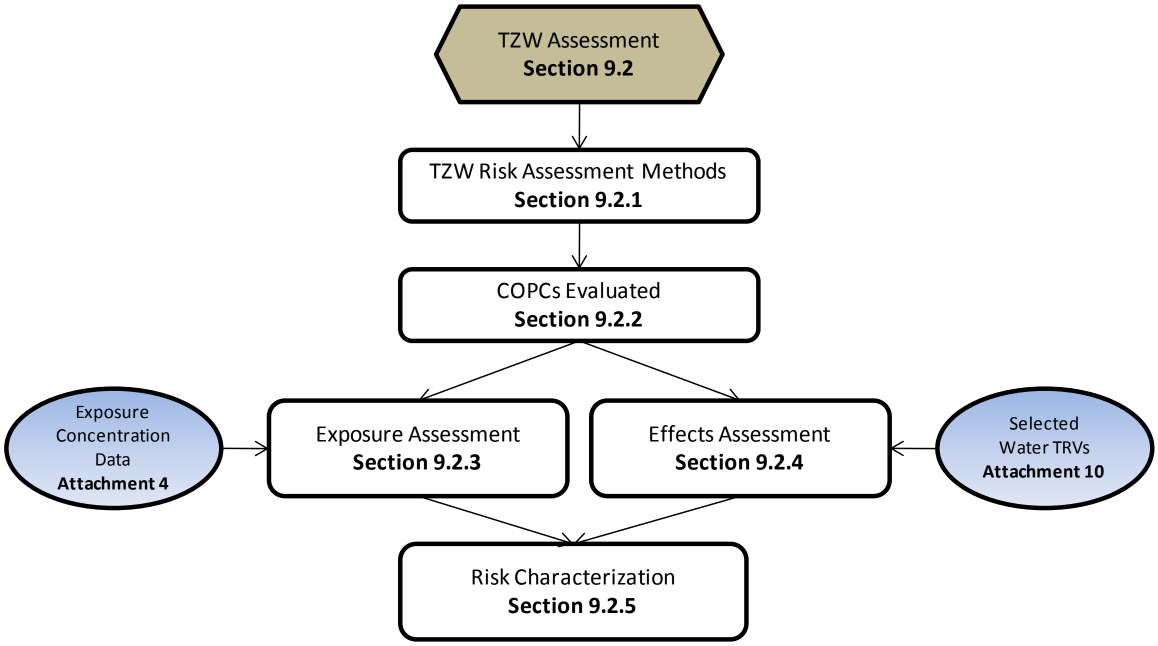


Figure 9-2. Overview of TZW Section Organization

9.2.1 TZW Risk Assessment Methods

As described in Section 6.6.1, TZW HQs were calculated by comparing COPC concentrations in individual TZW samples to chronic water TRVs. These TRVs were developed according to a hierarchy of water quality criteria and literature-based TRVs articulated in Attachment 10.

Baseline risk to amphibians was evaluated only for those areas of the Study Area that have potentially suitable amphibian habitat. COPCs were identified in the SLERA based on all TZW samples; for the baseline risk characterization, the analysis addresses only surface water samples collected from areas that offer potentially suitable amphibian habitat (Map 9-1). The comparison of surface water concentrations to water TRVs was conducted on an individual sample basis. Potentially unacceptable risks were identified by COPCs with HQs ≥ 1. Exposure data, effects data, and the quantitative risk results (i.e., magnitude, spatial distribution, and frequency of HQs) are discussed in the following sections. The relative strengths and uncertainties for both amphibian LOEs are evaluated together in the risk conclusions for amphibians (Section 9.3).

9.2.2 COPCs Evaluated

Fifty-four of the 58 TZW COPCs identified in the SLERA and refined screen (Attachment 5) are evaluated in the BERA. Four individual DDT metabolites identified in the SLERA (2,4′-DDD, 2,4′-DDT, 4,4′-DDD, and 4,4′-DDE) were evaluated as total DDx and were not evaluated individually; 4,4′-DDT was evaluated both individually and as total DDx because the TRV for all DDT-related compounds is based on 4,4′-DDT. Table 6-37 presents the detected TZW COPCs by site.

Fourteen TZW COIs could not be evaluated because no toxicological data are available to allow development of water TRVs (Table 6-38). The risks to amphibian receptors associated with exposure to these chemicals in TZW are therefore unknown. TRVs aere unavailable for individual dioxins and furans other than 2,3,7,8-TCDD. For fish and wildlife, dioxins and furans are evaluated as a toxicity-weighted sum based on the toxicity of each congener relative to 2,3,7,8-TCDD, using TEFs based on their common mechanism for toxicity. Because TEFs are not available for amphibians, no individual dioxin or furan (other than 2,3,7,8-TCDD) or dioxin group total could be evaluated.

By agreement with EPA, aluminum was not identified as a water COPC because its AWQC was developed using toxicity data from acidic waters and is not applicable to the circumneutral waters of the Study Area. Aluminum concentrations in background surface water and sediment were evaluated to determine whether a local source of aluminum is present within the Study Area (Section 6.5.5.3).

In addition, two TZW COIs were not retained as COPCs because no detected concentrations exceeded TRVs (although at least one DL exceeded a TRV): selenium and styrene. The potential risks to amphibians based on TZW data from these chemicals are unknown (see Table 5-11).

9.2.3 Exposure Assessment

This section presents the TZW exposure concentrations used to assess potential risks to amphibians. Section 9.2.3.1 discusses amphibian exposure areas. Section 9.2.3.2 describes how EPCs were derived. Section 9.2.3.3 summarizes uncertainty associated with TZW exposure. An overview of the sampling methods for all LWG- and non-LWG-collected TZW data used in this assessment is presented in the benthic risk assessment (Section 6.6.3.1). The RI Appendix C2 presents the process used to select these sites per agreement with EPA, ODEQ, and LWG. TZW sampling locations used in this assessment are shown on Map 9-1.

9.2.3.1 Exposure Areas

For amphibians, only TZW samples collected within amphibian habitat areas or quiescent areas were used to represent EPCs (Attachment 4, Part B; Map 9-1). The same exposure areas used for the amphibian surface water LOE (as described in Section 9.1.3.1) were used for the TZW LOE. Of the 192 TZW sample locations, 105 were excluded because they are outside quiescent areas that offer potential amphibian habitat (see Attachment 4, Part B).

9.2.3.2 TZW EPCs

EPCs in this assessment are represented by TZW concentrations in all individual TZW samples collected from amphibian exposure areas regardless of sampling method or depth.[149] TZW concentrations were compared to surface water TRVs to characterize risks

[149] All TZW samples evaluated in this BERA were from depths of 0 to 38 cm; however, the exact depth within that interval varied with sampling equipment (i.e.,peeper, Trident® probe, and Geoprobe).

to amphibians. A summary of the chemicals detected in shallow TZW from throughout the Study Area and the range of concentrations is presented in the benthic risk assessment (Section 6.6.3.2). All TZW data, by site, are presented in Attachment 4.

9.2.3.3 Uncertainty Associated with Ecological Exposure to TZW

The degree to which the samples are representative of exposure to TZW for organisms living in or on the sediment is a key uncertainty associated with this evaluation. Because the primary objective of RI groundwater pathway assessment was to evaluate whether transport pathways from upland contaminated groundwater plumes to the river were complete, TZW target analyte lists varied from site to site and were derived primarily based on COIs in the upland contaminated groundwater plumes. Therefore, not all COIs in sediments were analyzed in TZW samples. As described in Sections 4.4.3.1 and 6.1.5.2 of the draft final RI (Integral et al. 2011), there also may be other groundwater plumes in the Study Area that may be discharging into river sediments where TZW samples have not been collected.

Section 6.6.3.3 discusses the uncertainties associated with the benthic invertebrate exposure data for the TZW assessment. In short, in this focused study, TZW samples were collected from areas offshore of nine upland sites with known or likely pathways for discharge of contaminated upland groundwater to the Study Area. Because these areas include potential habitat for benthic invertebrates, benthic fish, amphibians, and aquatic plants, TZW for these receptors is considered a complete and significant pathway. These organisms reside in the sediment column or are in contact with the sediment surface; however, a number of studies suggest that for many benthic organisms, the water column provides relatively greater exposure to contaminants than does the sediment matrix (Hare et al. 2001). The proportion each matrix contributes to the exposure is influenced by how (and if) an organism irrigates its tube for respiration and waste removal, where in the sediment column it lives, its diet, and the configuration and construction of the burrow or tube wall.

The uncertainties summarized for benthic invertebrates apply to an even greater degree to amphibians. It is not uncommon for some species of amphibians to be associated with benthic habitat as larvae. Amphibian larvae are also known to burrow in muddy bottoms and live in the interstices of rocks (Wells 2007). However, the amphibian species most likely to use the Study Area are not associated with such habitat use. Red-legged frog, the representative amphibian species, would not have more than brief exposure to benthic habitats as adults because they live primarily in terrestrial habitats and feed mostly on terrestrial prey (see Section 9.0). Northern red-legged frog tadpoles eat epiphitic algae and prefer areas with dense vegetated cover (Dickman 1968; as cited in Lannoo 2005). Tadpoles of the Pacific treefrog, the other species identified in the Study Area, live in the water column feeding primarily on algae (Lannoo 2005). Amphibian eggs would not be in direct contact with TZW during development because they are typically laid near the surface of the water and are either attached to emergent vegetation as a single egg or deposited in a film acress the surface of the water (Sparling et al. 2000).

Given their life history, amphibian larvae likely have negligible exposure to porewater compared to surface water. Thus, the representativeness of the COPC concentrations in shallow TZW for purposes of estimating exposure and subsequent risk to amphibians is questionable and conservative to an uncertain degree.

9.2.4 Effects Assessment

TZW chemical concentrations were compared to the effects thresholds as part of the risk characterization process. At the direction of EPA (2008f), chronic water TRVs were developed for all TZW COPCs according to the hierarchy detailed in Attachment 10. Chronic water TRVs were developed through a review of WQS, criteria, published benchmarks, and toxicity data. The TRVs selected were approved by EPA for use in the BERA. Criteria for metals COPCs were hardness-adjusted when appropriate. If the published criteria for individual metals were based on dissolved concentrations, then the dissolved sample result was compared to the dissolved criterion; otherwise the total concentration for both the sample and criterion were used. Table 6-40 presents the TRVs for all TZW COPCs and their sources. These values were developed based on the sensitivities of fish and invertebrate species and are considered protective of all aquatic receptors, including amphibians.

As noted in Section 9.1.4, because the selected AWQC for 4,4′-DDT is based on protection of birds via ingestion of contaminated prey, risk estimates for aquatic receptors based on this TRV are associated with substantial uncertainty. An alternative TRV protective of aquatic organisms was developed in this BERA using methods consistent with those used for AWQC derivation. The alternative water TRV for DDx compounds was calculated as 0.011 µg/L. This alternative TRV is the appropriate metric for evaluating direct exposure of aquatic organisms to water because the AWQC (0.0010 µg/L) is based on the protection of brown pelican via ingestion of contaminated prey and cannot be used meaningfully to judge risk to aquatic organisms. Nevertheless, two sets of water HQs were derived: one using the alternative water TRV (0.011 µg/L) and one using the AWQC DDx water TRV (0.001µg/L). Because the alternative TRVs are more appropriate than the AWQC values for assessing risk to amphibians, they therefore were used as the primary line of evidence to determine risk conclusions.

9.2.5 Risk Characterization

This section presents the risk estimates for amphibians based on the TZW LOE. It also includes an evaluation of naturally occurring metals, and a list of COIs that could not be evaluated.

9.2.5.1 Risk Characterization Results

Individual HQs were calculated for all COPCs across all TZW samples within amphibian and plant exposure areas. The frequency with which individual samples have HQs ≥ 1 are shown in Table 9-4.

Table 9-4. Number of Individual TZW Samples in Amphibian and Aquatic Plant Exposure Areas with HQs ≥ 1

COPC	Samples with HQs ≥ 1	
	Number (Maximum HQ)	Percentage
Metals		
Barium (total)	49 of 49 (**1,100**)	100
Beryllium (total)	0 of 49 (0.78)	0
Cadmium (dissolved)	6 of 30 (**5.8**)[a]	20
Copper (dissolved)	1 of 22 (**1.3**)	4.5
Iron (total)	46 of 49 (**250**)	93.9
Lead (dissolved)	2 of 30 (**3**)	6.7
Magnesium (total)	6 of 49 (**7**)	12.2
Manganese (total)	49 of 49 (**550**)	100
Nickel (dissolved)	2 of 30 (**1.6**)	6.7
Potassium (total)	2 of 49 (**3.7**)	4.1
Sodium (total)	9 of 49 (**55**)	18.4
Zinc (dissolved)	1 of 30 (**14**)	3.3
PAHs		
2-Methylnaphthalene	3 of 37 (**3.4**)	8.1
Acenaphthene	2 of 37 (**3.3**)	5.4
Anthracene	3 of 37 (**3.6**)	8.1
Benzo(a)anthracene	9 of 37 (**8.5**)[a]	24.3
Benzo(a)pyrene	8 of 37 (**15**)[b]	21.6
Benzo(b)fluoranthene	0 of 37 (0.34)	0
Benzo(g,h,i)perylene	0 of 37 (0.66)	0
Benzo(k)fluoranthene	0 of 37 (0.37)	0
Chrysene	0 of 37 (0.27)	0
Dibenzo(a,h)anthracene	0 of 37 (0.078)	0
Fluoranthene	0 of 37 (0.7)	0
Fluorene	6 of 37 (**4.6**)	16.2
Indeno(1,2,3-cd)pyrene	0 of 37 (0.8)	0
Naphthalene	5 of 72 (**57**)[c]	6.9
Phenanthrene	7 of 37 (**4.6**)	18.9
Pyrene	0 of 37 (0.4)	0

Table 9-4. Number of Individual TZW Samples in Amphibian and Aquatic Plant Exposure Areas with HQs ≥ 1

	Samples with HQs ≥ 1	
COPC	**Number (Maximum HQ)**	**Percentage**
SVOCs		
1,2-Dichlorobenzene	1 of 56 (**1.9**)	1.8
1,4-Dichlorobenzene	0 of 50 (0.6)	0
Dibenzofuran	0 of 37 (0.25)	0
Pesticides		
4,4'-DDT	3 of 12 (**160**)[d]	25
Total DDx	8 of 12 (**280**)[c]	66.7
VOCs		
1,1-Dichloroethene	0 of 56 (0.17)	0
1,2,4-Trimethylbenzene	1 of 5 (**2**)	**20**
1,3,5-Trimethylbenzene	0 of 5 (0.37)	0
Benzene	0 of 56 (0.32)	0
Carbon disulfide	1 of 56 (**870**)[c]	1.8
Chlorobenzene	2 of 56 (**190**)	3.6
Chloroethane	1 of 56 (**3.4**)	1.8
Chloroform	2 of 56 (**21**)[e]	3.6
cis-1,2-Dichloroethene	0 of 56 (0.047)	0
Ethylbenzene	1 of 56 (**4.5**)	1.8
Isopropylbenzene	1 of 56 (**1.3**)	1.8
m,p-Xylene	0 of 56 (0.11)	0
o-Xylene	0 of 56 (0.44)	0
Toluene	1 of 56 (**2.9**)	1.8
Total xylenes	0 of 56 (0.77)	0
Trichloroethene	0 of 56 (0.7)	0
Petroleum Hydrocarbons[f]		
Gasoline fraction (aliphatic): C4-C6	2 of 22 (**1.3**)	9.1
Gasoline fraction (aliphatic): C6-C8	0 of 22 (0.8)	0
Gasoline fraction (aromatic): C8-C10	0 of 22 (0.52)	0

Table 9-4. Number of Individual TZW Samples in Amphibian and Aquatic Plant Exposure Areas with HQs ≥ 1

COPC	Samples with HQs ≥ 1 Number (Maximum HQ)	Percentage
Gasoline fraction (aliphatic): C10-C12	12 of 22 (**100**)	54.5
Conventionals		
Cyanide	1 of 2 (**23**)[c]	50
Perchlorate	3 of 17 (**19**)[c]	17.6

[a] An additional three samples had DLs greater than the TRV.

[b] An additional seven samples had DLs greater than the TRV

[c] One additional sample had a DL greater than the TRV.

[d] HQ presented is based on the alternative 4,4′-DDT TRV for protection of directly exposed aquatic organisms. An additional four non-detect samples had DLs greater than the TRV. These maximum HQs are based on unfiltered samples. Maximum HQs for filtered samples would be 2.8 for 4,4′-DDT (however, this contaminant was never detected) and 14.5 for total DDx. The alternative TRV is the appropriate value for evaluating direct exposure of aquatic organisms. However, when calculated based on the AWQC TRV, the 4,4′-DDT and total DDx HQs are ≥ 1 in 7 of 9 and 9 of 9 samples, respectively at one location (Arkema acid plant area; maximum 4,4′-DDT HQ = 1,800, and maximum total DDx HQ = 3,100).

[e] Two additional samples had a DL greater than the TRV.

[f] Because petroleum compounds are not CERCLA contaminants, gasoline-range hydrocarbons are not included in the final count of contaminants posing potentially unacceptable risk; they are included here because they may nonetheless contribute to risk.

AWQC – ambient water quality criteria
CERCLA – Comprehensive Environmental Response, Compensation, and Liability Act
COPC – contaminant of potential concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
DL – detection limit
EPC – exposure point concentration
HQ – hazard quotient
PAH – polycyclic aromatic hydrocarbon
SVOC – semivolatile organic compound
total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT, and 4,4′-DDT)
TZW – transition zone water
VOC – volatile organic compound

Bold identifies HQs ≥ 1.

The uncertainties associated with the TZW data as representive exposure data for amphibians is dicussed in Section 9.2.3.3 and Section 9.3. Actual exposure of amphibians to TZW is likely several orders of magnitude lower than represented by TZW COPC concentrations because amphibians in the Study Area have little or no contact with TZW.

9.2.5.2 Evaluation of Naturally Occurring Metals

Although there are many anthropogenic sources of metals, most metals measured in TZW are also common crustal elements in sediment. Among the most common metals associated with sediments, barium, iron, and manganese were detected in all TZW samples. These common metals are also associated with the highest HQs identified in the risk charactrization, but there is substantial uncertainty that their sources are anthropogenic. Sodium, potassium, and magnesium are also naturally present in water, but some industrial processes may elevate their concentrations in surface water.

The contribution of geochemical processes in sediments to the concentrations of selected metals in TZW was extensively evaluated in Appendix C2 of the draft final RI (Integral et al. 2011). Concentrations of iron and manganese in TZW are not well-correlated to potential anthropogenic source materials (i.e., petroleum hydrocarbons), suggesting that factors other than contamination in the sediment (e.g., naturally occurring organic materials) are contributing to concentrations measured in the TZW. Geochemical processes are also likely responsible for some percentage of the measured concentrations of barium in TZW, in addition to the contribution from migration of upland groundwater to the river.

By agreement with EPA, aluminum was not identified as a COPC because its AWQC was developed using toxicity data from acidic waters and is not applicable to the circumneutral waters of the Study Area. Aluminum was not included in the RI geochemical evaluation, but a background surface water concentration (established in Section 7.0 of the draft final RI (Integral et al. 2011)) is available to provide some context for TZW (since surface water is a component of shallow TZW). An upper-bound (UPL) background concentration for aluminum was 1,485 µg/L. The majority of the TZW concentrations were below this concentration.

9.2.5.3 COIs for Which Risks Cannot Be Quantified

Risks to amphibians cannot be quantified for 16 TZW COIs (aluminum, calcium, titanium, selenium, styrene, seven dioxin/furan congeners other than 2,3,7,8-TCDD, and four TPH [Table 6-42]). These are chemicals for which no TRV is available or whose maximum DL exceeds a TRV, but whose detected concentrations do not.

9.2.5.4 Summary of TZW Risk Evaluation

Thirty-two TZW COPCs with HQ ≥ 1 were identified for amphibians (Table 9-4).[150] The relative strengths and uncertainties for both amphibian LOEs are evaluated together in the risk conclusions for amphibians, Section 9.3.

9.3 RISK CONCLUSIONS

This section presents a summary of the overall risk conclusions for the amphibian risk assessment. Amphibian COPCs with HQs ≥ 1 in individual surface water or TZW samples were identified as posing potentially unacceptable risk. Risk conclusions incorporate the magnitude of HQs, spatial distribution and frequency of HQ ≥ 1, and the uncertainty of exposure and effects assumptions for estimating population-level risks to amphibians. As per the EPA Problem Formulation (Attachment 2), amphibians were selected as a surrogate receptor that is protective of reptiles; therefore, risk conclusions for amphibians are protective of reptiles. Thirty-three COPCs were identified as posing potentially unacceptable risk to amphibians (i.e., HQ ≥ 1) based on the surface water and

[150] Thirty-three TZW COPCs with HQs ≥ 1 were identified for amphibians. Because petroleum compounds are not CERCLA contaminants, gasoline-range hydrocarbons are not included in the final count of contaminants posing potentially unacceptable risk; they are mentioned because they may nonetheless contribute to risk.

TZW LOEs: 5 COPCs based on both LOEs, 1 COPC (BEHP) based on the surface water LOE only, and 27 COPCs based on the TZW LOE only. Risk conclusions associated with seven of these COPCs are discussed in Table 9-5 considering both LOEs[151]. Risk to amphibians from 27 COIs (including 19 surface water COIs and 16 TZW COIs) could not be evaluated because no appropriate screening-level TRVs were available or because DLs exceed the TRV (Tables 5-11 and 5-12; Table 6-35 for surface water COIs, and Table 6-42 for TZW COIs).

In general, most surface water samples were below effects thresholds. Those surface water samples with concentrations resulting in HQs ≥ 1 are below amphibian-specific thresholds or have HQs ≥ 1 during non-reproductive periods, indicating negligible risk to amphibian populations (Table 9-5). As discussed in Section 9.1.5.1, there is a high degree of uncertainty concerning the relevance of the selected TRVs for individual PAHs to amphibian populations because no amphibian-specific thresholds are available for comparison.

Risks to amphibians from the TZW COPCs with HQs ≥ 1 are negligible because of the lack of significant exposure in the Study Area. With notable exceptions, TZW HQs are low. Because amphibians are unlikely to be directly exposed to TZW, HQs < 100 probably indicate negligible risk to amphibians in the vicinity of the sample locations. Chemicals with HQs ≥ 100 are the naturally occurring metals barium, iron, and manganese; the pesticides 4,4′-DDT and total DDx; the PAHs benzo(a)anthracene, benzo(a)pyrene, and naphthalene; the VOCs carbon disulfide, chlorobenzene and trichloroethene; petroleum in the C10 to C12 range; and cyanide. Exposure to naturally occurring metals is similar to background and these metals pose negligible risk. The pesticides have TZW HQs ≥ 100 in samples from one area—the Arkema acid plant (2 of 14 HQs ≥ 100 for 4,4′-DDT, and 4 of 14 HQs ≥100 for total DDx). However, these pesticide risks may be overestimated because they are based on unfiltered samples where most of the DDx residue is apparently associated with the particulate content of the samples. The maximum HQ for filtered samples would be 2.8 for 4,4′-DDT (which was not detected) and 14.5 for total DDx. Carbon disulfide has an HQ ≥ 1 in one of three samples from the Gasco area. Chlorobenzene has HQs ≥ 100 in 2 of 14 samples from the Arkema acid plant area. Petroleum in the C10 to C12 range has an HQ ≥ 100 in 1 of 16 samples from the Exxon Mobil Oil area. Even these high TZW HQs pose negligible risk to amphibians in the localized areas because amphibians that are likely to occur in the Study Area are unlikely to use habitats where TZW samples were collected, indicating negligible risks to amphibian populations.

In Section 11, the COPCs for the amphibian population are evaluated alongside the COPCs for all other ecological assessment endpoints to reach overall ecological risk conclusions.

[151] As noted in Table 9.2, for amphibians, the surface water HQ < 1 for 4,4′-DDT based on the alternative TRV (the HQ ≥1 is for TZW only), and the HQ ≥ 1 for BEHP is for the surface water LOE only. However, BEHP and 4,4′-DDT have been included in Table 9-5 in addition to the five COPCs with HQs ≥1 for both amphibian LOEs.

Table 9-5. Summary of Amphibian Surface Water and TZW COPCs with HQs≥ 1

	Max HQ by LOE[a]			
COPC	Surface Water	TZW	Conclusion	Rationale
Metals				
Zinc	1.1	14	Negligible risk	Max surface water and TZW HQs are not indicative of ecologically significant risk. Risk is of limited spatial extent and all HQs are low (≤ 1.1 for surface water and ≤ 14 for TZW). Surface water HQ ≥ 1 for only 1 of 117 samples (collected in November during a non-reproductive period). HQ is ≥ 1 in only 1 of 30 TZW samples. Risk may be underestimated because the TRV is based on toxicity to fish and invertebrates, and the limited available data indicate amphibians may be more sensitive. TZW risk likely several orders of magnitude lower than reflected by HQs because of limited exposure potential.
PAHs				
Benzo(a)-anthracene	10	8.5	Negligible risk	Max surface water and TZW HQs are not indicative of ecologically significant risk. Risk is of limited spatial extent and all HQs are low (≤ 10 for surface water and ≤ 8.5 for TZW). Surface water HQ ≥ 1 in only 2 of 158 samples collected during non reproductive periods (July and winter); TRV is based on extrapolated Daphnia acute LC50; limited toxicological data indicate that amphibians may be less sensitive to PAHs than are fish and invertebrates. TZW risk likely several orders of magnitude lower than reflected by HQs because of limited exposure potential.
Benzo(a)pyrene	14	15	Negligible risk	Max surface water and TZW HQs are not indicative of ecologically significant risk. Risk is of limited spatial extent. Surface water HQ ≥ 1 in only 3 of 158 samples collected during non reproductive periods (July, November, and winter); TRV based on extrapolated Daphnia acute LC50; limited toxicological data indicate that amphibians may be less sensitive to PAHs than are fish and invertebrates. TZW risk likely several orders of magnitude lower than reflected by HQs because of limited exposure potential.
Naphthalene	50	57	Negligible risk	Max surface water and TZW HQs are not indicative of ecologically significant risk. Risk is of limited spatial extent. Surface water HQ ≥ 1 in only 1 of 159 samples collected during reproductive period (May); TRV based on risk to fish and invertebrates; limited toxicological data indicate that amphibians may be less sensitive to PAHs than are fish and invertebrates (all surface water concentrations are less than amphibian toxicity thresholds). TZW HQ≥ 1 in 5 of 72 samples. TZW risk likely several orders of magnitude lower than reflected by HQs because of limited exposure potential.

Table 9-5. Summary of Amphibian Surface Water and TZW COPCs with HQs≥ 1

COPC	Max HQ by LOE[a] Surface Water	Max HQ by LOE[a] TZW	Conclusion	Rationale
Phthalates				
BEHP	1.2	NA	Negligible risk	Max surface water HQ not indicative of ecologically significant risk. Risk is of limited spatial extent and all HQs are low (≤ 1.2 for surface water). HQ ≥ 1 in only 1 of 129 samples collected during non reproductive period (winter); TRV based on risk to fish, invertebrates, and amphibians.
Pesticides				
4,4′-DDT[b]	0.35	160	Negligible risk	Max TZW HQ not indicative of ecologically significant risk. Risk is of limited spatial extent and HQs are generally low (≤ 0.35 for surface water, and < 100 for all but two TZW samples). TZW HQ ≥ 1 in only 3 of 12 samples; TRV based on risk to fish, invertebrates, and amphibians. TZW risk likely several orders of magnitude lower than reflected by HQs because of limited exposure potential. Furthermore, maximum TZW risk is based on unfiltered samples. The maximum HQ for filtered samples would be 2.8 for 4,4′-DDT, which was not detected.
Total DDx[b]	1.8	280[c]	Negligible risk	Max surface water and TZW HQs are not indicative of ecologically significant risk. Risk is of limited spatial extent and HQs are generally low (≤ 1.8 for surface water and < 100 in all but four TZW samples). Surface water HQ ≥ 1 in 1 of 121 samples (sample N-qualified indicating interference from non-DDx chemical); TZW HQ ≥ 1 in 8 of 12 samples and > 100 in only 4, all from Arkema acid plant location; TRV based on risk to fish and invertebrates; available toxicity data indicate amphibians are less sensitive. TZW risk likely several orders of magnitude lower than reflected by HQs because of limited exposure potential. Furthermore, maximum TZW risk is based on unfiltered samples. The maximum TZW HQ for filtered samples would be 14.5 for total DDx.

Note: This table attempts to summarize the BERA's amphibian risk estimates and risk descriptions, the two major components of the risk characterization. Balancing and interpreting the different types of data evaluated in the BERA can be a major task requiring professional judgment. It can be difficult to prepare a concise summary of conclusions without losing important context, yet a concise summary is needed to help the risk manager judge the likelihood and ecological significance of the estimated risks (EPA 1997).

All the COPCs listed in this table have an HQ ≥ 1 in at least one LOE for at least one ecological receptor, and by definition pose potentially unacceptable risk. The likelihood and ecological significance of the potentially unacceptable risk may vary, though, from very low to very high. Therefore, the risk description may range from negligible to significant. For each receptor-COPC pair with a maximum HQ ≥ 1, this table provides maximum HQ by LOE, a synoptic risk description, and a very brief rationale for the risk description. This distillation of the body of knowledge presented in the BERA should not be taken out of context.

[a] HQs are presented only for water samples that exceed chronic TRVs. HQ s < 1 in all other water samples.

[b] HQ shown is based on the alternative 4,4′-DDT TRV for protection of directly exposed aquatic organisms. The alternative TRV is the appropriate value for evaluating direct exposure of aquatic organisms. However, when calculated based on the AWQC TRV (for protection of brown pelican via ingestion of contaminated prey) the surface water HQ ≥ 1 for 4,4′-DDT and total DDx (max HQs = 3.9 and 20, respectively). When calculated based on the AWQC TRV, TZW HQ ≥ 1 for 4,4′-DDT and total DDx at one location (Arkema acid plant area; max HQ = 1,800 and 3,100, respectively).

AWQC – ambient water quality criteria
BEHP – bis(2-ethylhexyl) phthalate
BERA – baseline ecological risk assessment
COPC – contaminant of potential concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
HQ – hazard quotient
LC50 – concentration that is lethal to 50% of an exposed population
LOE – line of evidence
PAH – polycyclic aromatic hydrocarbon
total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT, and 4,4′-DDT)
TRV – toxicity reference value
TZW – transition zone water

10.0 AQUATIC PLANT RISK ASSESSMENT

This section presents the draft BERA for the Portland Harbor aquatic plant community. As presented in the ecological CSM (Figure 3-2), three complete and significant exposure pathways were identified for the aquatic plant community. These pathways are direct contact with surface water, direct contact with sediment, and direct contact with TZW.[152] The surface water assessment is presented in Section 10.1. The direct sediment contact pathway is considered complete (Figure 3-2) but was not evaluated per EPA's Problem Formulation (Attachment 2) because no appropriate studies reporting adverse effects on aquatic plants from sediment-associated chemicals were identified in the literature.[153] The TZW evaluation is presented in Section 10.2. As required by EPA, TZW was screened against surface water screening values.

10.1 SURFACE WATER ASSESSMENT

Surface water COPCs were identified in the SLERA and refined screen using water TRVs based on AWQC or other TRVs available in the literature (Attachment 5). In this assessment, the same water TRVs were used to evaluate baseline risks to aquatic plants. All surface water samples taken from aquatic plant habitat areas were evaluated (Map 9-1).

- Section 10.1.1 presents the general approach used to assess risks to aquatic plants from surface water.
- Section 10.1.2 presents a summary of the COPCs evaluated.
- Section 10.1.3 presents an overview of how exposure concentrations were derived. All surface water chemical concentrations are presented in Attachment 4.
- Section 10.1.4 presents a summary of the effects data. Details on the development of the water TRVs are presented in Attachment 10.
- Section 10.1.5 presents the risk characterization results, COPCs, and associated uncertainties. These COPCs are further assessed in the aquatic plant risk conclusions (Section 10.3).

Figure 10-1 describes the layout of the surface water assessment section.

[152] As presented in Section 3, direct exposure of aquatic plants to TZW was evaluated as part of this BERA per EPA's Problem Formulation (Attachment 2), although there is some uncertainty associated with this pathway.

[153] EPA's ECOTOX database and Google® were searched using the terms aquatic, plant, sediment, toxicity, and phytotoxicity singly and in combination. Several sediment plant bioassays from contaminated sites (Lewis et al. 2001; Biernacki et al. 1997) were identified, but no acceptable sediment LOAELs were identified.

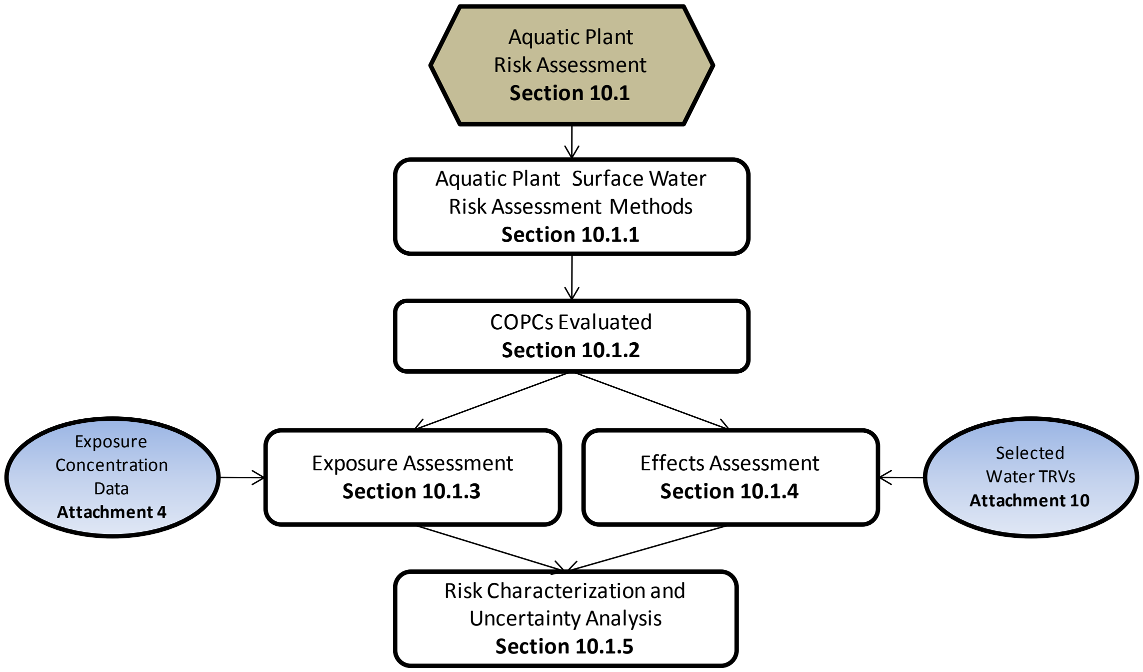


Figure 10-1. Overview of Aquatic Plant Surface Water Section Organization

10.1.1 Aquatic Plant Surface Water Risk Assessment Methods

Surface water HQs were calculated by comparing COPC concentrations in water samples to chronic water TRVs. These TRVs were developed from water quality criteria and literature-based TRVs, according to a hierarchy articulated in Attachment 10.

Baseline risk to aquatic plants was evaluated only for those areas of the Study Area that have potentially suitable aquatic plant habitat. COPCs identified in the SLERA were based on all surface water samples, whereas for the baseline risk characterization, surface water samples were restricted to those collected from within aquatic plant habitat areas (Map 9-1). The comparison of surface water concentrations to water TRVs was conducted on an individual sample basis, as directed by EPA direction (Attachment 2). Site-wide HQs were also calculated to characterize potential risks for the Study Area aquatic plant population as a whole. COPCs with HQs ≥ 1 for any individual surface water sample within an aquatic plant habitat area were identified as posing potentially unacceptable risk. The quantitative risk results (i.e., magnitude, spatial distribution, and frequency of HQs), the seasonal and sampling method patterns of HQs, and underlying uncertainties of exposure and effects data are presented in the risk characterization (Section 10.1.5). For all contaminants posing potentially unacceptable risk, the spatial distribution and magnitude of HQs and the associated exposure and effects assumptions were evaluated to arrive at risk conclusions for aquatic plants (Section 10.3).

10.1.2 COPCs Evaluated

Eleven of the 14 surface water COPCs identified in the SLERA and refined screen (Attachment 5) are evaluated in the BERA. Three individual DDT metabolites identified in the SLERA (2,4′-DDD, 2,4′-DDT, and 4,4′-DDD) were evaluated as part of total DDx and were not evaluated individually. Serving as the basis for the DDx TRV, 4,4′-DDT was evaluated both individually and as total DDx. All other COPCs were evaluated in this assessment (Table 6-30).

Nineteen surface water COIs were not evaluated in the SLERA and refined screen (Table 6-35) because no toxicological data were available. Risks to aquatic plants from these COIs could not be evaluated because no toxicological data are available; therefore, the risks to aquatic plants from these chemicals in surface water are unknown. Surface water thresholds are unavailable for individual dioxins and furans other than 2,3,7,8-TCDD. For fish and wildlife, dioxins and furans are evaluated as a toxicity-weighted sum based on the toxicity of each congener relative to 2,3,7,8-TCDD, using TEFs based on their common mechanism for toxicity. Because TEFs are not available for aquatic plants, no individual dioxin or furan (other than 2,3,7,8-TCDD) or dioxin group total could be evaluated.

Aluminum was not identified as a COPC as per agreement with EPA because the AWQC was developed using toxicity data from acidic waters and is not applicable to the Study Area. Aluminum concentrations in background surface water and sediment were evaluated to identify local sources of aluminum contamination, if any, within the Study Area (Section 6.5.5.3). Like aluminum, zinc is naturally occurring in the environment, and background zinc concentrations were also evaluated.

In addition, one COI (2,4′-DDE) was not retained as a COPC in the refined screen because no detected concentration exceeded its TRV (although at least one DL did). However, this COI was evaluated as a component of total DDx.

10.1.3 Exposure Assessment

This section presents the exposure concentrations used to evaluate risks to aquatic plants. An overview of all LWG- and non-LWG-collected surface water data (i.e., sampling events, rationale, and sample types) and general trends in COPC concentrations from the Study Area are presented in the benthic risk assessment (Section 6.5.3).

Amphibian and aquatic plant exposure areas (which are the same) were identified as the amphibian and aquatic plant habitat areas and quiescent areas (Map 9-1). Quiescent areas were defined as slow-moving, shallow waters in the Study Area potentially capable of sustaining the growth of plants adapted to living in or on aquatic environments (i.e., submerged or emergent plants). Among the submerged and emergent plants observed were water moss, grasses, sedges, smartweed, common rush, and cattails. The methods for surveying the submerged plant community are based on the *Washington Department of Ecology Aquatic Plant Sampling Protocol* (Parsons

2001). Any common plants not identified at the site were collected and identified later with the help of plant identification guidebooks (Cooke and Azous 1997; Guard 1995; Hitchcock and Cronquist 1973) and a local plant expert (2002). Aquatic plant habitats were identified based on professional judgment during the summer 2002 amphibian and aquatic plant reconnaissance survey (Integral et al. 2004a). The surface water samples collected from within these aquatic plant exposure areas were used as EPCs for aquatic plants.

Aquatic Plants in the Study Area

Aquatic plants were identified during the summer 2002 reconnaissance survey (Integral et al. 2004a), which was conducted to confirm the presence of and delineate potential habitats for amphibians and aquatic plants within the Initial Study Area (RM 3.5 to RM 9.2). Current physical conditions within the Study Area minimize the available habitat for aquatic plants and thus limit the establishment of dense submerged and emergent aquatic plant communities along the riverbanks. Piers and other overbank structures create large shaded areas, extensive shoreline modifications (e.g., riprap, sheet pile) cover available soil, and ship traffic and strong currents regularly erode or otherwise disturb nearshore habitats. Although aquatic and terrestrial plants, including submerged vegetation, emergent herbaceous and woody plants, shrubs, and trees, were observed at most of the locations during the survey, the emergent or aquatic plant communities along the Study Area shoreline are generally dominated by disturbance-tolerant native species such as cattails and common rush that are expected to exist in the habitat of an industrial harbor. Additional details are presented in Appendix B2 of the Programmatic Work Plan (Integral et al. 2004a).

Uncertainty is associated with the aquatic plant habitat areas because they were identified only within the boundaries of the Initial Study Area (RM 3.5 to RM 9.2). Potential aquatic plant habitat areas outside of the Initial Study Area but within the Study Area (RM 1.9 to RM 3.5 and RM 9.2 to RM 11.8) were not identified. Thus, these areas could not be evaluated, and risks to aquatic plants in these areas are unknown.

All surface water data, including data from aquatic plant habitat areas, are presented in Attachment 4. General trends in surface water COPC concentrations from samples within these exposure areas are described in the amphibian exposure assessment (Section 9.1). Inasmuch as the amphibian and aquatic plants habitat areas are the same, the trends in surface water concentrations are not repeated here.

Surface water EPCs in this assessment are represented by detected concentrations in all individual surface water samples collected within quiescent areas or amphibian and aquatic plant habitat areas. Surface water EPCs are also represented by site-wide UCL concentrations calculated using all surface water sampling locations collected within amphibian and aquatic plant habitat areas.[154] These site-wide UCLs were calculated for all surface water COPCs, except ethylbenzene and trichloroethene; only one sample located within an amphibian and aquatic plant habitat area was analyzed for VOCs (between RM 5.5 and RM 6.5). COPC surface water concentration data for all individual samples and calculated site-wide UCL concentrations are presented in Attachment 4. At locations where both XAD and peristaltic samples were collected and analyzed for organic COPCs, the results of the peristaltic samples (i.e., the low-resolution results) were removed from the dataset used to derive UCLs. These data were

[154] The site-wide UCL concentration was limited to surface water samples located within the amphibian and aquatic plant habitat areas identified between RM 3.5 and RM 9.2.

removed because COPC concentrations from XAD samples are based on high-resolution analyses with lower DLs and greater accuracy.

Surface water EPCs based on a UCL were calculated using ProUCL Version 4.0 software (EPA 2007f). EPA's ProUCL software tests the goodness of fit for a given dataset and then computes the appropriate 95^{th} UCL (as described in Section 7.1.3.1).

Surface water concentrations were compared to water TRVs to characterize risks to aquatic plants via exposure to surface water. Surface water COPC concentration data for all individual samples are presented in Attachment 4.

Uncertainty Associated with Surface Water Sampling Methods

Surface water sampling methods varied across sampling events, creating some uncertainty about data comparability. Surface water samples were collected both as single-point samples and as transect samples (vertical, horizontal, or both) using two types of sampling methods (XAD and the peristaltic method). Samples were collected over seven sampling events; and not all surface water locations were sampled during each event. Surface water transect samples are representative of a wider range of concentrations because the technique collects more water over a greater area and longer period. Horizontal transects were sampled at only two locations within the Study Area within amphibian and aquatic plant habitat areas (at RM 2.0 and at the mouth of Multnomah Channel) and thus are limited spatially.

The evaluation of transect, single-point, XAD, and peristaltic samples provides a larger dataset for estimating aquatic plant surface water EPCs. The advantage of having more data at least partially offsets the disadvantage of adding unquantified uncertainty about data comparability across sampling methods.

10.1.4 Effects Assessment

Per agreement with EPA (2008f), chronic water TRVs were developed for all surface water COPCs based on the hierarchy detailed in Attachment 10. Section 6.5.4 and Table 6-32 in the benthic risk assessment present the water TRVs developed for all surface water COPCs. Water TRVs were developed based on effects to primarily fish and invertebrate species. Significant uncertainties are therefore associated with the use of these as TRVs for evaluating risks to aquatic plants.

Because the selected TRVs for total PCBs (0.014 µg/L) and 4,4′-DDT (0.001 µg/L) are based on the protection of mammals and birds, respectively, risk estimates based on these TRVs for aquatic receptors are uncertain. Alternative TRVs specifically protective of aquatic receptors were developed in this BERA using methods consistent with those used for AWQC derivation. The alternative water TRV for 4,4′-DDT was calculated as 0.011 µg/L. The alternative water TRV for total PCBs was calculated as 0.19 µg/L. Derivation of these alternative TRVs is described in Section 6.5.4 and Attachment 10. No plant data were included in the datasets used to derive either alternative TRV. For evaluating direct exposure of aquatic organisms (i.e., plants) to water, the alternative TRVs are considered more appropriate than the AWQC-based TRVs; the total PCBs AWQC is based on the protection of mink via ingestion of contaminated prey, and the 4,4′-DDT AWQC is based on the protection of brown pelican via ingestion of contaminated prey. Note that for total PCBs, a slightly lower chronic value than the alternative TRV is reported in the AWQC document (EPA 1980c) for plants (lowest diatom value is 0.1 µg/L). For 4,4′-DDT, the reported lowest plant value is higher than the alternative TRV (lowest algae value is 0.3 µg/L) (EPA 1980a).

Two sets of water HQs were derived for total PCBs, 4,4′-DDT, and total DDx: one using the alternative water TRVs and one using the selected AWQC-based TRVs. The alternative TRVs are more appropriate than the AWQC values for assessing risk to aquatic plants and therefore were used as the primary line of evidence to determine risk conclusions.

Surface Water TRV Uncertainties

Surface water TRVs are based on the hierarchy presented in Attachment 10 and are intended to be protective of 95% of aquatic organisms; however, these TRVs may overestimate effects to aquatic plants. The uncertainty regarding the use of water quality TRVs for evaluation of plants is due in part to the relative insensitivity of aquatic plants to most aquatic chemicals compared to fish and invertebrates (herbicides being an obvious exception). The water TRVs for all COPCs were established by the AWQC and Tier II criteria, which are based on or included toxicological data for invertebrates (primarily *Daphnia* sp.), fish, bird, and amphibian species.

Limited plant and algae toxicity data were identified for some COPCs. Algae toxicity data are commonly used as a surrogate for aquatic plant data (e.g., Suter and Tsao 1996). One study indicated that algae are more sensitive than plants to 80% of chemicals tested (Fletcher 1990; as cited in Hoffman et al. 1995), so algae data may be conservatively protective of plants. However, sensitivity of plants to toxicants appears to vary widely among species and chemicals (Hoffman et al. 1995); thus, use of algae toxicity data to assess plant community risks is uncertain. The TRV uncertainties are discussed on a chemical-specific basis in Section 10.1.5.

The water TRV for DDx compounds based on the 4,4′-DDT AWQC is derived from the effects data for brown pelican. The total PCBs AWQC is based on the protection of mink via ingestion of contaminated prey. Alternative water TRVs were developed for total PCBs and DDx following the methods used to develop AWQC values and are more appropriate for evaluating risk to aquatic organisms (i.e., plants) directly exposed to surface water. Although both the selected and alterative water TRVs were used to derive water HQs, the alternative TRVs were used to draw risk conclusions.

10.1.5 Risk Characterization and Uncertainty Analysis

This section presents the risk estimates for aquatic plants based on the surface water LOE. An HQ approach was used to quantify risk estimates (Equation 6-1) following the risk characterization process described in Section 10.1.1. HQs were derived for all COPCs using Equation 6-1 to quantify surface water risk estimates. The EPC and TRV are represented by surface water concentrations.

Section 10.1.5.1 presents the risk characterization results and uncertainty evaluation. Section 10.1.5.2 presents an evaluation of background concentrations. Section 10.1.5.3 presents COIs for which risks to aquatic plants cannot be quantified. Section 10.1.5.4 summarizes surface water COPCs with HQ ≥ 1.

10.1.5.1 Risk Characterization Results and Uncertainty Evaluation

This section presents the risk estimates for aquatic plants based on the surface water LOE. An HQ approach was used to quantify risk estimates following the risk characterization process described in Section 10.1.1. HQs calculated for all surface water COPCs within aquatic plant exposure areas are the same as those presented for amphibians (Section 9.1.5 and Table 9-2). Samples with HQs ≥ 1 are shown on Map 9-2. Table 9-3 presents the HQs calculated from site-wide UCLs using the surface water samples located within the aquatic plant habitat areas (the same as the amphibian habitat areas).

The following COPCs have HQ ≥ 1 for aquatic plants: zinc, benzo(a)anthracene, benzo(a)pyrene, naphthalene, BEHP and total DDx. Site-wide amphibian and aquatic plant habitat HQ ≥ 1 occur for one COPC (naphthalene).

A discussion of these COPCs with HQs ≥ 1,[155] including an evaluation of the key uncertainties and the frequency and location and HQs ≥ 1, is presented below. Key uncertainties associated with the appropriateness of using water TRVs for assessing risks to aquatic plants are also discussed.

Metals – Zinc HQs ≥ 1 occur in only one sample (HQ = 1.1) and the site-wide HQ is < 1. Uncertainty is associated with the TRV for zinc (36.5 µg/L) because it is based on risks to fish and aquatic invertebrates. Toxicity data for 20 species of aquatic plants or algae were included in the zinc water quality criteria document (EPA 1987), with LOAELs ranging from 30 to 200,000 µg/L. Algae were both the most and least sensitive species tested. Wang et al. (1997) also reported highly variable toxicity values (from 10 to > 100,000 µg/L) for various plant species (including algae) as measured in laboratory toxicity tests. The 7-day zinc LCV identified for algae by Suter and Tsao (1996) is 30 µg/L,[156] based on *Selenastrum capricornutum,* as reported by Bartlett (1974). Given this LCV, the zinc TRV based on fish and invertebrate data appears to be protective of plants; however, the TRV is uncertain because of the variability in plant sensitivity.

PAHs and BEHP –Benzo(a)anthracene, benzo(a)pyrene, naphthalene, and BEHP HQs ≥ 1 occur in less than 2% of samples. The site-wide HQs for benzo(a)pyrene, benozo(a)anthracene, and BEHP are < 1, and the site-wide HQ for naphthalene is 3.2. Uncertainty is associated with the PAH and BEHP TRVs because they are based on toxicity data for invertebrates, fish, or amphibians. Toxicity to plants from PAHs via exposure to water has been measured in the parts per million range for both aquatic plants (i.e., duckweed) and algae (Douben 2003). Algae-specific toxicity data are available for naphthalene. The selected naphthalene Tier II-based TRV (12 µg/L) is

[155] A discussion of total PCBs is also provided.

[156] The algae-specific toxicity threshold for zinc (30 µg/L) is based on a hardness of 14.9 mg/L $CaCO_{3,}$ and the water TRV for zinc (36.5 µg/L) is based on a hardness of 25 mg/L $CaCO_3$).

based on risks to fish and aquatic invertebrates. Algae-specific water toxicity data for naphthalene indicate that aquatic plants may be less sensitive; however, PAH phytotoxicity increases with exposure to the sun's UV radiation. The naphthalene algae LCV presented in Suter and Tsao (1996) as reported by EPA (1980b) is 33,000 µg/L based on a *Chlorella vulgaris* 48-hour EC50, in which cell growth was inhibited. Aquatic plant-specific or algae-specific toxicity data are not available for benzo(a)anthracene, benzo(a)pyrene, and BEHP.

Total PCBs – Total PCB concentrations exceed AWQC in two samples from different locations (RM 3.7 and RM 6.7) in the Study Area, with a maximum HQ of 1.2. Both samples were collected during low-flow events. There is a high degree of uncertainty associated with the PCB AWQC-based TRV (0.014 µg /L) because it is based on risk to mink. With the alternative total PCB TRV of 0.19 µg/L, no samples exceed the TRV. The site-wide HQs using both the selected TRV and alternative TRV are < 1. Limited plant-specific PCB toxicological data are available. The AWQC document does not include any appropriate data for freshwater aquatic plants but does identify adverse effects in saltwater diatoms; the effects levels range from 0.1 µg/L for reduced diatom growth and species composition change to 100 µg/L for reduced diatom growth (EPA 1980c). The LCV reported for algae in Suter and Tsao (1996) is 0.144 µg/L, derived from a 24-hour study in which carbon fixation in *Scenedesmus quadricaudata* was reduced. The PCB water TRV is lower than the algae-specific threshold, indicating that risk to aquatic plants from PCBs in water may be overestimated. High PCB concentrations in algae and duckweed have been reported to affect photosynthesis and the viability of chloroplast structures; however, there is little evidence of PCB-induced effects in the cholorplasts of higher-level plants (Wang et al. 1997). High PCB concentrations in soil have been reported to affect growth in some terrestrial agricultural plants (Weber and Mrozek 1979). These limited data suggest that plants are likely less sensitive to PCBs than other ecological receptors.

DDx – Total DDx in only one sample (W001, RM 2.0) exceeds the TRV (HQ based on alternative TRV = 1.8). However, this result uses N-qualified data collected via peristaltic pump. The N-qualification indicates that the elevated concentration was likely due to analytical interference from a different chemical. Total DDx HQs based on non-N-qualified data (n = 20) range from 0.003 to 0.9. The peristaltic samples (i.e., the low-resolution results) have lesser accuracy than the XAD samples. The site-wide total DDx HQ using the alternative TRV is 0.17.

Toxicity data for four species of algae were included in the DDT quality criteria document (EPA 1980a), with LOAELs ranging from 0.3 to 800 µg/L. The lowest LOAEL (0.3 µg/L) was reported for growth and morphology effects in *Chlorella* sp. Wang et al. (1997) report phytotoxicity at much higher DDT concentrations (> 100,000 µg/L). The AWQC and alternative total DDx water TRVs are substantially lower than the reported algae-specific and phytotoxicity thresholds, indicating that risk to aquatic plants from DDx compounds in water is likely to be substantially overestimated.

10.1.5.2 Evaluation of Background Concentrations

By agreement with EPA, aluminum was not identified as a COPC because its AWQC was developed using toxicity data from acidic waters and is not applicable to the circumneutral waters of the Study Area. Background aluminum concentrations were established as part of the RI (see Section 7.0 of the draft final RI (Integral et al. 2011)). Background concentrations in sediment and surface water are compared in Attachment 11 and discussed in Section 6.5.5.3. Aluminum concentrations in sediment and surface water for the Study Area are generally similar to or below the background UCL and UPL. Aluminum and other trace elements are major constituents of the mineral fraction of sediment but contribute to the analytical chemical results because of the acid extraction step during analysis. Because aluminum is not biologically available or not toxic at naturally occurring concentrations generally found in surface water, aluminum is not expected to pose unacceptable risk to aquatic plants.

Zinc is also a naturally occurring crustal element in the environment. A background water concentration could not be established because the number of data points was limited (see Attachment 11). The Study Area UCL concentration of zinc (2.5 µg/L) is greater than the range of zinc concentrations detected in background[157] (1.4 to 2.2 µg/L). The study area UCL sediment zinc concentration (164 mg/kg dw) is greater than the background sediment UCL and UPL (79 and 110 mg/kg dw, respectively). These data indicate that zinc concentrations are elevated above background.

10.1.5.3 COIs for Which Risks Cannot Be Quantified

COIs for which risks to aquatic plants cannot be quantified based on surface water data are the same as those for benthic invertebrates listed in Table 6-35. These COIs are chemicals for which no TRV is available and include the following: aluminum, 4-chloroaniline, aniline, 2,4-DB, MCPP, and individual dioxin and furan congeners other than 2,3,7,8-TCDD.

10.1.5. Summary of Surface Water LOE

Six surface water COPCs with with HQs ≥ 1 were identified for aquatic plants: zinc, benzo(a)anthracene, benzo(a)pyrene, naphthalene, BEHP, and total DDx.

10.2 TZW ASSESSMENT

EPA's Problem Formulation (Attachment 2) calls for analysis of TZW[158] data relative to surface water effects thresholds for aquatic plants. TZW was evaluated in detail in Section 6.6 as an LOE for benthic invertebrates. Sections 7.4 and 9.2 evaluate these same data as LOEs for fish and amphibians, respectively.

[157] Zinc concentrations were detected in only 3 of 22 surface water samples included in the background dataset (see Section 7.0 of the draft RI).

[158] For the purpose of the BERA, TZW is the porewater associated with sediment matrix within the top 38 cm of the sediment column. TZW is composed of some percentage of both groundwater and surface water.

The TZW samples evaluated in this BERA were collected primarily during a 2005 sampling effort focusing on areas offshore of nine upland sites with known or likely pathways for discharge of contaminated upland groundwater to the Study Area. Sampling locations were selected at each of the nine study sites based on results of a groundwater discharge mapping field effort. The RI Appendix C2 presents the process used to select these sites per agreement with EPA, ODEQ, and LWG. The findings of the discharge mapping effort were considered in conjunction with relevant site data (e.g., hydrogeology, surface sediment texture delineation, distribution of COIs in upland groundwater and sediments) to identify zones of possible contaminated groundwater discharge. The TZW sampling locations selected for each site focus on the zones of possible groundwater plume discharge, as determined from the GWPA discharge mapping effort. Additional sampling locations were specified to provide comparative data for TZW quality outside of the potential discharge zones (Integral et al. 2011).

Because the primary objective of RI groundwater pathway assessment was to evaluate whether transport pathways from upland contaminated groundwater plumes to the river were complete, TZW target analyte lists varied from site to site and were derived primarily based on the COIs in the upland contaminated groundwater plumes. Therefore, not all COIs in sediments were analyzed in TZW samples. As described in Sections 4.4.3.1 and 6.1.5.2 of the draft final RI (Integral et al. 2011), there also may be other groundwater plumes in the Study Area that may be discharging into river sediments where TZW samples have not been collected.

The remainder of this section is organized as follows, as shown in Figure 10-2:

- Section 10.2.1 describes the general approach used to assess risks to aquatic plants from surface water.
- Section 10.2.2 summarizes the TZW COPCs evaluated. Some COPCs were not evaluated because no toxicity thresholds were available.
- Section 10.2.3 explains how exposure concentrations were estimated and describes uncertainties in those estimates. All TZW chemical concentrations are presented in Attachment 4.
- Section 10.2.4 summarizes the effects data. Details on the development of the water TRVs are presented in Attachment 10.
- Section 10.2.5 presents the risk characterization results and associated uncertainties.

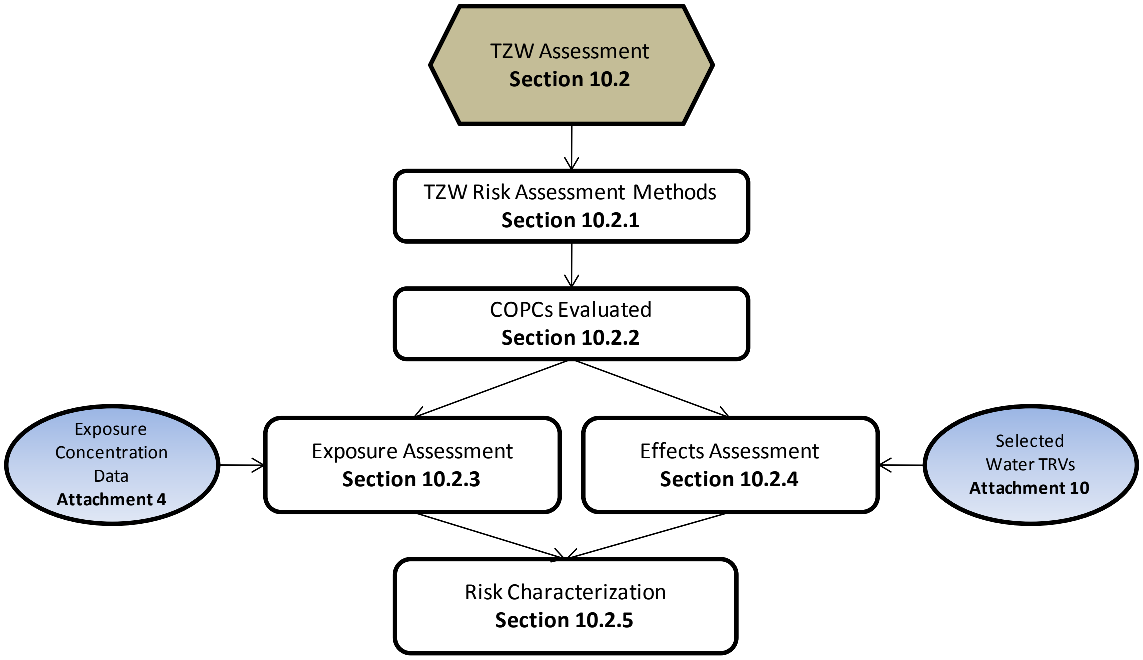


Figure 10-2. Overview of Aquatic Plant TZW Section Organization

10.2.1 TZW Risk Assessment Methods

As described in Section 6.6.1, TZW HQs were calculated by comparing COPC concentrations in individual TZW samples to chronic water TRVs. These TRVs were developed according to a hierarchy of water quality criteria and literature-based TRVs articulated in Attachment 10.

Baseline risk to aquatic plants was evaluated only for those areas of the Study Area that have potentially suitable aquatic plant habitat. COPCs identified in the SLERA are based on all TZW samples; by contrast, surface water samples for the present baseline risk characterization are restricted to those collected from areas offering potentially suitable aquatic plant habitat (Map 9-1). The comparison of surface water concentrations to water TRVs was conducted on an individual sample basis. Potentially unacceptable risks were identified by COPCs with HQs ≥ 1. Exposure data, effects data, and the quantitative risk results (i.e., magnitude, spatial distribution, and frequency of HQs) are discussed in the following sections. The relative strengths and uncertainties for both aquatic plant LOEs are evaluated together in the risk conclusions for aquatic plants (Section 10.3).

10.2.2 COPCs Evaluated

Fifty-four of the 58 TZW COPCs identified in the SLERA and refined screen (Attachment 5) are evaluated in the BERA. Four individual DDT metabolites identified in the SLERA (2,4′-DDD, 2,4′-DDT, 4,4′-DDD, and 4,4′-DDE) were evaluated as total DDx and not individually; 4,4′-DDT was evaluated both individually and as total DDx because the DDx TRV is based on 4,4′-DDT. Table 6-37 lists the detected TZW COPCs by site.

Fourteen TZW COIs could not be evaluated because no toxicological data are available to allow development of water TRVs (Table 6-38). The risks to aquatic plant receptors associated with exposure to these chemicals in TZW are therefore unknown. TRVs are unavailable for individual dioxins and furans other than 2,3,7,8-TCDD. For fish and wildlife, dioxins and furans are evaluated as a toxicity-weighted sum based on the toxicity of each congener relative to 2,3,7,8-TCDD, using TEFs based on their common mechanism for toxicity. Because TEFs are not available for aquatic plants, no individual dioxin or furan (other than 2,3,7,8-TCDD) or dioxin group total could be evaluated.

By agreement with EPA, aluminum was not identified as a COPC because its AWQC was developed using toxicity data from acidic waters and is not applicable to the circumneutral waters of the Study Area. Aluminum concentrations in background surface water and sediment were evaluated to determine whether a local source of aluminum contamination is present within the Study Area (Section 6.5.5.3).

In addition, two TZW COIs were not retained as COPCs because no detected concentrations exceeded TRVs (although at least one DL exceeded a TRV): selenium and styrene. Their potential risks to aquatic plants based on TZW data are unknown (see Table 5-11).

10.2.3 Exposure Assessment

This section presents the TZW exposure concentrations used to assess potential risks to aquatic plants. Section 10.2.3.1 discusses exposure areas. Section 10.2.3.2 describes how EPCs were derived. Section 10.2.3.3 summarizes uncertainty associated with TZW exposure. An overview of the sampling methods for all LWG- and non-LWG-collected TZW data used in this assessment is presented in the benthic risk assessment (Section 6.6.3.1). The RI Appendix C2 presents the process used to select these sites per agreement with EPA, ODEQ, and LWG. TZW sampling locations used in this assessment are shown on Map 9-1.

10.2.3.1 Exposure Areas

For aquatic plants, only TZW samples collected within aquatic plant habitat areas or quiescent areas were used to represent EPCs (Map 9-1). The same exposure areas used for the surface water LOE (as described in Section 10.1.3) were used for the TZW LOE. Of the 192 TZW sample locations, 105 were excluded because they are outside quiescent areas that offer potential aquatic plant habitat (see Attachment 4, Part B).

10.2.3.2 TZW EPCs

TZW EPCs in this assessment are represented by TZW concentrations in all individual TZW samples collected in the amphibian and aquatic plant exposure areas regardless of sampling method or depth sampled.[159] TZW concentrations were compared to surface water TRVs to characterize risks to aquatic plants via exposure to TZW. A summary of the chemicals detected in shallow TZW from throughout the Study Area and the range of concentrations is presented in the benthic risk assessment (Section 6.6.3.2). All TZW data, by site, are presented in Attachment 4.

10.2.3.3 Uncertainty Associated with Ecological Exposure to TZW

The degree to which the collected TZW samples are representative of exposure of organisms living in or on the sediment to TZW is a key uncertainty associated with the ecological evaluation of TZW. Because the primary objective of RI groundwater pathway assessment was to evaluate whether transport pathways from upland contaminated groundwater plumes to the river were complete, TZW target analyte lists varied from site to site and were derived primarily based on COIs in the upland contaminated groundwater plumes. Therefore, not all COIs in sediments were analyzed in TZW samples. As described in Sections 4.4.3.1 and 6.1.5.2 of the draft final RI (Integral et al. 2011), there also may be other groundwater plumes in the Study Area that may be discharging into river sediments where TZW samples have not been collected.

Section 6.6.3.3 discusses the uncertainties associated with the benthic invertebrate exposure data for the TZW assessment. In short, in this focused study, TZW samples were collected from areas offshore of nine upland sites with known or likely pathways for discharge of contaminated upland groundwater to the Study Area. Because these areas include potential habitat for benthic invertebrates, benthic fish, amphibians, and aquatic plants, TZW is considered a complete and significant pathway for these receptors.

Aquatic vascular plants may be directly exposed to TZW through their roots; however, aquatic vegetation habitat tends be present only outside areas of known or likely pathways for discharge of contaminated upland groundwater. Of the nine site locations where TZW samples were collected, Gunderson and Rhône-Poulenc were the only two where aquatic plants were mapped within 500 ft of a TZW sample (Map 9-1).

Furthermore, the aquatic plants found in the Study Area are restricted to water shallower than the depths at which TZW samples were collected (generally > 6 ft deep), thus further limiting exposure to TZW. According to the aquatic plant survey(Integral et al. 2004a), no submersed plants were found offshore in waters 2.4 to 3 m deep (depth surveyed); however, a few submersed plants were identified close to the waterline near shore in shallow water (generally shallower than the areas sampled for TZW).

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[159] All TZW samples evaluated in this BERA were within the 0 – 38 cm depth; however, the exact depth within the 0 – 38 cm horizon varied because of the different sampling equipment used to collect TZW (i.e.,peeper, Trident® probe, and Geoprobe).

Of the plant species identified in the LWR (Table 2-9), the obligate species are the most likely to be submerged in the river or in the riparian zone (Plant Conservation Alliance 2009; USDA FS 2011, 1990; Ecology 2011a, b; Clemson University 2011; USFWS 2011; WNHP 1999). Of the obligate species identified in Table 2-9, howellia (*Howellia aquatilis)* and wapato (*Sagittaria latifolia)* can live at water depths between 1 and 2 m (USFWS 2011; Ecology 2011b). Other obligate species typically live in shallower waters (Wisconsin DNR 2004; USDA FS 2011). Therefore, inclusion of all samples from throughout the quiescent areas likely overestimates the extent of potential exposure.

Aquatic plants were identified only within the boundaries of the Initial Study Area (RM 3.5 to RM 9.2). All TZW samples were also collected within the same boundaries.

10.2.4 Effects Assessment

TZW chemical concentrations were compared to the effects thresholds as part of the risk characterization process. At the direction of EPA (2008f), chronic water TRVs were developed for all TZW COPCs according to the hierarchy detailed in Attachment 10. Chronic water TRVs were developed through a review of WQS, criteria, published benchmarks, and toxicity data. The TRVs selected were approved by EPA Criteria for metals COPCs were hardness-adjusted when appropriate. If the published criteria for individual metals were based on dissolved concentrations, then the dissolved sample result was compared to the dissolved criterion; otherwise the total concentrations for both the sample and criterion were used. Table 6-40 presents the TRVs for all TZW COPCs and their sources. These values were developed based on the sensitivities of fish and invertebrate species and are considered protective of all aquatic receptors, including aquatic plants. In general, uncertainty regarding the use of water quality TRVs for evaluation of plants is likely to overestimate risk to plants because of their relative insensitivity compared with that of fish and invertebrates (herbicides being an obvious exception).

As noted in Section 10.1.4, because the selected AWQC for 4,4′-DDT is based on protection of birds via ingestion of contaminated prey, risk estimates for aquatic receptors based on this TRV are associated with substantial uncertainty. An alternative TRV protective of aquatic organisms was developed in this BERA using methods consistent with those used for AWQC derivation. The alternative water TRV for DDx compounds was calculated as 0.011 µg/L. This alternative TRV is the appropriate metric for evaluating direct exposure of aquatic organisms to water because the AWQC (0.0010 µg/L) is based on the protection of brown pelican via ingestion of contaminated prey and cannot be used meaningfully to judge risk to aquatic organisms. Nevertheless, two sets of water HQs were derived: one using the alternative water TRV (0.011 µg/L) and one using the AWQC DDx water TRV (0.001µg/L). Because the alternative TRVs are more appropriate than the AWQC values for assessing risk to aquatic plants, they therefore were used as the primary line of evidence to determine risk conclusions. Other uncertainties associated with effects data are presented in Section 6.5.4.

10.2.5 Risk Characterization

This section presents the risk estimates for aquatic plants based on the TZW LOE. It also includes an evaluation of naturally occurring metals and a list of COIs that could not be evaluated.

10.2.5.1 Risk Characterization Results

Individual HQs were calculated for all COPCs across all TZW samples within aquatic plant exposure areas. These areas are the same as amphibian exposure areas and the frequencies with which individual samples have HQs ≥ 1 are shown in Table 9-4.

The uncertainties associated with the TZW data as representive exposure data for aquatic plants is dicussed in Section 10.2.3.3. Actual exposure of aquatic plants to TZW is likely several orders of magnitude lower than represented by TZW COPC concentrations because aquatic plants in the Study Area likely have little or no contact with TZW. As discussed in Section 10.2.3.3, the assumption that plants are exposed throughout quiescent areas likely overestimates the extent of potential exposure. A more realistic estimate is that plants in water less than 6 ft deep may be exposed in the vicinity of the Gunderson or Rhône-Poulenc locations because emergent aquatic plants are not likely to exist in deeper water and no plants were identified within 500 ft of other TZW sampling locations.

As an uncertainty analysis, HQs were calculated using TZW samples collected within 500 ft of Gunderson and Rhône-Poulenc facilities in locations with a water depth of 6 ft or less. Water depths for Trident samples were determined based on the TZW field sampling report (Integral 2006a). River water depths were not reported for peeper samples in the field sampling report; a bathymetric layer (DEA 2003) was used to determine the water depth for these samples. This method also identified additional Trident samples that were possibly in less than 6 ft of water. These samples were also included in the uncertainty evaluation to be conservative. The frequencies with which HQs are ≥ 1 in individual samples from the Rhône-Poulenc and Gunderson sites in a water depth of 6 ft or less are shown in Table 10-1.

Table 10-1. Summary of TZW Samples with HQs ≥1 Collected Near Documented Aquatic Plant Areas

	Samples with HQs ≥ 1		
Analyte	**Number**	**Percentage**	**Range of HQs**
Gunderson			
Barium (total)	5 of 5	100%	1 – 40
Chloroethane	1 of 5	20%	3.4
Iron (total)	5 of 5	100%	1.9 – 57
Manganese (total)	5 of 5	100%	1.1 – 33
Rhône-Poulenc			
Barium (total)	1 of 1	100%	85

Table 10-1. Summary of TZW Samples with HQs ≥1 Collected Near Documented Aquatic Plant Areas

Analyte	Samples with HQs ≥ 1: Number	Samples with HQs ≥ 1: Percentage	Range of HQs
Iron (total)	1 of 1	100%	52
Manganese (total)	1 of 1	100%	7.9

Note: The sampling location IDs for Gunderson are GN03APR, GN03ATR, GN04BPR, and GN05ATR. The sample location ID for Rhône-Poulenc is R2RP02TR.

HQ – hazard quotient

ID – identification

TZW – transition zone water

10.2.5.2 Evaluation of Naturally Occurring Metals

Although there are many anthropogenic sources of metals, most of the metals measured in TZW are also common crustal elements in sediment. Among the most common metals associated with sediments, barium, iron, and manganese were detected in all TZW samples. These common metals are also associated with the highest HQs identified in the risk charactrization; there is substantial uncertainty that the source of these is anthropogenic. Sodium, potassium, and magnesium are also naturally present in water, but some industrial processes may elevate their concentrations in surface water.

The contribution of geochemical processes in sediments to the concentrations of selected metals in TZW was extensively evaluated in Appendix C2 of the draft final RI (Integral et al. 2011). Concentrations of iron and manganese in TZW are not well-correlated to potential anthropogenic source materials (i.e., petroleum hydrocarbons), suggesting that factors other than contamination in the sediment (e.g., naturally occurring organic materials) are contributing to concentrations measured in the TZW. Geochemical processes are likely responsible for some percentage of the measured concentrations of barium in TZW, in addition to the contribution from migration of upland groundwater to the river.

By agreement with EPA, aluminum was not identified as a water COPC because its AWQC was developed using toxicity data from acidic waters and is not applicable to the circumneutral waters of the Study Area. Aluminum was not included in the RI geochemical evaluation, but a background surface water concentration (established in Section 7.0 of the draft final RI (Integral et al. 2011)) is available to provide some context for TZW (since surface water is a component of shallow TZW). An upper-bound (UPL) background concentration for aluminum is 1,485 µg/L. The majority of the TZW concentrations are below this concentration.

10.2.5.3 COIs for Which Risks Cannot Be Quantified

Risks to aquatic plants cannot be quantified for 16 TZW COIs (aluminum, calcium, titanium, selenium, styrene, seven dioxin/furan congeners other than 2,3,7,8-TCDD, and four TPH [see Table 6-42]). These are chemicals for which no TRV is available or whose maximum DL exceeds a TRV, but whose detected concentrations do not.

10.2.5.4 Summary of TZW Risk Evaluation

Thirty-two TZW COPCs with HQs ≥ 1 were identified for aquatic plants (Table 9-4). [160] The relative strengths and uncertainties for both aquatic plant LOEs are evaluated together in the risk conclusions for aquatic plants, Section 10.3.

10.3 RISK CONCLUSIONS

This section summarizes conclusions of the aquatic plant risk assessment. Aquatic plant COPCs with HQs ≥ 1 in individual surface water or TZW samples were identified as contaminants posing potentially unacceptable risk to aquatic plants. Risk conclusions incorporate the magnitude of HQs, spatial distribution and frequency of HQs ≥ 1, and the uncertainty of exposure and effects assumptions. Thirty-three COPCs[161] were identified as posing potentially unacceptable risk to aquatic plants based on the surface water and TZW LOEs: 5 in both LOEs, 1 (BEHP) in the surface water LOE only, and 27 in the TZW LOE only. Risk conclusions for the aquatic plant community are summarized in Table 10-2. Risk conclusions associated with the COPCs having HQs ≥ 1 only for the TZW LOE are not presented in Table 10-2. Risks to plants from TZW are associated with high uncertainty due to uncertainty in both the exposure and effects data. TZW risks by area are discussed in greater detail in the benthic invertebrates risk conclusions section (Section 6.7.3). Risk to aquatic plants from 27 COIs, including 19 surface water COIs and 16 TZW COIs, could not be evaluated because no screening-level TRVs are available or because DLs exceed the TRV (Tables 5-11 and 5-12; Table 6-35 for surface water COIs, and Table 6-42 for TZW COIs).

[160] Thirty-three TZW COPCs with HQs ≥ 1 were identified for aquatic plants. Because petroleum compounds are not CERCLA contaminants, gasoline-range hydrocarbons are not included in the final count of contaminants posing potentially unacceptable risk; they are mentioned because they may nonetheless contribute to risk.

Table 10-2. Summary of Aquatic Plants Surface Water and TZW COPCs

COPC	Max HQ by LOE[a] Surface Water	TZW	Conclusion	Rationale
Metals				
Zinc	1.1	14	Negligible risk	Max surface water and TZW HQs are not indicative of ecologically significant risk. Risk is of limited spatial extent and all HQs are low (≤ 1.1 for surface water and ≤ 14 for TZW). HQ is ≥ 1 in only 1 of 117 surface water samples (collected in November during a non-reproductive period). HQ is ≥ 1 in only 1 of 30 TZW samples. Risk may be underestimated because the TRV is based on toxicity to fish and invertebrates whereas algae data indicate that aquatic plants toxicity to zinc is highly variable. TZW risk likely overestimated because HQs < 1 near documented aquatic plant locations.
PAHs				
Benzo(a)anthracene	10	8.5	Negligible risk	Max surface water and TZW HQs are not indicative of ecologically significant risk. Risk is of limited spatial extent and all HQs are low (≤ 10 for surface water and ≤ 8.5 for TZW). Surface water HQ ≥ 1 in only 2 of 158 samples collected during non reproductive periods (July and winter). TRV is based on extrapolated *Daphnia* acute LC50; plant-specific toxicity data were not available but data for other PAHs indicate plants likely less sensitive. TZW risk likely overestimated because HQs < 1 near documented aquatic plant locations.
Benzo(a)pyrene	14	15	Negligible risk	Max surface water and TZW HQs are not indicative of ecologically significant risk. Risk is of limited spatial extent. Surface water HQ ≥ 1 in only 3 of 158 samples collected during non reproductive periods (July, November, and winter). TRV based on extrapolated *Daphnia* acute LC50; plant-specific toxicity data are not available but data for other PAHs indicate plants likely less sensitive. TZW risk likely overestimated because HQs < 1 near documented aquatic plant locations.
Naphthalene	50	57	Negligible risk	Max surface water and TZW HQs are not indicative of ecologically significant risk. Risk is of limited spatial extent. Surface water HQ ≥ 1 in only 1 of 159 samples collected during reproductive period (May). TRV based on risk to fish and invertebrates; plant-specific toxicity data were not available but algae data indicate plants are less sensitive (all surface water concentrations are less than algae toxicity thresholds). TZW HQ ≥ 1 in 5 of 72 samples. TZW risk likely overestimated because HQs < 1 near documented aquatic plant locations.

Table 10-2. Summary of Aquatic Plants Surface Water and TZW COPCs

COPC	Max HQ by LOE[a] Surface Water	TZW	Conclusion	Rationale
Phthalates				
BEHP	1.2	NA	Negligible risk	Max surface water HQ is not indicative of ecologically significant risk. Risk is of limited spatial extent and all HQs are low (≤ 1.2 for surface water). HQ ≥ 1 in only 1 of 129 samples. TRV based on risk to fish, invertebrates, and amphibians; no plant-specific data are available. TZW risk likely overestimated because HQs < 1 near documented aquatic plant locations.
Pesticides				
4,4′-DDT	0.35	160	Negligible risk	Max TZW HQ is not indicative of ecologically significant risk. Risk is of limited spatial extent and HQs are generally low (≤ 0.35 for surface water and < 100 for TZW in all but two samples). TZW HQ ≥ 1 in only 3 of 12 samples. TRV based on risk to fish, invertebrates, and amphibians; algae toxicity data indicate plants are less sensitive (all surface water concentrations are less than algae toxicity thresholds). TZW risk likely overestimated because HQs < 1 near documented aquatic plant locations. Furthermore, maximum TZW risk is based on unfiltered samples. The maximum HQ for filtered samples would be 2.8 for 4,4′-DDT, which was not detected.
Total DDx[b]	1.8	280	Negligible risk	Max surface water and TZW HQs are not indicative of ecologically significant risk. Risk is of limited spatial extent and HQs are generally low (≤ 1.8 for surface water and < 100 for TZW in all but four samples). Surface water HQ ≥ 1 in only 1 of 121 samples (sample N-qualified, indicating interference from non-DDx chemical). TZW HQ ≥ 1 in 8 of 12 samples and > 100 in only 4 of 12 samples, all from Arkema acid plant area; TRV based on risk to fish and invertebrates; available toxicity data indicate plants are less sensitive. TZW risk likely overestimated because HQs < 1 near documented aquatic plant locations. Furthermore, maximum TZW risk is based on unfiltered samples. The maximum HQ for filtered samples would be 14.5 for total DDx.

Note: This table attempts to summarize the BERA's aquatic plant risk estimates and risk descriptions, the two major components of the risk characterization. Balancing and interpreting the different types of data evaluated in the BERA can be a major task requiring professional judgment. It can be difficult to prepare a concise summary of conclusions without losing important context, yet a concise summary is needed to help the risk manager judge the likelihood and ecological significance of the estimated risks (EPA 1997).

All the COPCs listed in this table have an HQ ≥ 1 in at least one LOE for at least one ecological receptor, and by definition pose potentially unacceptable risk. The likelihood and ecological significance of the potentially unacceptable risk may vary, though, from very low to very high. Therefore, the risk description may range from negligible to significant. For each receptor-COPC pair with a maximum HQ ≥ 1,this table provides maximum HQ by LOE, a synoptic risk description, and a very brief rationale for the risk description. This distillation of the body of knowledge presented in the BERA should not be taken out of context.

[a] HQs are shown only for water samples that exceed chronic TRVs. HQs < 1 in all other water samples.

[b] HQ shown is based on the alternative 4,4′-DDT TRV for protection of directly exposed aquatic organisms. Surface water HQ ≥ 1 for 4,4′-DDT and total DDx (max HQ = 3.9 and 20, respectively) when calculated using the AWQC-based TRV (for protection of brown pelican via ingestion of contaminated prey). TZW HQ ≥ 1 for 4,4′-DDT and total DDx at one location (Arkema acid plant area; max HQ = 1,800 and 3,100, respectively) when calculated using the AWQC-based TRV. The alternative TRV is considered more appropriate for evaluating direct exposure of aquatic organisms.

AWQC – ambient water quality criteria
BEHP – bis(2-ethylhexyl) phthalate
COPC – contaminant of potential concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
HQ – hazard quotient
LC50 – concentration that is lethal to 50% of an exposed population
PAH – polycyclic aromatic hydrocarbon
total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT, and 4,4′-DDT)
TRV – toxicity reference value
TZW – transition zone water

In general, surface water concentrations are below algae-specific thresholds or HQs ≥ 1 occur at low frequency and with low magnitude of exceedance, indicating negligible risks to the aquatic plant community. There is a high degree of uncertainty concerning the relevance of the selected TRVs for zinc, benzo(a)anthracene, benzo(a)pyrene, and BEHP to aquatic plants because either algae-specific thresholds are highly variable or no aquatic plant or algae-specific thresholds are available for comparison. In general, the use of water quality TRVs for evaluation of plants is likely to overestimate risk because plants are relatively insensitive compared to fish and invertebrates (herbicides being an obvious exception).

With notable exceptions, TZW HQs are low. Because the distribution of aquatic plants in the Study Area is highly restricted, HQs were evaluated only in the vicinity of areas with documented aquatic plants. In these areas, chemicals with HQs ≥ 1 include only the naturally occurring metals barium, iron, and manganese and the VOC chloroethane. Exposure to naturally occurring metals is similar to background and any risks are not likely due to anthropogenic sources. The single chloroethane HQ ≥ 1 is of low magnitude (3.3) and is not likely to pose ecologically significant risk.

In Section 11, results of the aquatic plant assessment are integrated with those of other ecological receptors to reach overall ecological risk conclusions.

11.0 ECOLOGICAL RISK CONCLUSIONS

Risk estimates in this BERA were calculated following CERCLA guidance (EPA 1997, 1998) and EPA's Problem Formulation (Attachment 2). The conclusions of the BERA, along with those of the BHHRA (Appendix F of the draft final RI (Integral et al. 2011)), are intended to provide information to risk managers on potentially unacceptable risks predicted under current conditions of the Study Area, as well as information on possible future approaches for protecting human health and the environment.

The BERA's risk conclusions are provided at the end of the risk assessment for each receptor group:

- In Section 6.7 for benthic community assessment endpoints
- In Section 7.6 for fish assessment endpoints
- In Section 8.3 for avian and mammalian assessment endpoints
- In Section 9.3for the amphibian assessment endpoint
- In Section 10.3 for the aquatic plant assessment endpoint

Consistent with ERAGs (EPA 1997) the foregoing risk conclusions identified the receptor-COPC pairs that, given the magnitude and extent of risk, are reasonably likely to result in adverse effects on the assessment endpoints selected to represent the valued ecological attributes of the Study Area. Section 11 does not recapitulate the analyses that went into drawing the risk conclusions. For that level of detail the reader is referred back to the aforementioned risk conclusion sections. The remainder of Section 11 is organized as follows:

- Section 11.1 presents a summary by receptor group and LOE of the 89[162] ecological COPCs identified as posing potentially unacceptable risk in this BERA based on HQ ≥ 1 for at least one receptor-LOE combination.
- Section 11.2 identifies COPCs identified as posing potentially unacceptable risks for ecological receptors in the Study Area that occur at concentrations similar to the sediment and surface water background levels defined in Section 7.0 of the draft final RI (Integral et al. 2011) or to tissue concentrations in four fish receptor species (i.e., juvenile Chinook salmon, brown bullhead, smallmouth bass, and lamprey ammocoetes) collected from the upriver reach of the Willamette River (RM 15.3 to RM 28.4).
- Section 11.3 combines the risk conclusions across all ecological receptor groups to provide a general overview of ecological risks and to identify the

[162] Ninety-one contaminants have HQs ≥ 1. Because petroleum compounds are not CERCLA contaminants, gasoline-range hydrocarbons and diesel-range hydrocarbons have been excluded from the final count even though they may be contributing to potentially unacceptable risk.

receptor-COPC pairs that, given the magnitude and extent of risk, are reasonably likely to result in adverse effects on the assessment endpoints.

Risk management recommendations from the LWG risk assessors to EPA risk managers, based on the results of the BERA, are presented in Section 12.

11.1 SUMMARY OF POTENTIALLY UNACCEPTABLE RISKS

Consistent with EPA Superfund ERA guidance (EPA 1997, 1998), potentially unacceptable risks were identified through an iterative process of analyzing the exposure and effects data for the various chemicals and ecological receptors, with increasing realism at each step in the process. For most receptors, several LOEs were evaluated (Section 3.3). For each LOE, risk characterization began with the SLERA (Section 5) and progressed iteratively through the final step in the risk characterization. Throughout the process, chemical-receptor pairs that showed the potential for adverse effects were further analyzed and those that did not were screened out. The final step in the process reflects the most realistic risk estimates. Potentially unacceptable risks were identified for each receptor-LOE-COPC combination based on the final step in the risk characterization.

Exposure data in the final step of the risk analysis were evaluated at the scale over which the receptors are likely to be exposed and, where pertinent, the variety of potentially contaminated prey the receptor may consume. For the least mobile receptors (e.g., benthic macroinvertebrates, sculpin, aquatic plants), exposure areas are no larger than the immediate area where samples were collected; for the most mobile receptors (e.g., white sturgeon, largescale sucker), the exposure areas encompass the entire Study Area. For moderately mobile receptors (e.g., smallmouth bass, mink) the Study Area is divided into several exposure areas each 1 to 3 miles long.

For all LOEs except sediment, numerical risk estimates were calculated as HQs (Equation 6-1). HQs were calculated separately for each receptor-LOE-COPC combination for each exposure area. Receptor-LOE-COPC combinations resulting in HQ $\geq$ 1 in the final step of the risk characterization in any exposure area were identified as posing potentially unacceptable risk. For the sediment LOE, a location was identified as posing potentially unacceptable risk to benthic invertebrates if the sediment was toxic or predicted to be toxic based on a sediment COPC concentration that exceeded a site-specific SQV.

Those chemicals for which exposure or effects data were insufficient to evaluate the risk were also identified as posing potentially unacceptable risk, although risk is unknown. Risk to benthic organisms, including clams and crayfish, could not be evaluated for 78 sediment COIs because either no relationship between sediment contaminants and toxicity was apparent in the site-specific dataset or too few data points were available to discern a relationship (Table 6-6 summarizes the selection of chemicals for evaluation of site-specifc toxicity). Other contaminants that could not be evaluated for their contribution to benthic community risks include 27 tissue COIs (Table 6-28), 19 surface water COIs (Table 6-35), and 16 TZW COIs (Table 6-42). Risk to fish from a number of

COIs could not be evaluated: 17 tissue-residue COIs (Table 7-13), 11 dietary COIs (Table 7-16), 5 surface water COIs (Table 7-40), and 9 TZW COIs (Table 7-43). Risk to birds and mammals from dietary exposure to 19 COIs could not be evaluated (Table 8-30). Risk to amphibians and aquatic plants from 27 COIs (including 19 surface water COIs and 16 TZW COIs) could not be evaluated (Tables 5-11, 5-12, and 6-35 for surface water; Table 6-42 for TZW). As per agreement with EPA (LWG 2010), these COIs are identified as chemicals for which no TRV is available as well as chemicals whose maximum DL exceeded a TRV but whose detected values did not.

Risk assessments are, by design, conservative in the face of uncertainty. However, not all uncertainties create a conservative bias. Some examples of uncertainties that could lead to underestimation of risk include unavailability of exposure or effects data; existing TRVs that might underestimate risk for untested sensitive species; synergistic interactions among the multiple chemicals; and metabolic processes that increase the toxicity of accumulated chemicals.

Table 11-1 tallies the COPCs (individual chemicals, sums, or totals) identified as posing potentially unacceptable risk for each assessment endpoint and Table 11-2 provides a more general summary for each ecological receptor group. In total, 89 CERCLA contaminants were identified as posing potentially unacceptable risk in this BERA based on HQ $\geq$ 1 for at least one receptor-LOE combination. The maximum HQs and numbers of samples resulting in HQ $\geq$ 1 for each receptor-LOE-COPC combination posing potentially unacceptable risk are presented in Attachment 19:[163]

- Benthic invertebrates – Eighty-three COPCs were identified via one or more of the sediment, tissue-residue, surface water, and TZW LOEs.164
- Fish – Fifty nine COPCs were identified using the tissue-residue, dietary-dose, surface water, and TZW LOEs.165
- Wildlife – Twelve COPCs were identified for birds using the dietary-dose and tissue-residue (egg) LOEs, and six COPCs were identified for mammals using the dietary-dose LOE.
- Amphibians – Thirty-three COPCs were identified using the surface water and TZW LOEs.166

[163] Counts of COPCs with HQs $\geq$ 1 are based on HQs derived using alternative surface water TRVs for total PCBs, 4,4′-DDT, and total DDx, as opposed to the AWQC-based TRVs.

[164] Eighty-five benthic invertebrate COPCs have HQs $\geq$ 1. Petroleum compounds are not CERCLA contaminants, and have been excluded from the final COPC count for sediment and TZW LOEs even though this chemical group may be contributing to potentially unacceptable risk.

[165] Sixty fish COPCs have HQs $\geq$ 1. Petroleum compounds are not CERCLA contaminants and have been excluded from the COPC count for the TZW LOE even though this chemical group may be contributing to potentially unacceptable risk.

- **Aquatic plants –** Thirty-three COPCs were identified using the surface water and TZW LOEs.[167]

The spatial extent, magnitude and potential ecological significance of TRV exceedances and the concordance among LOEs were considered to determine risk conclusions for contaminants posing potentially unacceptable risk. The analyses used to draw these conclusions are presented for each receptor group in Sections 6.7, 7.6, 8.3, 9.3, and 10.3. The main conclusions of the BERA by receptor group are briefly summarized below in Section 11.3.

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[166] Thirty-four amphibian COPCs have HQs ≥ 1. Petroleum compounds are not CERCLA contaminants and have been excluded from the COPC count for the TZW LOE even though this chemical group may be contributing to potentially unacceptable risk.

[167] Thirty-four aquatic plant COPCs have HQs ≥ 1. Petroleum compounds are not CERCLA contaminants and have been excluded from the COPC count for the TZW LOE even though this chemical group may be contributing to potentially unacceptable risk.

Table 11-1. COPCs with HQ ≥ 1 Organized by Assessment Endpoint and Line of Evidence for the Portland Harbor BERA

Line of Evidence	COPCs with HQ ≥ 1
Assessment Endpoint:[a] Benthic Invertebrate Survival, Growth, and Reproduction	
Macroinvertebrates (e.g., amphipods, isopods, bivalves, gastropods, oligochaetes, insects, decapods)	
Survival and biomass of *Chironomus dilutus* and *Hyalella azteca* exposed to site sediments compared with reference area sediments	Responses based on chemical mixtures; no individual COPCs identified
Concentrations in site sediment compared with effect levels derived from FPM and LRM models (i.e., SQVs) predicting reduced survival or biomass based on Portland Harbor surface sediment concentrations and toxicity reported for both *Hyalella* and *Chironomus* endpoints	6 metals, TBT, 19 individual PAHs or group sums, dibutyl phthalate, 3 SVOCs, 2 phenolic compounds, PCBs, 15 individual pesticides or group sums
Concentrations in site sediment compared with national consensus-based SQGs (PECs and related quotients), and effects-based SQGs (PELs, and related quotients)	8 metals, 14 individual PAHs or group sums, 2 PCBs, 9 individual pesticides or group sums
Concentrations in surface water compared with state WQS, national AWQC, or effects-based values derived from the literature that are protective of benthic macroinvertebrate survival, growth, and reproduction	Zinc, monobutyltin, benzo(a)anthracene, benzo(a)pyrene, naphthalene, BEHP, total DDx,[b] ethylbenzene, trichlorethene
Concentrations in shallow TZW compared with state WQS, national AWQC, or effects-based values derived from the literature that are protective of benthic macroinvertebrate survival, growth, and reproduction	14 metals, 16 individual PAHs, 3 SVOCs, the pesticides 4,4′-DDT[b] and total DDx,[b] 16 VOCs, gasoline–range hydrocarbons, cyanide and perchlorate
Empirical (field-collected) whole-body concentrations of epibenthic organisms compared with tissue TRVs	None
Steady-state estimates of laboratory-exposed whole-body concentrations in *Lumbriculus* compared with tissue TRVs	Arsenic, copper, zinc, TBT, PCBs, total DDx
Predicted (BSAF) whole-body concentrations of *Lumbriculus* compared with tissue TRVs	TBT, PCBs, total DDX
Bivalves (clams, mussels)	
Empirical (field-collected) whole-body concentrations in *Corbicula fluminea* and freshwater mussels compared with tissue TRVs	Copper, zinc, TBT, PCBs
Steady-state estimates of laboratory-exposed whole-body concentrations in *Corbicula fluminea* compared with tissue TRVs	TBT, BEHP, total DDx

Table 11-1. COPCs with HQ ≥ 1 Organized by Assessment Endpoint and Line of Evidence for the Portland Harbor BERA

Line of Evidence	COPCs with HQ ≥ 1
Predicted (BSAF) whole-body concentrations in *Corbicula fluminea* compared with tissue TRVs	Total PCBs, total DDx
Corbicula fluminea survival compared with control data from bioaccumulation tests	Responses based on chemical mixtures; no individual COPCs identified
Survival and biomass of *Chironomus dilutus* and *Hyalella azteca* exposed to site sediments, compared with reference sediments	Responses based on chemical mixtures; no individual COPCs identified
Concentrations in surface water compared with state WQS, national AWQC, or effects-based values derived from the literature that are protective of benthic macroinvertebrate survival, growth, and reproduction	Zinc, monobutyltin, benzo(a)anthracene, benzo(a)pyrene, naphthalene, BEHP, total DDx,[b] ethylbenzene, trichlorethene
Concentrations in shallow TZW compared with state WQS, national AWQC, or effects-based values derived from the literature that are protective of benthic macroinvertebrate survival, growth, and reproduction	14 metals, 16 individual PAHs, 3 SVOCs, the pesticides 4,4′-DDT[b] and total DDx,[b] 16 VOCs, gasoline-range hydrocarbons,[c] cyanide and perchlorate
Concentrations in site sediment compared with national consensus-based SQGs (PECs and related quotients) and effects-based SQGs (PELs and related quotients)	8 metals, 14 individual PAHs or group sums, 2 PCBs, 9 individual pesticides or group sums
Decapods (crayfish)[d]	
Empirical whole-body concentrations in crayfish compared with tissue TRVs	Copper
Predicted (BSAF or FWM) whole-body concentrations in crayfish compared with tissue TRVs	Total PCBs, total DDx
Concentrations in site sediment compared with national consensus-based SQGs (PECs and related quotients) and effects-based SQGs (PELs and related quotients)	8 metals, 14 individual PAHs or group sums, 2 PCBs, 9 individual pesticides or group sums
Concentrations in surface water compared with state WQS, national AWQC, or effects-based values derived from the literature that are protective of benthic macroinvertebrate survival, growth, and reproduction	Zinc, monobutyltin, benzo(a)anthracene, benzo(a)pyrene, naphthalene, BEHP, total DDx,[b] ethylbenzene, trichlorethene
Concentrations in shallow TZW compared with state WQS, national AWQC, or effects-based values derived from the literature that are protective of benthic macroinvertebrate survival, growth, and reproduction	14 metals, 16 individual PAHs, 3 SVOCs, the pesticides 4,4′-DDT[b] and total DDx,[b] 16 VOCs, gasoline–range hydrocarbons,[c] cyanide and perchlorate

Table 11-1. COPCs with HQ ≥ 1 Organized by Assessment Endpoint and Line of Evidence for the Portland Harbor BERA

Line of Evidence	COPCs with HQ ≥ 1
Assessment Endpoint:[a] Fish Survival, Growth, and Reproduction	
Omnivorous Fish (white sturgeon, largescale sucker[e])	
Empirical whole-body concentrations compared with tissue TRVs	Total PCBs
Dietary dose (including incidental sediment ingestion) compared with dietary TRVs	Copper
Concentrations in surface water compared with state WQS, national AWQC,[b] or effects-based values derived from the literature that are protective of fish survival, growth, and reproduction	No COPCs with HQs ≥ 1
Correlation of lesion prevalence with areas of contamination and/or comparison to lesion-based TRVs (if relevant to receptor species)	Inconclusive for PAHs
Invertivorous Fish (juvenile Chinook salmon,[f] peamouth, sculpin)	
Empirical whole-body concentrations compared with tissue TRVs	Copper, lead, total PCBs, total DDx
Predicted (BSAF or FWM) whole-body concentration compared with tissue TRVs (sculpin only)	Total PCBs, total DDx
Dietary dose (including incidental sediment ingestion) compared with dietary TRVs	Cadmium, copper, TBT
Concentrations in surface water compared with state WQS, national AWQC or effects-based TRVs reported in the literature	Zinc, monobutyltin, benzo(a)anthracene, benzo(a)pyrene, naphthalene, BEHP, total DDx[b], trichlorethene
Concentrations in shallow TZW compared with state WQS, national AWQC or effects-based TRVs reported in the literature (sculpin only)	14 metals, 16 PAHs, 3 SVOCs, the pesticides 4,4′-DDT[b] and total DDx,[b] 16 VOCs, gasoline-range hydrocarbons,[c] cyanide and perchlorate
Piscivorous Fish (northern pikeminnow, smallmouth bass)	
Empirical whole-body concentrations compared with tissue TRVs	Antimony, lead, total PCBs
Predicted (BSAF or FWM) whole-body concentrations compared with tissue TRVs (smallmouth bass only)	This LOE was not evaluated because empirical tissue data were available from all exposure areas.
Concentrations in surface water compared with reported state WQS, national AWQC,[b] or effects-based TRVs reported in the literature	Zinc, monobutyltin, benzo(a)anthracene, benzo(a)pyrene, naphthalene, BEHP

Table 11-1. COPCs with HQ ≥ 1 Organized by Assessment Endpoint and Line of Evidence for the Portland Harbor BERA

Line of Evidence	COPCs with HQ ≥ 1
Dietary dose (including incidental sediment ingestion) compared with dietary TRVs	Copper
Detritivorous Fish (Pacific lamprey ammocoete[f])	
Empirical whole-body concentration compared with tissue TRV	Copper
Concentrations in surface water compared with state WQS, national AWQC, or literature-based values that are protective of early life stages.	No COPCs with HQs ≥ 1[b]
Concentration in shallow TZW compared with state WQS, national AWQC, or effects-based values reported in the literature that are protective of early life stages[g]	14 metals, 16 PAHs, 3 SVOCs, the pesticides 4,4′-DDT[b] and total DDx,[b] 16 VOCs, gasoline range hydrocarbons,[c] cyanide and perchlorate
Assessment Endpoint:[a] Bird Survival, Growth, and Reproduction	
Invertivorous Birds (spotted sandpiper)	
Dietary dose (including incidental sediment ingestion) compared with dietary TRV	Copper, benzo(a)pyrene, dibutyl phthalate, total PCBs, PCB TEQ, total dioxin/furan TEQ, total TEQ, sum DDE, total DDx, aldrin
Omnivorous Birds (hooded merganser)	
Dietary dose (including incidental sediment ingestion) compared with dietary TRV	Total PCBs
Piscivorous Birds (osprey, bald eagle)	
Dietary-based approach incorporating food chain transfer of contaminants from appropriate fish species (assuming all exposure comes from prey fish) and incidental sediment ingestion	Lead, mercury, total PCBs
Measured concentrations in osprey eggs compared with egg- or embryo-based TRVs for DDT and metabolites, PCBs, and dioxin-like compounds	Total PCBs, PCB TEQ, total dioxin/furan TEQ, total TEQ, sum DDE[h]
Assessment Endpoint:[a] Mammal Survival, Growth, and Reproduction	
Aquatic-Dependent Mammals (mink, river otter)	
Dietary dose compared with dietary TRVs	Aluminum, lead, total PCBs, PCB TEQ, total dioxin/furan TEQ, total TEQ

Table 11-1. COPCs with HQ ≥ 1 Organized by Assessment Endpoint and Line of Evidence for the Portland Harbor BERA

Line of Evidence	COPCs with HQ ≥ 1
Assessment Endpoint:[a] Amphibian Survival, Growth, and Reproduction (frogs, salamanders)	
Concentrations in surface water compared with state WQS, national AWQC, or effects-based values reported in the literature that are protective of sensitive life stages	Zinc, benzo(a)anthracene, benzo(a)pyrene, naphthalene, BEHP, total DDx[b]
Concentrations in shallow TZW compared with state WQS, national AWQC, or effects-based values reported in the literature that are protective of sensitive life stages	11 metals, 8 PAHs, the SVOC 1,2-dichlorobenzene, the pesticides 4,4′-DDT[b] and total DDx,[b] 8 VOCs, gasoline-range hydrocarbons,[c] and the conventionals cyanide and perchlorate
Assessment Endpoint:[a] Aquatic Plant Survival, Growth, and Reproduction (phytoplankton, periphyton, macrophytes)	
Concentrations in surface water compared with state WQS, national AWQC, or effects-based values derived from the literature that are protective of sensitive life stages (e.g., germination, emergence, early life stage growth)	Zinc, benzo(a)anthracene, benzo(a)pyrene, naphthalene, BEHP, total DDx[b]
Concentrations in shallow TZW compared with state WQS, national AWQC, or effects-based values derived from the literature that are protective of sensitive life stages (e.g., germination, emergence, early life stage growth)	11 metals, 8 PAHs, the SVOC 1,2-dichlorobenzene, the pesticides 4,4′-DDT[b] and total DDx,[b] 8 VOCs, gasoline-range hydrocarbons,[c] and the conventionals cyanide and perchlorate

[a] The assessment endpoints for all receptors are based on protection and maintenance of their populations and the communities in which they live, except that the health of threatened or endangered species is to be protected at the level of the individual organism. Per the SOW, EPA's Problem Formulation (Attachment 2), and as stated in the Programmatic Work Plan (Integral et al. 2004b), the assessment endpoints were expressed as the survival, growth, and reproduction of each receptor group.

[b] Risk estimates for total PCBs, 4,4′-DDT, and total DDx for the surface water and TZW LOEs are based on the alternative total PCBs and 4,4′-DDT TRVs for protection of directly exposed aquatic organisms, rather than the selected AWQC-based TRVs. Additional exceedances occur using the AWQC-based TRVs and HQs, as presented in the surface water and TZW risk characterization sections for each receptor group. The alternative TRVs are considered more appropriate for evaluating direct exposure of aquatic organisms because the AWQC are based on protection of dietary risks to mammals and birds.

[c] The HQ for gasoline-range hydrocarbons is ≥ 1; however the COPC was not included in the counts of COPCs with HQs ≥ 1 because counts are based only on CERCLA contaminants.

[d] Although these LOEs are components of the benthic invertebrate community, the bivalve population and decapod population assessment endpoints are presented separately in this table. Evaluation of sediment toxicity to *Chironomus* and *Hyalella* and comparison of surface water and shallow TZW concentrations to TRVs were each conducted and presented only once as part of the benthic invertebrate community assessment. Similarly, comparison of sediment concentrations to published SQGs also occurred and was presented only once as part of the benthic community assessment.

[e] Carp is not a receptor of concern for the BERA but whole-body carp tissue was analyzed for dioxin-like chemicals, including PCB congeners; for these chemicals, carp is a surrogate for other omnivorous fish species.

[f] Juvenile Chinook salmon and Pacific lamprey ammocoetes were evaluated at the organism level because they have special status are (juvenile Chinook is federally threatened and Pacific lamprey is an Oregon state sensitive species of special concern to Tribes); effect thresholds based on reproduction are used as a surrogate for growth in juvenile Chinook salmon and Pacific lamprey ammocoetes.

[g] The TZW exposure pathway for fish receptors is considered complete and significant for only sculpin and lamprey ammocoetes. The ecological CSM shows a complete TZW exposure pathway for sucker, carp, and sturgeon but categorizes the pathway as insignificant.

[h] Bald eagle only based on extrapolation from osprey eggs and comparison to a NOAEL-based TRV. For osprey, all HQ < 1.

AWQC – ambient water quality criteria
BEHP – bis(2-ethylhexyl) phthalate
BERA – baseline ecological risk assessment
BSAF – biota-sediment accumulation factor
CERCLA – Comprehensive Environmental Response, Compensation, and Liability Act
COPC – contaminant of potential concern
CSM – conceptual site model
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
EPA – US Environmental Protection Agency
FWM – food web model
HQ – hazard quotient
LOE – line of evidence
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
PEC – probable effects concentration
PEL – probable effects level
SOW – scope of work
SQG – sediment quality guideline
SQV – sediment quality value
SVOC – semivolatile organic compound
TBT – tributyltin
TEQ – toxic equivalent
total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT, and 4,4′-DDT)
TRV – toxicity reference value
TZW – transition zone water
VOC – volatile organic compound
WQS – water quality standards

Table 11-2. Contaminants Posing Potentially Unacceptable Risk Organized by Receptor Group

COPC[a]	Benthic Invertebrates	Fish	Birds	Mammals	Amphibians	Aquatic Plants
Metals						
Aluminum				X		
Antimony		X				
Arsenic	X					
Barium	X	X			X	X
Beryllium	X	X				
Cadmium	X	X			X	X
Cobalt	X	X				
Copper	X	X	X		X	X
Iron	X	X			X	X
Lead	X	X	X	X	X	X
Magnesium	X	X			X	X
Manganese	X	X			X	X
Mercury		X	X			
Nickel	X	X			X	X
Potassium	X	X			X	X
Sodium	X	X			X	X
Vanadium	X	X				
Zinc	X	X			X	X
Butyltins						
Monobutyltin	X	X				
Tributyltin	X	X				
PAHs						
2-Methylnaphthalene	X	X			X	X
Acenapthene	X	X			X	X
Anthracene	X	X			X	X
Benzo(a)anthracene	X	X			X	X
Benzo(a)pyrene	X	X	X		X	X
Benzo(b)fluoranthene	X	X				
Benzo(g,h,i)perylene	X	X				
Benzo(k)fluoranthene	X	X				
Chrysene	X	X				
Dibenzo(a,h)anthracene	X	X				
Fluoranthene	X	X				

Table 11-2. Contaminants Posing Potentially Unacceptable Risk Organized by Receptor Group

COPC[a]	Benthic Invertebrates	Fish	Birds	Mammals	Amphibians	Aquatic Plants
Fluorene	X	X			X	X
Ideno(1,2,3-cd) pyrene	X	X				
Naphthalene	X	X			X	X
Phenanthrene	X	X			X	X
Pyrene	X	X				
Phthalates						
BEHP	X	X			X	X
Dibutyl phthalate			X			
SVOCS						
1,2-Dichlorobenzene	X	X			X	X
1,4-Dichlorobenzene	X	X				
Dibenzofuran	X	X				
PCBs						
Total PCBs	X	X	X[c]	X		
PCB TEQ			X[c]	X		
Dioxins/furan TEQ			X[c]	X		
Total TEQ			X[c]	X		
VOCs						
1,1-Dichloroethene	X	X				
1,2,4-Trimethylbenzene	X	X			X	X
1,3,5-Trimethylbenzene	X	X				
Benzene	X	X				
Carbon disulfide	X	X			X	X
Chlorobenzene	X	X			X	X
Chloroethane	X	X			X	X
Chloroform	X	X			X	X
cis-1,2-Dichloroethene	X	X				
Ethylbenzene	X	X			X	X
Isopropylbenzene	X	X			X	X
Toluene	X	X			X	X
Trichloroethene	X	X				
m,p-Xylene	X	X				
o-Xylene	X	X				
Total xylenes	X	X				

Table 11-2. Contaminants Posing Potentially Unacceptable Risk Organized by Receptor Group

COPC[a]	Benthic Invertebrates	Fish	Birds	Mammals	Amphibians	Aquatic Plants
Pesticides						
Aldrin			X			
4,4'-DDD	X					
sum DDE			X			
4,4’-DDT	X	X			X	X
Total DDx	X	X	X		X	X
Other Chemicals						
Cyanide	X	X			X	X
Perchlorate	X	X			X	X

[a] The COPCs listed in this table are CERCLA contaminants. Several additional contaminants may also contribute to potentially unacceptable risk. These contaminants include TPH, ammonia, and sulfides.

BEHP – bis(2-ethylhexyl) phthalate
CERCLA – Comprehensive Environmental Response, Compensation, and Liability Act
COPC – chemical of potential concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
HCH – hexachlorocyclohexane
HQ – hazard quotient
PAH –polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
TEQ – toxic equivalent
total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT, and 4,4′-DDT)

11.2 BACKGROUND AND UPRIVER CONCENTRATIONS

For all contaminants posing potentially unacceptable risk, Attachment 11 presents a comparison of background and Study Area 95th percentile UCLs in sediment and surface water. For aluminum, dibutyl phthalate, benzyl alcohol, and alpha-endosulfan, background sediment UCLs are the same as or higher than Study Area UCLs. The background surface water UCL concentration is higher than the Study Area UCL only for aluminum. Attachment 11 also includes a comparison of concentrations in fish tissue from the upriver reach and the Study Area for all fish tissue-residue and wildlife dietary contaminants posing potentially unacceptable risk. Although fish tissue data from the upriver reach are insufficient to allow calculation of UCLs, their concentrations are similar to those in the Study Area for aluminum, mercury, and copper, as presented in Section 7.1.5.

Background concentrations for sediment and surface water, and upriver concentrations for fish tissue provide context for Study Area risk predictions but were not used to discount risks or influence risk estimates. Where background concentrations exceed screening-level TRVs or upriver fish tissue concentrations exceed tissue TRVs, upriver or regional sources may be contributing to unacceptable risks in the Study Area.

11.3 ECOLOGICAL RISK CONCLUSIONS

The risk conclusions across all ecological receptor groups are combined and briefly summarized in this section to provide a general overview of ecological risks and to identify the receptor-COPC pairs that, given the magnitude and extent of risk, are reasonably likely to result in adverse effects on the assessment endpoints that were selected in the Problem Formulation to represent the valued ecological attributes of the Study Area. To reiterate, the analyses used to draw these conclusions are presented in Sections 6.7, 7.6, 8.3, 9.3, and 10.3, and are not repeated here. For example, this section (11.3) contains statements with qualitative adjectives like “limited” or “moderate” when describing the spatial extent of exposure to a COPC at concentrations yielding HQs ≥ 1. Statements such as, “uncertainty in the tissue-residue TRV is more likely to over- than underpredict risk” are made without repeating the supporting evidence. In cases such as these, the reader interested in the details should refer back to the risk conclusions section for the relevant receptor group. The main conclusions of the BERA by receptor group are presented in Sections 11.3.1 through 11.3.5. Section 11.3 closes with a brief synopsis of potential future benthic community risks in erosional sections of the Study Area.

11.3.1 Benthic Invertebrate Community

COPCs occur at concentrations that are projected to pose unacceptable benthic risks for about 7% of the Study Area. Unlike other ecological receptors, for which risk was evaluated on a chemical-specific basis, risk to the benthic invertebrate community was evaluated in large part by considering exposure to the mixture of chemicals present in the Study Area sediments, using toxicity tests and multivariate predictive models based on the toxicity test results. Point-by-point assessment of potential effects on benthic organisms using data from toxicity testing, modeling, and benthic tissue-residue analyses indicates that metals, TBT, PAHs, several SVOCs, two phenolic compounds, dibutyl phthalate, total PCBs, total DDx, and other pesticides pose potentially unacceptable risk. Several other contaminants (TPH, ammonia, and sulfides) may also contribute to potentially unacceptable risk at some areas. A WOE was assessed to identify contaminants that were most likely posing unacceptable risk. Based on that evaluation, the primary COPCs in sediment that likely pose potentially unacceptable risk to the benthic community or populations are PAHs, PCBs, and total DDx. Although other contaminants may also contribute to unacceptable risk, their distribution and magnitude of risk tends to be represented by the distribution and magnitude of primary COPCs. One exception is the certain contaminants associated with the localized TZW investigation areas. In these areas, VOCs, cyanide, and perchlorate may also pose potentially unacceptable risks; however, these contaminants often co-occur with PAHs and DDx.

The phenolic compound 4-methylphenol may also be contributing to benthic community risk. The analysis conducted for the BERA shows that the sediment exposure pathway is sufficient to be of concern for 4-methylphenol. Widely distributed throughout the Study Area, this contaminant is found in both contaminated and otherwise uncontaminated areas. Methylated phenols are readily biodegraded under aerobic conditions, and 4-methylphenol is expected to have a half-life in sediment on the order of days. That

4-methylphenol was found suggests the presence of ongoing sources; however, whether and to what extent the source is degradation of historical contamination versus influx from ongoing point or non-point discharges is not known.

Sediment profile images of the surface sediment suggest that the physical environment (sediment grain size, transport regime, bottom slope) in the Study Area can explain the presence of early colonizing, transitional, and mature benthic communities in 90% of the images evaluated. In these cases, the successional stage matched the expected community structure based on the physical regime and habitat characteristics. In the vast majority of cases, mature benthic communities occurred in fine-grained depositional environments; early colonizing or transitional communities were found in less physically stable areas (for example, with steep slopes, active sediment transport, high rates of deposition, or physical disturbance). In the 31 (of 377) cases where the community successional stage was not as might be predicted by the physical environment, about two-thirds (19) occur between RM 5.0 and RM 9.0. The greatest combined area associated with potentially unacceptable risk to the benthic community was found in this same reach, suggesting possible chemical toxicity, among other potential factors, as the reason for the presence of lower successional stages. These qualitative results suggest that overall, the benthic community in the Study Area is typical of a large river system that is strongly influenced by physical processes. Impacts from sediment contamination appear to be limited to certain depositional areas that have received historical releases of contamination.

11.3.2 Fish

The fish assessment endpoints are survival, growth and reproduction of omnivorous, invertivorous, and piscivorous fish, as well as survival and growth of detritivorous fish. The assessment endpoints are based on protection and maintenance of populations and the communities in which they live, except for Pacific lamprey ammocoete and juvenile Chinook salmon, which, as special status species, are to be protected at the organism level.

Total PCBs were found to pose low risk to populations of piscivorous fish and the small-home-range invertivorous fish sculpin. Total PCB tissue-residue HQs ≥ 1 were calculated for smallmouth bass, northern pikeminnow, and sculpin samples from locations throughout the Study Area (max HQ = 9.4). HQs < 1 for juvenile Chinook salmon and peamouth show that risk to sculpin does not imply risk to invertivorous fish with larger home ranges. Together, the low Study Area-wide tissue-residue HQ of 1.6 for largescale sucker in combination with HQs < 1 for most omnivorous fish samples and with uncertainty in effects data indicate that risk to omnivorous fish is negligible.

The potential for adverse effects on all of the fish assessment endpoints from total PCBs was assessed to be low: the other LOE for PCBs—surface water—resulted in HQs < 1,[168] tissue-residue HQs ≥ 1 occurred over only a moderate spatial extent (or in relatively few samples for large-home-range fish), and uncertainty in the tissue-residue TRV is more likely to overpredict than underpredict risk. The tissue-residue TRV for total PCBs is conservative because it is based partially on uncertain toxicity data, including field data from contaminated sites where other contaminants were also present, suggesting that the TRV reflects toxicity from chemicals other than PCBs.

The spatial extent of dietary risk to juvenile Chinook salmon from cadmium encompasses a substantial portion of the Study Area. However, the assumption that juvenile Chinook consume benthic invertebrates, rather than the pelagic prey they are known to eat, overestimates exposure. The selected TRV also very likely overestimates risk because it is 3 orders of magnitude below the lowest salmon-specific NOAEL.

The spatial extent of dietary and tissue-residue risk from copper to several fish (sculpin, juvenile Chinook salmon, Pacific lamprey ammocoetes, northern pikeminnow, largescale sucker, and juvenile white sturgeon) also encompasses a substantial portion of the Study Area. The copper-fish TRVs are highly uncertain. The dietary TRV could not be replicated in subsequent studies, and the tissue-residue TRV is within the range of copper nutritional requirements for some (but not all) fish species. Furthermore, predictions of risk to fish based on tissue concentrations copper is highly uncertain because fish regulate this essential metal.

Several COPCs in TZW were identified as posing risk to individual fish, but not their populations. Benthic fish, including burrowing fish (lamprey ammocoetes) and fish that feed on benthic organsims (sculpin), have relatively low exposure to porewater compared with surface water because of their feeding habits and respiratory requirements. For this reason concentrations of COPCs in shallow TZW likely overestimate exposure, to an uncertain degree. Because TZW exceedances are localized, none of the TZW COPCs is likely to pose risk to Study Area benthic invertebrate or fish populations. However, 38 TZW COPCs,[169] 6 metals (barium, iron, manganese, sodium, vanadium, and zinc), 16 PAHs (2-methylnaphthalene, acenaphthene, anthracene, benzo(a)anthracene, benzo(a)pyrene, benzo(b)fluoranthene, benzo(g,h,i)perylene, benzo(k)fluoranthtene, chrysene, dibenzo(a,h)anthracene, fluoranthene, fluorene, indeno(1,2,3-cd)pyrene, naphthalene, phenanthrene, pyrene), 2 SVOCs (1,2-dichlorobenzene, 1,4-dichlorobenzene), the pesticides 4,4′-DDT and total DDx, 10 VOCs (benzene, carbon disulfide, chlorobenzene, chloroform, cis-1,2dichloroethene, ethylbenzene, o-xylene, toluene, total xylenes and trichloroethene), cyanide and perchlorate have high

[168] When calculated using the alternative TRV for protection of directly exposed aquatic organisms rather than the AWQC, which is based on protection of mink through dietary exposure.

[169] Petroleum hydrocarbons were evaluated as an uncertainty and gasoline-range aliphatic hydrocarbons (C10-C12) have HQ > 10 over a limited spatial extent and also pose potentially unacceptable risk to individual lamprey.

concentrations in localized areas that could adversely affect Pacific lamprey ammocoetes at those locations. The magnitude of risk to individual lamprey from these COPCs is unknown however, because the TRVs were derived to be protective of the most sensitive species and are likely to overpredict risk to Pacific lamprey which has been shown to have average or lower sensitivity than most aquatic species for several chemicals causing toxicity from different modes of action (Andersen et al. 2010). Three of the 38 COPCs (excluding petroleum hydrocarbons.) with HQs > 10 are naturally occuring metals (barium, iron, and manganese) and there is substantial uncertainty as to whether their source is anthropogenic.

Risk to fish from other COPCs that resulted in HQs ≥ 1 in the final step of the risk characterization were found unlikely to result in ecologically significant adverse effects on the fish populations. The rationale for this conclusion is that TZW exposure assumptions likely overestimate risk, the TRV overestimates risk, and the great majority of samples result in HQs < 1 (indicating a limited spatial extent of potentially unacceptable risk).

11.3.3 Wildlife

The avian assessment endpoints are survival, growth, and reproduction of invertivorous, omnivorous, and piscivorous birds. The mammalian assessment endpoint was survival, growth, and reproduction of aquatic-dependent mammals. The assessment endpoints are based on protection and maintenance of populations and the communities in which they live, except for threatened or endangered species, which are to be protected at the organism level.

Total PCBs pose the primary risk. Mink and river otter HQs ≥ 1 throughout the Study Area (mink HQ = 19 to 33, river otter HQ = 21 to 31) indicate that PCBs pose ecologically significant risk of reduced reproductive success to populations of both receptors in the Study Area. While this BERA has established that PCBs pose the potential for adverse effects, the true effect of PCB exposure on Study Area populations is still unknown because of a number of uncertainties. These include quantifiable uncertainties about dietary exposure and about PCB dose-response, and quantifiable uncertainty about the level of effect associated with a population-level response. These uncertainties have not been fully examined in the BERA.

Reproductive success in spotted sandpiper, bald eagle, and osprey might also be reduced because of PCB exposure, as indicated by spotted sandpiper and bald eagle HQs ≥ 1 throughout the Study Area (max HQ = 12 for sandpiper and 3.9 for eagle) and by less widespread osprey HQs ≥ 1 (max HQ = 4.4). Overall, a greater degree of uncertainty is associated with PCB risk estimates for birds than for mammals because of uncertainty about exposure and uncertainty in the effects data. Uncertainty is higher for otter than for mink because otter-specific effects data are lacking.

Total TEQ exposure also poses ecologically significant risk of reduced reproductive success to populations of mink (with HQs up to 12). Total TEQ risk to birds and otter is low, considering the WOE for eagle and osprey, the more limited spatial extent of TRV exceedances for sandpiper, and the low magnitude of HQs for river otter. PCBs are responsible for the majority of total TEQ exposure, in that PCB TEQ HQs generally constitute the majority of the total TEQ HQs. For example, mink total TEQ HQs are ≥ 1 in 16 of 109 potential prey samples; of these samples, PCB TEQ HQs are ≥ 1 in 15 samples and total dioxin/furan TEQ HQs are ≥ 1 in only 4 samples. As is the case for total PCBs, a greater degree of uncertainty is associated with total TEQ risk estimates for birds and otter than for mink because of uncertainties in both exposure and effects data for birds and uncertinaty in effects data for otter.

Osprey egg data indicate that DDx compounds pose negligible risk to osprey and low to negligible risk of reduced reproductive success to individual bald eagles within limited portions of the Study Area. The only other wildlife receptor with a DDx HQ ≥ 1 is the spotted sandpiper. DDx compounds pose negligible risk to the spotted sandpiper population because the HQs are of low magnitude, span a limited spatial extent, and based on uncertainties in exposure and effects that likely cause overestimates of risk.

The spatial extent of copper HQ ≥1 in sandpiper encompasses a large portion of the Study Area; however, risk is negligible. Only one prey item (laboratory-exposed worms) had tissue concentrations associated with an HQ ≥ 1. Copper HQs based on a mixed-species diet are < 1. Additionally, the selected TRV was below the lowest bounded literature-reported NOAEL for birds.

Risk to wildlife from other COPCs that resulted in HQ ≥ 1 in the final step of the risk characterization were found unlikely to result in ecologically significant adverse effects on the receptor populations because the HQs are of low magnitude, span a limited spatial extent, and are based on uncertainties in exposure and effects that likely cause an overestimate of risk.

11.3.4 Amphibians

The amphibian assessment endpoints are survival, growth, and reproduction of amphibians. The assessment endpoints are based on protection and maintenance of populations and the communities in which they live, except for threatened or endangered species, which are to be protected at the organism level. For all COPCs with HQs ≥ 1, the risk to amphibian populations was assessed to be negligible. COPCs in surface water samples resulting in HQ ≥ 1 were found at concentrations below amphibian-specific thresholds or were collected during non-reproductive periods (when amphibians may not be present in the Study Area). For the TZW LOE, the great majority of samples result in HQs < 1, indicating limited spatial extent of exceedance. Although risk to amphibians from TZW is highly uncertain, it is likely to be negligible because significant exposure to Study Area TZW by this receptor group is unlikely.

11.3.5 Aquatic Plants

The aquatic plant assessment endpoints are survival, growth, and reproduction of aquatic plants. The assessment endpoints are based on protection and maintenance of populations and the communities in which they live, except for threatened or endangered species, which are to be protected at the organism level.For all COPCs with HQs ≥ 1, the risk to aquatic plant populations was assessed to be negligible. The same COPCs whose surface water HQ is ≥ 1 were found in the great majority of samples to have HQ < 1 and at concentrations generally below algae-specific thresholds. For the TZW LOE, the great majority of samples result in HQs < 1, indicating limited spatial extent of exceedance.

11.3.6 Potential Future Risks to the Benthic Community

Risk to the benthic community was assessed both for current conditions in the Study Area and estimated future conditions. The future condition assessment is based on the maximum bed change scenario presented in the draft RI (Map 3.4-7) and a sample-by-sample evaluation of changes in status of predicted risk in the erosional areas based on comparison to site-specific SQVs. Attachment 18 presents the approach and results of the current and future risk predictions in the erosional areas of the Study Area. For the majority of erosional sediments (approximately 60%), there was no change of status in predicted risk to the benthic community (i.e., the sediment quality was similar at the erosional depth and the surface). This finding is not surprising because the erosional sediments are predicted to be primarily sands. Of the remaining erosional sediments, approximately 24% is predicted to be more contaminated in the future. The last 16% of the erosional area is predicted to be cleaner after the erosional event.

12.0 RISK MANAGEMENT RECOMMENDATIONS

This section presents the LWG's ecological risk management recommendations to develop and evaluate remedial alternatives that are protective of ecological resources. Risk management recommendations are provided in four main parts:

- Section 12.1 presents recommended COCs for populations of fish and wildlife receptors.[170]
- Section 12.2 presents recommendations regarding contaminants present in TZW. TZW risk management recommendations are presented separately from those for other exposure media because the TZW LOE focuses on a spatially limited set of nine TZW sampling areas; the other exposure media (sediment, tissues, and surface water) were evaluated Study Area-wide. Furthermore, the TZW sampling areas were selected to capture information at locations with known or likely pathways for ongoing sources (discharge of upland contaminated groundwater), whereas the other exposure media were investigated because they represent complete exposure pathways to ecological receptors from contaminated sediment. Thus, both the nature and extent of risk as well as the alternatives for addressing them are unique for TZW.
- Section 12.3 presents risk management recommendations for protection of the benthic invertebrate community. As the BERA's benthic risk conclusions rely heavily on LOEs that do not identify specific COPCs (i.e., empirical measurements of sediment toxicity, predictions of sediment toxicity based on multivariate statistical models, and benthic community data from SPI imagery), this section recommends methodologies for delineating benthic AOCs and for evaluating the degree to which remedial action alternatives protect the benthic community.
- Section 12.4 summarizes the risk management recommendations.

12.1 Recommendation of COCs for Study Area Populations of Fish and Wildlife Receptors

In this section, the entire set of contaminants identified as posing potentially unacceptable risk to fish and wildlife receptors is evaluated. The purpose of the evaluation is to identify the COPCs for fish and wildlife receptors to use in the FS to develop and evaluate remedial alternatives that are protective of ecological resources. This subset of COPCs constitutes the recommended COCs.

As discussed in Section 3, the assessment endpoints for most of the ecological receptors identified in EPA's Problem Formulation are for protection of the populations of fish, birds, mammals, and amphibians, and for protection of communities of benthic

[170] Where secondary benthic LOEs support these recommendations for fish and wildlife receptors, they are identified.

invertebrates and aquatic plants. The exceptions are that assessment endpoints for special status species identified in EPA's Problem Formulation (i.e., bald eagle, juvenile Chinook salmon, and Pacific lamprey ammocoetes) are for protection at the level of the organism.

The COC recommendations provided in this section are intended to address Study Area-wide risks to receptor populations. These recommendations are also intended to be protective of the aquatic plant community and receptors assessed at the organism level, except risk to Pacific lamprey ammocoetes from TZW exposure. Recommendations regarding risks from exposure to contaminants posing potentially unacceptable risk in TZW are presented in Section 12.2. Recommendations regarding identification of benthic risk areas and related protectiveness are provided in Section 12.3.

The remainder of Section 12.1 is presented in three main parts:

- Section 12.1.1 presents the rationale for COC recommendations.
- Section 12.1.2 applies that rationale to recommend COCs.
- Section 12.1.3 provides additional recommendations for the contaminants posing potentially unacceptable risk that are recommended as COCs. This includes recommendations about which receptors of concern should be considered along with the COCs to assess the protectiveness of potential remedies in the FS analysis of alternatives.

12.1.1 Rationale for COC Recommendations

COCs will be used to develop and evaluate remedial alternatives that are protective of ecological resources. The FS will also evaluate whether remedial alternatives for these COCs address the full list of contaminants posing potentially unacceptable risk.

The COC recommendations took into account one or more of the following factors:

- How often, where, and in which media risk thresholds were exceeded
- The ecological relevance (strengths and weaknesses) of the exposure estimates used to calculate HQs
- The toxicological effects associated with the TRV
- The magnitude of the exceedance
- Whether a relationship was found between COPC concentrations in co-located sediment and tissue concentrations (for small-home-range species)
- The relative strength and concordance among LOEs used to evaluate risks
- Comparison of Study Area concentrations with available background or upriver data

Some of these factors are strongly risk-based (e.g., the toxicological effects associated with the TRV, and the relative strength and concordance among LOEs), whereas others are more directly related to practical FS considerations (e.g., whether a relationship was found between COPC concentrations in co-located sediment and tissue concentrations for small-home-range species, and comparison with available background or upriver data).

12.1.2 COC Recommendations

Table 12-1 summarizes the contaminants posing potentially unacceptable risk in this BERA and whether they are recommended as COCs for fish and wildlife receptors. Contaminants posing potentially unacceptable risk based on the TZW LOE are discussed in Section 12.2. Areas and contaminants posing potentially unacceptable risk to the benthic community are discussed in Section 12.3; however, where benthic tissue-residue and surface water LOEs support the selection of COCs for protection of fish and wildlife, they are noted. Nineteen COPCs with at least one HQ ≥ 1 have been identified in this BERA for fish and wildlife receptors.[171,172] The set consists of seven metals, two butyltins, three PAHs, two phthalates, PCBs, dioxins/furans, two pesticides, and one VOC. The specific rationale for COC recommendations—based on the seven factors identified in Section 12.1—follows Table 12-1.

Table 12-1. COC Recommendations for All Receptor Group-LOE Pairs with an HQ ≥ 1

COPC	Receptor Group-LOE Pairs Resulting in HQ ≥ 1
Contaminants Recommended as COCs	
PCBs	
Total PCBs	Benthic invertebrate – tissue residue (clam, worm)
	Fish – tissue-residue (sucker, sculpin, bass, pikeminnow)
	Mammal – diet (mink, river otter)
	Bird – diet (sandpiper, osprey, bald eagle, merganser)
	Bird – tissue-residue (osprey, bald eagle)
Dioxins/Furans	
Total TEQ[a]	Mammal – diet (mink, river otter)
	Bird – diet (sandpiper)
	Bird – tissue residue (osprey, bald eagle)

[171] PCB TEQ and dioxin/furan TEQ are not included in this count because they are components of the total TEQ.

[172] Risk management recommendations for the benthic community assessment endpoints and the TZW LOE are handled separately and are not included in this COPC count.

Table 12-1. COC Recommendations for All Receptor Group-LOE Pairs with an HQ ≥ 1

COPC	Receptor Group-LOE Pairs Resulting in HQ ≥ 1
Contaminants Not Recommended as COCs	
Inorganic Metals	
Aluminum	Mammal – diet (mink)
Antimony	Fish – tissue residue (bass)
Arsenic	Benthic invertebrate – tissue residue (worm)
Cadmium	Fish – diet (sculpin, Chinook)
Copper	Benthic invertebrate – tissue residue (clam, crayfish, worm)
	Fish – diet (sucker, sturgeon, Chinook, peamouth, sculpin, pikeminnow)
	Fish – tissue-residue (sculpin, Chinook, lamprey, pikeminnow)
	Birds – diet (sandpiper)
Lead	Fish – tissue-residue (peamouth, bass)
	Birds – diet (osprey)
	Mammals – diet (mink)
Zinc	Benthic invertebrates – surface water, benthic invertebrate tissue residue (clam, mussel, worm)
	Fish – surface water (sculpin, bass, pikeminnow)
	Amphibians – surface water
	Aquatic plants – surface water
Organometals	
Mercury	Fish – diet (sculpin)
Monobutyltin	Benthic invertebrates – surface water
	Fish – surface water (sculpin, bass, pikeminnow)
	Birds – diet (bald eagle)
TBT	Benthic invertebrate (clam and worm tissue residue)
	Fish – diet (sculpin)
PAHs	
Benzo(a)anthracene	Benthic invertebrates – surface water
	Fish – surface water (sculpin, bass, pikeminnow)
	Amphibians – surface water
	Aquatic plants – surface water

Table 12-1. COC Recommendations for All Receptor Group-LOE Pairs with an HQ ≥ 1

COPC	Receptor Group-LOE Pairs Resulting in HQ ≥ 1
Benzo(a)pyrene	Benthic invertebrates – surface water
	Fish – surface water (sculpin, bass, pikeminnow)
	Birds – diet (sandpiper)
	Amphibians – surface water r
	Aquatic plants – surface water
Naphthalene	Benthic invertebrates – surface water
	Fish – surface water (sculpin, bass, pikeminnow)
	Amphibians – surface water
	Aquatic plants – surface water
Phthalates	
BEHP	Benthic invertebrates – surface water, tissue residue (worms)
	Fish – tissue residue (sculpin, bass,)
	Fish – surface water (sculpin, bass, pikeminnow)
	Amphibians – surface water
	Aquatic plants – surface water
Dibutyl phthalate	Birds – diet (sandpiper)
Pesticides	
Aldrin	Birds – diet (sandpiper)
Total DDx[b]	Benthic invertebrates – surface water, tissue residue (clam, worm)
	Fish –tissue residue (sculpin)
	Fish – surface water (sculpin)
	Birds – diet (sandpiper)
	Birds – tissue residue (bald eagle)
	Amphibians – surface water
	Aquatic plants – surface water
4,4′-DDD	Benthic invertebrate – tissue residue (worms)
VOCs	
Ethylbenzene	Benthic invertebrates – surface water
Trichloroethene	Benthic invertebrates – surface water
	Fish – surface water (sculpin)

[a] Total TEQ includes risk estimates for PCB TEQ and total dioxin/furan TEQ.

[b] Total DDx includes risk estimates for the additional DDx components that were also evaluated independently (sum DDE, 4,4′-DDE, and 4,4′-DDT). Risk estimates for the surface water LOE are based on the alternative 4,4′-DDT TRVs for protection of directly exposed aquatic organisms, rather than the AWQC-based TRV. The alternative TRV is considered more appropriate for evaluating direct exposure of aquatic organisms because the AWQC is based on protection of dietary risks to birds.

AWQC – ambient water quality criterion
BEHP – bis(2-ethylhexyl) phthalate
COC – contaminant of concern
COPC – contaminant of potential concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
HQ – hazard quotient
LOE – line of evidence
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
SVOC – semivolatile organic compound
TBT – tributyltin
TEQ – toxic equivalent
total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT, and 4,4′-DDT)
TRV – toxicity reference value
VOC – volatile organic compound

12.1.2.1 Recommended COCs

PCBs

Total PCBs is recommended as a COC because exposure poses a risk of ecologically significant adverse effects to mink and river otter populations. It also poses risk of ecologically significant adverse effects to spotted sandpiper, osprey, sculpin, and smallmouth bass populations and risk of adverse effects to bald eagles.The benthic tissue-residue LOE also supports the selection of PCBs as a COC. These additional risks are lower than the risk to mink and river otter populations. Further risk management recommendations regarding total PCBs are provided in Section 12.1.3.

Dioxins/Furans

Total TEQ is recommended as a COC because exposure poses a risk of ecologically significant adverse effects to mink populations. Total TEQ also poses risk of adverse effects to river otter, spotted sandpiper, and osprey populations and to bald eagles. These latter risks are lower than the risk to the mink population. Further risk management recommendations regarding dioxins/furans are provided in Section 12.1.3.

12.1.2.2 COPCs Not Recommended as COCs

Inorganic Metals

None of the seven metals with HQ ≥ 1 is recommended as a COC for assessing potential remedy protectiveness of ecological receptors. The rationales for exclusion are as follows:

- Aluminum poses potentially unacceptable risk only for mink. For the following reasons, it is not recommended as a COC:
  - Aluminum exceeds the dietary TRV only based on sediment ingestion, no prey samples exceed the effects threshold.
  - TRV is based on exposure of mice to a highly soluble ionic form of aluminum with higher bioavailability than typically found in the diet or drinking water.

- Study Area sediment and surface water concentrations are similar to background.

- Antimony poses potentially unacceptable risk based only on the tissue-residue LOE for smallmouth bass. For the following reasons, it is not recommended as a COC:
  - Low frequency of TRV exceedance (1 of 32 [3.1%] smallmouth bass samples)
  - Weakness of the exposure estimate (the single composite sample is an outlier for both antimony and lead, suggesting that a fish in the sample might have swallowed a fishing sinker)[173]
  - Weakness of the effects estimate (TRV is based on a single study with a generic ACR applied)
  - Absence of relationship between concentrations in sediment and co-located tissue samples (Windward 2009b)
- Discordance between the weaker tissue-residue LOE and the stronger surface water LOE (surface water TRV based on numerous exposure data and moderately sized Tier II effects dataset). Arsenic poses potentially unacceptable risk to benthic invertebrates based only on the tissue-residue LOE. It is not recommended as a COC for two reasons:
  - Low frequency of exceedance of the TRV (2 of 35 samples)
  - Low magnitude of the exceedance (maximum HQ = 1.5)
- Cadmium poses potentially unacceptable risk based only on the dietary LOE for juvenile Chinook salmon and sculpin. For the following reasons, it is not recommended as a COC:
  - Low frequency of TRV exceedance in sculpin prey samples (9 of 111 [8.1%] prey samples, with maximum HQ = 2.2; and 1 of 1,348 [< 0.1%] sediment samples)
  - Weakness of the Chinook exposure estimate (juvenile Chinook were conservatively presumed to feed predominantly on benthic organisms; this feeding strategy is contrary to the literature, which shows they feed predominantly on pelagic organisms)
  - Uncertainty about the toxicological effects associated with the TRV (rockfish LOAEL setting the TRV is 2 to 3 orders of magnitude below the nine NOAELs from other studies, including four NOAELs and two LOAELs for salmonids)

[173] Antimony can be mixed with lead as a hardener for lead-based products (ATSDR 1992). For example, one fish tackle supplier notes that fishing sinkers contain 94% lead and 6% antimony for hardness and color (Blue Ocean Tackle 2011).

   - Low magnitude of juvenile Chinook salmon dietary HQ (3.5 assuming mixed prey diet) when taking into account the likelihood that both exposure and effects are overestimated (per the two previous items)
   - Discordance of the dietary LOE with the surface water and tissue-residue LOEs (the cadmium AWQC is based on a very large dataset so is the strongest LOE; the tissue-residue LOE is weak because fish sequester or otherwise bioregulate inorganic metals)

- Copper poses potentially unacceptable risk based on the fish tissue-residue, fish dietary, sandpiper dietary, and the benthic invertebrate tissue-residue LOEs. For the following reasons, copper is not recommended as a fish COC:

   - Weakness of the tissue-residue LOE for inorganic metals (fish can actively bioregulate copper tissue concentrations; invertebrates sequester copper and in the case of crayfish, copper forms the basis of their hemoglobin)
   - Irreproducible toxicological effects associated with the dietary TRV (selected LOAEL could not be replicated in subsequent tests with the same species)
   - Selected LOAEL is barely above range of nutritional requirements found in the literature for some fish species
   - Discordance of the tissue and dietary LOEs with the stronger water LOE (which is based on numerous exposure data and a very large AWQC dataset showing that fish are not among the most sensitive species; absence of HQ ≥1 via the water LOE is the strongest evidence for drawing risk conclusions)
   - Similarity of fish tissue concentrations in the Study Area and upriver

  For the following reasons, copper is not recommended as a shorebird COC:

   - Unlikely ecological significance of prey organism TRV exceedance (tissue-residue HQ ≥ 1 in only one prey item, laboratory-exposed worms; HQs < 1 for a mixed-species diet).
   - The selected TRV is less than the lowest bounded literature-reported NOAEL for birds.
   - Low magnitude of TRV exceedance (maximum HQ = 1.3) considering the likely overestimates of exposure and effects (per the two previous items)

  For the following reasons, copper is not recommended as a benthic invertebrate COC:

   - Low magnitude of TRV exceedance (maximum HQ = 2.6)
   - Weakness of the tissue-residue LOE for inorganic metals (invertebrates sequester copper and in the case of crayfish, copper forms the basis of their hemoglobin)

- Lead poses potentially unacceptable risk based on the tissue-residue LOE for peamouth and smallmouth bass, and on the dietary LOE for osprey and mink. It is not recommended as a fish COC for the following reasons:
  - Low frequency of tissue TRV exceedance (2 of 32 [6.2%] smallmouth bass and 1 of 4 [25%] peamouth samples)
  - Weakness of the exposure estimate (smallmouth bass concentration yielding high HQ [280] is an outlier for both antimony and lead in the same sample, suggesting that a fish in the composite sample might have swallowed a fishing sinker)
  - Discordance of tissue-residue LOE with dietary and water LOEs (based on a very large dataset, the lead AWQC is the strongest LOE; the tissue-residue LOE is weak because fish generally can sequester or otherwise bioregulate inorganic metals; the dietary LOE is more likely to overpredict than underpredict risk)

  Lead is not recommended as a bird or mammal COC because the only sample yielding an HQ ≥ 1 is the same outlier smallmouth bass sample as identified for antimony above

- Zinc poses potentially unacceptable risk for fish (sculpin, bass, pikeminnow), amphibians, and aquatic plants based only on the surface water LOE. It poses a potentially unacceptable risk to benthic invertebrates based on the surface water and tissue-residue LOEs. It is not recommended as a COC for the following reasons:
  - Low frequency of surface water TRV exceedance for all receptors (1 of 167 samples [< 1%], with maximum HQ = 1.2)
  - Discordance of the stronger surface water LOE with the weaker tissue-residue and dietary LOEs for fish (surface water toxicity data were sufficient to derive AWQC; tissue-residue LOE is weak because fish generally can sequester or otherwise bioregulate inorganic metals; the dietary LOE is relatively weak because the TRV is based on only two studies)
  - The tissue-residue LOE for benthic invertebrates is a weak LOE

Organometals

- Mercury poses potentially unacceptable risk based on the dietary LOE for sculpin and bald eagle. It is not recommended as a fish COC because the dietary TRV was exceeded in only 1 of 1,345 sediment samples (< 0.001%) and in no tissue samples. Mercury is not recommended as an eagle COC for the following reasons:
  - Discordance between the dietary and tissue-residue LOEs

- Possible overestimate of bald eagle exposure when using osprey exposure as a surrogate because of greater proportion of terrestrial prey in the bald eagle diet
- Low HQ (maximum HQ = 1.7) given the discordant LOEs and possibility that exposure is overestimated (per the previous two items)
- Higher concentrations in upriver fish tissue than in Study Area fish tissue

- Monobutyltin poses potentially unacceptable risk based on the surface water LOE. It is not recommended as a COC for three reasons:
  - Low frequency of surface water TRV exceedance (1 of 167 samples [< 1%])
  - Likely overestimate of toxicological effects associated with the TRV (which is based on the more toxic TBT)
  - Low magnitude of exceedance (HQ = 1.2) considering the likely overestimate of effects and limited spatial extent of HQ ≥ 1 (per the previous two items)
- TBT poses potentially unacceptable risk based on the dietary LOE for sculpin and tissue-residue LOE for benthic invertebrates. It is not recommended as a COC for fish for the following reasons:
  - Single dietary TRV exceedance (based on 1 lab worm sample of 81 prey samples [1.2%] and only when combined with sediment ingestion)
  - Low magnitude of exceedance (maximum HQ = 1.0)
  - Uncertainty about toxicological effects associated with the TRV (reproduction success was reduced at the TRV, but not dose-responsive)
  - Discordance of dietary LOE with the tissue-residue and water LOEs (TBT tissue residue is noted to be reliable predictor of toxicity and is the strongest LOE(Meador et al. 2002a))

  It is not recommended as a COC for benthic invertebrates because of the following:
  - The TRV was exceeded in empirical bioaccumulation samples only at one location.
  - While predicted tissue residues exceeded the TRV more frequently, the moderate strength of the regression was highly influenced by the one high value in the dataset. The predicted tissue residues are uncertain and not supported by empirical data.
  - The TRV is uncertain due to the inclusion of imposex—the endpoint that defined the lower distribution of the SSD, which set the TRV

PAHs[174]

Benzo(a)anthracene, benzo(a)pyrene, and naphthalene pose potentially unacceptable risk to benthic invertebrates, fish, amphibians, and aquatic plants based on the surface water LOE. Benzo(a)pyrene poses potentially unacceptable risk to spotted sandpiper based on the dietary LOE. None of these three individual PAHs is recommended as a COC for assessing potential remedy protectiveness of ecological receptors.[175]

- Benzo(a)anthracene is not recommended as a COC for two reasons:
  - Low frequency of surface water TRV exceedance (2 of 245 samples [< 1%], both between RM 6.4 and RM 6.5[176]
  - Discordance of surface water LOE with dietary LOE for fish (benzo(a)anthracene did not screen in as a fish COPC by the dietary LOE)
- Benzo(a)pyrene is not recommended as a COC based on the surface water LOE for two reasons:
  - Low frequency of surface water TRV exceedance (3 of 122 [2.4%] near-bottom surface water samples, all from RM 6.4 to RM 6.5)[177]
  - Discordance of the surface water LOE with the dietary LOE for fish (benzo(a)pyrene did not screen in as a fish COPC by the dietary LOE)
- Benzo(a)pyrene is not recommended as a COC based on the bird dietary LOE for two reasons:
  - Low frequency of dietary TRV exceedance for spotted sandpiper (1 of 27 [3.7%] lab worm samples assuming lab worm-only diet; all HQs < 1 for clam-only diet)
  - Low magnitude of exceedance (maximum HQ = 1.6) considering potential overestimate of exposure by presuming lab worm-only diet
- Naphthalene is not recommended as a COC for two reasons:
  - Low frequency of surface water TRV exceedance (10 of 268 [3.7%] samples, all from west side of RM 6.4 to RM 6.5 during a single sampling event [the May 2005 non-LWG sampling event])[178]

[174] Risk management recommendations regarding PAHs as they relate to risks from the TZW LOE and benthic AOCs are discussed separately in Sections 12.2 and12.3, respectively.

[175] In the TZW LOE, however, concordance of surface water and TZW exceedances at RM 6.4 to RM 6.5 supports identification of benzo(a)anthracene, benzo(a)pyrene and naphthalene as COCs for this location (see Section 12.2).

[176] In the TZW LOE, however, concordance of surface water and TZW exceedances at this sampling location support identification of benzo(a)anthracene as a COC for this location (see Section 12.2).

[177] In the TZW LOE, however, concordance of surface water and TZW exceedances at this sampling location support identification of benzo(a)pyrene as a COC for this location (see Section 12.2).

  - Discordance of the surface water LOE with the dietary LOE for fish (naphthalene did not screen in as a fish COPC by the dietary LOE)

Phthalates

Neither of the two phthalates is recommended as a COC:

- BEHP poses potentially unacceptable risk based on the benthic invertebrate and fish tissue-residue and surface water LOEs. It is not recommended as a COC for several reasons:
  - Low frequency of surface water TRV exceedance (2 of 190 samples [1.1%])
  - Low frequency of fish tissue-residue TRV exceedance (1 of 38 sculpin samples [2.6%], 2 of 32 smallmouth bass samples [6.3%]) and low frequency of the benthic invertebrate tissue-residue TRV exceedance (1 of 35 clam samples or 3%)
  - Low magnitude of exceedance for fish tissue TRV (maximum HQ = 2.9) and for benthic invertebrate TRV (maximum HQ = 2.8)
  - Absence of toxicological effects associated with the tissue TRV (which is based on an unbounded NOAEL)
  - Absence of relationship between concentrations in co-located sediment and tissue samples
- Dibutyl phthalate poses potentially unacceptable risk based on the dietary LOE for spotted sandpiper. It is not recommended as a COC for several reasons:
  - Low frequency of dietary TRV exceedance (1 of 28 clam samples [3.6%], no worm samples)
  - Low magnitude of dietary TRV exceedance (maximum HQ = 1.4 for clam-only diet; maximum HQ < 1 for worm-only diet)
  - Absence of a relationship between concentrations in co-located sediment and tissue samples
  - Higher sediment concentrations in background than in Study Area

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[178] In the TZW LOE, however, concordance of surface water and TZW exceedances at this sampling location support identification of naphthalene as a COC for this location (see Section 12.2).

Pesticides

None of the three organochlorine pesticides is recommended as a COC for assessing potential remedy protectiveness of ecological receptors:

- Aldrin poses potentially unacceptable risk based on the dietary LOE for spotted sandpiper. It is not recommended as a COC for two reasons:
  - Low frequency of dietary TRV exceedance (1 of 27 lab worm samples [3.7%])
  - Low magnitude of exceedance (maximum HQ = 1.4 based on the only lab worm sample that yields an HQ ≥ 1; HQ < 1 for clam-only and mixed diets)
- Total DDx poses potentially unacceptable risk based on the tissue-residue LOE for sculpin and benthic invertebrates; the dietary LOE for spotted sandpiper; the egg LOE for bald eagle; and the surface water LOE for the benthic community, sculpin, amphibians, and aquatic plants. The rationale for exclusion from the list of recommended COCs varies with LOE.[179] DDx is not recommended as a COC for the following reasons:
  - Low frequency of TRV exceedance (1 of 170 samples [<1%]) in surface water based on N-qualified data, indicating interference from another analyte
  - Low frequency of exceedance in empirical benthic tissue residue (2 of 35 worm samples or 6%)
  - Low frequency of exceedance in predicted benthic tissue residues (up to 15 samples of 1,128 or 1.3%) and approximately half of which are based on N-qualified data
  - Low frequency of TRV exceedance (2 of 27 lab worm samples [7.4%]) used in the dietary LOE for sandpiper
  - Low magnitude of exceedance of TRV for sandpiper diet (maximum HQ = 1.4 assuming lab worm-only diet; HQ < 1 for all clam-only and mixed diets)
  - Questionable relevance of estimated exposure for the bird egg LOE for bald eagle (there is significant uncertainty about the source of DDx residues in the osprey eggs collected from the Study Area because the adults overwinter in Mexico and Central America, nesting and laying eggs shortly after returning to the lower Willamette (Henny et al. 2004)
  - Potential risk of adverse effects on bald eagles is present because NOAEL HQs are ≥ 1 in eggs from two of five exposure areas;because both were below the LOAEL, there is no empirical evidence of potential risk.

[179] Total DDx and 4,4′-DDT are recommended as TZW COCs in the TZW sampling area at ~ RM 7.4W (see Section 12.2).

  - All egg total DDx concentrations were below the recommended effects threshold reported in Elliott and Harris (2001\2002) based on a comprehensive review of the available bald eagle toxicological effects data
  - Absence of relationship between concentrations in osprey egg samples and nearby sediment (NOAEL HQ ≥ 1 in eggs from two of five exposure areas, but NOAEL HQ < 1 in eggs from where sediment DDx concentrations were highest)
  - Discordance of LOEs (mixed species dietary NOAEL HQs < 1 in all exposure areas)
- 4,4′-DDD poses potentially unacceptable risk based on the tissue-residue LOE for benthic invertebrates. This contaminant is not recommended as a COC for the following reasons:
  - Low frequency of TRV exceedance (1 of 35 samples or < 3%)
  - Low magnitude of the exceedance (HQ = 1.2)

VOCs

Two VOCs (ethylbenzene and trichloroethene) measured in surface water exceeded their respective TRVs; however, neither is recommended as a COC based on the following rationale:

- Low frequency of exceedance; TRV exceeded in 1 of 23 (4%) samples collected from ~ RM 6.5 (west bank) during one sampling event
- Low magnitude of exceedance of the TRV for ethylbenzene (HQ = 1.6)

12.1.3 Risk Management Recommendations for Recommended COCs

Based on the information presented in Section 12.1, total PCBs and total TEQ pose the primary risks to fish and wildlife. The remainder of this section provides additional risk management recommendations for these recommended COCs:

- Section 12.1.3.1 recommends the use of mink to evaluate total PCB and total TEQ remedies.
- Section 12.1.3.2 examines relationship between PCB and TEQ risk.
- Section 12.1.3.3 discusses potential problems with the use of the bird egg LOE as an evaluation tool for potential remedies.

12.1.3.1 Receptors of Concern for Purposes of Assessing the Protectiveness of Potential Remedies in the FS Analysis of Alternatives

Total PCBs is recommended as a COC because exposure poses a risk of ecologically significant adverse effects to mink and river otter populations. Total PCBs also poses lower risk of ecologically significant adverse effects to benthic invertebrates, spotted sandpiper, osprey, sculpin, and smallmouth bass populations and to bald eagles. Total

TEQ is recommended as a COC because exposure poses a risk of ecologically significant adverse effects to mink populations. Total TEQ also poses lower risk of adverse effects to river otter, spotted sandpiper, and osprey populations and to bald eagles.

For the dietary LOE, HQs are a function of food and sediment ingestion rates relative to the organism's body weight, the COPC concentrations in prey and sediment, and the TRV. Of the receptors at risk from PCBs and total TEQ via the dietary LOE, mink has the lowest TRVs. The bird PCB LOAEL TRV is higher than that of mink by a factor of 16, and the bird total TEQ LOAEL TRV is higher than that of mink by a factor of 64, indicating that risk to mink occurs at lower dietary doses.

Given the same sediment and prey data, dietary risk estimates for mink will always be higher and more widespread than those for the other receptors. Food and sediment ingestion rates as a function of body weight are higher for mink than for otter; and they are higher for birds than for mink (by a factor ranging from 1.3 for osprey to 7 for spotted sandpiper). However, the difference in TRVs (for both total PCBs and total TEQ) more than offsets the difference in ingestion rates. Although a mink population is not known to be present in the Study Area, mink are assumed to forage in all areas of the Study Area and to prey on small- and large-home-range fish. Analysis of remedial alternatives for mink will thus be protective of other receptors in the Study Area potentially affected by PCBs and dioxins.

Predicted mink risk is based on species-specific effects data, making mink risk predictions a relatively strong basis for risk management decisions. This is not the case for the other receptors (predicted risks are not based on species-specific effects data), whose conclusions therefore provide a less certain basis for risk management recommendations. Because the available data suggest that mink are quite sensitive to PCBs and dioxins/furans, and probably more so than the other receptors at risk, the mink population should be the receptor of concern when assessing ecological risk reduction for the remedial alternatives (for total PCBs and total TEQ).

Because protection of other receptors by mink is contingenent on the habitat use, prey, and home-range assumptions used for the BERA, any alteration of these assumptions for analysis of uncertainties in the FS should be examined to ensure that protection of all receptors at risk from PCBs and TEQ are still protected under alternate assumptions for mink.

Because the relationship between sediment contamination and bird egg tissue concentrations is highly uncertain, the tissue-residue LOE has limited utility as a tool for assessing the protectiveness of potential remedies in the FS analysis of alternatives. This is discussed further in Section 12.1.3.3.

12.1.3.2 Relationship Between PCB and TEQ Risk

Total TEQ is the sum of multiple PCB and dioxin/furan congeners, each weighted by their toxicity relative to that of the most toxic congener (2,3,7,8-TCDD). TEQ concentrations for birds and mammals were calculated as the sum of individual PCB and dioxin/furan congener concentrations weighted by their TEFs. The PCB TEQ is the TEF-weighted sum of only the dioxin-like PCB congener concentrations, the total dioxin/furan TEQ is the TEF-weighted sum of only the dioxin/furan congener concentrations, and total TEQ is the sum of the PCB TEQ and the total dioxin/furan TEQ. TEF values for a given congener generally fall within a range of about an order of magnitude for mammals (Sanderson and Van den Berg 1999); TEFs for birds are more uncertain (Van den Berg et al. 1998). Because of this uncertainty, TEQ risks may be over- or underestimated.

As with total PCBs, mink is the receptor most sensitive to dioxins/furans and subject to the greatest spatial extent of TEQ risk in the Study Area. PCBs are responsible for the majority of total TEQ risk, in that PCB TEQ HQs generally constitute the majority of the total TEQ HQs. For example, of the 15 (out of 109) potential prey samples with mink total TEQ HQ ≥ 1, 7 exceed the TRV for PCB TEQ but only 4 exceed the TRV for total dioxin/furan TEQ (see Attachment 17). No individual samples result in an exceedance of both the PCB TEQ TRV and the dioxin/furan TEQ.

Because total TEQ risk is largely driven by PCB, and redundant with total PCB risk (with the four exceptions noted above), and because adverse effects in mink are better correlated with total PCB exposures than with TEQ exposures (Fuchsman et al. 2007), the FS analysis of alternatives should focus primarily, but not exclusively, on evaluating whether remedies protect the mink population from risk due to exposure to total PCBs.

12.1.3.3 Bird Egg LOE and the FS

PCBs and total TEQ pose low risk to birds based on the tissue-residue LOE. It is recommended that the bird egg LOE not be used to develop and evaluate remedial alternatives in the FS. Risk to osprey and bald eagle based on the egg LOE cannot be directly compared with dietary risks. Egg tissue concentrations might reflect exposure to contaminated prey from the Study Area. Alternatively, inasmuch as osprey lay eggs shortly after returning to the Study Area from overwintering in Mexico and Central America, the egg residues might reflect exposure to contaminants outside of the Study Area. Furthermore, the bioaccumulation relationship from prey to egg is not well-characterized, rendering predictions based on this relationship highly uncertain.

A statistical evaluation was conducted to determine if a relationship between fish tissue and bird egg tissue concentrations in the Study Area could be expressed using biomagnification regressions (BMRs). A BMR expresses the relationship between fish prey and bird egg tissue concentrations based on co-located data rather than based on an average ratio. BMRs were calculated based on the method by Burkhard (2009) using co-located (within 1 mile) composite fish tissue and egg concentrations from seven locations throughout the Willamette River (Henny et al. 2003; 2009). Several possible linear tissue-sediment models were screened. No significant relationship (i.e., no BMR) could be found for any bird egg COPC based on the the criteria of a significantly positive slope at a $p = 0.05$ and an $r^2 > 0.030$, except total TEQ ($r^2 = 0.52$). For total TEQ, application of the BMR to the Study Area requires extrapolation outside of the dataset, thus rendering the relationship uncertain. The implication is that the available dataset is insufficient to estimate a reliable BMR.

Because mink is the receptor most sensitive to PCBs and dioxins/furans, it is recommended that from an ecological risk management perspective, FS analyses should focus primarily on the mink dietary risk reduction associated with the remedial alternatives.

12.2 TZW RISK MANAGEMENT RECOMMENDATIONS

The TZW LOE was used to assess risks to benthic invertebrate, benthic fish (i.e., sculpin and lamprey ammocoetes), aquatic plant, and amphibian populations and communities. Pacific lamprey are identified in EPA's Problem Formulation as a "species of special concern" with direction to assess risk at the organism level. Measured TZW concentrations exceed water TRVs in all of the TZWsampling areas; by EPA's direction individual lamprey ammocoetes are exposed to potentially unacceptable risk. The degree to which TZW poses potentially unacceptable risk to individual lamprey ammocoetes is uncertain. Lamprey ammocoete toxicity testing has demonstrated their relative insensitivity to toxicants across six modes of action (Andersen et al. 2010). It is probable that the BERA overestimates both lamprey ammocoete exposure and effects, to an unquantified degree.

The TZW samples evaluated in this assessment were collected primarily during a 2005 sampling effort focused offshore of nine[180] upland sites with known or likely pathways for discharge of upland contaminated groundwater. The primary objective of the RI groundwater pathway assessment was to evaluate whether transport pathways from upland contaminated groundwater plumes to the river were complete. Therefore, TZW target analyte lists varied from site to site and were derived primarily based on the COIs in the upland contaminated groundwater plumes. Consequently, not all COIs in sediments were analyzed in TZW samples. As described in Sections 4.4.3.1 and 6.1.5.2 of the draft final RI (Integral et al. 2011), there also might be other groundwater plumes in the Study Area discharging into river sediments where TZW samples have not been collected.

TZW sampling focused on sites with contaminated groundwater pathways that were a potential concern. Where these groundwater pathways are confirmed to be a concern, they will be addressed through source control. Source controls should be in place prior to implementation of sediment remedies, particularly those associated with upland sources (EPA 2002b, 2005a) in order to prevent recontamination. These source control actions will reduce contaminant flux to the river and accelerate recovery. Source controls will reduce baseline risk by intercepting ongoing contaminant migration. While the residual contaminated groundwater plumes may remain near the mudline, they will attenuate over time. Because source controls should precede the sediment remedy, the magnitude of

[180] The area offshore of the Arkema site was divided into two areas (the acid plant area and the chlorate plant area).

potential risk identified in the BERA should be diminished when the sediment remedy is implemented.

The TZW LOE was evaluated by comparing TZW COPC concentrations in individual samples to water effect thresholds. EPA directed the LWG to assume that benthic organisms would be exposed to undiluted shallow (0 to 38 cm) TZW, an assumption that the LWG found to be highly conservative. As discussed in Section 6.6.3.3, actual TZW exposure is probably much lower because of feeding habits, burrowing behavior, avoidance of low oxygen levels at the TZW sample depths, and low food content in sediments at the TZW sample depths.

It is recommended that only those TZW COPCs with HQ ≥ 100 be considered as COCs to develop and evaluate remedial alternatives that are protective of ecological resources.[181] This recommendation is based on two factors. First, by definition any contaminant with HQ ≥ 1 poses potentially unacceptable risk, but the evidence presented in Section 6.6.3.3 strongly supports the position that the potential for unacceptable risk at HQs < 10 is very small. Therefore, a factor of 10 was applied to account for the evidence that benthic receptors are not directly exposed to undiluted TZW. Second, EPA guidance (EPA 2005a) states that remedies should be evaluated under the assumption that sources of COPCs to the groundwater plume have been controlled. The effect of source control should be to reduce the potential flux of groundwater COPCs into the shallow transition zone prior to sediment remediation. An additional factor of 10 was applied to account for the control of COPC sources.

Almost all metals measured in TZW are common crustal elements. Barium, iron, and manganese are among the most common metals associated with sediments. These same metals are also associated with the highest HQs in the risk characterization, but there is substantial uncertainty that their source is ubiquitously anthropogenic. It is recommended that TZW concentrations of these metals not be used to assess remedy effectiveness.

Given the foregoing, TZW COC recommendations for each site are provided in Table 12-2.

[181] There is uncertainty associated with 4,4′-DDT and total DDx as COCs because HQs based on filtered samples are less than 100. This suggests that the risk from DDx compounds in TZW may be lower than indicated by the maximum concentrations in unfiltered samples due to lower bioavailability of the particulate bound fraction of the contaminant.

Table 12-2. COC Recommendations for COPCs with HQs ≥ 100 at TZW Sampling Areas

	Maximum HQ ≥ 100									
		Arkema								
COPC	ARCO	Acid Plant	Chlorate Plant	Exxon Mobil	Gasco	Gunderson	Kinder Morgan	Rhône-Poulenc	Siltronic	Willbridge
Contaminants Recommended as TZW COCs										
Benzo(a)anthracene					120				1,200	
Benzo(a)pyrene					210				2,700	
Naphthalene					260				1,100	
4,4′-DDT		160[a]								
Total DDx		280[a]								
Chlorobenzene		190								
cis-1,2-Dichloroethene									110	
Trichloroethene									1,900	
Cyanide					4,400				130	
Carbon disulfide					870					
Contaminants Not Recommended as TZW COCs										
Barium (total)		610	1,100					170		
Iron (total)		110	250	110	130				180	120
Manganese (total)			550	150	130			130		110
Gasoline-range aliphatic hydrocarbons C10-C12[b]					540				150	

[a] Maximum HQs are based on unfiltered samples. Maximum HQs for filtered samples would be 2.8 for 4,4′-DDT (however, this contaminant was never detected) and 14.5 for total DDx.

[b] Petroleum hydrocarbons may contribute to risks to ecological receptors; however, petroleum is not considered a CERCLA contaminant.

CERCLA –Comprehensive Environmental Response, Compensation, and Liability Act
COC – contaminant of concern
COPC – contaminant of potential concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
HQ – hazard quotient
total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT, and 4,4′-DDT)

Potential remedies should be evaluated in the FS for the degree to which they protect benthic invertebrate communities and individual Pacific lamprey ammocoetes from risk due to contaminated groundwater discharge, assuming that groundwater source control measures have been implemented.

12.3 BENTHIC RISK MANAGEMENT RECOMMENDATIONS

The primary LOE for identifying benthic community risks is based on sediment toxicity (both measured and predicted based on multivariate statistical models [FPM and LRM]); however, the risk assessment methodologies are designed to address chemical mixtures. The results are correlative and do not conclusively identify contaminants causing toxicity.[182] Contaminants whose sediment concentrations, when considered as a group (i.e, in aggregate), appear to help explain the observed toxicity based on the FPM and LRM are presented in Table 12-3.[183]

Table 12-3. Contaminants Potentially Contributing to Benthic Risk Based on Predicted Sediment Toxicity LOE

Contaminant	
Metals	
Cadmium	Lead
Chromium[b]	Mercury[b]
Copper	Silver
PAHs	
2-Methylnaphthalene	Dibenzo(a,h)anthracene
Acenaphthene	Fluoranthene
Acenaphthylene	Fluorene
Anthracene	Indeno(1,2,3-cd)pyrene
Benzo(a)anthracene	Phenanthrene
Benzo(b)fluoranthene	Pyrene
Benzo(b+k)fluoranthene	Total HPAHs
Benzo(g,h,i)perylene	Total LPAHs[b]
Benzo(k)fluoranthene	Total PAHs

[182] Risk conclusions based on the secondary benthic LOEs—tissue residue, surface water, and TZW—can identify COCs and are noted in Sections 12.1 and 12.2, where these LOEs support the identification of COCs.

[183] The contaminant list is a combination of SQVs derived using the FPM and the LRM. Each SQV has a different reporting basis depending on the normalization selected for the model. All FPM SQVs are dry-weight normalized. LRM SQVs used a number of different normalizations including dry-weight, organic carbon, percent fines and combinations of normalizations.

Table 12-3. Contaminants Potentially Contributing to Benthic Risk Based on Predicted Sediment Toxicity LOE

Contaminant	
Chrysene	
Phthalates	
Dibutyl phthalate	
SVOCs	
Benzyl alcohol	Dibenzofuran[b]
1,2-Dichlorobenzene	Carbazole[b]
Phenols	
4-Methylphenol[a]	Phenol
PCBs	
Total PCBs[b]	
Pesticides	
2,4′-DDD	beta-HCH
4,4′-DDD	delta-HCH[b]
4,4′-DDE	Dieldrin
4,4′-DDT	Endrin
Sum DDD[b]	Endrin ketone
Sum DDE	cis-Chlordane
Sum DDT	Total endosulfan[b]
Total DDx	
Petroleum Hydrocarbons	
Diesel-range hydrocarbons	

[a] All SQVs derived from the FPM are less than the apparent effect threshold and therefore may contribute to false predictions of toxicity.

[b] FPM SQVs based on one or two endpoints are less than the apparent effect threshold and may contribute to false predictions of toxicity

COPC – contaminant of potential concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
FPM – floating percentile model
HCH – hexachlorocyclohexane
HPAH – high-molecular-weight polycyclic aromatic hydrocarbon
LOE – line of evidence
LPAH – low-molecular-weight polycyclic aromatic hydrocarbon
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
SVOC – semivolatile organic compound
total DDx – sum of all six DDT isomers (2,4′-DDD, 4,4′-DDD, 2,4′-DDE, 4,4′-DDE, 2,4′-DDT, and 4,4′-DDT)

Because the primary benthic LOE (bioassay results) does not identify the cause of the empirical toxicity (i.e., specific COPCs or other factors), the risk management recommendations focus on two other questions:

1. Where were potentially unacceptable benthic community risks occurring in the Study Area at the time of the BERA data collection?
2. What tools from the BERA can be used in the FS analysis of alternatives to assess the effectiveness of potential remedies on protecting the benthic community?

The remainder of this section is arranged around these two questions. Section 12.3.1 outlines the guidelines EPA provided about how to answer them. Section 12.3.2 answers the first question by presenting recommended benthic AOCs. Section 12.3.3 answers the second question by recommending tools by which to assess the effect of potential remedies on the benthic community in the FS analysis of alternatives.

12.3.1 EPA Guidelines for Evaluating Benthic Risk in the Feasibility Study

The LWG and EPA have been working on benthic risk management recommendations since early 2010, following guidelines EPA in an April 21, 2010 letter (EPA 2010a). The guidelines provide direction for evaluating benthic risk in the draft FS. Specifically, EPA described its primary goals for the FS analysis of alternatives for benthic assessment endpoints:

- Define areas that pose unacceptable risk to the benthic community
- Define the areas and volume of contamination that may pose risk to the benthic community
- Evaluate remedial action alternatives and effectiveness (did it meet the RAO)

The letter also provided guidelines for evaluating remedy effectiveness:

- All benthic SQGs in the March 24, 2010 list will be included in the analysis. If specific SQGs are found to be inconsistent with other LOEs listed below, EPA will review the analysis and determine whether these should be included in the draft FS.[184]
- Sediment toxicity bioassays will form the primary LOE for this analysis. The sediment toxicity LOE will include level 2 (moderate) and level 3 (severe) effects for all endpoints (chironomus [sic] biomass and mortality and hyalella [sic] biomass and mortality).

[184] The SQVs have subsequently been revised based on additional modeling and negotiations between the LWG and EPA, as documented in item 11 of Attachment B to a January 12, 2011, LWG letter to EPA (LWG 2011a), the attachment to a February 25, 2011, RI/FS schedule letter from EPA to the LWG (Humphrey 2011), and the LWG's March 9, 2011, draft response (LWG 2011b) to EPA's February 25, 2011, letter.

- The analysis will consider the number and degree of exceedance of SQGs.
- The analysis will consider other LOEs such as TZW compared to ambient water quality criteria for the protection of aquatic life and benthic tissue TRVs.
- The analysis will consider the presence/absence of nearby sources and examine benthic community structure (e.g., via sediment profile imaging and related information).
- The analysis will consider data quality and data density issues for the SQGs.

The LWG's implementation of these guidelines is known by EPA and the LWG as the "comprehensive benthic approach." Developed by the LWG after receiving the EPA's April 21, 2010, directives and guidelines (EPA 2010a), the comprehensive benthic approach was first presented informally to EPA (Eric Blischke and Burt Shephard) by the LWG (John Toll and Jim McKenna) on July 20, 2010, to elicit early feedback. It was formally presented to EPA during the September 29, 2010, LWG Small Technical Group Benthic Toxicity AOPCs Meeting with EPA. Item 11 in Attachment B to the LWG's January 12, 2011, letter to EPA (LWG 2011a), and the attachment to EPA's February 25, 2011, response letter to the LWG (Humphrey 2011) document the decision to proceed with an updated version of the comprehensive benthic approach.

12.3.2 Recommended Benthic Areas of Concern for FS Evaluation

Recommended benthic AOCs, based on the LWG's application of the comprehensive benthic approach upon completion of the draft final BERA, are shown on Maps 12-1a and 12-1b. Sediment toxicity bioassays form the primary LOE for the comprehensive benthic approach used to delineate the recommended benthic AOCs, as per the EPA April 21, 2010, guidelines (EPA 2010a). Predicted toxicity (based on multiple sets of SQVs) and tissue residues (both empirical and predicted) provide secondary LOEs to identify benthic risk areas. TZW and surface water were used as supporting LOEs.

SPI data were not used in the development of AOCs because the sampling program was not designed to link SPI image locations with toxicity sampling locations and in turn allow an assessment of the relationship between benthic community successional stage and contaminant effects. Details of the approach used to identify recommended benthic AOCs are as follows:

- Locations with empirical bioassay results indicating significant toxicity were identified.
  - One toxicity endpoint (*Chironomus* biomass or growth, *Hyalella* biomass or growth) exceeding an L3 threshold or two endpoints exceeding an L2 endpoint were considered significant toxicity.

- Locations where significant sediment toxicity is predicted based on sediment chemistry exceeding an MQ of 0.7 or a pMax of 0.59 were identified.
  - Sampling locations where both the MQ and the pMax thresholds were exceeded were considered toxic.
  - Sampling locations where neither the MQ or pMax threshold was exceeded were considered non-toxic.
  - Sampling locations where the models disagreed (i.e., either the MQ or the pMax threshold was exceeded, but not both) were considered uncertain.
- Locations where empirical tissue residues or, in the absence of empirical tissue residue data, predicted tissue residues exceeded their TRVs were identified.
  - The evidence of risk provided by measured or predicted exceedance of metals TRVs was considered weak because of species-specific differences in metals sequestration or other bioregulation.
  - The evidence of risk provided by predicted exceedance of the TBT TRV was considered weak because of high uncertainty in the TBT bioaccumulation model.
- TZW exceedance areas with HQs > 100 were delineated.
- All LOEs were overlaid on a map.
  - Areas where two or more adjacent empirical bioassay sampling locations indicate significant toxicity were identified as benthic AOCs.
  - Areas where risks were identified at two or more adjacent sampling locations based on chemistry LOEs (predicted toxicity, empirical or predicted bioaccumulation) or a combination of bioassay and chemistry LOEs were identified as benthic AOCs.
  - TZW exceedance areas were identified as benthic AOCs.
- Boundaries of the benthic AOCs split the distance between sampling locations exceeding criteria and surrounding clean sampling locations except where:
  - Other physical features were present (e.g., pier, channel edge, property boundary), in which case the boundary was drawn at the physical features.
  - The nearest sampling sampling location was at a distance greater than 200 ft, in which case the boundary was drawn at a subjective distance less than halfway to nearest sampling location.

12.3.3 Benthic Assessment Tools for the FS Analysis of Alternatives

Bioassays cannot form the primary LOE for the FS analysis of alternatives, because the analysis is of potential future conditions. Therefore, the sediment chemistry LOE, as applied in the comprehensive benthic approach, will have to be used to judge protectiveness of potential remedies. The comprehensive benthic approach uses concordance between an MQ based on the site-specific SQVs and the predicted pMax to identify benthic risk areas. EPA selected the MQ threshold of 0.7 and the pMax threshold of 0.59 that the LWG used in defining benthic AOCs. These same thresholds should be used to evaluate the protectiveness of potential remedies. The analysis of alternatives should also consider whether and how much natural recovery would occur prior to implementing active remedies. Per EPA guidance (EPA 2002b, 2005a), the analysis should presume that source control measures will be in place.

12.4 SUMMARY OF RISK MANAGEMENT RECOMMENDATIONS

The purpose of the risk management recommendations provided in Section 12 is to identify COCs, receptors, and AOCs that the LWG considers necessary and sufficient to develop and evaluate remedial alternatives that are protective of ecological resources. The FS will also evaluate whether remedial alternatives for these COCs, receptor and AOCs address the full list of contaminants potentially posing unacceptable risk.

In summary, the following are recommended as receptor-COC pairs of concern for futher consideration in the FS:

- For non-benthic invertebrate receptors, total PCBs and total TEQ are the recommended COCs. Mink is the recommended receptor of concern. Most of the contaminants posing potentially unacceptable risk were not recommended as COCs for the non-benthic receptors based on risk characterization considerations (magnitude, spatial extent, and ecological significance of HQs ≥ 1). This list includes all the metals, butyltin, phthalate, pesticide, and VOC COPCs.
- For aquatic receptors exposed via TZW, 4,4′-DDT, total DDx,[185] chlorobenzene, benzo(a)anthracene, benzo(a)pyrene, naphthalene, carbon disulfide, cyanide, cis-1,2-dichloroethene, and trichloroethene are the recommended COCs. These recommendations presume that contaminated groundwater source control measures will be implemented prior to sediment remedies. ODEQ is working with upland property owners to implement contaminated groundwater source control measures prior to sediment remedies.
- For benthic organisms, recommended benthic AOCs were mapped by applying the comprehensive benthic approach based on EPA's April 21, 2010, guidelines for assessing benthic risk in the FS (EPA 2010a). The FS analysis of alternatives will have to rely on the predicted toxicity metrics to evaluate potential remedies and should take into account sediment quality changes that will take place before active implementation of remedies.

[185] There is uncertainty associated with 4,4′-DDT and total DDx as COCs because HQs based on filtered samples are less than 100. This suggests that the risk from DDx compounds in TZW may be lower than indicated by the maximum concentrations in unfiltered samples because of the lower bioavailability of the particulate-bound fraction of the contaminant.

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