PORTLAND HARBOR RI

APPENDIX G
BASELINE ECOLOGICAL RISK ASSESSMENT

DRAFT

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Prepared for
The Lower Willamette Group

Prepared by
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RECOMMENDED FOR INCLUSION IN ADMINISTRATIVE RECORD

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LIST OF ACRONYMS

Acronym	Definition
ACR	acute-to-chronic ratio
AET	apparent effects threshold
ANOVA	analysis of variance
AOC	Administrative Order on Consent
AOPC	area of potential concern
ASTM	American Society for Testing and Materials
AVS	acid volatile sulfide
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
BMF	biomagnification factor
BMR	biomagnification regression
BSAF	biota-sediment accumulation factor
BSAR	biota-sediment accumulation regression
bw	body weight
CERCLA	Comprehensive Environmental Response, Compensation, and Liability Act
CFD	cumulative frequency distribution
cfs	cubic feet per second
COC	chemical of concern
COI	chemical of interest
COPC	chemical of potential concern
CRD	Columbia River Datum
CSL	cleanup screening level
CSM	conceptual site model
DDD	dichlorodiphenyldichloroethane

Acronym	Definition
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
EF	exceedance factor
EPA	US Environmental Protection Agency
EPC	exposure point concentration
EqP	equilibrium partitioning
ERA	ecological risk assessment
ERL	effects range – low
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
FS	feasibility study
FWM	food web model
GIS	geographic information system
HCH	hexachlorocyclohexane
HPAH	high-molecular-weight polycyclic aromatic hydrocarbon
HHRA	human health risk assessment
HQ	hazard quotient
ID	identification
IWC	integrated water column
J-qualifier	estimated concentration

Acronym	Definition
LAET	lowest apparent effects threshold
LC50	concentration that is lethal to 50% of an exposed population
LC10	concentration that is lethal to 10% of an exposed population
LCR	Lower Columbia River
LCV	lowest chronic value
LD50	dose that is lethal to 50% of an exposed population
LOAEL	lowest-observed-apparent-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
MDD	minimum detectable difference
N-qualifier	presumptive evidence of a compound
NB	near bottom
NMFS	National Marine Fisheries Service
NN-interpolation	natural neighbors interpolation
NOAEL	no-observed-apparent-effect level
NS	near surface
O&M	operation and maintenance
OC	organic carbon
ODEQ	Oregon Department of Environmental Quality
ODFW	Oregon Department of Fish and Wildlife
% TOC	percent total organic carbon
PAH	polycyclic aromatic hydrocarbon
PBRA	potential benthic risk area
PCB	polychlorinated biphenyl
PEC	probable effects concentration
PEL	probable effects level

Acronym	Definition
PIT	passive integrated transponder
PRE	preliminary risk evaluation
PRG	preliminary remediation goal
PTO	petrogenic (petroleum compound)
PYO	pyrogenic (petroleum compound
RG	remediation goal
RI	remedial investigation
RM	river mile
RSET	Regional Sediment Evaluation Team
SCRA	site characterization and risk assessment
SEM	simultaneously extracted metals
SHA	special habitat area
SIR	sediment ingestion rate
SL	screening level
SL1	screening level 1
SL2	screening level 2
SLERA	screening-level ecological risk assessment
SMB	smallmouth bass
SMDP	scientific/management decision point
SMS	Washington State Sediment Management Standards
SOW	scope of work
SPI	sediment profile imaging
SQG	sediment quality guideline
SQS	sediment quality standards
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
TCDD	tetrachlorodibenzo-*p*-dioxin
TBT	tributyltin

Acronym	Definition
TEC	threshold effects concentration
TEF	toxic equivalency factor
TEL	threshold effects level
TEQ	toxic equivalent
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 oxygen penetrates the sediment surface; 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 chemical 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 fish prey tissue and the concentration of the chemical in piscivorous bird egg tissue using co-located data pairs
biota-sediment accumulation factor	the ratio of the concentration of a chemical 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 chemical in the tissue of an organism and the concentration of the chemical in sediment using co-located data pairs
bioturbation	the disturbance of sediment by the actions of organisms living on or in the bottom
chemical of concern	the subset of chemicals of potential concern with exposure concentrations that are based on ecologically relevant assumptions and exceed toxicity reference values
chemical of interest	chemical detected in the Study Area for all exposure media (i.e., surface water, transition zone water, sediment, and tissue)
chemical of potential concern	the subset of chemicals of interest with maximum detected concentrations that are greater than screening-level effect thresholds
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 chemical 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)
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)

Term	Definition
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 chemical and effects on ecological receptors
effect threshold	a level of chemical 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 chemical 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
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

Term	Definition
hazard quotient	the quotient of the concentration of a chemical 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 chemical that causes a measured response that negatively affects an organism
macroinvertebrate	invertebrate large enough to be seen by the naked eye
macropthalmia	lamprey pupae (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 chemical 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	chemical-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 chemical 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 chemicals of potential concern are identified by comparing maximum chemical concentrations to screening level effect thresholds
sediment quality guideline	a published sediment concentration associated with an adverse effect to benthic 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
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 low and high site-specific SQGs; toxicity thresholds are based on the reference thresholds
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
toxicity threshold	a level of chemical exposure of a receptor above which a particular effect is expected to occur or below which no effect is expected to occur
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 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). 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 the US Environmental Protection Agency (EPA) for conducting the RI/FS. The approach is also consistent with EPA guidance for conducting ERAs (EPA 1997a, 1998).

The overall purpose of the BERA is to determine if deleterious ecological effects from exposure to uncontrolled releases of hazardous substances to the Willamette River may be occurring at the Study Area under current conditions. In this event, the BERA provides information to risk managers to support management decisions on how to protect ecological receptors. Although a preliminary assessment of areas of the river that may be associated with unacceptable risks is made in the draft BERA, the areas that may require remediation (i.e., areas of potential concern [AOPCs]) will be determined in the FS, following the development of sediment preliminary remediation goals (PRGs) that are protective of both ecological receptors and human health.

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 BERA assumed that the entire Study Area represents potential habitat in estimating risks; further evaluation of specific habitat areas should be another key component considered in determining future risk management decisions.

The evaluation of potential unacceptable risks to ecological receptors at the Portland Harbor Superfund Site has been an ongoing and iterative process that has involved 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 formed the basis of the risk estimates in this draft BERA. The risk estimates evaluate ecological receptors under worst-case exposure scenarios (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) and 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, with the exception that organism health was designated by EPA as the assessment endpoint for juvenile Chinook salmon, Pacific lamprey ammocoetes, and bald eagle.

The draft BERA identified ecological chemicals of concern (COCs). COCs come from a longer list of chemicals of interest (COIs). The ERA reduces the list of COIs by following a set of fairly standard, conservative assumptions to identify chemicals of potential concern (COPCs). The risk assessment methods presented in EPA's Problem Formulation were used to evaluate all COPCs in the draft BERA. Additional risk analysis methods, based on more ecologically relevant assumptions, were used in the draft BERA to help focus risk conclusions and identify the COCs that may pose unacceptable risks to ecological receptors.

Key findings of the draft BERA include the following:

- In total, 31 COCs (as individual chemicals, sums, or totals) were identified for the Study Area. The majority of COCs identified in the draft BERA were determined to pose no unacceptable risks to ecological populations or communities.
- Unacceptable ecological risks are primarily from four chemicals: polychlorinated biphenyls (PCBs), dioxins and furans, DDx, and polycyclic aromatic hydrocarbons (PAHs).
- Bioaccumulation of PCBs by receptors and their prey poses the most significant ecological risks of all COCs evaluated. Mink were identified as the ecological receptor at greatest risk and as having the largest spatial extent of risk. Most unacceptable ecological risks from other COC-receptor pairs are spatially co-located with mink PCB risks. The unacceptable ecological risks for other COC-receptor pairs likely would be reduced or eliminated as a collateral benefit of mink PCB remedies. The extent of the collateral benefit will become apparent as AOPCs are developed for the FS.
- Other areas of risk were identified for the benthic invertebrate community based on exceedances of PAH and DDx sediment quality guidelines (SQGs), primarily in scattered areas of the middle reach of the Study Area (River Mile [RM] 5.0 to RM 8.0).
- The spatial evaluation identified approximately 5% of the Study Area as posing unacceptable risk to the benthic community (i.e., Level 3 SQGs or exceeding tissue TRVs).
- Surface water and transition zone water (TZW) results do not identify any unique ecological risk areas, but they do provide useful information for evaluating potential remedies in the FS.
- Exposure to mercury concentrations in fish was found to potentially pose unacceptable risk to individual bald eagles; however, mercury contamination is a greater Willamette River issue requiring watershed-scale risk management.

- Anthropogenic background is an important factor, in addition to site-specific releases, contributing to unacceptable ecological risk in the Study Area.
- Unacceptable risks of the re-exposure of buried contaminated sediment within the Study Area are low on a harbor-wide scale.

These findings provide the framework for analyzing and comparing alternatives to remediate areas of elevated risk to ecological receptors during the FS.

The draft BERA identified uncertainties associated with the risk conclusions. Following the methods of EPA's Problem Formulation, the identification of COCs was conducted using conservative methods and assumptions. This applied even during COC identification, with the consequence that not all COCs pose unacceptable ecological risk. The most important conservative assumption in the draft BERA's COC selection process was the assumption that effects on organisms translate into effects on ecological populations and communities. Populations compensate for individual losses through a variety of ecological processes, so individual-level risks do not necessarily imply population-level risks.

Other examples of conservatism include assumptions about chemical bioavailability and assumptions that reduced effect thresholds (toxicity reference values [TRVs]) to levels that, for example in the case of essential metals, had to be readjusted upward because they were below nutritional requirements. To account for the conservative assumptions used to identify COCs, additional considerations were used to identify only those COCs that potentially pose unacceptable risks to ecological populations and communities. These additional considerations included spatial analysis and further evaluation of the ecological relevance of exposure assumptions, as well as the relevance of selected toxicity thresholds to assessing risks to populations or communities.

An additional uncertainty is the extent of co-occurrence of chemicals (present in a form that can exert a toxic effect) and ecological receptors. For most ecological receptors, all exposure areas throughout the Study Area were assumed to provide some type of habitat; however, a lack of habitat in some areas would contribute to an overestimation of exposure. Risk managers should consider these and other uncertainties when applying the results of the predicted risks in this draft BERA to risk management decisions.

The following sections outline a generalized overview of the problem formulation that provided a framework for the draft BERA, as well as the overall risk characterization conclusions for risks to individual receptor groups (i.e., 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 Ecological PRE (Windward 2005a), and in Appendix G of the Comprehensive Round 2 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 draft BERA.

ES.1.1 Identification of COPCs

The draft BERA followed the steps and procedures laid out in the Problem Formulation document for defining ecological COPCs. Using chemical data for biological tissue, surface sediment, surface water, and TZW, over 100 chemicals, including metals and various organic compounds were identified as COPCs. The following COPCs were identified for each media type:

- **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 –** Nine dietary COPCs were identified for fish, and twenty-five dietary COPCs were identified for birds and/or mammals. Diet COPCs were defined using both tissue and sediment data and were evaluated as part of the fish and wildlife risk evaluation**.**
- **Surface water –** Fifteen COPCs were identified. Surface water COPCs were evaluated as part of the benthic invertebrate, fish, amphibian, and aquatic plant risk evaluation.
- **TZW –** Fifty-nine COPCs were identified. TZW COPCs were evaluated as part of the benthic invertebrate, fish, amphibian, and aquatic plant risk evaluation.

ES.1.2 Refined Conceptual Site Model

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

- **Benthic invertebrate community**[1] – 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[2]

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

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

- **Piscivorous fish populations –** smallmouth bass and northern pikeminnow
- Detritivorous fish individuals – Pacific lamprey ammocoetes
- Sediment-probing invertivorous bird populations – spotted sandpiper
- Omnivorous bird populations[3] – hooded merganser
- Piscivorous bird populations[4] – osprey and bald eagle
- 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)

Consistent with the Problem Formulation, for all receptors and receptor groups evaluated at the community or population level, the lowest-observed-adverse-effect-levels (LOAELs) were used to define COCs. The draft BERA points out that organism-to-population and organism-to-community extrapolation is a source of uncertainty; in many cases, the use of LOAELs will identify COCs that do not pose unacceptable ecological risk to populations or communities (Forbes et al. 2001).

ES.1.3 Analysis Plan

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

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 draft BERA, COPCs were next evaluated on receptor-specific ecologically relevant exposure scales. For dietary risks to fish and wildlife, exposure estimates were also determined using multiple prey species and prey portions reported in the literature. Exposure concentrations were based both on empirical chemical concentrations quantified in the analytical laboratory, and, for some LOEs (i.e., the tissue-residue LOE, the dietary LOE for shorebirds, and the bird-egg LOE), exposure concentrations were also based on predicted values. Toxicity to benthic invertebrates was predicted based on sediment chemical exceedances of site-specific SQGs.

Effects assessment – The effects assessment used two general approaches. For most ecological receptors, the effects of COPCs were assessed through comparison of chemical concentrations in environmental media to chemical- and medium-specific TRVs or SQGs. EPA (2008f) specified the TRVs that were used in the draft BERA. However, some TRVs

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

[4] Bald eagles were evaluated at the individual level.

selected by EPA are associated with significant uncertainty; these TRVs were further evaluated as part of the risk characterization process. The second approach used sediment toxicity bioassays as a direct measure of the effects of sediment chemistry on the survival and growth 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 SQGs; several published sets of SQGs were also evaluated to predict unacceptable risks to the benthic community. All sets of SQGs were tested to establish their reliability as predictors of benthic toxicity in the Study Area. The SQG sets were required to meet an 80% overall reliability criterion, with not more than 20% false positive predictions of benthic toxicity (i.e., predicting toxicity when there was none) and not more than 20% false negative predictions (i.e., predicting no toxicity when there actually was toxicity). Only one of the two models and three sets of published low SQGs met these criteria and were used to evaluate risks in this draft BERA.

Risk characterization and uncertainty analysis – Risk characterization was conducted primarily using hazard quotients (HQs) that were calculated by dividing media-specific concentrations by their respective TRV or SQG. The risk characterization integrated information on contaminant exposure and effects to estimate risks and identify COCs and estimate risks. In total, 31 ecological COCs (as individual chemicals, sums, or totals) were identified in this draft BERA. Uncertainties were evaluated for all LOE and were considered in formulating risk conclusions. The draft BERA conclusions are indeterminate for COCs with data limitations. In some cases, such as where the spatial extent of TRV exceedances is small, or where TRVs are based on very conservative assumptions, a conclusion that a chemical poses negligible risk to a receptor is warranted even for COCs.

For TZW, only a screening-level assessment was conducted; COCs were not identified using this LOE. The TZW collected from the Study Area represents a snapshot of only the limited areas where the data were collected. Furthermore, the ability of benthic organisms to limit their exposure to anoxic porewater diminishes the ecological relevance of a TZW exceedance. The screening-level assessment was sufficient to conclude that potential effects from TZW exposure would generally occur in locations where effects are also predicted from sediment exposure.

ES.2 BENTHIC INVERTEBRATE RISK ASSESSMENT

Twenty-seven COCs (individual chemicals, sums, or totals) were identified for benthic invertebrates based on site-specific SQG, tissue TRV, and surface water TRV exceedances. These COCs included seven metals (aluminum, arsenic, cadmium, copper, mercury, silver, and zinc), tributyltin, butyltin, individual PAHs (benzo[a]anthracene, benzo[a]pyrene, and naphthalene) and PAH sums (total benzofluoranthenes, total high-molecular-weight PAHs, and total low-molecular-weight PAHs), bis (2-ethylhexyl) phthalate (BEHP), phenol, benzyl alcohol, carbazole, total PCBs, two volatile organic compounds (trichloroethene and ethylbenzene), and 5 pesticides (DDx, dieldrin, endrin, endrin ketone, and delta-hexachlorocyclohexane). TZW was also evaluated by comparisons with TRVs for water to identify COPCs; however, as previously stated, no attempt was made to further characterize TZW exposure. COCs do not necessarily indicate the potential for unacceptable risk to

benthic populations or communities because they were defined on the basis of point-by-point exceedances of organism-level effect thresholds, not on population- or community-level endpoints, and this point-by-point spatial scale is not relevant for assessing risks to populations or communities. Rather, these 27 COCs represent a set of chemicals whose sediment concentrations are statistically correlated (as a set) with benthic toxicity.

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. The SPI data analysis was used as corroborative evidence in the draft 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 limited to depositional areas that have received historical releases of contamination.

The potential for benthic risk was determined as follows. First, empirical sediment COC concentrations were mapped, then the chemical concentrations were interpolated, and the resulting estimated chemical concentrations were used to compared with SQGs[5] to predict site-specific risk. Next, these sediment chemistry maps were used (with bioaccumulation models[6]) to predict where organisms might accumulate COCs to concentrations above tissue TRVs. Areas where estimated sediment concentrations exceeded SQGs or where empirical or predicted tissue concentrations exceeded TRVs were identified as potential benthic risk areas (PBRAs).[7] The certainty regarding predictions of unacceptable benthic community risk within PBRAs containing sediment that underwent empirical toxicity testing was evaluated by examining the distribution of chemical concentrations in the non-toxic samples with the toxic samples. Differences or similarities were used to classify the certainty of the predicted risks as probable, uncertain, or unlikely. Empirical tissue-residue data were mapped as points, allowing a visual assessment of concordance across LOEs. Water TRV exceedances were considered along with sediment SQG and tissue TRV exceedances; they were found to co-occur with areas of SQG exceedances. Based on the spatial analysis, PAHs, PCBs, and DDx are the COCs that were found to pose unacceptable risks to the benthic invertebrate community.

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[5] The low and high SQGs were used to identify three sediment effect categories: Level 1 (below the low SQG), Level 2 (between the high and low SQGs), and Level 3 (above the high SQG).

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

[7] The qualifier "potential" is used because the measurement endpoints used to delineate the PBRAs are organism-level endpoints, and the assessment endpoints are the benthic community or populations of benthic species.

The spatial evaluation identified approximately 5% of the Study Area as posing unacceptable risk to the benthic community (i.e., Level 3 or exceeding the tissue TRV). Those areas where benthic unacceptable risk was associated with PCBs were coincident with areas of PCB risks to mink and other higher-trophic-level receptors. Other areas contributing to unacceptable benthic risk are from PAH SQG exceedances on the west side of the river between approximately RM 6.1 and RM 6.6, in portions of the navigational channel between approximately RM 5.1 and RM 5.8, and in a small area along the eastern shoreline at approximately RM 2.8, and DDx exceedances in scattered locations on the western side of the river between approximately RM 6.9 and RM 7.6.

ES.3 FISH RISK ASSESSMENT

Three primary quantitative LOEs were used to characterize risks to fish: the tissue-residue LOE, the dietary-dose LOE, and the surface water 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. Benthic fish exposure to TZW was also evaluated by identifying COPCs; however, as previously stated, no attempt was made to further characterize TZW exposure.

Sixteen COCs were identified for at least one fish species, including six metals, (aluminum, cadmium, copper, lead, mercury, and zinc), tributyltin, butyltin, three individual PAHs (benzo[a]anthracene, benzo[a]pyrene, and naphthalene), BEHP, total PCBs, trichloroethene, ethylbenzene, and DDx. Risk conclusions for each fish COC were determined by evaluating the risk estimates and the reliability of each LOE. Overall, there were negligible risks to the fish (based on species-specific assessment endpoints) in the Study Area from COCs identified in the draft BERA because the spatial extent of TRV exceedances was small, or the TRVs used to calculate HQs were based on organism-level effects that do not translate into population-level responses.

ES.4 WILDLIFE RISK ASSESSMENT

Dietary exposure risks to wildlife receptors were evaluated using two LOEs. Dietary dose was used as an LOE for all six wildlife receptors. Both prey tissue ingestion and incidental sediment ingestion were reflected in dietary-dose estimates. Egg tissue residue, estimated from maternal dietary dose, was used as a second LOE for bald eagle and osprey. The bird-egg LOE for evaluating risk to piscivorous birds (i.e., bald eagle and osprey) is associated with a high degree of uncertainty because the available biomagnification factor data and field-based TRVs are unreliable. As an unreliable predictor of risks to piscivorous birds, the bird-egg LOE was not used to draw risk conclusions. Only the dietary-dose LOE (based on more reliable exposure and effects assumptions) was used to determine risk conclusions for bald eagle and osprey.

Nine COCs (copper, lead, and mercury, benzo[a]pyrene, dibutyl phthalate, total PCBs, total TEQ, aldrin, and DDx) were identified for at least one bird receptor based on dietary-dose or bird-egg LOEs, and three COCs (lead, total PCBs and total TEQ) were identified for mink or river otter. Of these COCs, the only likely unacceptable risks to wildlife are due to PCBs.

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. Mink are estimated to have a greater risk than river otter because of their higher metabolic requirements. Spotted sandpipers and bald eagles might also face increased risk of reduced reproductive success because of PCB exposure. Overall, however, the unacceptable risk to birds from PCB exposure is uncertain because of the underlying dietary assumptions used to derive risk estimates (i.e., that the spotted sandpiper diet consists only of laboratory-exposed worms and that 100% of the bald eagle diet is composed of fish from the Study Area).

Except for shorebirds, exposure and risk estimates are based on the assumption that all of the Study Area provides some type of habitat for each of the receptor species and/or its prey. A lack of actual habitat in some areas of the Study Area would contribute to an overestimation of exposure. This factor should be considered in developing risk management decisions for each of the AOPCs to be identified as part of the FS. Remedies that address mink risks would likely mitigate any potential risks to bird receptors, as well as river otters.

ES.5 AMPHIBIAN/REPTILE AND AQUATIC PLANT RISK ASSESSMENT

Eight COCs were identified for aquatic plants and amphibians based on the surface water LOE (i.e., aluminum, zinc, benzo[a]anthracene, benzo[a]pyrene, naphthalene, BEHP, total PCBs, and DDx); however, it was concluded that none of these identified COCs are expected to pose unacceptable risks to aquatic plant communities and amphibian populations in the Study Area. In general, surface water concentrations of COCs were below algae- or amphibian-specific thresholds, and/or COC concentrations in water exceeded thresholds by a low magnitude and at low frequency, which indicates low to negligible risks. Furthermore, surface water concentrations of several COCs generally had HQs > 1.0 during non-reproductive periods (when amphibians may not be present in the Study Area), indicating negligible risks to amphibian populations. There is some uncertainty concerning the relevance of selected TRVs for aquatic plants and amphibians because aquatic plant-, algae-, or amphibian-specific thresholds for several COCs are limited or not available for comparison. Because amphibians had been selected as the surrogate for reptiles, risk conclusions for amphibians also apply to reptiles.

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. 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 Remedial Investigation/Feasibility Study Programmatic Work Plan (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 (AOC) (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 1997a, 1998) or as directed by EPA.

The evaluation of potential unacceptable risks to ecological receptors at the Portland Harbor Superfund Site has been an ongoing and iterative process that has involved 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:

- *Programmatic Work Plan for the Portland Harbor RI/FS (Programmatic Work Plan)* (Integral 2005)
- *Portland Harbor Superfund Site Ecological Risk Assessment: Comprehensive Synopsis of Approaches and Methods* (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 risks were made based on preliminary datasets in two documents:

- *Portland Harbor RI/FS, Ecological Preliminary Risk Evaluation* (Ecological PRE) (Windward 2005a)
- *Portland Harbor RI/FS Comprehensive Round 2 Site Characterization Summary and Data Gaps Analysis Report* (Comprehensive Round 2 Report) (Integral et al. 2007)

Data from the Study Area were collected in three sampling rounds (Rounds 1, 2, and 3) concurrent with the production of documents describing the ERA process and presenting an evaluation of 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 in order 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.

Figure 1-1 presents an overview of how the development and production of the BERA fits in with the overall RI/FS process for the Portland Harbor Superfund site.

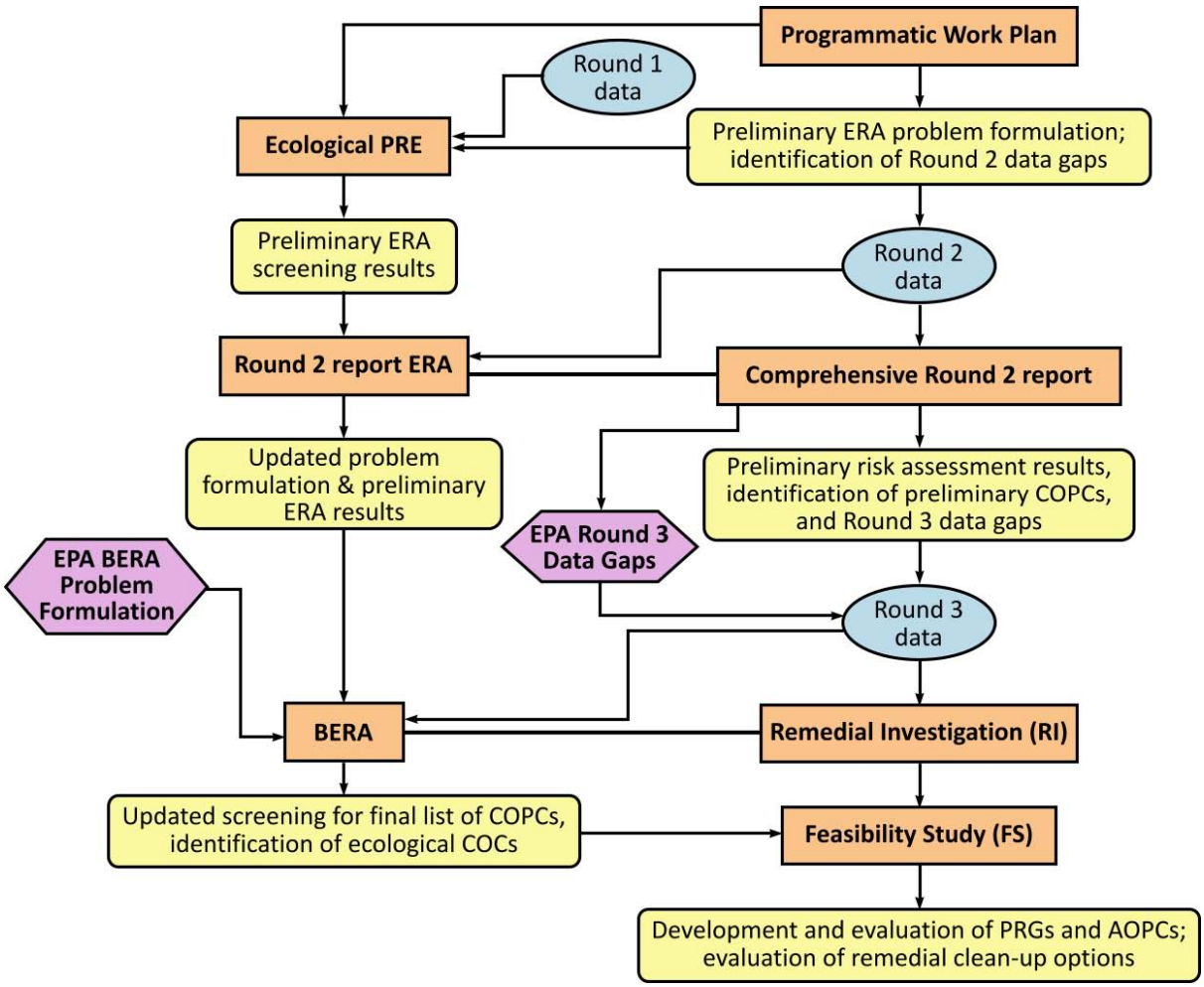


Figure 1-1. ERA Process as Part of the Portland Harbor RI/FS

The RI initially focused on the stretch of the Willamette River 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 *Programmatic Work Plan for the Portland Harbor RI/FS* (Integral 2005) refers to that initial Study Area as the "ISA." In the Comprehensive Round 2 Report (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 enlarged to include the area of the river between RM 11 and 11.8. The term "Study Area" was retained for the BERA and includes the 10-mi stretch of the river between approximately RM 1.9 and RM 11.8.

The BERA has two broad objectives:

- Identify unacceptable risks posed by chemical 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 at Portland Harbor, provide information that risk managers can use to set cleanup levels protective of ecological receptors.

This document identifies ecological chemicals of concern (COCs) for which an exceedance of acceptable ecological risk thresholds was found or predicted. A subset of COCs is also identified as those COCs that contribute most significantly to potential unacceptable ecological risks. The spatial distribution of risks is also presented to provide a basis for identifying areas of potential concern (AOPCs) that will be evaluated in the subsequent FS.

The BERA will be used to support the development, in the FS, of chemical thresholds to be used as 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 based on background sediment quality, technical feasibility, and other risk management considerations. EPA will identify the final remediation goals (RGs) for the site in the Record of Decision, following completion of the FS.

Each subsequent section of this document includes an introduction that is a summary of the section (Sections 2.0 though 5.0) or a summary of the overall process used in the risk assessment of each receptor group (Sections 6.0 through 10.0). The introductory summaries also provide a roadmap to assist the reader and outline the organization of the section and associated appendices. Text boxes are also included in subsequent sections to provide the reader with key points on the overall ecological picture of the Study Area and to highlight key uncertainties.

The remainder of this document is organized as follows:

- Section 2.0 – Site Description – This section presents general information about the ecological habitat types and species present in the LWR.
- Section 3.0 – BERA Problem Formulation – This section presents a summary of 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 data from the Study Area used in the BERA.
- Section 5.0 – Identification of Chemicals of Potential Concern (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.

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).[8]
- 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 an analysis of the sediment quality guidelines' (SQGs') ability to predict benthic toxicity.

[8] 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 8 provides the biota-sediment accumulation regression (BSAR) and biomagnification regression (BMR) data analysis methods and results.
- Attachment 9 presents the revised aquatic biota tissue toxicity reference value (TRV) derivation process provided by EPA (2008f) and selected tissue TRVs.
- Attachment 10 presents a summary of how ecological thresholds for COPCs in surface water and TZW were derived.
- Attachment 11 presents a comparison of background chemical concentrations in sediment and surface water to Study Area concentrations.
- Attachment 12 presents the sample-by-sample and individual prey component risk evaluation for fish.
- Attachment 13 presents the details on the exposure assumptions used as part of the fish dietary line of evidence (LOE).
- Attachment 14 summarizes the literature-derived fish dietary and wildlife TRVs.
- Attachment 15 provides the results of the lamprey toxicity tests.
- Attachment 16 presents the details on the exposure assumptions used as part of 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.

2.0 SITE DESCRIPTION

Although the BERA focuses specifically on chemical 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 develop a Conceptual Site Model 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 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, providing some 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 the width and depth of the river, 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 whereas coarse sediments (i.e., sand, gravel) are indicative of higher-energy environments.

The numerous organisms that use the Lower Willamette River (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, its abundance, and its 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, or 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 fish, such as salmon and lamprey, and provides habitat for numerous resident fish species (more than 40 species have been collected in numerous historical and recent studies) 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 have been identified that use or may use the LWR, including opportunistic piscivores.

Conditions within the LWR provide limited suitable 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. Current conditions in the LWR prevent the widespread development of dense, submerged, and emergent plant communities along the riverbanks because of high turbidity and the presence of riprap and other bank modifications. More detailed information on habitats and organisms using the LWR is presented in the following subsections: Habitat Types in the Lower Willamette River (Section 2.1), and Species Presence and Habitat Use (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 (deeper than 20 ft [6m], with an average depth of 39 ft [12 m] ± 10 ft [3 m] and a maximum depth of 78 ft [24 m]) provides foraging habitat for fish and wildlife that feed mainly in the water column. 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.

There are three types of benthic habitats in the open water areas of the LWR: 1) unconsolidated sediments (sands and silts) in the deeper water (greater than approximately 20 ft [6 m]) Columbia River Datum [CRD]) of the navigation channel and lower channel slopes; 2) 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); and 3) developed shoreline areas (e.g., rock riprap, sheet pile, bulkheads) (Figure 2-2). 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 impacts of navigation, natural sediment deposition, bed load transport/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 that are 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 due to the greater physical disturbance in these areas. The hard surfaces of the developed shoreline provide habitat for an epibenthic community.

Figure 2-1. Nearshore In-Water Habitat

Figure 2-2. In-Water Bulkhead Structure

2.1.2 Bank and Riparian Habitat

In 2007-2008, the City of Portland updated its natural resource inventory of the 12 mi 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 based upon six classes of attributes

defined by the City of Portland, including wildlife movement corridor, large wood/channel dynamics, organic inputs, nutrient cycling and food web,[9] stream flow 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 reported that vegetated[10] riprap (25%), unclassified fill (21%), and beach (23%) were the dominant bank types in the North Reach of the Willamette River (Broadway Bridge to the Columbia River) (City of Portland 2008) (Map 2-1). The bank types classified in the inventory were identified based on physical characteristics and were not associated with a specific elevation.

Figure 2-3. Riprap Bank

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

[10] 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). The 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 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 habitats for juvenile largescale suckers (yearling and sub-yearling) and peamouth (Wydoski and Whitney 2003). The peamouth remains nearshore during winter months, moving to deeper waters in the summer months. The shallow waters with abundant plants and woody debris available for cover are the preferred habitats for 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, with the exception of areas such as Harborton Wetlands, Oaks Bottom, Forest Park, and Powers Marine Park. Therefore, although isolated wildlife habitat areas along the LWR corridor exist, 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 and aquatic and semi-aquatic mammals. Significant habitat sites in the Study Area included 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 (SHAs) 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 that occur within and along the river. The following subsections present an overview of the various aquatic or river-dependent biological communities in the LWR. The species present and their habitat use are also described.

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) or so small they live among the sand grains below the sediment surface (e.g., meio- or microfauna). Benthic macroinvertebrate communities in rivers tend to be dominated by members of the phylum Arthropoda, which includes insects, mites, amphipods, copepods, and crayfish (Figure 2-7). Other phyla such as Mollusca (e.g., clams, mussels, and snails), Annelida (e.g., oligochaete worms, polychaete worms, and leeches), and Platyhelminthes (flatworms) are important members of the benthic community. 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), glean organic matter from deposited material they consume (deposit feeders), prey on other invertebrates and small fish (predators such as some oligochaetes and crayfish), and parasitize other organisms. Lifestyles are also diverse and reflect various strategies to adapt to changes in environmental conditions. The River Continuum Concept 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). The benthic community in the LWR is dominated by organisms that filter feed (e.g., clams, amphipods[11]) or deposit feed (e.g., tubificid worms, chironomid larvae, and amphipods) (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 shoreline[12] (e.g.,

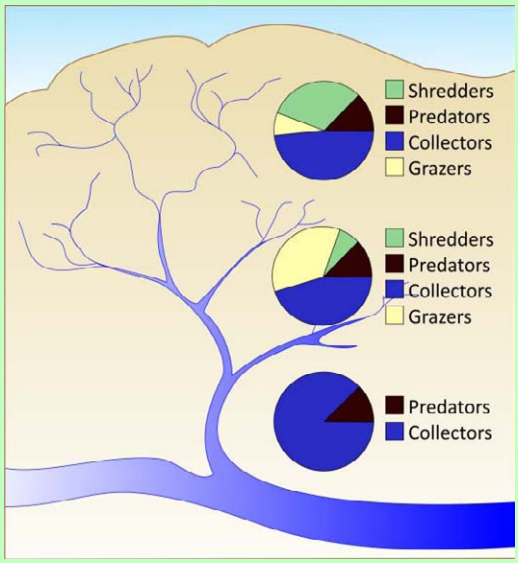
[11] *Corophium* sp. is the dominant benthic amphipod in the LWR and can feed on both deposited and suspended organic material.

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

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.

What is the River Continuum Concept?

Chemical changes in the environment (such as those evaluated in this risk assessment) are but one of many ecological factors that affect benthic communities. Knowing something 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 the headwaters to its mouth. As a river widens and deepens in its course downstream, parts of the bottom are removed from the photic zone and the influence of the riparian zone as a local source of food (including particulate carbon) and shading effects is lessened. 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, there is a decrease in production through photosynthesis 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 are organisms that 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, caddisfly larvae) feeding on the periphyton, which accumulates on larger structures such as stones, wood, or large aquatic plants. Shredders only make up a small percentage of the invertebrates in lower sections of rivers because of reduction in the 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 oligochaetes, many (~35) species of chironomid larvae (midges), amphipods (particularly *Corophium* sp.) 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 mollusks.

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 that are 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).

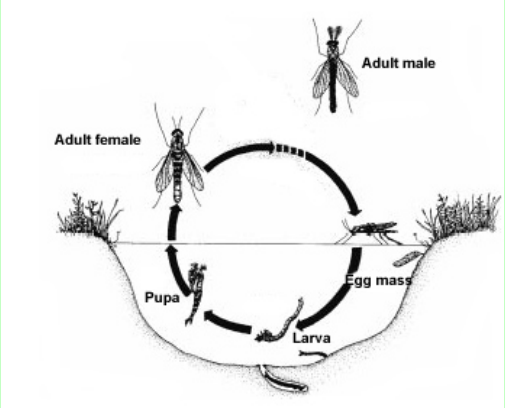


Figure courtesy of the University of Florida Entomology & Nematology Department

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, and 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

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

Phylum	Class or Order	Scientific Name	Common Name
		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
		Aulodrilus pluriseta	tubificid worm
		Aulodrilus sp.	tubificid worm
		Branchiura sowerbyi	tubificid worm
		Limnodrilus hoffmeisteri	tubificid worm
		Quistradrilus multisetosus	tubificid worm
		Chaetogaster sp.	naidid 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

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

Phylum	Class or Order	Scientific Name	Common Name
		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
	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

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

Phylum	Class or Order	Scientific Name	Common Name
		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
		Nanocladius sp.	midge
		Orthocladius Complex	midge
		Parachironomus sp.	midge
		Paracladopelma sp.	midge
		Parakiefferiella sp.	midge
		Paralauterborniella nigrohalteris	midge
		Paraphaenocladius sp.	midge
		Paratanytarsus 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
		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. 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, depending on the species. Some chironomid species can cling to rocks, aquatic macrophytes, and other hard substrates; whereas 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 inches [<4 cm]) clams are taxonomically and functionally related to the native freshwater mussels and fingernail clams (Asiatic clams are placed in the same suborder, Corbiculacea, with fingernail clams) found in the LWR. These bivalves all share similar feeding strategies (filter and surface detrital feeding) and physiological mechanisms for exposure to and uptake of chemicals in sediment.

The Asiatic clam is tolerant of a variety of environmental conditions, preferring sand or gravel substrates in medium to larger rivers. Compared to less resilient native mussels and clams, Asiatic clams are more successful in drainage systems subject to periodic anthropogenic or natural disturbances. 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. This species' low age to maturity, high fecundity, and dispersal mechanism 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 and other invertebrate predators, and fish (e.g., carp, bluegill, and sturgeon) (Thorp and Covich 2001).

Their widespread distribution and abundance in the LWR makes them a useful species for environmental monitoring and investigations of environmental quality.

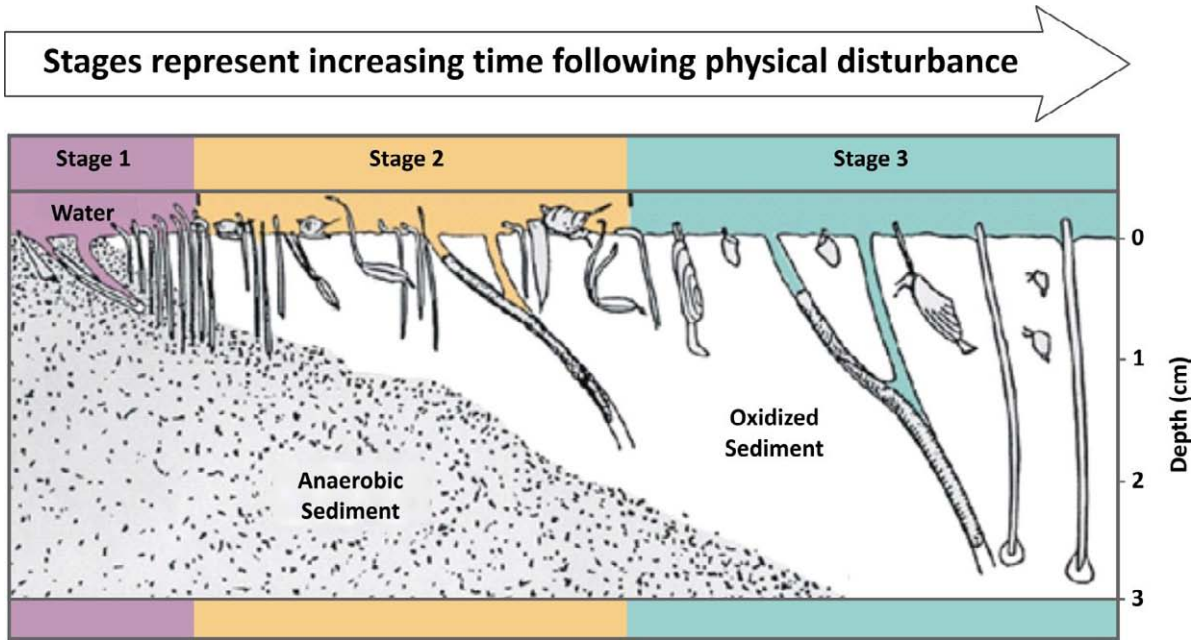
Mayflies and caddisflies, which can be found in many microhabitats (e.g., burrowing in sediment, clinging to the undersides of rocks) in the LWR, use various feeding strategies, including grazing on algae and diatoms, filtering particles from the water column, and preying on other organisms. Crayfish 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 2 to 5 chironomid taxa at each location. Oligochaetes had a similar distribution with a total of 12 taxa found in the Study Area, usually with 3 to 6 taxa per 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 of the two species present). Three groups of crustaceans (amphipods, isopods, and ostracods) were found during the survey, usually in low densities (i.e., fewer than five organisms per sample) with only *Corophium* sp., 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 summer of both 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* sp. 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 can indicate physical instability of an environment or continual exposure to contaminants (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 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 or within the navigation channel). 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

Biological Community	Physical Regime					
	Highly Depositional	Depositional	Erosional/ Transport	Mixed	Unknown[a]	Total
Early	64	48	66	31	10	219
Transitional	4	5	8	5		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 due to the presence of debris.

[b] Indeterminate successional stages were primarily associated with coarse-grained sediments or debris fields where 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 that is used 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, nadid 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, reflecting both their small size and the limited depth to which oxygen can occur in 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 in 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 and/or feed up to 20 cm below the sediment surface. These mature Stage 3 communities are responsible for the mixing of surface sediments 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 and/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 at are present at the SWI. This station is highly depositional and considered a Stage 1 community.

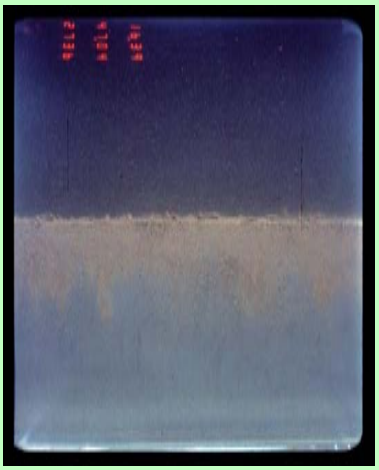


Image to the left shows gray, fine sandy silt with abundant amphipod tubes at the sediment-water interface. The dense assemblage of amphipod tubes, the sequestering of fine-grained sediment, and resultant colonization is a classic Stage 2 assemblage.

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. 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 presence of Stage 1 infauna in this segment of the river appeared to be related to physical disturbance from higher current velocity. The fines that were carried through the middle segment appeared to be deposited in the lower reaches of the Study Area (RM 2.0 to RM 5.0), resulting in a fine-grained region that offered suitable habitat for the colonization of Stage 3 infauna. 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 successfully colonize and survive.

During the summer of 2002, LWG collected crayfish for chemical analysis at 23 locations throughout the Study Area (SEA et al. 2003). The crayfish were not identified to species 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 potential to collect 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 and was collected at all 33 locations in the Study Area. Two other larger mollusks, tentatively identified in the field as western pearlshell mussel (*Margaritifera falcata*) and winged floater (*Anodonta nuttalliana*), were collected at 17 and 2 locations, respectively. Gastropod snails (Pleuroceridae) were 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 and numerous resident fish, including recreational species such as bass and sturgeon.

Piscivorous birds, aquatic mammals, and some fish 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. Chemicals within the system can directly affect fish species and adversely affect fish populations and can also be transferred through the food web through prey consumption by higher-trophic-level aquatic fauna.

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

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

The following subsections provide detailed information on prey and habitat preferences and site use of fish species within these feeding guilds based on numerous studies, including Farr and Ward (1993), Fishman (1999), Beak (2000), and North et al. (2002), or on a comprehensive study that compiled published and unpublished literature on fish use of Portland Harbor (Ellis Ecological 2000). Table 2-3 presents the fish species known to be present or that have been historically observed in the LWR. 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
Largescale sucker[b]	*Catostomus macrocheilus*	resident
Brown bullhead[b]	*Ameiurus nebulosus*	resident

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

Species	Scientific Name	Resident or Migratory[a]
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
Northern pikeminnow[b]	*Ptychocheilus oregonensis*	resident
Smallmouth bass[b]	*Micropterus dolomieui*	resident

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

Species	Scientific Name	Resident or Migratory[a]
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 chemicals 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 of these fish are native to the region and are resident species that 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, and consumes diatoms, detritus, crustaceans, and snails as an adult and is known to consume large amounts of sediment during feeding (CBFWA 1996). It 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 to be from 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 old). (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)).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 that are caught and released to track their movements. One juvenile white sturgeon that was 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*) are two introduced members of the Ictaluridae family that 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 are tolerant of 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 commonly captured in several studies (Farr and Ward 1992; Beak 2000; Hughes and Gammon 1987) but were rare in other studies (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 that are 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). Bluegill have similar habitat preferences as the pumpkinseed, preferring low gradient, low velocity areas with abundant pools and aquatic vegetation (Stuber et al. 1982). Pumpkinseed and bluegill are both benthopelagic species and have been caught in the LWR (Farr and Ward 1992; Beak 2000; Tetra Tech 1995). Goldfish are a benthic feeder that prefers 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, whereas, 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). American shad prefer to spawn in broad flats or shallow water of large rivers, and juvenile shad remain in fresh water for their first summer, moving to marine waters in the fall (Stier and Crance 1985). While in fresh water, juveniles 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). All of 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 mollusks (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 diets consist of aquatic insects, crustaceans, snails, and fish eggs as adults (Wydoski and Whitney 2003).

What type of habitat do sculpin prefer?

Because sculpin 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 chemicals, they represent key species to examine.

Various sculpin (*Cottus*) species inhabit the LWR; seven species have been reported to occur in the Willamette River: prickly sculpin, reticulate sculpin, riffle sculpin, mottled sculpin, Paiute sculpin, shorthead sculpin, and torrent sculpin. These sculpin species have a similar benthic habitat preference that is unique among fish living in the LWR. Sculpin generally prefer shallow water and tend to forage within a very small home range. Some sculpin species may be found in sandy areas (i.e., prickly, reticulate, and riffle sculpin); however, 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 chemicals.

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 and collected from within areas containing moderate cover. The size of sculpin collected from the Study Area ranged from approximately 3.5 to 7 inches in length.

The threespine stickleback is native to the LWR. Threespine stickleback can live in both fresh water and marine systems but spawn in freshwater habitats. It is a benthic feeder in

fresh water, and consumes 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 for foraging. Starry flounder are benthic feeders consuming crabs, mollusks, and small fish (Orcutt 1950).

The warmouth is exotic to the region. Warmouth prefer backwater habitats with slow-moving water and dense vegetation and are known to be adversely affected by channelization (McMahon et al. 1984a). Juvenile warmouth feed on protozoa, bacteria, and zooplankton; and 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). In general, many of these species are anadromous, hatching in fresh water and then migrating to salt water before returning again 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 these species are grouped with invertivores because their juvenile stages 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 (Healy 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 (Healy 1991). While in fresh water, juvenile Chinook salmon feed on aquatic insect larvae and terrestrial insects (Wydoski and Whitney 2003; Healy 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 and 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 Oregon Department of Fish and Wildlife (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 only for a short period of time (Domina 1997). This discrepancy is probably due to the different locations sampled and different methods used to observe the fish.

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, 13.8 km/day, 11.0 km/day, and 7.2 km/day for steelhead, coho, yearling Chinook, and sub-yearling Chinook, respectively (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 lower 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 those 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 fish play a key role in the dynamics of the aquatic community. Because of their high trophic status, these fish have a greater potential for biomagnifying chemicals such as polychlorinated biphenyls (PCBs), dichlorodiphenyltrichloroethanes (DDTs), and mercury than many other species. The diets of piscivorous fish in the LWR have been shown to be similar (Fishman 1999). Of the piscivores listed in Table 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 or 4 years for males and females, respectively (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). Northern pikeminnow was the most commonly caught fish out of six common species in the LWR (Ward and Nigro 1992).

Smallmouth bass, largemouth bass, and black and white crappie are members of the Centrarchidae family and are piscivorous or semi-piscivorous fish. All four species are benthopelagic, consuming fish, crayfish and other crustaceans, mollusks, and worms as adults and insect larvae and zooplankton as juveniles (Wydoski and Whitney 2003; Turner 1966; George and Hadley 1979). All four species have been introduced to the Willamette River (i.e., are not native species). The largemouth bass has a longer life span (i.e., 12 to 16 years) than smallmouth bass (approximately 10 years) (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 bass inhabit warm shallow waters with abundant plants and woody debris available for cover (Wydoski and Whitney 2003). 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); whereas 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 and require moderate-to-large riverine systems with abundant shallow vegetated areas for all life stages, and prefer rocky areas in rivers or below falls for spawning (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 LWR 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 present in freshwater systems, although they can tolerate brackish water (Krieger et al. 1983). Yellow perch are reported to have small home ranges based on tagging studies and reports from large lakes(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 and 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.

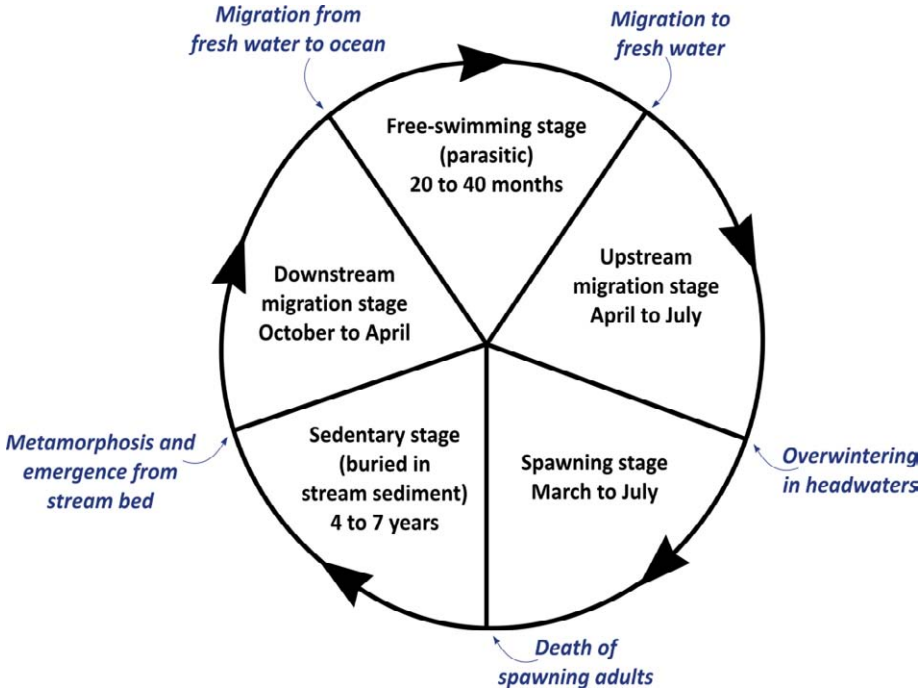


Figure 2-9. Typical Lifecycle of a Pacific Lamprey

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 and 4 to 6 years, respectively) (Kostow 2002a, b). Pacific lamprey are 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 that they spend in the Study Area is unknown. Adult lamprey do not feed in fresh water; their growth 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, which become ectoparasitic as adults, 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, Pacific lamprey fills an ecological niche not otherwise represented in this risk assessment. The Pacific lamprey (*Entosphenus tridentatus)* is anadromous, which means that the early life stages are spent in fresh water; the lamprey then migrate out to the ocean from which they return as adults to spawn in fresh water. Adult lamprey are briefly present in the LWR from April to July during their upstream spawning migration. Spawning takes place in headwater streams the following spring when water temperatures are between 10 and 15°C. They migrate up the rivers by swimming upstream briefly and then resting while sucking rocks. The lamprey spawn in gravel and sandy bottoms by depositing 10,000 to 100,000 very small eggs. 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, sedentary, and survive by filtering food particles such as detritus, diatoms, and algae. The juvenile lamprey will stay burrowed in the mud for about 4 to 6 years, only 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 appears 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 changes 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 live in habitat areas in the LWR. Birds that use the LWR are from various feeding guilds, each filling a distinct role in the ecosystem. Many of the bird species using LWR habitats migrate beyond the United States, so are protected under the Migratory Bird Treaty Act (16 USC 703-712). Mammals that use the LWR are predominantly piscivorous; however, their diet may also include amphibians and aquatic invertebrates. Birds and mammals provide a pathway for energy and nutrients to be transferred from the aquatic to the terrestrial ecosystem and 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 chemicals through biomagnification of chemicals up the food chain. A summary of the presence of and habitat use by wildlife species in the Study Area is presented 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 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 herons, northern shovelers, pintails, mallards, wood ducks, coots, widgeons, gulls, and cormorants (Adolfson et al. 2000).

Bird species that were 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 presents the aquatic and semi-aquatic bird species that may breed along the LWR. Table 2-5 presents a list of bird species that may occur seasonally or for which the LWR represents only part of their habitat.

Table 2-4. Resident Bird Species that Potentially Breed 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 that Potentially Breed 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. Birds that are Seasonally or Minimally Associated with Aquatic Habitat in the Study Area

Species	Scientific Name
Herbivore	
Aleutian Canada goose	*Branta canadensis leucopareia*
American wigeon	*Anas Americana*
Canvasback	*Aythya valisineria*
Gadwall	*Anas strepera*
Northern pintail	*Anas acuta*
Northern shoveler	*Anas clypeata*
Ring-necked duck	*Aythya collaris*
Tundra swan	*Cygnus columbianus*
Diving 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*
Wilson's phalarope	*Phalaropus tricolor*
Ring-billed gull	*Larus delawarensis*

Table 2-5. Birds that are Seasonally or Minimally Associated with Aquatic Habitat in the Study Area

Species	Scientific Name
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*
Piscivore	
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).

2.2.3.1.1 Herbivorous Birds

Two common herbivores using the Study Area are Canada geese (*Branta canadensis*) and mallards (*Anas platyrhynchos*). Canada geese are common in the vicinity of the Study Area throughout the year (Puchy and Marshall 1993). Some individuals are year-round residents, while others merely overwinter in the area. 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 in the area. 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 will also consume invertebrates to meet their metabolic requirements.

2.2.3.1.2 Carnivorous and Omnivorous Birds

Seven diving omnivorous and carnivorous birds are residents of the LWR (Table 2-4). Resident omnivores include: cinnamon teal (*Anas cyanoptera*), wood duck (*Aix sponsa*), and American coot (*Fulica americana).* The cinnamon teal is a fairly common breeding duck found throughout Oregon (Puchy and Marshall 1993); they typically overwinter south of Oregon, but some individuals remain in Western Oregon throughout the year (Csuti et al. 2001). Cinnamon teal is a dabbling duck that forages on a mix of aquatic plants and aquatic invertebrates, such as mollusks, midges, and larvae (Ehrlich et al. 1988) in vegetated shoreline areas. 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). Wood ducks are perching ducks that prefer to nest in woodland habitats, often in trees and snags near water (Ehrlich et al. 1988). They feed in shallow water, mainly foraging on seeds and aquatic plants, but they are also known to eat aquatic insects (Csuti et al. 2001).

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

Four species of diving carnivores that may use the Study Area are the American dipper (*Cinclus mexicanus*), common merganser (*Mergus merganser*), hooded merganser (*Lophodytes cucullatus*), and pied-billed grebe (*Podilymbus podiceps*). Common and hooded mergansers are both locally common breeders in the Willamette Valley with some being year-round residents (Csuti et al. 2001). Mergansers prefer to nest in tree cavities in close proximity to 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 therefore, eat more aquatic invertebrates (e.g., crustaceans, aquatic insects) than do common mergansers (Csuti et al. 2001), but they are also known to feed on small fish (Bendell and McNicol 1995).

American dippers are mostly year-round residents (Puchy and Marshall 1993). American dippers generally prefer smaller, fast-flowing streams but are 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 other birds because of their feeding behavior and prey. 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*). In June 2004, a shorebird reconnaissance survey of the LWR was conducted, and 28 shorebird beaches were identified (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). They feed on insects and benthic invertebrates such as crustaceans, mollusks, 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 both 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 a habitat along intertidal beach areas. Sandpipers use their specialized beaks to probe in beach sediments for invertebrate prey (including epibenthic and infaunal invertebrates) and require some herbaceous vegetation along beach areas for building 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 sandpiper, upland sandpiper, stilt sandpipers, purple sandpipers, western sandpipers) indicate a relatively small home range for sandpipers, ranging from approximately 0.5 to 5 mi (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 local common breeders in the Willamette Valley, and some are present year-round (Puchy and Marshall 1993). Spotted sandpipers have been observed using habitat in the LWR, including the Study Area. In June 2004, a shorebird reconnaissance survey was conducted, and 28 shorebird beaches, representing potential sandpiper habitat, were identified (Saban and Andersen 2004). These beaches were characterized by sandy stretches of intertidal sediment. Some shorebird beaches had upland vegetation present. The longest continuous beaches characterized by gentle slopes were located at 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. Other smaller beaches were located at interspersed locations throughout the Study Area.

2.2.3.1.4 Piscivorous Birds

Resident piscivorous birds that feed from the Willamette River in the vicinity of the Study Area include osprey (*Pandion haliaetus*), double-crested cormorants (*Phalacrocorax auritus*), herons (both the green heron [*Butorides virescens*] and great blue heron [*Ardea herodias*]), American bitterns (*Botaurus lentiginosus*), bald eagles (*Haliaeetus leucocephalus*), and belted kingfishers (*Ceryle alcyon)*. Consumption of secondary aquatic consumers, such as invertivorous fish, gives piscivorous birds the highest potential exposure to biomagnifying chemicals. Few species from this guild feed solely on fish, but fish make up the majority of the diets for all eight species of piscivorous birds 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 these species are potentially exposed to chemicals 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 south to western Mexico and Central America (Martell et al. 2001). 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. 2008). In 1993, one osprey nest was observed between RM 0 and RM 26; and in 2001, 10 nests were observed, several of which were located within the Study Area boundaries (Henny et al. 2008).

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 of osprey have been monitored in recent years along the Willamette River system. Osprey populations in the Willamette River and lower Santiam River (a major tributary that flows into the Willamette River south of Salem) have increased at an annual rate of 13.7% from 1993 until 2001 (Henny et al. 2008).

Osprey populations in the LWR have undergone similar trends as the rest of the Willamette River populations. Between RM 0 and RM 26, the number of osprey nests has increased; one nest was observed in 1993, and ten nests were observed 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. (2008) reported in 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 productive rates of 0.7 and 0.8 young per nest that have been reported to be the minimum required rate to maintain stable bald eagle and osprey populations (Henny et al. 2008; 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 which is 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 in areas 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 anywhere 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 mollusks (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 will 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 further 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 are also opportunistic and 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 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 presented in Table 2-6. These aquatic or semi-aquatic mammals 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 that Potentially Use the Lower Willamette River

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

Source: Csuti et al. (2001)

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 mollusks. 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, mollusks, and small mammals and 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 use Portland Harbor for migrating upstream to their preferred feeding areas.

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. Most of the native amphibians (e.g., long-toed salamander, northern red-legged frog and Pacific tree frog) that may be found in the Study Area prefer undisturbed areas that offer ephemeral wetlands with emergent vegetation and shallow waters. They are preyed upon by the more ubiquitous 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 a suitable place for either species to reside. 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) are special-status species.[13] 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).

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

[b] Identified during LWG sampling activities.
[c] Oregon State listed as a critical species on the ODFW sensitive species list (ODFW 2005).

An amphibian/reptile reconnaissance survey was conducted to confirm the presence of amphibians within the Study Area. Areas of likely amphibian habitat were identified based on known bank conditions from prior field efforts, information from the Willamette River Natural Resource Inventory (City of Portland 2008), and observations made in the field. Multiple sites within the initial Study Area (between RM 3.5 and RM 9.2) were visited over a 3-day period in June of 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 2001 Amphibian/Reptile Reconnaissance Survey

The results of the survey are presented in Table 2-8 and in Map 2-5. As a result of the survey, 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
Downstream from St. John's Bridge (both	No amphibians or reptiles observed

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

Location	Reconnaissance Survey Results
west and east bank) between RM 6 and RM 8	
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

2.2.5 Aquatic Plants

Aquatic plant communities are used by ecological receptors for nesting habitat, breeding habitat, and refuge. Aquatic plant communities also provide food for herbivores and play a role in the cycling of nutrients. Chemicals in the ecosystem may affect plant species and may adversely affect plant communities and/or be transferred from plants through the food web through consumption by higher-trophic-level species. The current conditions in the LWR prevent the widespread development of dense, submerged, and emergent plant communities along the riverbanks due to high turbidity and the presence of riprap and other bank modifications.

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), and a list of species that were observed during those surveys is presented 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

Table 2-9. Plant Species of the LWR

Common Name	Scientific Name	Wetland Indicator Status[a]
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
Willamette daisy[c]	*Erigeron decumbens*	NA

Table 2-9. Plant Species of the LWR

Common Name	Scientific Name	Wetland Indicator Status[a]
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
FACW – facultative wet; moderate to high probability of occurrence in regional wetlands
FAC – facultative; moderate probability of occurrence in regional wetlands
FACU – facultative upland; low to moderate probability of occurrence in regional wetlands
NA – status not available
Varies – status varies by species

[b] Exotic species.

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

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

[e] Federal species of concern.

[f] Listed as threatened (federal).

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

Another qualitative survey was performed by Adolfson et al. (2000). The survey included an inventory of fish and wildlife habitats along the shoreline of the LWR and identified 10 distinct habitat classes: bottomland forest, foothill savanna, conifer forest, meadow, shrub, emergent wetland, beach, rock outcrop, open water, and unvegetated/disturbed. Although all of these habitats are present in the vicinity of the LWR, the bottomland forest, emergent wetlands, beach, and open water habitat classes are the most common, occurring along the shoreline within the Study Area. Historically, bottomland forests were an important component of the Willamette River floodplain system (Sedell and Froggatt 1984), but they have been reduced to a portion of their former extent (Adolfson et al. 2000). Emergent wetlands also exist in a few small remnant patches in areas adjacent to the shoreline. Beach habitats throughout the LWR typically consist of narrow shoreline areas with sand substrate and are dominated by a variety of annual grasses and perennial shrubs. Open-water habitats exist throughout the LWR in tributaries, sloughs, and side channels and are often dominated by aquatic plant species from bottomland forest, emergent wetland, and scrub/shrub plant communities (Adolfson et al. 2000). 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 CERCLA (EPA 1997a), 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.

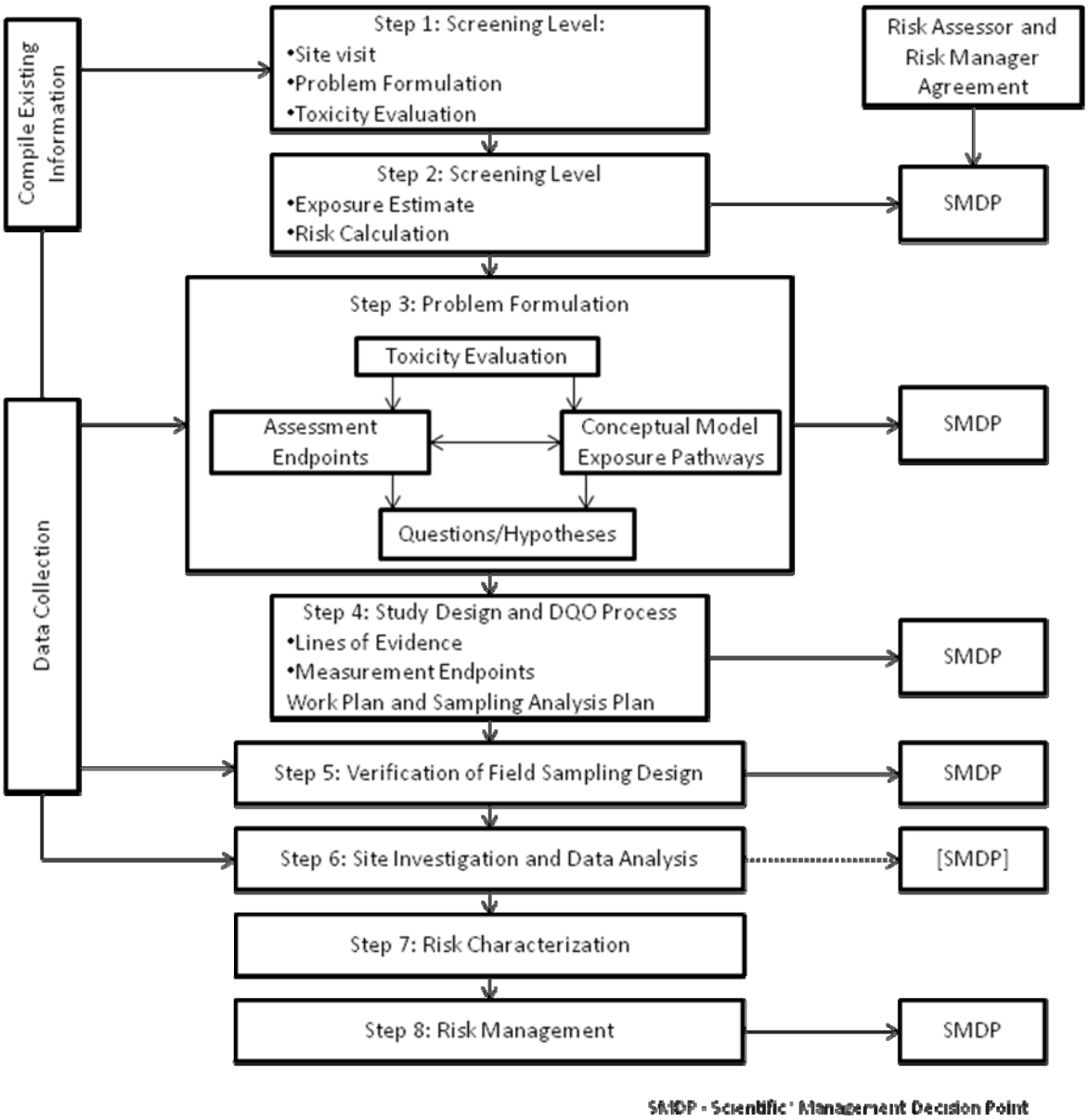


Figure 3-1. BERA 8-Step Process for Superfund (EPA 1997a)

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

- 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 conceptual site model (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 in Appendix G of the Comprehensive Round 2 Report (Integral et al. 2007). Discussions between the LWG and EPA led to EPA developing a Problem Formulation to provide a framework for completing the BERA that addressed implementation of the five steps identified above and accounted for data and information collected to date. Detailed information on the BERA Problem Formulation is presented in Attachment 2,[14] and the elements of the Problem Formulation are summarized as follows:

- 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 following the procedures outlined in the Problem Formulation (Attachment 2), Rounds 1, 2, and 3 data were screened to compile the BERA COPCs using the complete dataset identified for the BERA (Section 4.0). Although the SLERA and refined screen are a part of the BERA Problem Formulation process, this step was conducted using the BERA dataset and therefore, a general summary of the SLERA and refined screen and the ecological COPCs that were identified for each ecological receptor group are presented in a separate section (Section 5.0) following the summary of the BERA dataset (Section 4.0). Details on the SLERA and refined screen are presented in Attachment 5.

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 1997a), along with the assessment endpoints, exposure pathways, and risk

[14] 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 clarification was needed as to how the Problem Formulation was implemented in the BERA.

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/testable hypotheses for the BERA.

Consistent with EPA Superfund guidance (EPA 1997a), 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:

- Their societal and cultural significance (i.e., species valued by society or that have special regulatory status – threatened or endangered)
- Their ecological significance (i.e., species that serve a unique ecological function)
- Their potential level of exposure to likely COPCs at the site (i.e., site usage)
- Their relative ability to bioaccumulate likely COPCs at the site
- Their sensitivity to likely COPCs at the site
- The 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**[15] – 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[16]
- **Piscivorous fish populations** – smallmouth bass and northern pikeminnow
- **Detritivorous fish individuals** – Pacific lamprey ammocoetes
- **Sediment-probing invertivorous bird populations** – spotted sandpiper
- **Omnivorous bird populations**[17] – hooded merganser

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

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

- **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, with the exception that the health of threatened or endangered species is to be protected at the individual organism level. 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. For all receptors and receptor groups evaluated at the community or population level, lowest-observed-adverse-effect-levels (LOAELs) were used to derive risk estimates, which is appropriate for a BERA. LOAEL TRVs provide a basis for evaluating whether exposure concentrations are at or above a level that may cause an effect on survival, growth, or reproduction of some individual organisms in experimentally exposed populations or communities. This follows the conventional practice in ecological risk assessment of using organism-level TRVs defined in this manner to evaluate the potential effects on populations or communities; however, organism-to-population and organism-to-community extrapolation is a source of uncertainty and, in many but not all cases, a source of conservatism in risk estimates. Whether the organism-to-population or organisms-to-community extrapolation is a source of conservatism depends in large part on the level of effect represented by the TRV, which varies because of differences in the toxicological databases for different chemicals and receptor groups (Solomon et al. 2008; Suter 2007; Posthuma et al. 2002; Pastorok et al. 2001). This uncertainty is further evaluated in the individual ecological receptor group risk assessment sections (Sections 6.0 through 10.0) and in the risk conclusions section (Section 11.0).

- Two LOEs, surface water and shallow (< 38 cm) TZW toxicity, were used to characterize risks to aquatic plants and amphibians. Risks to benthic invertebrates were assessed using several LOEs: sediment toxicity (measured in laboratory tests, predicted based on comparison to site-specific or generic sediment quality guidelines [SQGs]), surface water toxicity, TZW toxicity, and tissue residue (both measured and predicted). Risks to fishes were assessed using four LOEs: surface water toxicity, tissue residues (both empirical and predicted), dietary toxicity, and fish condition. One LOE, dietary toxicity, was used to characterize risks to birds and mammals. In addition, risks to osprey and bald eagles were assessed based on predicted accumulations of chemicals in eggs.

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[17] Belted kingfisher was evaluated in the uncertainty assessment.

The ecological CSM illustrates the pathways that chemicals 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 relative to their exposure potential: complete and significant, complete and significance unknown, complete and insignificant, and incomplete. 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 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. 2007; Integral et al. 2004b), is presented in Attachment 2. The ecological CSM consists of: 1) a diagram that describes contaminant sources within the Study Area, 2) illustrations of how contaminants are transported from their sources to the environmental media to which ecological receptors are exposed, and 3) descriptions of how 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. This simplified CSM presents only those exposure media that were evaluated in the BERA, per the assessment endpoints and measurement endpoints identified by EPA (see Section 3.3). The simplified CSM also presents tissue as a separate exposure medium; whereas, in the refined ecological CSM (Attachment 2) exposure to tissue is identified as a dietary exposure route under each abiotic exposure medium.

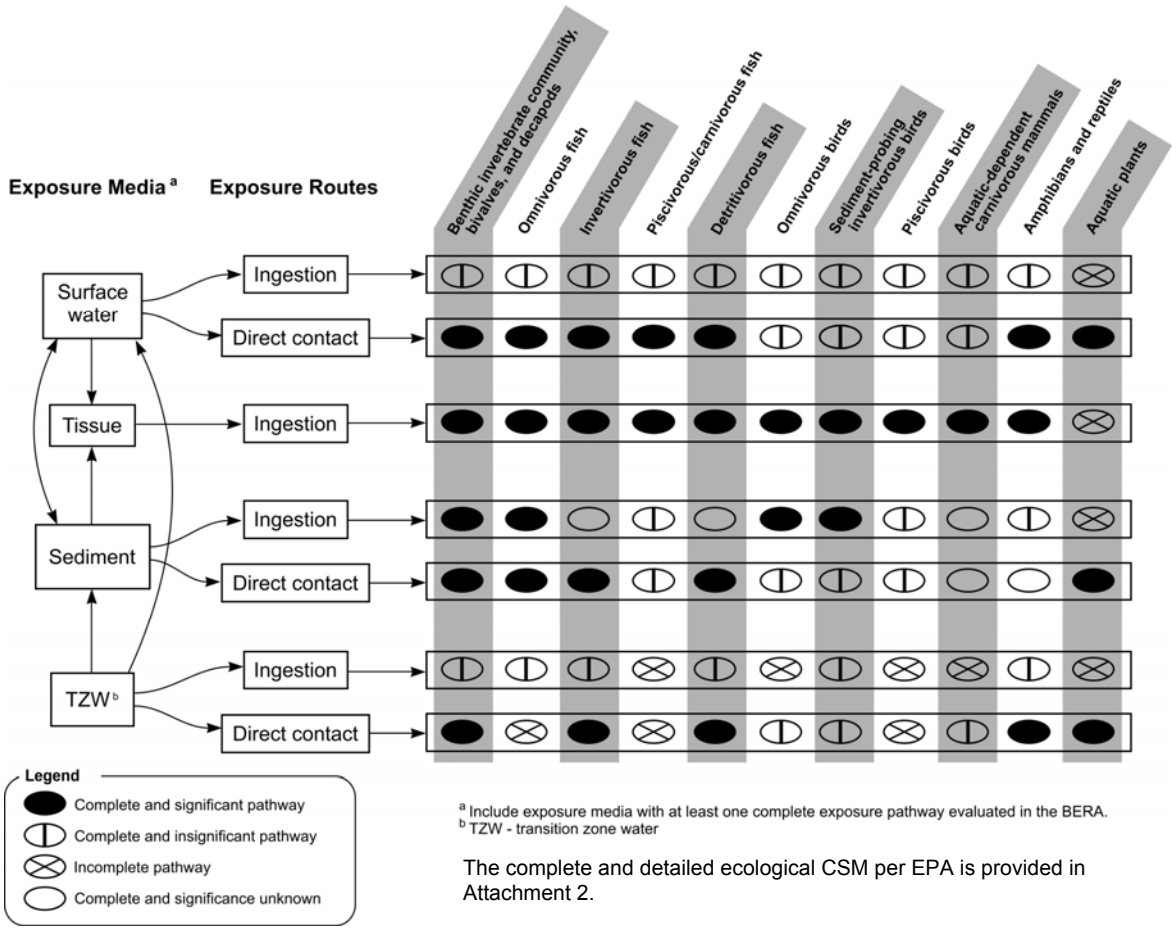

Figure 3-2. Simplified Ecological CSM

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 as implemented in this BERA. Assessment endpoints are characteristics of selected environmental receptors that are to be protected (EPA 1997a). Questions and hypotheses to test suspected interactions between receptors and contaminants lead to the selection of measurement endpoints that quantify exposure to chemicals 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, relative to reference area sediments
		Bulk surface sediment contaminant concentrations	Concentrations in site sediment compared to effect levels 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 to national consensus-based SQGs (TECs, PECs and related quotients), mechanistic-based SQGs (EqP), and effects-based SQGs (PELs, TELs, ERLs, ERMs, AETs, LRMs, RSET SLs, and related quotients)
		Water contaminant concentrations	Surface water concentrations compared to state WQS, national AWQC, or effects-based values derived from the literature that are protective of benthic macroinvertebrate survival, growth, and reproduction
			Shallow TZW concentrations compared to state WQS, national AWQC, and 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 concentration of epibenthic organisms relative to tissue TRVs
			Steady-state estimates of laboratory-exposed whole-body concentrations in *Lumbriculus* relative to tissue TRVs
			Predicted (BSAF[b]) whole-body concentration of *Lumbriculus* relative to 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 relative to tissue TRVs
			Steady-state estimates of laboratory-exposed whole-body concentrations in *Corbicula fluminea* relative to tissue TRVs
			Predicted (BSAF[b]) whole-body concentrations in *Corbicula fluminea* relative to tissue TRVs
		Survival and growth of clams used in bioaccumulation testing	*Corbicula fluminea* survival and growth relative to 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, relative to reference sediments
		Water contaminant concentrations[c]	Surface water concentrations compared to state WQS, national AWQC, or effects-based values derived from the literature that are protective of benthic macroinvertebrate survival, growth, and reproduction
			Shallow TZW concentrations compared to state WQS, national AWQC, and 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 to national consensus-based SQGs (TECs, PECs and related quotients), mechanistic-based SQGs (EqP), and effects-based SQGs (PELs, TELs, ERLs, ERMs, AETs, LRMs, RSET SLs, 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 relative to tissue TRVs
			Predicted (BSAF or FWM) whole-body concentrations in crayfish relative to tissue TRVs
		Bulk sediment contaminant concentrations[c]	Concentrations in site sediment compared to national consensus-based SQGs (TECs, PECs and related quotients), mechanistic-based SQGs (EqP), and effects-based SQGs (PELs, TELs, ERLs, ERMs, AETs, LRMs, RSET SLs, and related quotients)
		Water contaminant concentrations[c]	Surface water concentrations compared to state WQS, national AWQC, or effects-based values derived from the literature that are protective of benthic macroinvertebrate survival, growth, and reproduction
			Shallow TZW concentrations compared to state WQS, national AWQC, and effects-based values derived from the literature that are protective benthic macroinvertebrate survival, growth, and reproduction
Fish			
Survival, growth, and reproduction of omnivorous fish	Carp,[d, e] white sturgeon, largescale sucker[d, f, g]	Water contaminant concentrations[h]	Surface water concentrations compared to 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 concentration relative to 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 to 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]	Water contaminant concentrations[h]	Concentration in surface water relative to state WQS, national AWQC or effects-based TRVs reported in the literature
			Concentration in shallow TZW relative to 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 concentration relative to tissue TRVs
			Predicted (BSAF or FWM) whole-body concentration relative to 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 to dietary TRVs
Survival, growth, and reproduction of piscivorous fish	Northern pikeminnow, smallmouth bass	Water exposure contaminant concentrations[h]	Concentration in surface water relative to 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 concentration relative to tissue TRVs
			Predicted (BSAF or FWM) whole-body concentration relative to 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 to 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
Survival and growth of detritivorous fish	Pacific lamprey ammocoete[j]	Water contaminant concentrations[h]	Compare surface water concentrations to state WQS, national AWQC, or literature-based values that are protective of early life stages.
			Concentration in TZW relative to state WQS, national AWQC, or effects-based values reported in the literature that are protective of early life stages[l]
		Fish tissue contaminant concentrations	Empirical whole-body concentration relative to 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 to 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 to 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 or modeled)	Predicted concentrations in eggs (based on appropriate model) compared to egg or embryo-based TRVs for DDT and metabolites, PCBs, and dioxin-like compounds
Mammals			

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 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 to dietary TRV
Amphibians			
Survival, growth, and reproduction of amphibians	Frogs, salamanders	Water contaminant concentrations from amphibian assessment areas	Surface water concentrations compared to state WQS, national AWQC, or effects-based values reported in the literature that are protective of sensitive life stages TZW concentrations compared to 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	Water contaminant concentrations from aquatic plant assessment areas	Surface water concentrations compared to 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 concentrations compared to state WQS, national AWQC, and 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, with the exception 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 an SL 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 will be used (Meador et al. 2002).

[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 chemicals, including PCB congener analysis, and is a surrogate for other fish species for these chemicals.

[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 will be 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 (chemical 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 individual level because they are federally threatened (juvenile Chinook) and an Oregon State sensitive species that are of special concern to tribes (Pacific lamprey); Effect thresholds based on reproduction used as a surrogate for growth for 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 to suckers, carp, and sturgeon but categorizes the pathway as insignificant.

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

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

AET – apparent effects threshold
AWQC – ambient water quality criteria
BERA – baseline ecological risk assessment
BSAF – biota-sediment accumulation factor
CSM – conceptual site model
DDT – dichlorodiphenyltrichloroethane
EqP – equilibrium partitioning
ERA – ecological risk assessment
ERL – effects range – low
ERM – effects range – medium
FWM – food web model
LRM – logistic regression model
OC – organic carbon
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
PEC – probable effects concentration
PEL – probable effects level
RSET – Regional Sediment Evaluation Team
SL – screening level
SQG – sediment quality guideline
TBT – tributyltin
TEC – threshold effects concentration
TEF – toxic equivalency factor
TEL – threshold effects level
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 chemicals via pathways identified in the CSM, effects from that exposure, or both. EPA (1998) ERA guidelines 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). Some modification of the language has been made 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 benchmarks 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 benchmarks 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 in shallow TZW greater than the toxicity thresholds for survival, growth, or reproduction of invertivorous or detritivorous fish?
- 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[18] fish?
- 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 benchmarks for survival, reproduction, or growth?
- Are detritivorous fish more or less sensitive to waterborne chemicals than the species used to develop existing water quality criteria and TRVs?
- 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?

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?
- 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?

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

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 1997a, 2004a). The general BERA analysis plan was developed by EPA and is presented as part of the Problem Formulation (Attachment 2). Details on the analytical approaches used to evaluate risks to each ecological receptor group are provided in the effects and exposure sections of 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 various media (i.e., sediment, water, and tissue) were based on concentrations in environmental samples on 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 chemical- and media-specific TRVs or SQGs for COPCs at the site. These TRVs provide estimates of chemical 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 benthic invertebrates exposed to sediments.

Risk characterization – As part of the risk characterization, information on contaminant exposure and effects was integrated to estimate risks to the receptors. Several 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: 1) the location and spatial extent of site contamination exceeding adverse effect thresholds; and 2) the magnitude of or degree to which adverse effect thresholds are exceeded.

Weight of evidence framework – When multiple LOEs are available for assessing risk to an ecological receptor, a framework is needed to reconcile any inconsistencies that arise among the LOEs as well as to determine the reliability of all available LOEs for a given COC-receptor pair. EPA's Problem Formulation provided one such weight-of-evidence (WOE) framework. However, EPA recognized that the WOE framework, as proposed in the Problem Formulation (Attachment 2), 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 to lead to different weights on the TRV LOE for different COPCs). Because of this limitation, this proposed WOE framework was not used in the BERA. EPA (2009a) 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: 1) incorporate various exposure and effects scenarios in the risk estimation process that capture the range of uncertainties in assumptions; 2) express numeric risk calculations as point estimates with statistical measures of uncertainty (e.g., confidence limits, percentiles); and 3) conduct a 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 RI. Details on all SCRA data collected are presented in Section 2.0 and 5.0 of the draft 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 to RM 11.8, defined as the Study Area. The following data are included in the BERA dataset:

- Chemistry data of 1,469 surface sediment samples
- Chemistry data of 315 whole-body fish and invertebrate tissue samples and stomach contents
- Chemistry data of 313 surface water samples
- Chemistry data of 192 TZW samples collected adjacent to nine sites
- Sediment toxicity data with *Chironomus dilutus* and *Hyalella azteca* conducted with 269 surface sediment samples
- Sediment toxicity data from bioaccumulation testing of *Corbicula fluminea* and *Lumbriculus* sp. based on sediments collected from 33 stations.

Qualitative data were also collected from the Study Area to support the BERA and provide additional information for characterization of the ecological setting of the Study Area and for characterizing risks to ecological receptors:

- Qualitative fish health field observation data
- Benthic macroinvertebrate (infaunal and epifaunal) community structure data
- Aquatic plant and amphibian/reptile habitat reconnaissance survey
- Shorebird beach habitat reconnaissance survey

Surface sediment and fish tissue data collected from the downstream reach (RM 0 to RM 1.9), Multnomah Channel, and 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 to 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 of 45 surface sediment samples
- Chemistry data of 11 of 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 of 22 surface sediment samples
- Chemistry data of 19 whole-body fish tissue samples and stomach contents
- Bioassay data with *Chironomus dilutus* and *Hyalella azteca* 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 RI.

More detailed descriptions of the BERA dataset used to calculate risk estimates to ecological receptors are presented in Section 4.1. Non-Study Area data from the upstream reach, Multnomah Channel, or 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.

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, 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/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 risks to ecological receptors. These data are briefly summarized at the end of this section.

Table 4-1. Overall Summary of BERA Dataset

Medium	Data Types	Number of Samples
Study Area Data (RM 1.9 – RM 11.8)		
Surface sediment	Chemical 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	Chemical concentrations in field -collected clams, multiplate 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, laboratory-exposed worm and clam tissue[a]	315
Surface water	Chemical concentrations in LWG surface water collected using a peristaltic pump and XAD-2 Infiltrex™ 300 system (column and filter) and surface water collected from 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
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 azteca*	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 azteca*	22

Table 4-1. Overall Summary of BERA Dataset

Medium	Data Types	Number of Samples
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

[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 RI 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

NA – not applicable

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]	Chemicals 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	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	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]	Chemicals 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
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
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. 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.

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

All non-LWG data included in the SCRA database (see Appendix A of the draft RI) 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 that were 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 from these sampling events are presented in Attachment 4 and Attachment 6, respectively.

4.1.2 Biota Tissue

Tissue chemistry data from the Study Area included in the BERA dataset were selected from various LWG sampling events to evaluate risks to selected ecological receptors of concern. Table 4-3 presents a summary of the fish and invertebrate tissue samples included in the BERA dataset that were collected from within the Study Area.[19] Benthic tissue in the BERA dataset included field-collected tissue samples for crayfish,[20] clams (*Corbicula fluminea*), freshwater mussels (Western pearlshell [*Margaritifera falcata*] and winged floater [*Anodonta nuttalliana*]), and epibenthic invertebrates and zooplankton collected with multiplate samplers. 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.

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[19] 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).

[20] 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.

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

Sampling Event Name and Period	Species	No. of Samples[a]	Chemicals Analyzed in Tissue
Round 1A tissue sampling, June 2002	Juvenile Chinook salmon	6	PCB Aroclors, metals, PAHs, organochlorine pesticides, phenols, phthalates, SVOCs
Round 1 tissue sampling, July to November 2002	Black crappie[b]	4	PCB Aroclors, dioxins and furans, metals, PCB congeners, organochlorine pesticides, SVOCs
	Brown bullhead[b]	6	PCB Aroclors, dioxins and furans, metals, PAHs, PCB congeners, organochlorine pesticides, phenols, phthalates, SVOCs
	Carp[c]	6	PCB Aroclors, dioxins and furans, metals, PAHs, PCB congeners, organochlorine pesticides, phenols, phthalates, SVOCs
	Clam	3	PCB Aroclors, butyltins, metals, PAHs, organochlorine pesticides, phenols, phthalates, SVOCs
	Crayfish	27	PCB Aroclors, dioxins and furans, metals, PAHs, PCB congeners, organochlorine pesticides, phenols, phthalates, SVOCs
	Largescale sucker	6	PCB Aroclors, metals, PAHs, organochlorine pesticides, phenols, phthalates, SVOCs
	Northern pikeminnow	6	PCB Aroclors, metals, organochlorine pesticides, SVOCs
	Peamouth	4	PCB Aroclors, metals, organochlorine pesticides, SVOCs
	Sculpin	27	PCB Aroclors, dioxins and furans, metals, PAHs, PCB congeners, organochlorine pesticides, phenols, phthalates, SVOCs
	Smallmouth bass	14	PCB Aroclors, dioxins and furans, metals, PAHs, PCB congeners, organochlorine pesticides, phenols, phthalates, SVOCs
Round 2A benthic tissue, 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

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

Sampling Event Name and Period	Species	No. of Samples[a]	Chemicals Analyzed in Tissue
Round 2A juvenile Chinook salmon tissue sampling, May 2005	Juvenile Chinook salmon (stomach contents)	5	PAHs, PCB congeners, organochlorine pesticides, SVOCs
	Juvenile Chinook salmon	9	Butyltins, dioxins and furans, metals, PAHs, PCB congeners, organochlorine pesticides, phenols, phthalates, SVOCs
Round 2A multiplate tissue sampling, September 2005	Invertebrates (collected on multiplate sampler)	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 (n=3) and macropthalmia[d] (n=2)	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	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 Name and Period	Species	No. of Samples[a]	Chemicals Analyzed in Tissue
Round 3B biota sampling, August – 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]	9	Butyltins, dioxins and furans, metals, PAHs, PCB congeners, organochlorine pesticides, phenols, phthalates, SVOCs
	Sculpin	12	Butyltins, dioxins and furans, metals, PAHs, PCB congeners, organochlorine pesticides, phenols, phthalates, SVOCs
	Smallmouth bass[e]	18	Butyltins, dioxins and furans, metals, PAHs, PCB congeners, organochlorine pesticides, phenols, phthalates, SVOCs

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

[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 furan and dioxin-like PCB congener data 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.

BERA – baseline ecological risk assessment

BHHRA – baseline human health risk assessment

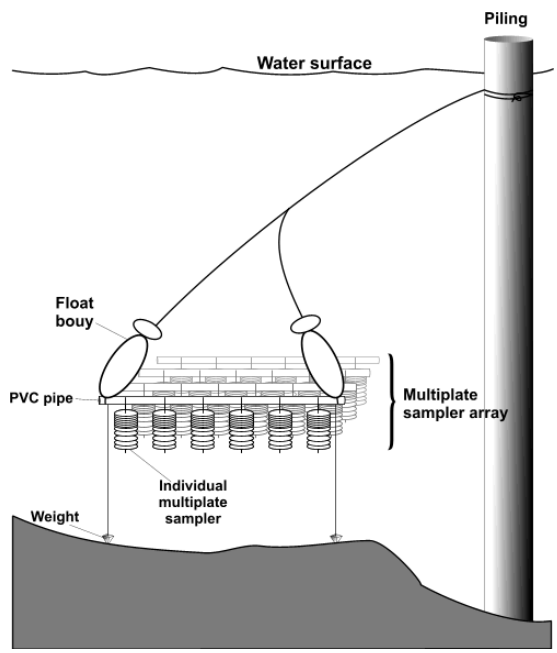
PAH – polycyclic aromatic hydrocarbon

PCB – polychlorinated biphenyl

SVOC – semivolatile organic compound

What is a multiplate sampler?

Multiplate samplers were used to provide an artificial substrate for passive collection of invertebrates. They were EPA-style, round Hester-Dendy samplers that consisted of 14 3-in.-diameter plates made of tempered hardboard separated by spacers. The space between the individual plates varies 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. The multiplate sampler arrays were anchored with a double-buoy system to maintain 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 either a tag line 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.



All fish tissue samples were collected in the field. Data represent whole-body tissue composite samples. Fish tissue composite samples 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 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 chemical 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-mi stretches in Rounds 1 and 3, respectively.

- Largescale sucker, peamouth, northern pikeminnow tissue composite samples included individual fish collected over 2-mi stretches in Round 1.
- Carp tissue composites included individual fish collected over 3- and 4-mi stretches in Round 1 and Round 3, respectively.
- Black crappie and brown bullhead tissue composites included individual fish collected over 3-mi stretches in Round 1.
- Juvenile (pre-breeding) white sturgeon tissue composite samples included individual fish collected over 3-mi stretches in Round 3.

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 Surface Water

Surface water data collected from the Study Area used in the BERA dataset includes all Round 2 and Round 3 LWG-collected data, as well as non-LWG data. Table 4-4 presents a summary of the surface water samples included in the BERA dataset. Study area data from all three sampling events conducted during Round 2 and all four sampling events conducted during Round 3 were included in the BERA dataset. Map 4-14 presents all the Study Area surface water sampling locations included in the BERA dataset.

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

Sampling Event	Sampling Period	Sampling Method	Number of Samples[a]	Chemicals Analyzed in Surface Water
LWG-Collected Data				
Round 2A, surface water event 1	November to December 2004	Peristaltic pump	25	PCB Aroclors, butyltins, herbicides, metals, PAHs, organochlorine pesticides, phenols, phthalates, SVOCs
		XAD	8[b]	Dioxins and furans, PAHs, PCB congeners, organochlorine pesticides, phthalates, SVOCs
Round 2A, surface water event 2	March 2005	Peristaltic pump	27	PCB Aroclors, butyltins, herbicides, metals, PAHs, organochlorine pesticides, phenols, phthalates, SVOCs
		XAD	8[b]	Dioxins and furans, PAHs, PCB congeners, organochlorine pesticides, phthalates, SVOCs

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

Sampling Event	Sampling Period	Sampling Method	Number of Samples[a]	Chemicals Analyzed in Surface Water
Round 2A, surface water event 3	July 2005	Peristaltic pump	25	PCB Aroclors, butyltins, herbicides, metals, PAHs, organochlorine pesticides, phenols, phthalates, SVOCs
		XAD	8[b]	Dioxins and furans, PAHs, PCB congeners, organochlorine pesticides, phthalates, SVOCs
Round 3, surface water event 1	January 2006	Peristaltic pump	3	Butyltins, herbicides, metals, PAHs, phenols, phthalates, SVOCs
		XAD	2[b]	Dioxins and furans, PAHs, PCB congeners, organochlorine pesticides, SVOCs
Round 3, surface water event 2	September 2006	Peristaltic pump	12	Butyltins, herbicides, metals, PAHs, phenols, phthalates, SVOCs
		XAD	12[b]	Dioxins and furans, PAHs, PCB congeners, organochlorine pesticides, SVOCs
Round 3, surface water event 3	November 2006	Peristaltic pump	40	Butyltins, herbicides, metals, PAHs, phenols, phthalates, SVOCs
		XAD	38[b]	Dioxins and furans, PAHs, PCB congeners, organochlorine pesticides, SVOCs
Round 3, surface water event 4	January to March 2007	Peristaltic pump	44	Butyltins, herbicides, metals, PAHs, phenols, phthalates, SVOCs
		XAD	38[b]	Dioxins and furans, PAHs, PCB congeners, organochlorine pesticides, SVOCs
Non-LWG-Collected Data				
Siltronic Supplemental In-River transition	May 2005	Geoprobe	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
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
SVOC – semivolatile organic compound
VOC – volatile organic compound
XAD – XAD-2 Infiltrex™ 300 system

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 chemical 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 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. 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.

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.[21]
- **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 twelve single-point locations were sampled.

[21] 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).

- **Round 3, Event 4 (January – March 2007[22])** – 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 twelve 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 RI). 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: 1) a peristaltic pump, or 2) an Infiltrex™ 300 system with an XAD-2 resin column (XAD). Surface water samples collected using the peristaltic pump were collected using a low volume of water (10 to 20 L) over a short period of time (i.e., 30 minutes) and were analyzed for chemical (i.e., metals, pesticides, herbicides, and semivolatile organic compounds [SVOCs]) and conventional parameters. Surface water samples that were analyzed for hydrophobic organic compounds (i.e., dioxins and furans, PCBs, polycyclic aromatic hydrocarbons [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.

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

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: a peristaltic pump (as shown on the right) and Infiltrex™ 300 system with an XAD-2 resin column (XAD sampling). 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 feet above the river bottom) or from near the surface (1 to 3 feet 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 analyses were collected using a Standard MasterFlex peristaltic pump that was 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 container. One container was made of polycarbonate and was used for collecting samples for trace metals, butyltins, and conventional analyses. The second container was made of stainless-steel or glass and was used to collect samples to be analyzed for organic compounds. 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 chemicals. 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.

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 for relevant ecological receptors (i.e., benthic invertebrates, fish, amphibians, and aquatic plants). The chemistry results from the Study Area surface water samples are presented in Attachment 4.

4.1.4 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 sample collected using a Geoprobe adjacent to the Siltronic site. During Round 2, TZW sampling was conducted by LWG between October 3 and December 2, 2005, to collect samples of groundwater discharge to the LWR. Table 4-5 presents a summary of the shallow (0 to 38 cm) TZW samples. Map 4-15 presents all the Study Area TZW sampling locations included in the BERA dataset.

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

Sampling Site	Sampling Method	No. of Shallow Samples[a]	Chemicals 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 facility	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
	Tridentprobe (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

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

Sampling Site	Sampling Method	No. of Shallow Samples[a]	Chemicals Analyzed in Transition Zone Water
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
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.
PAH – polycyclic aromatic hydrocarbon
SVOC – semivolatile organic compound
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 that is equipped with temperature, conductivity, and water sampling probes. With the Trident probe, TZW is collected through a small-diameter, Teflon®-coated, stainless steel probe with a port on the end covered by steel mesh. The Geoprobe® 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).

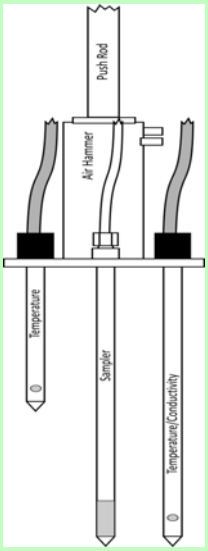
Shallow TZW data were used to estimate TZW exposure concentrations for relevant ecological receptors (i.e., benthic invertebrates, sculpin, lamprey ammocoetes, aquatic plants, and amphibians). The chemistry results from the Study Area TZW samples are presented in Attachment 4.

How was transition zone water collected?

The transition zone is the interstitial area in which groundwater and surface water both 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 that is equipped with temperature, pH, conductivity, and water sampling probes. Oxidation reduction potential was 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.



4.1.5 Qualitative/Reconnaissance Level Data

Several reconnaissance level surveys were conducted in support of the BERA but were not used directly to calculate risk estimates. The results of these surveys are further described in other sections of this BERA. The data were used 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:

- **Qualitative fish health field observation data** – Internal and external fish condition observations were collected during Round 3 sampling of juvenile white sturgeon in February and March 2007. A summary of the field observations are presented as part of the fish risk assessment in Section 7.0.
- **Benthic macroinvertebrate (infaunal and epifaunal) community data** – 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. Multiplate samplers were analyzed for tissue and taxonomic analysis of colonizing epifaunal species. The results of the 2002 grab and multiplate sampling events are presented in Attachment B2 of Appendix B of the Programmatic Work Plan (Integral et al. 2004b) and the results of the 2005 multiplate sampling event are presented in the Round 2 Sampling of Invertebrates

using Multiplate Samplers Field Sampling Report[23] (Windward 2005b). These data were used to help characterize the benthic invertebrate community in the Study Area as presented in Section 2.0.

- **Aquatic plant and amphibian/reptile reconnaissance survey** – 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. The results of this survey are presented in detail in Attachment B2 of Appendix B of the Programmatic Work Plan (Integral et al. 2004b). The habitat areas identified during this survey are presented as part of the amphibians and aquatic plant risk assessments in Sections 9.0 and 10.0, respectively.
- **Shorebird beach habitat reconnaissance survey** – 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 during this survey are presented as part of the wildlife risk assessment in Section 8.0.

4.2 NON-STUDY AREA DATA

Per EPA, data collected from outside the boundaries of the Study Area were also evaluated in the BERA. Data from the following areas were evaluated in the BERA:

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

Tables 4-6 and 4-7 present the sediment and tissue data, respectively, from the downstream reach, Multnomah Channel, and downtown reach that were evaluated in the BERA. These non-Study Area sediment and tissue data were compared to 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.

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

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

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
Multnomah Channel	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
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.
BERA – baseline ecological risk assessment
LWR – Lower Willamette River
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
RM – river mile
SVOC – semivolatile organic compound

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 that were appropriate to represent reference conditions for use 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 RI.

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 – RM 28.4). Eighteen stations were sampled during Round 2 sampling in November 2004, and four additional stations were sampled during Round 3 sampling 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 Attachments G4 and G6, respectively.

4.3.2 Biota Tissue

Fish tissue chemistry data collected during four LWG sampling events included samples from the upriver reach (Table 4-8). Whole-body tissue composites from upriver locations were collected for the following fish species: juvenile Chinook salmon, smallmouth bass, brown bullhead, and Pacific lamprey (as ammocoetes and macropthalmia). A single juvenile Chinook salmon stomach content composite sample collected at RM 17 was also included in the upriver dataset. The chemistry results from the fish 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; sample count includes field replicates.

BERA – baseline ecological risk assessment

LWR – Lower Willamette River

RM – river mile

5.0 IDENTIFICATION OF COPCS

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

A comprehensive COPC screen was conducted again in this BERA based on 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 (i.e., maximum detected concentration or maximum DL) surface sediment, tissue residue, surface water, and TZW concentrations, and dietary doses were compared to screening-level thresholds provided by EPA. The second tier was a refined screen of those chemicals that passed through the first tier. In the refined screen, maximum detected concentrations were compared to 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 for the different receptors groups, across all LOEs for each receptor group:

- **Invertebrates –** 106 COPCs, including 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, 18 volatile organic compounds (VOCs), 3 total petroleum hydrocarbons (TPH), cyanide, and perchlorate
- **Fishes –** 72 COPCs, including 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) –** 23 COPCs, including 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, including 6 metals, butyltin, 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 those chemicals 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.

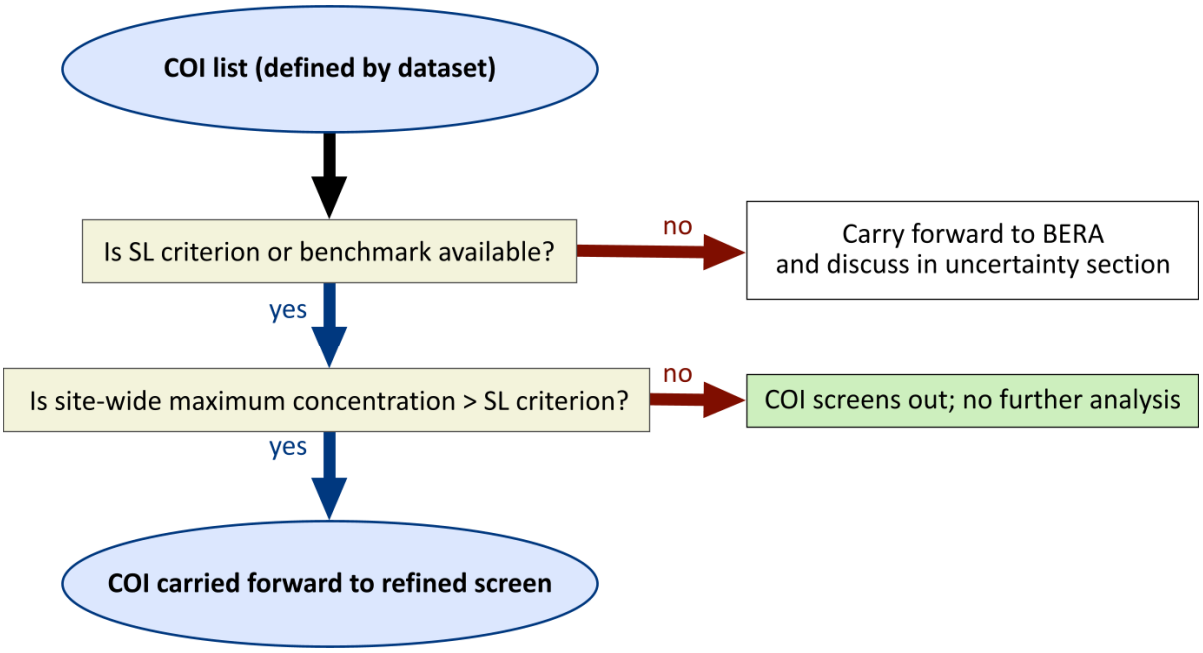


Figure 5-1. The SLERA Process – Step One for Identifying COPCs

In the SLERA, all chemicals that were detected in each medium (i.e., surface sediment, benthic invertebrate tissue, fish tissue,[24] surface water, and shallow TZW) from the Study Area in the BERA dataset were identified as media-specific COIs. COIs for the wildlife dietary assessment were those chemicals detected in both surface sediment and fish or

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

invertebrate tissue samples. 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 invertebrate and fish tissue, COIs were screened on a receptor-specific basis. Maximum COI concentrations (i.e., maximum detected values or maximum DLs) were compared to media-specific (or dietary) screening-level thresholds. COIs were not evaluated further 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-tier screening 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 to the respective screening-level thresholds. If detected concentrations did not exceed screening-level thresholds, the COI was not evaluated further.

Step 3 – For those COIs that were 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: 1) medium, 2) magnitude of exceedance, and 3) 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.

Step 4 – Tissue or diet COIs that were retained in Step 2 or 3 that were also identified as nutritionally essential were compared to available nutritional information. The tissue or diet COI was eliminated if the maximum detected concentration was less than the "nutritionally essential" concentration.[25]

[25] Ultimately, the nutritional role of selected metals was not used to eliminate any COIs as COPCs; Step 4 was not implemented due to the lack of definitive information with high certainty, as required by EPA.

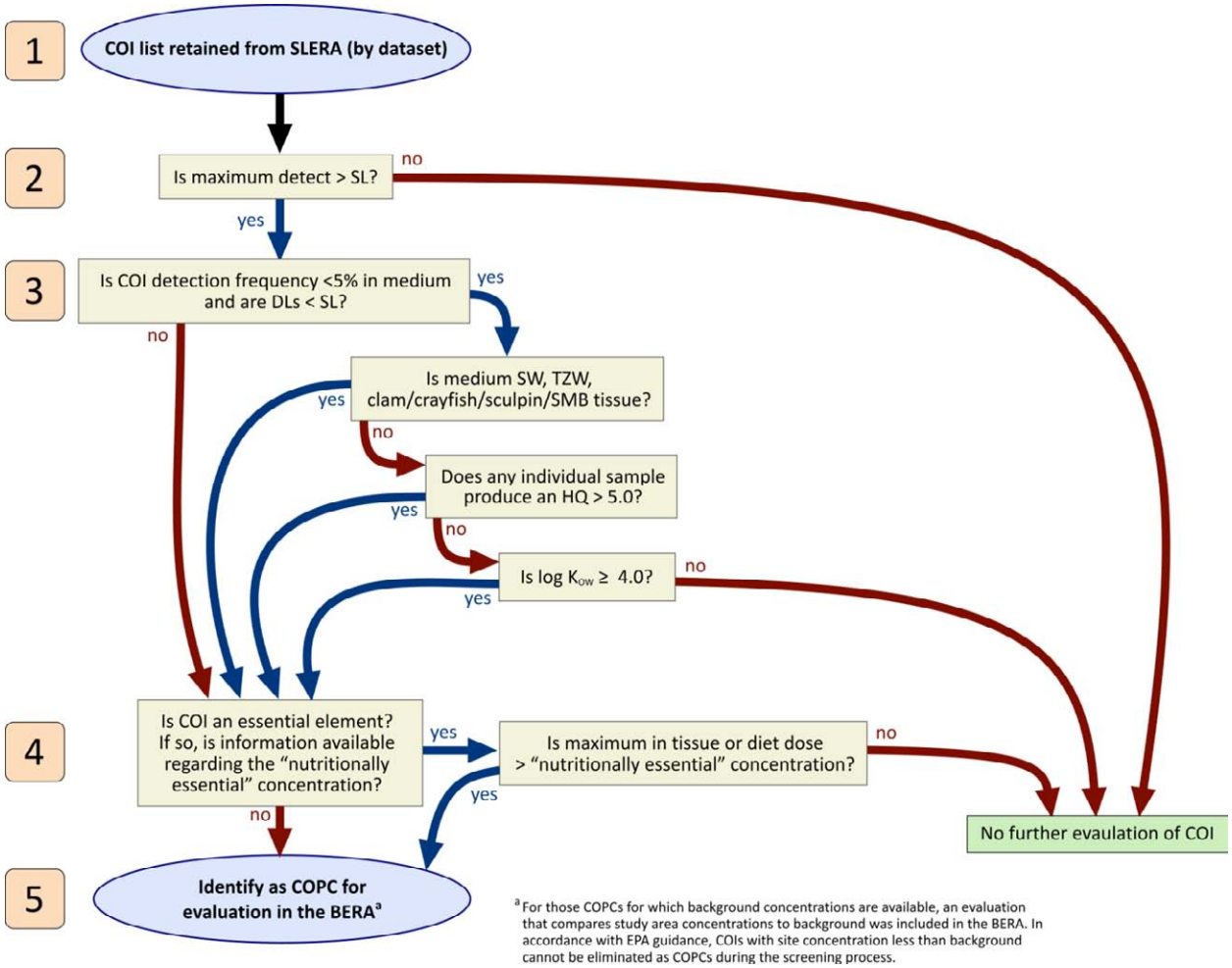


Figure 5-2. The Refined Screening Process – Step Two for Identifying COPCs

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 and fish tissue, and receptor-specific dietary scenarios. All COIs that did not screen out based on 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.

5.2 BENTHIC INVERTEBRATE COPCS

The SLERA and refined screen identified COPCs for benthic invertebrates in four media types: surface sediment, tissue, surface water, and shallow TZW using the process outlined above. These COPCs were further evaluated in the benthic risk assessment (Section 6.0).

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

- **Surface sediment** – The lowest of the SQGs provided by EPA were used to screen surface sediment data. The TPH SQGs were derived by EPA and its partners using the Alaska TPH TRVs (EPA 2008a).
- **Tissue** – Aquatic tissue-residue screening-level benchmarks were based on either 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 benchmarks were 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 benchmarks was developed by EPA and its partners (EPA 2005f) for data evaluation in the Ecological PRE. The tissue residue screening threshold for 2,3,7,8-tetrachlorodibenzo-*p*-dioxin (TCDD) was based on Isensee (1978).
- **Surface Water and TZW** – Surface water and TZW screening-level benchmarks were represented by chronic water TRVs . Water TRVs were developed using a selection hierarchy, agreed upon by LWG and EPA, of water quality regulatory benchmarks and literature-based thresholds, including national and proposed State of Oregon AWQC, Tier II values (Suter and Tsao 1996), final chronic values for individual PAH compounds (Table 3-4 of EPA 2003d), 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.

This screening step resulted in 106 COPCs being identified for benthic invertebrates based on the SLERA and refined screening steps across all four media types. These COPCs are presented in Table 5-1 and include 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, 18 VOCs, 3 TPH, cyanide, and perchlorate. Additional information about the identification of COPCs through the SLERA and refined screen for each LOE applicable to benthic invertebrates is presented in Attachment 5.

Table 5-1. Benthic Invertebrate COPCs

	Media			
COPC	**Sediment**	**Tissue**	**Surface Water**	**TZW**
Metals				
Aluminum		X	X	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			X
Silver	X			
Sodium				X
Vanadium				X
Zinc	X	X	X	X
Butyltins				
Butyltin 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

	Media			
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	
Butylbenzyl 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

	Media			
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			X	X
2,4′-DDT			X	X
4,4′-DDD	X	X	X	X
4,4′-DDE	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
Acrolein				X
Benzene				X

Table 5-1. Benthic Invertebrate COPCs

	Media			
COPC	**Sediment**	**Tissue**	**Surface Water**	**TZW**
Carbon disulfide				X
Chlorobenzene				X
Chloroethane				X
Chloroform				X
Ethylbenzene			X	X
Isopropylbenzene				X
Styrene				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 Chemicals				
Cyanide				X
Perchlorate				X

BEHP – bis(2-ethylhexyl) phthalate
COPC – chemical of potential concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
HCH – hexachlorocyclohexane
HPAH – high-molecular-weight polycyclic aromatic hydrocarbon
LPAH – low-molecular-weight polycyclic aromatic hydrocarbon
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
SVOC – semivolatile organic compound
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

5.3 FISH COPCS

The SLERA and refined screen identified COPCs for fish in three media types: tissue, surface water, and shallow TZW using the process outlined above. 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 across each media type. Screening thresholds are presented in Attachment 5. The following screening thresholds were used to derive fish COPCs for each media/LOE:

- **Tissue** – Aquatic tissue-residue screening-level benchmarks were based on either a fifth percentile LOAEL (derived from Appendix B of the Ecological PRE (Windward 2005a) or from Dyer et al. (2000)), or benchmarks were calculated as the product of EPA AWQC and a BCF. The approach for developing aquatic tissue-residue screening-level benchmarks was developed by EPA and its partners (EPA 2005f) 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 benchmarks were derived for prey tissue and sediment based on dietary dose screening-level thresholds (expressed as mg/kg bw/day). Fish dietary dose screening-level thresholds were provided by EPA (2008f) and based on no-observed-apparent-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 benchmarks were represented by chronic water TRVs. TRVs were developed using a selection hierarchy, agreed upon by LWG and EPA, of water quality regulatory benchmarks and literature-based thresholds, including national and proposed State of Oregon AWQC, Tier II values (Suter and Tsao 1996), final chronic values for individual PAH compounds (Table 3-4 of EPA 2003d), 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.

Seventy-two COPCs were identified for fish based on the SLERA and refined screening steps across all media and for the diet LOE. These COPCs are presented in Table 5-2 and include 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. Additional information about the identification of COPCs through the SLERA and refined screen for each LOE applicable to fish is presented in Attachment 5.

Table 5-2. Fish COPC

	Media/LOE			
COPC	**Tissue**	**Diet**	**Surface Water**	**TZW**
Metals				
Aluminum	X		X	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				
Butyltin 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

Table 5-2. Fish COPC

	Media/LOE			
COPC	**Tissue**	**Diet**	**Surface Water**	**TZW**
Chrysene				X
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			
SVOCs				
1,2-Dichlorobenzene				X
1,4-Dichlorobenzene				X
Dibenzofuran				X
PCBs				
Total PCBs	X			
Pesticides				
2,4′-DDD				X
2,4′-DDT				X
4,4′-DDD	X			X
4,4′-DDE				X
4,4′-DDT	X			X
Total DDx	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

Table 5-2. Fish COPC

COPC	Media/LOE			
	Tissue	Diet	Surface Water	TZW
Chloroform				X
cis-1,2-Dichloroethene				X
Ethylbenzene				X
Isopropylbenzene				X
Styrene				X
Toluene				X
Trichloroethene				X
m,p-Xylene				X
o-Xylene				X
Total xylenes				X
TPH				
Gasoline-range hydrocarbons				X
Other Chemicals				
Cyanide				X
Perchlorate				X

BEHP – bis(2-ethylhexyl) phthalate
COPC – chemical 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)
TZW – transition zone water
VOC – volatile organic compound

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 for each LOE:

- **Diet** – Receptor-specific diet-based screening level benchmarks were derived for prey tissue and sediment based on dietary dose screening-level thresholds (expressed as mg/kg bw/day). Wildlife dietary dose screening-level thresholds were provided by EPA (2008f) and were based on EPA's ecological soil screening level

(Eco SSL) documents or based on 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** – Bird egg tissue-residue screening-level benchmarks were derived for prey tissue using bird egg tissue-residue thresholds and piscivorous bird egg biomagnification factors (BMFs). Bird egg tissue thresholds derived as dietary dose screening-level thresholds were provided by EPA (2008f) and were recommended by EPA (2008j) 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 5-3 and include: 11 metals, 4 individual PAHs or PAH sums, 2 phthalates, total PCBs, dioxin TEQ, PCB TEQ, total TEQ, and 3 pesticide or pesticide sums. Additional information about the identification of COPCs through the SLERA and refined screen for each LOE applicable to wildlife is presented in Attachment 5.

Table 5-3. 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	X
Selenium	X		X
Thallium	X		
Zinc	X		
PAHs			
Benzo(a)pyrene	X		
Total HPAHs	X		X
Total LPAHs	X		
Total PAHs	X		
Phthalates			
BEHP	X		

Table 5-3. Wildlife COPCs

COPC	LOE		
	Bird Diet	**Bird Egg**	**Mammal Diet**
Dibutyl phthalate	X		
PCBs and Dioxins			
Dioxin TEQ	X	X	X
Total PCBs	X	X	X
PCB TEQ	X	X	X
Total TEQ	X	X	X
Pesticides			
Aldrin	X		
Sum DDE	X	X	
Total DDx	X		X

BEHP – bis(2-ethylhexyl) phthalate
COPC – chemical 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
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)
TEQ – toxic equivalent

5.5 AMPHIBIAN AND AQUATIC PLANT COPCS

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

Screening thresholds are presented in Attachment 5. Screening thresholds in surface water and TZW were used to derive amphibian and aquatic plant COPCs. Surface water and TZW screening-level benchmarks were represented by chronic water TRVs. Water TRVs were developed using a selection hierarchy agreed upon by LWG and EPA of water quality regulatory benchmarks and literature-based thresholds including: national and proposed State of Oregon AWQC, Tier II values (Suter and Tsao 1996), final chronic values for individual PAH compounds (Table 3-4 of EPA 2003d), 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.

Sixty-five COPCs were identified for amphibians and aquatic plants through two LOEs. These COPCs are presented in Table 5-4 and include 16 metals, butyltin, 16 individual PAHs, BEHP, 3 SVOCs, total PCBs, 6 pesticide or pesticide sums, 18 VOCs, gasoline-range hydrocarbons, cyanide, and perchlorate. Additional information about the identification of COPCs through the SLERA and refined screen for each LOE applicable to amphibians and aquatic plants is presented in Attachment 5.

Table 5-4. Amphibian and Aquatic Plant COPCs

	LOE	
COPC	**Surface Water**	**TZW**
Metals		
Aluminum	X	X
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
Vanadium		X
Zinc	X	X
Butyltins		
Butyltin 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

Table 5-4. Amphibian and Aquatic Plant COPCs

	LOE	
COPC	**Surface Water**	**TZW**
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	X	X
2,4′-DDT	X	X
4,4′-DDD	X	X
4,4′-DDE		X
4,4′-DDT	X	X
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

Table 5-4. Amphibian and Aquatic Plant COPCs

COPC	LOE	
	Surface Water	TZW
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 Chemicals		
Cyanide		X
Perchlorate		X

BEHP – bis(2-ethylhexyl) phthalate
COPC – chemical 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)
TZW – transition zone water
VOC – volatile organic compound

6.0 BENTHIC INVERTEBRATE RISK ASSESSMENT

This section presents the 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 populations, and decapod (crayfish) populations. The three main components for characterizing risks to benthic macroinvertebrates were 1) an assessment of actual or predicted sediment toxicity, a process that included sediment toxicity testing, the development of site-specific SQGs, and the comparison of Study Area sediment concentrations to site-specific and generic national/regional SQGs; 2) a tissue-residue assessment in which both empirical and predicted chemical concentrations in Study Area tissue were compared to tissue-residue TRVs; and 3) a surface water assessment in which chemical concentrations in water were compared to water TRVs derived for the protection of aquatic organisms. In addition, risks to benthic macroinvertebrates from exposure to TZW were assessed at nine areas 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		
Media	**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/regional and site-specific SQGs	X	Included in benthic community evaluation	Included in benthic community evaluation
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 to tissue TRVs	X	X	NE
	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

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

		Benthic Invertebrates		
Media	**LOE**	**Benthic Community[a]**	**Bivalves**	**Crayfish**
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.

LOE – line of evidence
NE – not evaluated
SQG – sediment quality guideline
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 derived using a reference envelope approach developed for EPA by MacDonald and Landrum (2008) for use in the BERA. These direct measures of toxicity were considered to be a primary line of evidence with low uncertainty 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 SQGs were derived from the available paired toxicity and chemistry data for surface sediments. Two numerical modeling techniques were evaluated for this purpose: the floating percentile model (FPM) (Avocet 2003) and the logistic regression model (LRM) (Field et al. 1999). Using all the endpoints measured in the toxicity tests, both models determined which samples were significantly toxic based on comparisons with the corresponding negative control results and the reference thresholds. Model-derived site-specific SQGs that could reliably predict toxicity were carried forward herein to help assess benthic risks.[26] Chemical concentrations that exceeded their respective site-specific SQGs indicated the potential for benthic toxicity. Although direct measures of toxicity provide more certainty for benthic risk assessment, the comparison of sediment chemical concentrations to site-specific SQGs provides increased spatial resolution in the absence of toxicity test data.

At EPA's direction, sample-by-sample sediment chemistry data also were compared to selected sets of regional and national SQGs, to mean SQG quotient thresholds, and to equilibrium partitioning sediment benchmarks (ESBs). In the BERA, these sediment quality

[26] The FPM met specified model reliability criteria and was used to develop site-specific SQGs, which, in turn were used to predict the presence or absence of adverse effects at stations where only chemistry data were available. The LRM did not meet the reliability criteria and could not be used to develop site-specific SQGs. The reliability analysis is presented in Section 6.2.4.

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 those SQGs were derived using any information from Portland Harbor and a number of them were derived from marine and estuarine toxicity tests and sediment chemistry. Using the same performance criteria applied in the development of site-specific SQGs, the following generic SQGs were evaluated for reliability in predicting toxicity in Portland Harbor sediment.

- Threshold effects levels (TELs) and probable effects levels (PELs) (Smith et al. 1996)
- Threshold effects concentrations (TECs) and probable effects concentrations (PECs) (MacDonald et al. 2000)
- Sediment quality standards (SQS) and cleanup screening levels (CSLs) (Ecology 1995)
- Effects range – low (ERL) and effects range – median (ERM) (Long et al. 1995)
- Lower screening level 1 (SL1), and upper screening level 2 (SL2), from the RSET program (RSET 2006)
- Mean quotients of selected regional and national SQGs (Long et al. 2006)
- ESBs for PAH mixtures, nonionic organic compounds, gamma-hexachlorocyclohexane (HCH), endrin, and dieldrin (EPA 2003c, b, 2008k)

None of the generic SQGs could reliably predict toxicity in Portland Harbor sediments (Attachment 7); therefore, the generic SQGs were not used in risk characterization for the BERA.

In the line of evidence based on benthic invertebrate tissue residues, sample-by-sample data were compared to TRVs focused on reduced survival, growth, and reproduction. The TRVs were selected from published studies and approved by EPA for use in the BERA. A TRV exceedance was used to indicate the likelihood of deleterious effects on benthic invertebrates.

Three classes of tissue concentrations 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 BSARs derived from site-specific tissue and sediment data were 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.[27] Uncertainty associated with the laboratory-exposed

[27] The usefulness of applying the tissue-residue approach to metals is highly uncertain (EPA 2007e).

benthic invertebrates is greater because the duration of the test could not fully replicate field conditions. Although the greatest uncertainty is associated with 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 water quality standards (WQS), AWQC, Tier II, or other water quality values approved by EPA for use in the BERA. In the TZW assessment porewater sample results were compared to surface water TRVs.

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. Although a LOE not formally selected for the BERA, this 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. The methods used to characterize risk for each LOE are those 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. Additional analyses to further refine the risk estimates in support of risk management decisions are also presented, as allowed by EPA's Problem Formulation (Attachment 2).

Individuals interested in the overall conclusions regarding the benthic community, including a synoptic analysis of uncertainty, are referred to Section 6.7, which presents the final determination of COCs, 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.

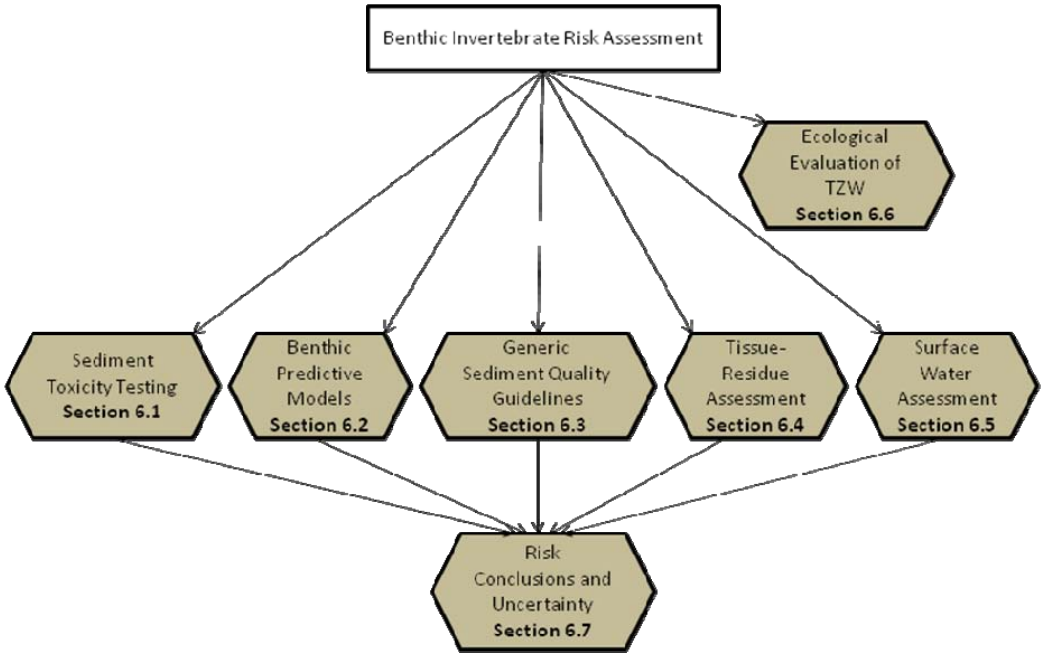


Figure 6-1. Overview of Benthic Invertebrate Risk Assessment Section Organization

6.1 SEDIMENT TOXICITY TESTING

The adverse effects on macroinvertebrates caused by exposure to surface sediments were assessed by performing two sediment toxicity tests: the 10-day survival and growth test using the midge *Chironomus dilutus* (formerly *C. tentans)* and the 28-day survival and growth test using the amphipod *Hyalella azteca*. The toxicity tests were conducted using 269[28] surface sediment samples collected within the Study Area (RM 1.9 to RM 11.8), 2 samples collected in the downtown reach (at RM 12.1), and 22 samples collected in the upriver reach (RM 15.4 to RM 25.5). As described in EPA's Problem Formulation, the growth endpoint was determined as the total mass of survivors in a sample. Because this non-traditional endpoint combines the effects of mortality and reduced growth, reduced growth is not an independent LOE from the one based on mortality. ASTM (2000) identified mean survival and mean biomass per surviving test organisms as the primary endpoints for the *C. dilutus* and *H. azteca* tests and noted that the combined endpoint based on total biomass of survivors has not been routinely applied in sediment testing (Section 4.1.1 of ASTM 2000). Toxicity was established based on statistically significant differences ($p \leq 0.05$) between test and negative control sample responses and exceedances of reference

[28] Since the toxicity tests were conducted, locations where 13 of the 269 surface sediment samples had been collected have been dredged. These 13 samples were therefore excluded from the sediment toxicity assessment. However, these samples were retained for the interpretive models (Section 6.2) and the reliability analysis of the generic SQGs (Section 6.3).

thresholds based on survival and growth in the upriver reach. The survival and growth of *Corbicula fluminea* from the 33 bioaccumulation tests conducted in Round 2 were used to qualitatively assess potential toxicity to bivalves. Those results were considered qualitative because the *C. flunimea* test was designed to evaluate bioaccumulation rather than toxicity, and the data are not sufficient to allow the results to be evaluated statistically. Crayfish will be addressed in the risk conclusions because receptor-specific toxicity testing was not conducted. This section presents the sediment toxicity assessment for macroinvertebrates (Section 6.1.1) and for bivalves (Section 6.1.2).

The details of the macroinvertebrate assessment are presented as follows:

Section 6.1.1.1 presents the reference envelope approach (MacDonald and Landrum 2008). Two reference thresholds were developed: low thresholds to identify samples that are unlikely to be associated with adverse effects and high thresholds to identify samples that are likely to be associated with adverse effects. Samples with toxicity that fell between the low and high thresholds are in a "gray area," where the likelihood of toxicity is considered to be uncertain. Additional details on the reference envelope approach are presented in Attachment 6.

Section 6.1.1.2 presents the toxicity assessment based on the sediment toxicity tests. The 256[29] sediment samples from the Study Area were evaluated based on the low and high reference thresholds.

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 growth and mortality.

Section 6.1.2.2 presents the bivalve sediment toxicity assessment and the uncertainty analysis.

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

[29] This number excludes the 13 sediment samples whose locations have since been dredged

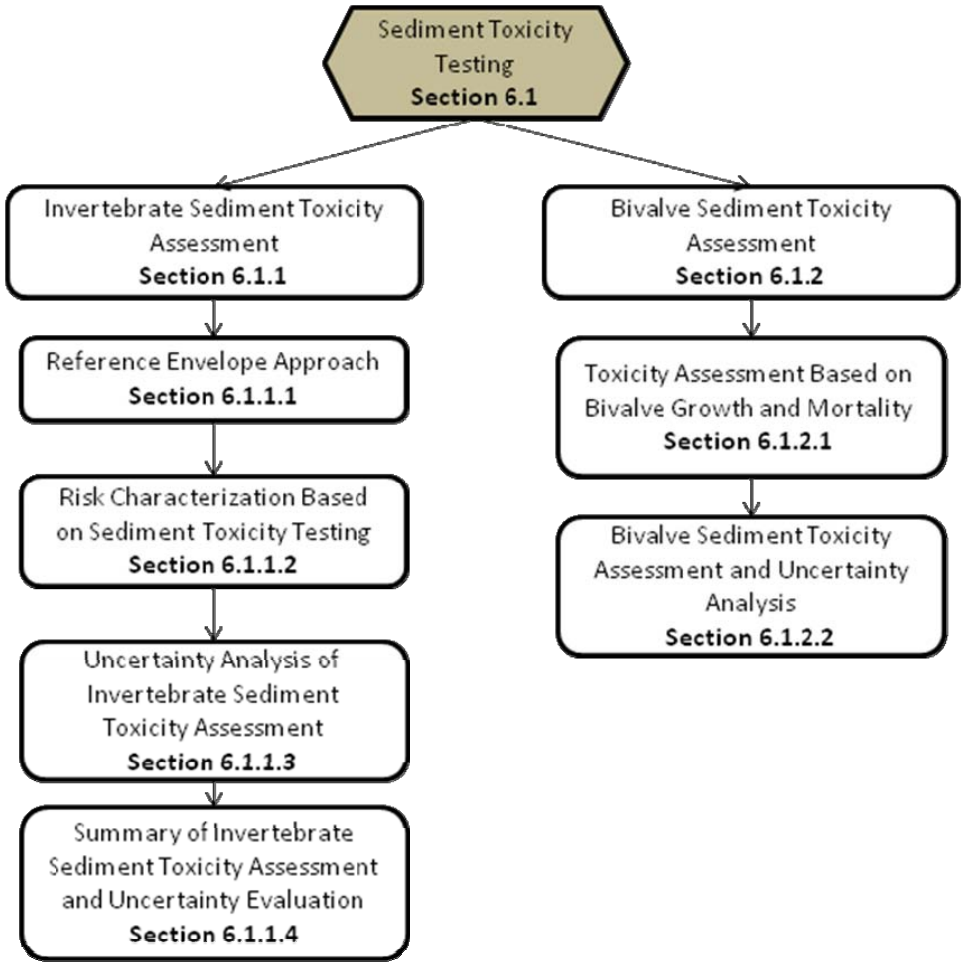


Figure 6-2. Overview of Sediment Toxicity Testing Section Organization

6.1.1 Invertebrate Sediment Toxicity Assessment

This section presents the reference envelope approach (Section 6.1.1.1), the evaluation of the sediment toxicity tests from the Study Area (Section 6.1.1.2), and the uncertainties associated with the biological effects levels (Section 6.1.1.3).

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). In this approach, both the negative control and an effect threshold representing reference conditions were used to identify sediment samples that are likely or unlikely to be associated with adverse effects to benthic invertebrates.

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 such that values that fall outside of the reference envelope are considered different from reference. The reference envelope can be used to identify adverse effects by bounding the minimum and maximum values, tolerance limits on some percentage of the empirical data distribution, confidence intervals around a mean, or upper and lower percentiles. The derivation of the reference envelope for toxicity responses for Portland Harbor followed the approach recommended in MacDonald and Landrum (2008) (Attachment 6). The methodology used differs from the reference envelope methods originally developed by Hunt et al. (1998) and (Smith 2002), which entails calculating a tolerance limit on a percentile of the reference data. For example, rather than selecting the fifth percentile of the reference area biomass data as the critical threshold, the methodology calls for calculating the lower tolerance limit for this threshold. Samples would then be considered toxic if there was greater than a 95% chance that the sample biomass was equal to or less than that expected for the lowest 5% of the reference area biomass data.

Twenty-six toxicity tests performed on sediment samples collected at 22 upriver reach locations were evaluated for inclusion in the reference envelope. Five biological and chemical criteria were used to select the reference area dataset, including toxicity test standard performance criteria, control-adjusted response rate criteria based on the National Sediment Inventory (EPA 2004b), and three chemical criteria based on the Regional Sediment Evaluation Team (RSET): interim SL1, PEC, mean PEC-Q_{dw}, and sum of the ESB toxicity units (TUs) for PAHs ($\sum$ESB-TU_{PAHs}). The criterion based on simultaneously extracted metals (SEM) and acid volatile sulfide (AVS) recommended by MacDonald and Landrum 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[30] upriver reach locations met all the criteria and the associated toxicity test data were included in the reference envelope evaluation. In addition, toxicity test data from two locations from the upstream end of the Study Area (RM 10.6 and RM 11.2) were included in the reference envelope evaluation per the request of EPA. Map 6-1 presents the locations of the 18 samples used to derive the reference envelope.

Toxicity test data based on the mean endpoint response at each station were fitted to the statistical distribution that best described the data (as determined by @Risk software). Upper percentiles (95th) were derived for the two mortality endpoints, and lower percentiles (5th) were derived for the two growth endpoints[31] to represent the normal range of reference

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

[31] Per EPA's Problem Formulation, the growth endpoint was determined as the total mass of survivors in a sample.

responses. Based on pre-selected levels of magnitude of toxicity (further details are presented in Attachment 6), a 10% difference from reference was used as the low reference threshold (i.e., low threshold +10%), and a 20% difference from reference was used as the high reference threshold per MacDonald and Landrum (2008). Table 6-2 presents the reference values and the low and high reference thresholds used in the BERA.

Table 6-2. Biological Effects Levels Based on the Reference Envelope

Test and Endpoint	Reference Value (%)[a]	Low Threshold (%)[b]	High Threshold (%)[b]
Chironomus dilutus mortality	8.1	18.1	28.1
Chironomus dilutus growth[c]	11.3	21.3	31.3
Hyalella azteca mortality	13.3	23.3	33.3
Hyalella azteca growth[c]	32.9	42.9	52.9

[a] These values were based on the 95th percentile for the control-adjusted (test-control) mortality endpoints and the fifth percentile for the control-adjusted (test/control) growth endpoints. Values are expressed as % increase in mortality or % reduction in growth.

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

[c] The thresholds are expressed as percent reduction in growth. As an example, if the total biomass in a test sediment is reduced by more than 31.3% compared to the total biomass in the negative control then it is regarded a hit based on the *Chironomus* high-reference threshold.

The results of the amphipod and midge toxicity tests were compared to batch-specific negative controls to identify sediment samples with significantly greater responses (one-tailed, parametric or non-parametric *t*-test, with $\alpha = 0.05$ comparison-wise). The toxicity test results for samples with significantly greater mortality or reductions in growth were then compared to the low and high reference thresholds derived from reference envelope samples to identify those site samples that exhibited toxicity. A toxicity test result below the low reference threshold was considered a non-adverse effect (Level 1, per EPA's definition); results above the high reference threshold were considered indicative of adverse affects (Level 3, per EPA's definition); results between the low and high reference thresholds were considered uncertain (Level 2, per EPA's definition). EPA further classified toxicity test results that were not significantly different from negative controls as Level 0. These classifications are applicable to individual organisms or sediment samples and cannot be directly linked to population level effects without consideration of spatial or temporal scale.

6.1.1.2 Toxicity Assessment Based on Sediment Toxicity Testing

By agreement between EPA and the LWG, the reference envelope approach by MacDonald and Landrum (2008) (Attachment 6) supersedes the approach provided in EPA's Problem Formulation (Attachment 2). Table 6-3 summarizes the toxicity tests results for the 269[32]

[32] Includes the 13 stations that have since been dredged.

toxicity test sampling locations in the Study Area based on a comparison with the negative control and the reference thresholds. The results were grouped into categories of not significantly different ($p > 0.05$) from the control and significantly different ($p \leq 0.05$) from the control. The latter category was divided into groups showing responses below the low reference threshold, showing responses in between the low and high reference thresholds, and showing responses greater than the high reference threshold. The majority of the sampling locations (82 to 93% depending on endpoint) had toxicity test responses below the low thresholds, whereas 6 to 13% of the stations had toxicity test responses above the high thresholds, indicating adverse effects to benthic invertebrates. Maps 6-2 through 6-5 present the results of the two toxicity tests for the 269 stations in the Study Area. It should be noted that dredging occurred at 13 toxicity test sampling locations in the Study Area after the toxicity testing had been performed, and data from those locations are no longer representative of current conditions. The individual toxicity test results are presented in Attachment 6.

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

	Number of Sampling Locations				
	Chironomus		Hyalella		Classification Based on Any One Endpoint
Category	Mortality	Growth[a]	Mortality	Growth[a]	
Level 0: Not significantly different ($p > 0.05$) from negative control	204[b]	146[b]	171	54	37
Level 1: Significantly different ($p \leq 0.05$) from control and ≤ low reference threshold	23	70	69[b, c]	179[b, c]	169
Level 2: Significantly different ($p \leq 0.05$) from control and > low reference threshold and ≤ high reference threshold	11[d]	10[d]	2	11	20
Level 3: Significantly different ($p \leq 0.05$) from control and > high reference threshold	18[e]	30[e, f]	14[e]	12[e, f]	30[e]

[a] The growth endpoint was defined as the total mass of survivors in a sample.

[b] The two downtown stations included in the reference envelope fell within these categories.

[c] One of the two downtown stations between RM 11.8 and 15.3 not included in the Study Area had significantly increased adverse effects compared to the negative control for the *Hyalella* mortality and growth endpoints.

[d] Of the 13 sampling locations that were later dredged, one sample had *Chironomus* and *Hyalella* mortality > low threshold and ≤ high threshold.

[e] The 13 sampling locations that were later dredged had samples that exceeded the high threshold as follows: *Chironomus* mortality – 6 samples, *Chironomus* growth – 8 samples, *Hyalella* mortality – 3 samples, *Hyalella* growth – 3 samples, and any one endpoint – 8 samples.

[f] Growth in sampling locations with 100% mortality was not statistically evaluated; However, these locations were included in the group in which growth exceeded the high threshold.

6.1.1.3 Uncertainty Analysis of Invertebrate Sediment Toxicity Assessment

The statistical power of the high and low reference thresholds was evaluated for the four toxicity test endpoints. Power was adequate to detect a difference from reference conditions for all thresholds and endpoints except for *Chironomus* mortality and biomass at the low threshold (i.e., 10% difference greater than the reference condition). The average minimum detectable difference (MDD) for *Chironomus* mortality was 31.7% (±11.8%) (Table 6-4), which resulted in 29% of the samples having inadequate power to detect that difference (Table 6-5). However, of those samples with inadequate power, only four samples exceeded the numeric threshold without being identified as significantly different ($p \leq 0.05$) (i.e., most differences were within the range of reference differences). The average MDD for *Chironomus* biomass was 35.3% (±14.5%), which resulted in 42% percent of the samples having inadequate power to detect that difference. However, only one sample exceeded the reference threshold without being identified as significantly different ($p \leq 0.05$). Hence, the statistical power analysis indicated that the quality of the toxicity test data was adequate to detect a difference from reference conditions for the majority of the thresholds and endpoints.

Table 6-4. Minimum Detectable Difference by Endpoint

Test Endpoint	Low Threshold	High Threshold	Mean MDD (min – max)
Chironomus mortality	18.1	28.1	31.7 (9.9 – 76.0)
Chironomus growth	21.3	31.3	35.3 (11.4 – 89.9)
Hyalella mortality	23.3	33.3	22.4 (6.5 – 64.4)
Hyalella growth	42.9	52.9	10.5 (4.5 – 24.9)

MDD – minimum detectable difference

Table 6-5. Number of Samples Exceeding the Reference Threshold That Were Not Identified as Significantly Different from Negative Controls

Test Endpoint	Low Threshold	High Threshold
Chironomus mortality	4	0
Chironomus growth	1	0
Hyalella mortality	0	0
Hyalella growth	0	0

Uncertainties Associated with Derivation of Thresholds

The reference envelope thresholds used in the benthic risk assessment were derived by selecting the 5th percentile of the reference area control-adjusted growth endpoints and the 95th percentile of the reference area control-adjusted mortality endpoints and adding 10 or 20% for the low and high toxicity thresholds, respectively. At another Superfund site, the Calcasieu (Louisiana) Estuary Superfund project, the reference envelope value was calculated as the lower 97.25th % prediction limit of the untransformed reference survival data. This value was then multiplied by 1.1 and 1.2 to derive the low and high toxicity thresholds. Using this approach and the Portland Harbor reference envelope stations, the low toxicity thresholds would be higher for all endpoints except for *Hyalella* mortality, and the high toxicity thresholds would be lower for all endpoints except for *Chironomus* growth (see table below).

Test Endpoint	Threshold	Lower Prediction Limit Thresholds (%)	BERA Thresholds (%)
Chironomus mortality	reference	9.3	8.1
	low	18.4	18.1
	high	27.4	28.1
Hyalella mortality	reference	12.8	13.3
	low	21.5	23.3
	high	30.3	33.3
Chironomus growth	reference	18.8	11.3
	low	27.0	21.3
	high	35.1	31.3
Hyalella growth	reference	39.0	32.9
	low	45.1	42.9
	high	51.2	52.9

6.1.1.4 Summary of Invertebrate Sediment Toxicity Assessment and Uncertainty Evaluation

In the invertebrate sediment toxicity assessment, up to 30 toxicity test sampling locations of the 256 locations in the risk dataset were identified as Level 3, having reduced survival or growth based one or more toxicity test endpoint and exceeding the high reference thresholds. No adverse effects were identified at 206 stations, across all endpoints (i.e., Level 0 and Level 1 in Table 6-3). Twenty stations were categorized as Level 2 by exceeding the low reference thresholds for one or more of the four endpoints but not exceeding the high thresholds. The chironomid biomass endpoint defined all but one of the adversely affected stations; chironomid mortality was the next most frequent endpoint to identify an impacted station. Amphipod endpoints agreed with each other in almost all cases, but identified less than half of the impacted stations. The statistical power analysis indicated that the quality of the toxicity test data was adequate to detect a difference from reference conditions for the majority of the thresholds and endpoints.

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. The potential for sediment toxicity based on these tests was presented in Section 6.1.1.2 and is considered applicable to the assessment of the potential for impacts on bivalves. Section 6.1.2.1 presents the qualitative toxicity assessment based on the growth and mortality measured during the bioaccumulation tests with *Corbicula fluminea*, as directed in EPA's Problem Formulation. Section 6.1.2.2 presents the toxicity assessment and uncertainty analysis.

6.1.2.1 Toxicity Assessment Based on Bivalve Growth and Mortality

The growth and mortality data from the bioaccumulation test with *C. fluminea* performed for the tissue-residue assessment provide a direct measure of the exposure of and 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, and the data area are not sufficient to allow the results to be evaluated statistically. *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, the survival of all test sediments and negative controls ranged from 97 to 100% (a total of 177 and 183 clams were exposed in the two tests). An estimate of the growth rate for clams exposed to the 33 sediment samples was made based on final tissue weights for test and control samples compared to the initial control biomass estimates. Clams exposed to 24 of the sediment samples had growth similar to or better than the control clams (80% or greater of the growth in the negative control group). Clams exposed to sediment samples collected at nine locations had less growth than those in the control group (60 to 79% of the growth in the negative control group). These nine samples were located between RM 1.9 to RM 2.2 on the eastern shore, along the east bank at RM 4.8, on the west bank at RM 6.9 and RM 6.3, in Willamette Cove and Reidell Cove, at the mouth of Swan Island Lagoon, and on the eastern bank at RM 8.2 and RM 10.

6.1.2.2 Bivalve Sediment Toxicity Assessment and Uncertainty Analysis

The uncertainties associated with this assessment are high because bioaccumulation tests are not designed to measure growth and because they use late-stage juvenile organisms whose growth has slowed, rather than early-stage juveniles with rapid growth. Initial organism biomass is estimated only to determine stocking densities and ensure sufficient tissue mass for chemical analysis at the end of the test, which introduces potential error and uncertainty. Therefore, growth rates cannot be accurately determined as part of the bioaccumulation test.

The growth estimates were calculated as the final biomass divided by the initial estimated control biomass, and the growth in the test sediments was then compared to the growth in the negative control group. Differences could not be statistically tested because the biomass measurements were not replicated; thus differences are difficult to interpret.

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 stations, results from 14 of the 33 bioaccumulation sediment sampling areas were compared with those from 25 toxicity testing sampling locations. Toxicity test samples were aggregated according to "nearest neighbor" for comparison to the equivalent bioaccumulation toxicity results. Table 6-6 presents a comparison of the toxicity tests results and bioaccumulation test results.

Table 6-6. Comparative Agreement Among Clam and Other Invertebrate Toxicity Test Endpoints

	Toxicity Test Aggregates		
Bioaccumulation Test	**Non-Toxic[a]**	**Indeterminate[a]**	**Toxic[b]**
Mortality ≤ 20% (n = 14)	11	3	0
Mortality > 20% (n = 0)	0	0	0
Growth reduction ≤ 20% (n = 9)	8	1	0
Growth reduction > 20% (n = 5)	3	2	0

a Aggregate identified as indeterminate if some of the toxicity tests indicated adverse effects and others did not.
b Non-toxic was defined as ≤ low reference thresholds, and toxic was defined as > high reference thresholds based on the reference envelope approach.

Mortality results, measured as part of the bioaccumulation tests, agreed with the co-located/nearby sediment toxicity results 79% of the time (11 out of 14 aggregates). At the remaining three aggregates, some toxicity tests indicated toxicity based on the mortality endpoint, but others did not, so the agreement was indeterminate. The bioaccumulation test results based on the growth endpoint were in agreement with the toxicity test results 57% of the time (8 out of 14 aggregates; the remaining 6 aggregates either disagreed or were indeterminate). Based on this analysis, the toxicity tests appear to be good predictors for the clam mortality endpoint. Because of the deficiencies and uncertainties associated with the growth data no reliable conclusion can be drawn; however, in no case did the bioaccumulation or the toxicity tests suggest that the sediments were toxic.

6.2 BENTHIC PREDICTIVE MODELS

To assess sediment toxicity within the Study Area where toxicity tests were not conducted, two models to develop a predictive relationship between surface sediment chemistry and toxicity responses (the FPM and the LRM) were selected following extensive discussion and review with EPA and other stakeholders. The FPM and the LRM predict relationships

from different perspectives, although their goals are similar: to identify the principal chemicals that appear to be most strongly associated with sediment toxicity and to develop a predictive relationship based on empirical data (i.e., sediment chemistry and toxicity test data) that allows for SQG development and interpretation of sediment chemistry data. The FPM focuses on the chemicals that are most likely associated with detected toxicity, and establishes SQGs for those chemicals by minimizing prediction errors (e.g., false positives and false negatives) and optimizing predictive reliability within the model. The LRM focuses on developing statistical models (using logistic regression) that describe the relationship between the probability of toxicity and the concentrations for each chemical. In addition to developing predictive statistical models for individual chemicals, the LRM can also be used to combine multiple chemicals into a single logistic curve that provides a probability of toxicity for the chemical suite being considered.

The LRM failed the reliability analysis and the model was therefore not used to identify COCs for benthic invertebrates or derive SQGs. Attachment 6 presents a detailed description of the LRM and the reliability analysis outcomes based on Portland Harbor data.

Four Useful Reliability Parameters

The false negative rate is the number of toxicity test sampling locations that were predicted to be non-toxic by the SQGs, but were toxic based on the toxicity test results, divided by the number of samples that were actually toxic. Hence, this error rate identifies the proportion of toxic samples that was erroneously deemed as non-toxic.

The false positive rate is the number of toxicity test sampling locations that were predicted to be toxic by the SQGs, but were non-toxic based on the toxicity test results, divided by the number of samples that were actually non-toxic. Hence, this error rate identifies the proportion of non-toxic samples that was erroneously deemed as toxic.

The predicted no-hit reliability rate is the number of correctly predicted non-toxic locations divided by the total number of predicted non-toxic locations. This reliability rate identifies the proportion of locations that were predicted to be non-toxic that were correctly predicted.

The overall reliability rate is the number of correctly predicted sampling locations (both true negatives and true positives) divided with the total number of sampling locations. This reliability rate identifies the overall proportion of correctly predicted locations.

Toxicity Test Result	**Result Predicted by SQG**		
	No-Hit	**Hit**	
No-hit	True no-hit (negative)	False hit (positive)	**Total true no-hits (negative)**
Hit	False no-hit (negative)	True hit (positive)	**Total true hits (positive)**
	Total predicted no-hits	**Total predicted hits**	**Total sample size**

Two effects thresholds were developed to evaluate risks to benthic invertebrates from exposure to surface sediments. The low effects thresholds were developed with the narrative intent of identifying chemical concentrations below which adverse effects on benthic invertebrates are not expected to occur (equivalent to EPA Levels 0 and 1). The narrative intent of the high effects thresholds was to identify concentrations above which adverse effects on benthic invertebrates are likely to occur (i.e., Level 3 responses). The overall goal was to identify areas that potentially pose unacceptable risks to benthic invertebrates and areas that are likely to exhibit no adverse effects on benthic invertebrates. Where there is separation between the low and high effects thresholds, risks are considered uncertain (considered Level 2, by EPA definition).

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

Section 6.2.1 presents a summary of the data used in the models.

Section 6.2.2 presents the FPM and further refinements of the selection of chemicals used in this model.

Section 6.2.3 presents the derivation of site-specific SQGs using the FPM

Section 6.2.4 presents the risk characterization based on the site-specific SQGs

Figure 6-3 presents a flowchart of the organization of the benthic interpretive models section.

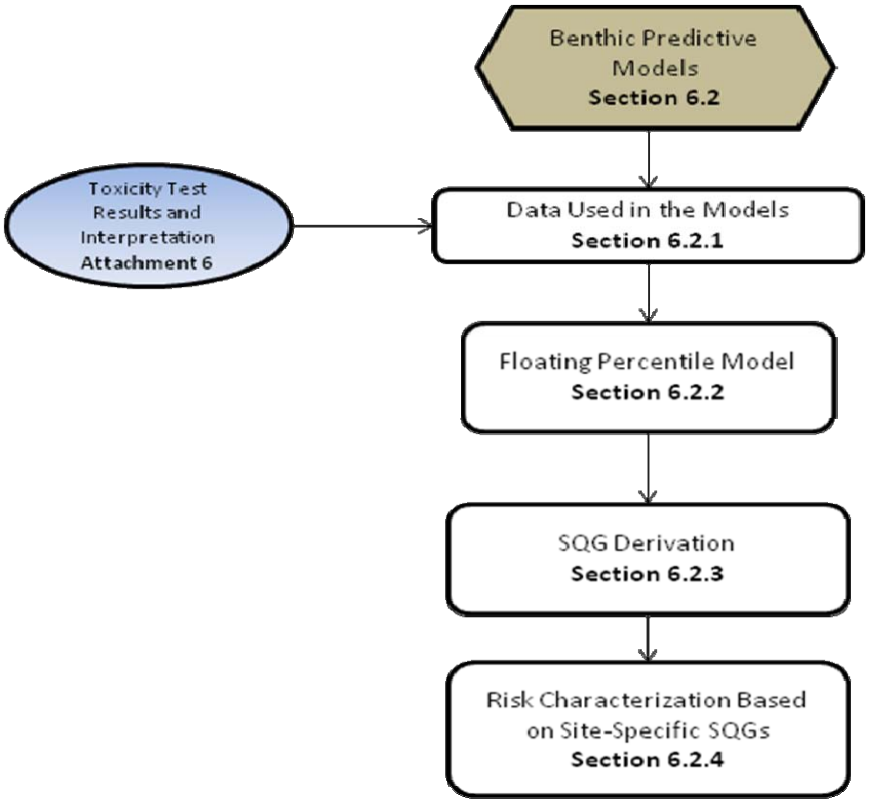


Figure 6-3. Overview of the Benthic Interpretive Models Section Organization

6.2.1 Data Used in the Models

The predictive models were developed using the 293 samples with co-located sediment chemistry and toxicity test data. Of these surface sediment samples, 269 samples were collected within the Study Area, 2 samples were collected in the downtown reach, and 22 samples were collected in the upriver reach. Thirteen of the toxicity test sampling locations in the Study Area were dredged after toxicity testing had been performed and are not part of the BERA risk dataset; however, these sampling locations have been retained for use in model development. The toxicity test data included the 10-day sediment toxicity test measuring survival and growth in *Chironomus dilutus* and the 28-day sediment toxicity test measuring survival and growth 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). Separate models were developed for each of the four endpoints (i.e., *Chironomus* mortality and growth and *Hyalella* mortality and growth). The BERA surface sediment chemistry data used in the models are summarized in Section 4.0.

For model development, only detected values were used because undetected chemistry values cannot be used to develop a predictive relationship between sediment chemistry and sediment toxicity (Avocet 2003). One exception was non-detects that were part of a chemical group total (e.g., total PCBs). In these cases, one-half the detection limit was used

in the group totals (summation rules are presented in Attachment 3). Chemical data qualified with N, NJ, and NJT[33] qualifiers were included in the models, at EPA's direction. Chemicals with fewer than 30 detected values were excluded from the modeling effort because at least 30 data points are needed to create a usable distribution for the development of SQGs, based on analyses of other datasets from Oregon and Washington (Avocet 2003).

Certain conventional parameters (specific gravity, liquid limit, individual grain size, and total solids) were screened out of both models because they are not considered contaminants. However, other conventional parameters, including percent fines, bulk sediment ammonia, and sulfides, were retained in the two models because they enhanced the model performance. Percent fines was not considered a contaminant, and because ammonia and sulfides are natural sediment constituents that may cause toxicity, they were not addressed as contaminants. The analytes screened out as a result of the above factors are presented in Table 6-7.

Table 6-7. Chemicals Screened Out Prior to Model Development

Individual Chemicals with Fewer than 30 Detected Values[a]	Number of Detects
1,2,3-Trichloropropane	1
1,2,4-Trichlorobenzene	6
1,2-Dichlorobenzene	5
1,2-Dichloroethane	1
1,4-Dichlorobenzene	11
2,3,4,5-Tetrachlorophenol	5
2,3,4,6 and 2,3,5,6-Tetrachlorophenol coelution	7
2,3,5,6-Tetrachlorophenol	3
2,4,5-Trichlorophenol	6
2,4,6-Trichlorophenol	19
2,4-D	6
2,4-DB	1
2,4-Dichlorophenol	2
2,4-Dimethylphenol	1
2-Chlorophenol	1
2-Methylphenol	3

[33] 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).

Table 6-7. Chemicals Screened Out Prior to Model Development

Individual Chemicals with Fewer than 30 Detected Values[a]	Number of Detects
4-Chloro-3-methylphenol	6
4-Chloroaniline	1
4-Nitroaniline	1
Acetone	4
Aniline	16
Aroclor 1221	2
Aroclor 1242	15
Aroclor 1268	12
Benzene	19
Chlorobenzene	13
Endosulfan sulfate	3
Gasoline-range hydrocarbons	21
Heptachlor	20
Heptachlor epoxide	6
Hexachlorobutadiene	24
Hexachloroethane	27
Isopropylbenzene	21
m,p-Xylene	17
MCPA	2
MCPP	1
Mirex	7
n-Nitrosodiphenylamine	10
Oxychlordane	22
o-Xylene	23
Styrene	1
Toluene	5
Benzoic acid	26
Bis(2-chloroethyl) ether	2
Carbon disulfide	12
Chloroform	15
Chromium hexavalent	3
Dimethyl phthalate	19
Di-n-octyl phthalate	5
Endrin aldehyde	6

Table 6-7. Chemicals Screened Out Prior to Model Development

Individual Chemicals with Fewer than 30 Detected Values[a]	Number of Detects
Ethylbenzene	14
Methyl tert-butyl ether	6
Methylethyl ketone	20
Total xylenes	25
Trichloroethene	6

[a] Analytes were detected at least once. An analyte not listed in the table and not retained for model development was never detected.

D – dichlorophenoxyacetic acid

DB – dichlorophenoxy)butyric acid

MCPA – 2-methyl-4-chlorophenoxyacetic acid

MCPP – methylchlorophenoxypropionic acid

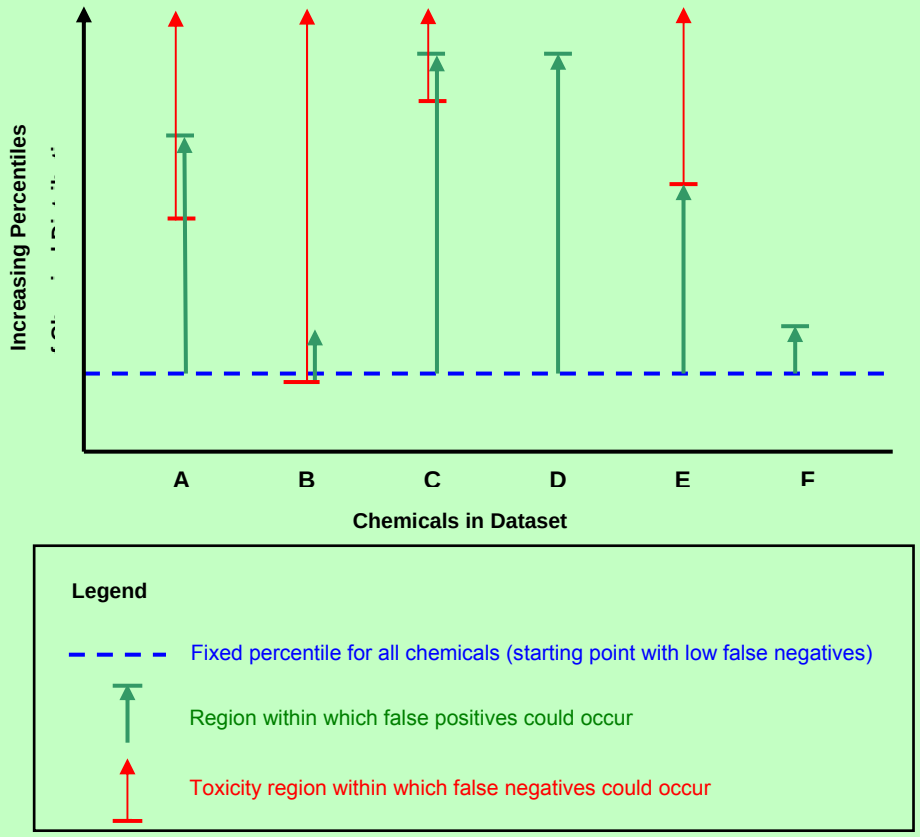
Use of NJ-Qualified Data in the Predictive Models

In the first modeling effort, NJ-qualified surface sediment chemistry data were not included because of the uncertainties (regarding both the presence and quantity of the chemical) associated with these data. The results of the two predictive models were presented in the *Interpretive Report: Estimating Risks to Benthic Organisms Using Predictive Models Based on Sediment Toxicity Tests* (Windward et al. 2006). EPA later questioned the exclusion of NJ-qualified data and stated that "Generally, EPA recommends including the N, NJ, and NJT values for modeling purposes" (EPA 2006b). LWG agreed to revise the surface sediment chemistry dataset by including the NJ-qualified data, and they 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.

6.2.2 Floating Percentile Model

The FPM uses a threshold approach to develop site-specific SQGs. The model allows the user to select an optimal false negative rate (i.e., rate associated with erroneously concluding the sediments are not toxic) and then adjusts the candidate SQG values until false positive rates (i.e., rates associated with erroneously concluding that sediments are toxic) are decreased to their lowest possible level while retaining the same low false negative rate. This adjustment process is unlike most other existing SQG sets, which require the SQGs for all chemicals to be based on the same percentile of the hit (i.e., exceedance of an effects threshold) or no-hit distribution. After the 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. In this manner, optimized site-specific SQGs can be developed for a number of different target false negative rates, allowing the trade-offs between false negatives and false positives to be evaluated relative to risk management costs. The modeling was done using the automated FPM Microsoft Excel® spreadsheets provided by RSET (Anderson 2008) and the designation of toxicity was based on the reference envelope approach.

Floating Percentile Model



The basic concept behind the FPM is to select an optimal percentile of individual chemical concentrations within a multiple chemical dataset that, collectively, accurately predict toxicity. The selected percentile represents a chemical-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 chemical's overall concentration distribution. The green vertical line for each chemical 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 chemicals. The threshold for each individual chemical 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 chemicals is maximized. In the figure, the onset of toxicity within a chemical distribution varies by chemical and may even occur at the minimum value (e.g., Chemical B) or not within the measured range (Chemical D). Once each chemical has been individually adjusted upward, the false positive rate will have been significantly reduced while a low false negative rate is retained. Most chemicals should be at or near their actual toxicity range, rather than at a level arbitrarily assigned by a fixed percentile.

After the removal of chemicals with < 30 detected concentrations and selected conventional parameters, the list of chemicals was refined for FPM development using the following criteria:

- Chemicals not summed into any totals were included (e.g., metals, phthalates, and conventionals).
- Chemicals represented as totals or sums were included (e.g., total PAHs and sum dichlorodiphenyldichloroethanes [DDDs]).
- Chemicals included in totals were excluded (e.g., individual PAHs, PCBs, and xylenes).

This reduced list of chemicals was processed through the FPM analysis of variance (ANOVA) program to test for differences between the hit and no-hit concentrations for each endpoint. The ANOVA was performed on 48 chemicals; 36 chemicals with significant differences between toxic and non-toxic distributions for at least one endpoint were carried forward for development of SQGs (Table 6-8).

Table 6-8. Chemicals Retained in the Floating Percentile Model After ANOVA Analysis

Chemical	Number of Detections	Retained After ANOVA Analysis?
Metals		
Aluminum	293	No
Antimony	224	Yes
Arsenic	293	Yes
Cadmium	291	Yes
Chromium	292	Yes
Copper	293	Yes
Lead	293	Yes
Mercury	289	Yes
Nickel	281	No
Selenium	120	No
Silver	293	Yes
Zinc	293	Yes
Butyltins		
Butyltin	70	No
Dibutyltin	74	Yes
Tetrabutyltin	34	No
Tributyltin	73	No

Table 6-8. Chemicals Retained in the Floating Percentile Model After ANOVA Analysis

Chemical	Number of Detections	Retained After ANOVA Analysis?
PAHs		
Sum of 34 ESB PAHs	288	Yes
Total benzofluoranthenes	280	Yes
Total HPAHs[a]	287	Yes
Total LPAHs[b]	283	Yes
Total PAHs[a,b]	288	Yes
SVOCs		
Benzyl alcohol	61	Yes
Carbazole	205	Yes
Phthalates		
BEHP	209	No
Butylbenzyl phthalate	96	No
Diethyl phthalate	44	Yes
Phenols		
4-Methylphenol	123	Yes
Pentachlorophenol	75	No
Phenol	88	Yes
PCBs		
Total PCBs	231	Yes
Dioxins and Furans		
Total dioxins furans	70	No
Pesticides		
Aldrin	85	No
delta-HCH	52	Yes
Dieldrin	48	Yes
Endrin	31	Yes
Endrin ketone	70	Yes
Sum DDD	280	Yes
Sum DDE	269	Yes
Sum DDT	242	Yes
Total chlordane	235	Yes
Total DDx	285	Yes
Total endosulfan	85	No

Table 6-8. Chemicals Retained in the Floating Percentile Model After ANOVA Analysis

Chemical	Number of Detections	Retained After ANOVA Analysis?
TPH		
PYO-PTO	41	Yes
Conventionals		
Ammonia	292	Yes
Sulfide	240	Yes
Total percent fines	293	Yes
Total organic carbon	293	Yes

[a] Includes benzo(a)anthracene, benzo(a)pyrene, benzo(b)fluoranthene, benzo(b+k)fluoranthene, benzo(g,h,i)perylene, benzo(j+k)fluoranthene, benzo(k)fluoranthene, chrysene, dibenzo(a,h)anthracene, fluoranthene, indeno(1,2,3-cd)pyrene, pyrene

[b] Includes 2-methylnaphthalene, acenaphthene, acenaphthene, acenaphthylene, anthracene, fluorine, naphthalene, phenanthrene

ANOVA – analysis of variance
BEHP – bis(2-ethylhexyl) phthalate
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
ESB – equilibrium partitioning sediment benchmark
HCH – hexachlorocyclohexane
HPAH – high-molecular-weight polycyclic aromatic hydrocarbon
LPAH – low-molecular-weight polycyclic aromatic hydrocarbon
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
PYO – pyrogenic (petroleum compound)
PTO – petrogenic (petroleum compound)
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
-

6.2.3 SQG Derivation

The selection of the SQGs from the FPM involved a series of model runs and subsequent reliability analyses. The reliability analyses were based on achieving specific goals for a set of SQGs. Goals presented in the draft Washington State freshwater guidelines (Avocet 2003) were adopted for use in Portland Harbor. Specifically, both false negatives and false positives should be below 20%, and the overall reliability should be above 80%. In addition, predicted no-hit reliability should be above 90% in order to have greater confidence in defining a sampling location as having no adverse effects (Avocet 2003).

Model runs and selection of the final SQGs are described in the following steps.

1. For each of the eight endpoints (*Chironomus* and *Hyalella* mortality and growth, for low and high reference thresholds), the set of chemicals in Table 6-8 was processed in the FPM model in three separate runs. The first run involved requesting SQGs for a single optimal false negative rate of 20%, the second run involved requesting SQGs for a range of false negative rates (10, 20, and 30%) and retaining only the final set of SQGs for the 20% false negative rate, and the third run involved entering

the minimum SQG for each chemical from the 10, 20, and 30% runs into the model, and computing the error and reliability rates for that set of SQGs.

From these three sets of SQGs, the best-performing set of SQGs for each endpoint and toxicity threshold was selected. The best-performing set for each endpoint had the highest overall reliability rate and the most balanced false positive and false negative rates. Each of the best-performing sets of SQGs for each endpoint met all the reliability criteria, with false negative and false positive rates ranging from 5 to 20% and from 4 to 20%, respectively, overall reliabilities > 80%, and predicted no-hit reliabilities ranging from 96 to 100%.

2. The lowest low SQG was selected from all the low reference threshold endpoints and the lowest high SQG was selected from all the high reference threshold endpoints to represent the initial low and high SQG for each chemical.

3. All error and reliability rates for each set of the initial SQGs were calculated using the appropriate pooled (low or high reference threshold) endpoint (i.e., if any individual endpoint indicated toxicity, the pooled endpoint was coded as a hit).

4. Chemicals were removed from the list of SQGs if one of the following conditions occurred:

   - The SQG was equal to the maximum concentration that occurred within the model dataset
   - The maximum no-hit concentration was higher than the maximum hit concentration
   - Removal caused no change in any of the overall error and reliability rates (tested through trial and error) or SQGs for other chemicals (indicated that the chemical was redundant and correlated with other chemicals).

These steps result in a paired set of site-specific SQGs in which the low SQGs identified the low toxicity thresholds and the high SQGs identified the high toxicity thresholds. The high site-specific SQGs met the overall error and reliability rates described above (Table 6-9) (Avocet 2003). The set of low SQGs did not achieve the desired false positive rate, but because the goal for this endpoint is to designate non-toxic areas, incorrectly predicting toxicity above this threshold was not considered as important as guaranteeing that false negative rates were maintained. However, the high false positive rate for the low SQG set had the effect of lowering the overall reliability rate below 80%.

Table 6-9. Overall Error and Reliability Rates for Low and High SQG Sets from FPM

Reliability Parameters	Low Threshold	High Threshold
%False negatives	20.0	19.4
%False positives	39.5	14.8

Hit reliability	80.0	80.6
No-hit reliability	60.5	85.2
Predicted hit reliability	34.3	43.3
Predicted no-hit reliability	92.2	96.9
Overall reliability	64.5	84.6

FPM – floating percentile model
SQG – sediment quality guideline

The final set of site-specific SQGs is presented in Table 6-10. The SQGs for most chemicals were set at their apparent effects thresholds (AETs) (maximum no-hit concentration) (Barrick et al. 1988)for the dataset; thus, they were associated with false positive rates of 0%. False positive rates for chemicals whose SQGs were set lower than the AET were all less than 15% for the low SQGs and 8% for the high SQGs.

Table 6-10. Final Set of Site-Specific SQGs

	Selected SQGs		False Positive Rate	
Chemical	**Low Toxicity Threshold**	**High Toxicity Threshold**	**Low Toxicity Threshold**	**High Toxicity Threshold**
Metals (mg/kg dw)				
Cadmium	0.714	3.51	6%	0%
Copper	76.9	562	15%	0%
Mercury	0.155	0.722	11%	0%
Silver	1.72	1.72	0%	0%
PAHs (µg/kg)				
Total benzofluoranthenes	53,000	53,000	0%	0%
Total HPAHs	610,000	610,000	0%	0%
Total LPAHs	2,000	18,000	8%	2%
SVOCs (µg/kg)				
Benzyl alcohol	36	36	0%	0%
Carbazole	1,100	1,100	0%	0%
Phenols (µg/kg)				
Phenol	120	120	0%	0%
PCBs (µg/kg)				

Table 6-10. Final Set of Site-Specific SQGs

Chemical	Selected SQGs		False Positive Rate	
	Low Toxicity Threshold	High Toxicity Threshold	Low Toxicity Threshold	High Toxicity Threshold
Total PCBs [a]	300	500	13%	8%
Pesticides (µg/kg)				
delta-HCH	1.29	2.35	3%	0%
Dieldrin	21.5	21.5	0%	0%
Endrin	20.7	20.8	0%	0%
Endrin ketone	8.5	8.5	0%	0%
Total DDx	218	218	5%	4%
Conventionals				
Ammonia (mg/kg)	132	276	10%	0%
Sulfide (mg/kg)	29.1	38.5	5%	4%

[a] Total PCBs are based on Aroclors.

DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
HCH – hexachlorocyclohexane
HPAH – high-molecular-weight polycyclic aromatic hydrocarbon
LPAH – low-molecular-weight polycyclic aromatic hydrocarbon
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
SVOC – semivolatile organic compound
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)

6.2.4 Risk Characterization Based on Site-Specific SQGs

As stated previously, site-specific SQGs were derived only from the FPM because the LRM failed to meet the established reliability criteria. Two sets of site-specific SQGs were derived using the FPM, one based on high reference thresholds and one based on low reference thresholds. The site-specific SQGs based on the high reference threshold have similar narrative intent as the generic high SQGs (PEL, PEC, ERM, CSL, and SL2), and the site-specific SQGs based on the low reference threshold have similar narrative intent as the generic low SQGs (TEL, TEC, ERL, SQS, and SL1).

6.2.4.1 Comparison of Study Area Concentrations to Site-Specific SQGs

SQGs that could reliably predict toxicity were derived for 16 chemicals or chemical sums (four metals, three PAHs, two SVOCs, phenol, total PCBs, and five pesticides or pesticide sums) and two conventional (ammonia and sulfides) parameters (Table 6-10). For some of the chemicals, the SQGs for the high and low effects thresholds were the same value, providing a clear separation between predicted toxic and non-toxic sediment samples. For

other chemicals, the two SQGs were separated by some concentration range, creating a larger uncertainty in the prediction of toxic and non-toxic samples.

Using site-specific SQGs, toxicity was predicted for 1, 213 surface sediment sampling locations in the BERA database for which there were only chemistry data. Toxicity could not be assessed at 37 locations (3.1% of all samples) because chemical analyses were not performed for the site-specific SQGs. Chemical concentrations less than the low SQGs identified samples that would be unlikely to cause adverse effects (Level 0 or 1) and those locations where no low SQGs were exceeded are presented in Map 6-6. Concentrations exceeding the high threshold SQGs were identified as likely to cause adverse effects on benthic invertebrates (Level 3) (Map 6-7). In places where sediment chemical concentrations exceeded the low SQGs but not the high SQGs, adverse effects to benthic invertebrates are uncertain and were classified as Level 2.

The areas that were predicted to be toxic or where toxicity was uncertain were mapped based on a spatial analysis of the chemistry sample exceedances of the high and low SQGs. The spatial analysis used a natural neighbors spatial interpolation (NN-interpolation)[34] statistical algorithm (de Smith et al. 2008) to estimate exceedance areas based on individual sample points. Where too few detected values were available to allow interpolation, exceedances were displayed as points. Maps 6-8 through 6-24 present the areas (or sample points) that may represent toxicity to benthic organisms based on exceedance of any one high SQG (Level 3). Uncertain toxicity areas (Level 2 exceedances) are also shown.

Unacceptable benthic community risks from exposure to metals was limited in both the spatial extent and magnitude of the exceedances. Sediment concentrations of cadmium, copper, mercury, and silver were categorized as Level 3 in localized areas of the river (fewer than 20 sediment samples exceeded the high SQGs for these metals, and few locations were contiguous). Exceedance factors (EFs) ranged from 1.0 to 75, but almost all EFs for metals were less than 3.0.

Unacceptable benthic community risks from PAHs were predicted in channel or western shoreline sediments between RM 5.0 and RM 6.8, with a few additional areas along the eastern shoreline at RM 2.8, RM 4.6, and RM 4.9. Total benzofluoranthenes, total high-molecular-weight PAHs (HPAHs), and total low-molecular-weight PAHs (LPAHs) exceeded their respective SQGs; areas exceeding the total benzofluoranthene and total HPAH SQGs were included within the areas defined by total LPAH exceedances. LPAHs also exhibited the greatest magnitude of exceedance relative to the high SQG.

[34] 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 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.

Other SVOCs exceeding their respective SQGs included benzyl alcohol, carbazole, and phenol. SQG exceedances by these chemicals were limited to only a few samples scattered throughout the Study Area. The samples that exceeded the carbazole SQG also exceeded the total LPAH SQG. Level 3 phenol exceedances were also found within the LPAH Level 3 exceedance areas.

Sediment concentrations of PCBs that were higher than the total PCB SQG occurred at the eastern shoreline between RM 2.0 and RM 2.5, the southwestern portion of International Slip, smaller individual locations between RM 4.0 and RM 4.5 on the eastern shoreline, RM 6.9 along the eastern shoreline, RM 7.6 along the western shoreline, portions of Swan Island Lagoon, the western shoreline between RM 8.8 and RM 9.3, and a section of the channel and eastern shoreline between RM 11.0 and RM 11.8.

Few pesticides, with the exception of total DDx, had a sufficient number of detected values to allow spatial interpolation. Individual sediment samples exceeding the SQGs for 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.

6.2.4.2 Comparison of Non-Study Area Concentrations to Site-Specific SQGs

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 SQGs, where available. The number of detected concentrations exceeding site-specific SQGs from each of these non-Study Area reaches, as well as the Study Area, are presented in Table 6-11. Site-specific SQGs were exceeded in at least one sediment sample collected outside the Study Area for the following COPCs: mercury, silver, LPAHs, benzyl alcohol, carbazole, and total PCBs.

Table 6-11. Study Area and Non-Study Area Surface Sediment Exceedances of Site-Specific SQGs

	Number of Site-Specific SQG Exceedances/ Total Number of Samples			
COPCa	**Downstream Reach**	**Multnomah Channel**	**Downtown Reach**	**Study Area**
Metals				
Cadmium	0/38	0/17	0/38	5/1,092
Copper	0/38	0/17	0/38	6/1,102
Mercury	1/38	0/17	3/38	10/1,089
Silver	0/38	0/17	4/38	15/1,090
PAHs				
Total benzofluoranthenes	0/38	0/17	0/38	12/1,045

Table 6-11. Study Area and Non-Study Area Surface Sediment Exceedances of Site-Specific SQGs

COPCa	Number of Site-Specific SQG Exceedances/ Total Number of Samples			
	Downstream Reach	Multnomah Channel	Downtown Reach	Study Area
HPAHs	0/38	0/17	0/38	8/1,150
LPAHs	0/38	0/17	1/38	39/1,150
SVOCs				
Benzyl alcohol	1/38	0/17	3/38	4/970
Carbazole	0/38	0/17	1/38	16/973
Phenols				
Phenol	0/38	0/17	0/37	3/1028
PCBs				
Total PCBs	1/38	0/17	0/38	24/845
Pesticides				
delta-Hexachlorocyclohexane	0/38	0/17	0/38	5/825
Dieldrin	0/38	0/17	0/38	2/823
Endrin	0/38	0/17	0/38	2/677
Endrin ketone	0/38	0/17	0/38	4/828
Total DDx	0/38	0/17	0/38	51/873
Conventionals				
Ammonia	NA	NA	NA	4/200
Sulfide	NA	NA	NA	11/198

[a] Only COPCs with site-specific SQGs are presented.

COPC – chemical of potential concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
HPAH – high-molecular-weight polycyclic aromatic hydrocarbon
LPAH – low-molecular-weight polycyclic aromatic hydrocarbon
NA – not analyzed
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
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)

6.3 GENERIC SEDIMENT QUALITY GUIDELINES

The effects of sediment-associated chemicals on the benthic community were also assessed for chemistry-only stations by comparing Portland Harbor sediment chemistry data to generic regional and national SQGs as described in the EPA's Problem Formulation (Attachment 2). 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, representing different narrative intents, are typically selected based on a preponderance of evidence or

distributional characteristics of the paired effects/concentration data. Theoretical SQGs, calculated based on equilibrium partitioning theory, have been developed for selected non-ionic organic compounds, including PAHs, dieldrin, and endrin (EPA 2003b, c, d, 2008k). Per EPA's Problem Formulation (Attachment 2), the generic SQGs were used to evaluate the potential for toxicity and subsequent risk to the benthic community, along with site-specific SQGs derived from Portland Harbor sediment chemistry and toxicity test data.

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 presents the risk characterization using the selected sets of low SQGs that passed the reliability analysis.

Figure 6-4 presents a flowchart of the generic SQGs section organization.

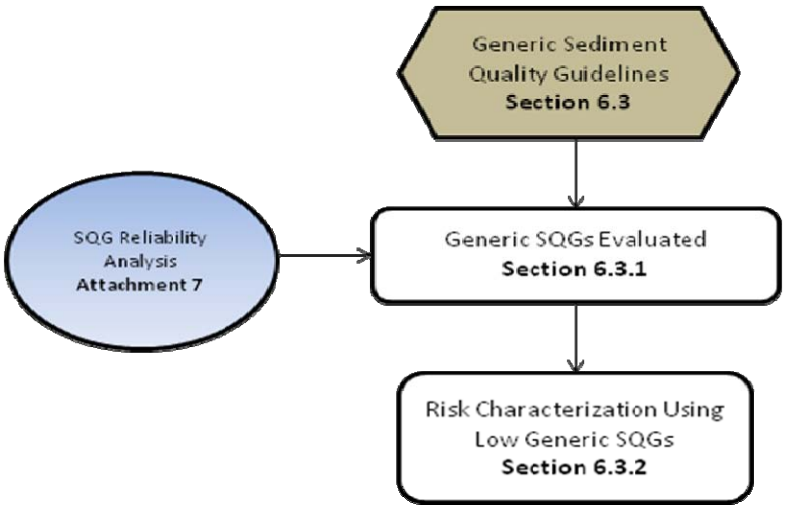


Figure 6-4. Overview of Generic Sediment Quality Guidelines Section Organization

6.3.1 Generic SQGs Evaluated

Several compendia of empirical SQGs have been published, 11 of which were included in this assessment. These SQGs come in pairs of "low toxicity" (low SQGs) and "high toxicity" (high SQGs) thresholds with differing purposes or narrative intent. In general, the low SQGs are sediment chemical concentrations below which adverse effects on benthic invertebrates are not expected to occur, and the high SQGs are concentrations above which adverse effects on benthic invertebrates are somewhat likely to occur. Concentrations from the low up to at least the high SQGs are in the range where toxicity is uncertain. Table 6-12 identifies the generic SQGs that were used in this assessment and describes the basis and narrative intent of each. The narrative intent of site-specific SQGs is described in Section 6.2.

Table 6-12. Generic SQGs, Their Derivation, and Narrative Intent

SQG	Toxicity Threshold	Narrative Intent	Source
TEC	Low	The consensus-based TECs were intended to identify the concentration of sediment-associated contaminants below which adverse effects on sediment-dwelling organisms are not expected to occur. The TECs were derived as geometric means of SQGs (including TELs and ERLs) in the literature with similar narrative intent. The SQGs are derived from a combination of freshwater and marine toxicity tests.	Macdonald et al. (2000)
TEL	Low	The TELs were intended to estimate the concentrations of chemicals below which adverse biological effects only rarely occurred. The TELs were derived by calculating the geometric mean of the 15th percentile of the effect dataset and the 50th 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)
SL1	Low	The SL1s correspond to concentrations below which adverse effects on benthic organisms would not be expected. The SL1s were derived using the FPM and freshwater toxicity tests with both mortality and growth endpoints. These SQGs are derived based on a regional (Western Washington and Oregon) biological effects database.	RSET (2006)
ERL	Low	The concentrations below the ERLs represent minimal-effects thresholds that were intended to estimate conditions below which effects would be rarely observed. The ERLs were derived as the 10th percentile of a database composed of multiple studies, species, and effects endpoints (predominantly mortality). These SQGs are derived from a national biological effects database that includes other sediment SQGs, marine toxicity tests, and benthic field studies.	Long et al. (1995)
SQS	Low	The SQS corresponds to a concentration that will result in no adverse effects, including no acute or chronic adverse effects on biological resources and no significant health risk to humans. SQS are generally based on the LAETs (see LAET). These SQGs are derived from a biological effects database from Puget Sound, Washington (for estuarine, not freshwater sediments).	Ecology (1995)
LAET	Low	AET is the sediment contaminant concentration above which statistically significant ($p \leq 0.05$) adverse biological effects (relative to appropriate reference conditions) would always be expected. LAET is the lowest of four AETs derived from four marine toxicity tests (amphipod mortality, echinoderm and oyster abnormality, and bacterial luminescence) and benthic community data. The biological effects database is from Puget Sound, Washington.	Gries and Waldow (1996)
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	MacDonald et al. (2000)

Table 6-12. Generic SQGs, Their Derivation, and Narrative Intent

SQG	Toxicity Threshold	Narrative Intent	Source
		with similar narrative intent. The SQGs are derived from a combination of freshwater and marine toxicity tests.	
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)
SL2	High	The SL2 corresponds to a concentration at which minor adverse effects may be observed in the more sensitive groups of benthic organisms. SL2s were derived using the FPM and freshwater toxicity tests with both mortality and growth endpoints. These SQGs are derived from a regional (Western Washington and Oregon) biological effects database.	RSET (2006)
ERM	High	The concentrations equivalent to or above the ERMs represent thresholds above which effects would frequently occur. The ERMs were derived as the 50th percentile of a database composed of multiple studies, species, and effects endpoints (predominantly mortality). These SQGs are derived from a national biological effects database that includes other sediment SQGs, marine toxicity tests, and benthic field studies.	Long et al. (1995)
CSL	High	The CSLs establish a minor adverse effects threshold above which adverse effects are expected to occur. CSLs are generally based on the second lowest AETs (see LAET). CSLs are used to define potential cleanup areas to be remediated under SMS. These SQGs are derived from a biological effects database from Puget Sound, Washington (for estuarine, not freshwater sediments).	Ecology (1995)

AET – apparent effects threshold
CSL – cleanup screening level
ERL – effects range – low
ERM – effects range – median
FPM – floating percentile model
LAET – lowest apparent effects threshold
PEC – probable effects concentration
PEL – probable effects level
RSET – Regional Sediment Evaluation Team
SL1 – screening level 1
SL2 – screening level 2
SMS – Washington State Sediment Management Standards
SQS – sediment quality standards
TEC – threshold effects concentration
TEL – threshold effects level

Compendia of 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/SQG) for each chemical in the compendium (for a particular sediment sample) and then averaging to get the mean SQG quotient. Mean SQG quotients are sometimes considered to be a better measure of sediment quality than are individual chemical SQGs because they integrate multiple chemicals (Long et al. 2006).

The term "narrative intent" refers to the specific predictions associated with exposure to sediment chemical concentrations below, between, or above SQGs. All SQGs (e.g., individual chemical SQGs and mean SQG quotients) must have a narrative intent in order to be used appropriately. The concept of narrative intent is essential for understanding what the comparison of sediment chemistry data with any particular measurement endpoint means, in terms of the potential risk posed by that sediment to the benthic community. Understanding narrative intent also is essential for comparing predictions across LOEs (generic and site-specific) to arrive at overall conclusions about benthic community risk from exposure to chemicals in sediments.

In the SLERA and the refined screening process, the maximum surface sediment concentrations were compared to the lowest of any of the SQGs provided by EPA. The screening process identified 70 COPCs for benthic invertebrates. The generic SQGs, mean quotients, and ESBs for PAH mixtures, neutral organic compounds, and pesticides for the 70 COPCs were evaluated to identify chemicals or locations in the Study Area that may cause adverse effects on the benthic community. As with the site-specific SQGs, the reliability of each set of generic SQGs, mean quotients, and ESBs was assessed using the same reliability criteria of ≤ 20% false negative and false positive rates and an overall reliability of ≥ 80%. Attachment 7 presents the reliability assessment. None of the five sets of high SQGs met the acceptability criteria, and they were therefore not used to assess the surface sediment chemistry data from the Study Area. Similarly, the high SQG mean quotients and the ESBs for PAH mixtures, the sum of PAHs and narcotic non-ionic organics, and pesticides could not meet the acceptability criteria. Three sets of SQGs (TECs, TELs, and ERLs) met the acceptability criterion for false negative rates but not false positive rates. Given that the narrative intent of the low generic SQGs is to define areas of no or low toxicity, the false positive error rate failure does not greatly affect their use for the purpose of identifying unimpacted areas (i.e., the purpose for which they are intended).

6.3.2 Risk Characterization Using Low Generic SQGs

Map 6-6 presents the sediment chemistry stations in the Study Area identified as having no adverse effects on the benthic community, based on the non-exceedance of one or more sets of the low SQG (TECs, TELs, and ERLs) that met the acceptability criterion. The results at these stations provide a conservative estimate of non-toxic sediments from a benthic toxicity perspective.

6.4 TISSUE-RESIDUE ASSESSMENT

One LOE for evaluating risks to the benthic invertebrate community and bivalve and crayfish populations was 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; this section compares the COPC concentrations in benthic invertebrate tissue to BERA tissue TRVs to identify COCs.

Three types of benthic invertebrate tissue information (field-collected data, laboratory-exposed organism data, and predicted values) were included in the assessment. 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*) was performed to estimate tissue concentrations for other common sediment-exposed benthic invertebrates. Laboratory exposure of the Asiatic clam (*Corbicula fluminea*) was performed (Windward and Integral 2005b). In areas without co-located tissue-residue data, tissue concentrations were predicted for field-collected clams and crayfish and for laboratory-exposed clams and worms; the concentrations were predicted from individual sediment samples for those chemicals 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).

Assessment Based on Individual Samples

In the tissue-residue assessment, risks to the benthic invertebrate community and 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 > 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 although ODEQ calls this a population risk threshold, exceedance of such a threshold may or may not result in a change in a true population-level endpoint (e.g., density, growth rate, or age structure of the population).

The comparison of tissue concentrations to TRVs was conducted on an individual sample basis per EPA's Problem Formulation (Attachment 2). The point-by-point comparisons to TRVs make it possible to look at the spatial distribution of TRV exceedances. However, EPA's Problem Formulation (Attachment 2) also directed the LWG to identify COPCs as COCs if tissue-residue concentrations exceeded the TRVs in any one benthic tissue sample. This conservative definition incorrectly implies that TRV exceedances in a single or isolated tissue sample pose an unacceptable risk to the benthic community, bivalve population, or crayfish population. This issue is addressed in the risk conclusions (Section 6.7), which also evaluates the importance of benthic invertebrate COCs across multiple LOEs.

The details of the tissue-residue assessment are presented as follows:

- Section 6.4.1 presents 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 Sections 6.4.2. Tissue-residue concentrations, when detected or predicted, for each COPC are used as evidence of exposure. All tissue-residue data are presented in Attachment 4.
- Baseline tissue TRVs (Section 6.4.3), 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 6.4.4 presents the risk characterization results, uncertainties, and COCs identified for the tissue-residue LOE. In Section 6.7, these COCs are compared with the risk characterization for the other LOEs (i.e., sediment toxicity [measured or predicted by SQGs], and surface water) to identify COCs for which the preponderance of evidence indicates unacceptable risk to benthic invertebrates in the Study Area.

Figure 6-5 presents a flowchart of the benthic tissue-residue assessment section organization.

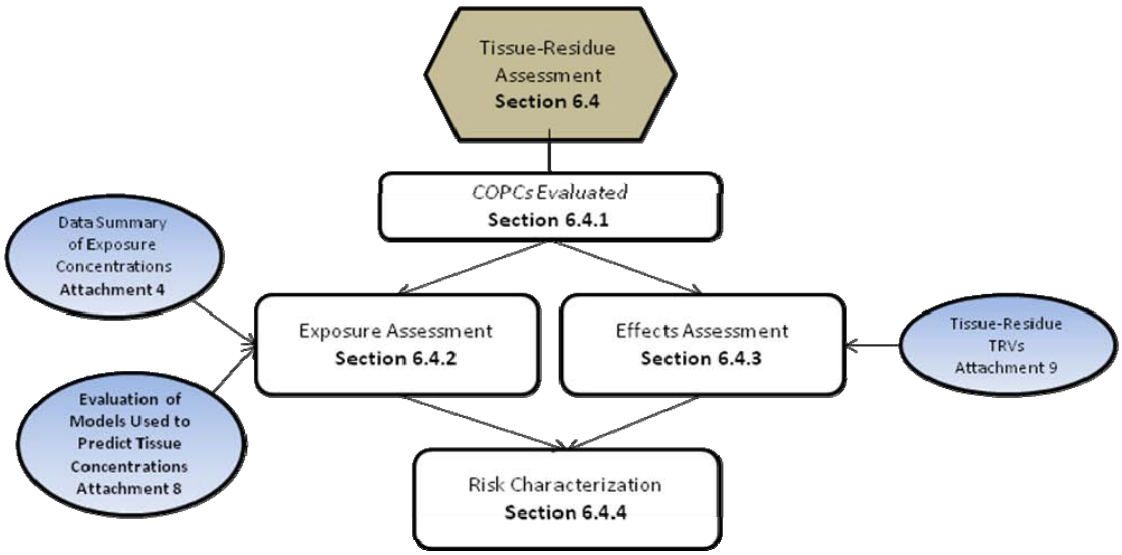


Figure 6-5. Overview of Benthic Tissue-Residue Assessment Section Organization

6.4.1 COPCs Evaluated

Eleven COPCs for benthic invertebrate tissue residues were identified in the SLERA and refined screen (Table 6-13). 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 RI), and thus it was not

considered to be a site-related chemical. Six COIs were not evaluated in the SLERA and refined screen (Table 6-14) because no toxicological data were available.

Table 6-13. 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
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 – chemical of potential concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
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 6-14. Benthic Invertebrate Tissue COIs with No TRVs

COI	
Metals	
Manganese	
SVOCs	
Benzoic acid	Bis(2-chloroethoxy) methane
Benzyl alcohol	Nitrobenzene
Phenols	
4-Nitrophenol	

COI – chemical of interest
SVOC – semivolatile organic compound
TRV – toxicity reference value

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.

6.4.2 Exposure Assessment

This section presents the exposure assessment using tissue-residue data collected in the field as well as tissue residues derived in laboratory bioaccumulation testing or predicted using BSA) presented in Attachment 8.

6.4.2.1 Empirical Tissue EPCs

Empirical tissue exposure point concentrations were derived from both field-collected and laboratory-exposed benthic invertebrates. COPC concentrations in composite tissue samples collected from the Study Area represent EPCs for clams, mussels, crayfish, and epibenthic invertebrates. The field sampling locations are shown on Maps 4-3 through 4-5. 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 account for 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.

6.4.2.2 Predicted Tissue EPCs

Field data or steady state-adjusted laboratory data on tissue residue concentrations were used to represent EPCs. Tissue data were available from field-collected and laboratory-exposed clams, field-collected crayfish, and laboratory-exposed worms.[35] EPCs also were predicted, using either a mechanistic bioaccumulation model or site-specific BSARs. 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 preliminary remediation goals (PRGs) for the FS. The models are presented in the draft bioaccumulation modeling report for the Portland Harbor RI/FS Windward (2009).

The mechanistic model was available for predicting total PCB, pesticide, and dioxin and furan concentrations.[36] 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 for benthic invertebrate tissue COPCs that met appropriate regression analysis assumptions, had a statistically significant positive slope ($p < 0.05$), had an $r2 > 0.30$ and were not modeled mechanistically. Windward (2009) 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 tributyltin (TBT) in laboratory-exposed clams and laboratory-exposed worms met the criteria noted above (Table 6-15). BSARs were used to predict TBT concentrations in laboratory-exposed worms; however, the laboratory-exposed clam TBT BSAR was rejected because it failed to predict correctly the empirical field clam tissue TBT data (see Attachment 8). A BSAR that is unable to predict empirical, field-collected data was judged to be an inappropriate tool for predicting tissue concentrations elsewhere. Therefore, only the laboratory-exposed worm TBT BSAR was used to predict tissue concentrations; it is the only information available for estimating TBT bioaccumulation in benthic infaunal consumers.

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[35] Tissue concentrations in mussels and epibenthic organisms were not predicted using BSARs or the mechanistic model, as these receptors are not directly associated with sediment.

[36] Because dioxins and furans were not classified as tissue COPCs for any benthic invertebrate receptors, the mechanistic model was not used to predict dioxin and furan benthic tissue concentrations.

Table 6-15. Summary of Benthic Invertebrate Tissue COPCs and Selected Models Used to Predict Tissue Concentrations

	Field-Collected				Laboratory-Exposed			
	Clam		Crayfish		Clam		Worm	
COPC	Predicted Tissue?	Selected Model	Predicted Tissue?	Selected Model	Predicted Tissue?	Selected Model	Predicted Tissue?	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[c]	NA	Yes	BSAR
Phthalates								
BEHP	No[b]	NA	No[b]	NA	No[a]	NA	No[a]	NA
Dibutyl phthalate	No[b]	NA	No[b]	NA	No[b]	NA	No[b]	NA
PCBs								
Total PCBs	Yes	Mechanistic model	Yes	Mechanistic model	Yes	Mechanistic model	Yes	Mechanistic model
Pesticides								
Total DDx	Yes	Mechanistic model	Yes	mechanistic model	Yes	Mechanistic model	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 2009), 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 sediment and tissue detected concentration data pairs were available (n=5).

[c] The laboratory-exposed clam TBT BSAR was rejected because it fails at predicting the empirical field clam tissue TBT data. A BSAR that was unable to predict empirical, field collected data was judged to be an inappropriate model for predicting tissue concentrations elsewhere.

BEHP – bis(2-ethylhexyl) phthalate
BSAR – biota-sediment accumulation regression
COPC – chemical 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

Interestingly, field-exposed clams did not show a significant relationship between sediment and tissue TBT concentrations, but laboratory exposed clams did. 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 6-20). 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.[37] 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 ≤ 21 days. The chemical-specific ACRs are presented in Table 6-21. 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), which 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, was calculated, and the 10th percentile of the SSD was selected as the LOAEL. Table 6-16 presents the benthic tissue-residue LOAELs for the 10 COPCs.

[37] The decisions and compromises made during the negotiation process are captured in the e-mails presented in Attachment 1.

Table 6-16. 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) is 20 times lower than the sublethal effect threshold (3 mg/kg) proposed for protection of juvenile salmonid prey ((Meador 2000a), which was based on reduced growth in multiple species (Meador 2000b). Use of this TRV likely overpredicts toxicity.
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: fifteen acceptable toxicity studies were identified. The test organisms included annelids, amphipods, daphnids, decapods, and insects. The PCBs included Aroclors 1016, 1242, 1254, 1268, 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 (eight) of toxicological studies based on crustaceans and annelids was available.

BEHP – bis(2-ethylhexyl) phthalate
COPC – chemical of potential concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene

DDT – dichlorodiphenyltrichloroethane
LOAEL – lowest-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

Tissue-Residue TRV Uncertainties

Species sensitivity distributions could be developed for 7 of the 10 COPCs based on 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 for these chemicals.

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) 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.

6.4.4 Risk Characterization

This section presents the risk estimates for benthic invertebrates based on the tissue-residue LOE. Hazard quotients (HQs) were derived for COPC-receptor pairs using the following equation:

$$HQ = \frac{EPC}{TRV} \qquad \textbf{Equation 6-1}$$

Where:

HQ = hazard quotient
EPC = exposure point concentration
TRV = toxicity reference value

For the tissue-residue LOE, the EPC and TRV are represented by tissue-residue concentrations expressed as mg/kg ww.

6.4.4.1 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).

COPCs with HQs > 1.0 for any individual benthic invertebrate tissue sample were identified as COCs. For all COCs, 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 COCs and associated uncertainties were further examined to arrive at risk conclusions for benthic invertebrates (Section 6.7).

6.4.4.2 Risk Characterization Results and Uncertainty Evaluation

Table 6-17 presents the frequency of COPC HQs > 1.0 for all benthic invertebrate tissue samples that were field-collected or laboratory-exposed.[38] The COPCs were identified for each individual receptor (e.g., field-collected clams, mussels, crayfish, and epibenthic invertebrates, and laboratory-exposed clams and worms) in the SLERA (Attachment 5). Eight of the ten COPCs had at least one HQ > 1.0.

Table 6-17. Number of Individual Benthic Invertebrate Empirical Samples with LOAEL HQs > 1.0

	Number of Samples with HQ > 1.0 (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	2/35 (1.5)
Cadmium	0/38	0/7	NA	NA	NA	0/35
Copper	32/38 (1.8)	NA	32/32 (2.6)	0/2	0/35	1/35 (2.6)
Zinc	34/38 (2.2)	5/7 (1.7)	NA	NA	NA	27/35 (1.3)
Butyltins						
Tributyltin	1/34 (3.5)	NA	NA	NA	1/35 (4.5)	1/35 (11)
Phthalates						
BEHP	NA	NA	NA	NA	1/35 (2.8) [a]	NA
Dibutyl phthalate	0/38	NA	NA	NA	NA	0/35[a]
PCBs						
Total PCBs	1/41 (2.0)	NA	0/32	NA	NA	8/35 (7.5)[a]
Pesticides						
4,4′-DDD	0/41	NA	NA	NA	0/35 [a]	1/35 (1.2) [a]
Total DDx	0/41	NA	NA	NA	1/35 (2.2) [a]	2/35 (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.

[38] Tissue residues for neutral organic compounds in laboratory-exposed organisms were based on steady-state adjusted concentrations.

BEHP – bis(2-ethylhexyl) phthalate
COPC – chemical 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)
TRV – toxicity reference value

Table 6-18 presents the frequency of HQs > 1.0 for all benthic invertebrate tissue-residue concentrations that were predicted for the four COPCs with a significant sediment-tissue relationship.

Table 6-18. Number of Individual Sediment Samples Predicted to Have LOAEL HQs > 1.0

	Number of Samples Predicted to Have HQs > 1.0 (Maximum HQ)		
COPC	**Clam[a]**	**Crayfish**	**Worm**
Butyltins			
Tributyltin	NA	NA	27/272 (149)
PCBs			
Total PCBs	5/1,100 (12)	20/1,100 (20)	15/1,100 (19)
Pesticides			
Total DDx	12[b]/1,128 (6.7)	13[c]/1,128 (9.1)	15[d]/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] Six of the twelve predicted sediment samples with HQs > 1.0 were based on N-qualified data.
[c] Six of the thirteen predicted sediment samples with HQs > 1.0 were based on N-qualified data.
[d] Six of the fifteen predicted sediment samples with HQs > 1.0 were based on N-qualified data.

COPC – chemical of potential concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect 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)

Bold identifies HQs > 1.0.

From the empirical and predicted tissue-residue data, eight chemicals whose HQs > 1.0 were identified as tissue-residue COCs for the benthic invertebrate community: arsenic, copper, zinc, TBT, BEHP, total PCBs, 4,4′-DDD, and total DDx. For bivalves, six chemicals were identified as tissue-residue COCs: copper, zinc, TBT, BEHP, total PCBs, and total DDx. For crayfish, copper, total PCBs, and total DDx were identified as tissue-residue COCs. Maps 6-25 through 6-29 show the HQs by location for both empirical data and predicted tissue-residue data for all COCs except arsenic and BEHP. The arsenic HQs > 1.0 were located at RM 3.7 (east bank) and RM 7.4 (east bank), and the BEHP HQ > 1.0 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 > 1.0.

Four metals (arsenic, cadmium, copper, and zinc) were identified as benthic community COCs using the tissue-residue LOE. Two key principles (EPA 2007) should be considered, before concluding that metals pose an unacceptable risk to the benthic community based on the tissue-residue LOE:

- 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 was discussed in Section 6.4.2.2, 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. Consequently, the empirical tissue residue and sediment chemistry data indicate, at the most, possible localized metals risks to subpopulations of benthic organisms in a few discrete locations. They do not indicate unacceptable risk to the benthic community in Portland Harbor.

Risks to benthic invertebrates from arsenic in the Study Area are likely low. Arsenic had only two HQs > 1.0 (east bank RM 3.7 and east bank RM 7.4), based on laboratory-exposed worm tissues. Copper and zinc had HQs > 1.0 throughout the Study Area based on field-collected and laboratory-exposed tissues. However, both copper and zinc are essential nutrients for invertebrates and are also metabolically regulated metals. Table 6-19 presents the tissue-residue concentration for copper and zinc in the benthic invertebrates. The maximum tissue-residue concentrations for both metals are within a factor of 4 of the nutritional thresholds,[39] except for copper in laboratory-exposed worms (9.2). These copper and zinc concentrations are likely within the range that organisms are able to actively regulate. Because of these factors, the risk estimates are highly uncertain.

[39] Nutritional thresholds were provided by EPA (2008d) and are discussed in Attachment 9.

Table 6-19. Copper and Zinc Concentrations in Tissue Compared to Nutritional Values

Tissue Type by Chemical	Mean (mg/kg ww)	Range (mg/kg ww)	Nutritional Value (mg/kg ww)	Mean Ratio of Observed to Nutritional Value	Maximum Ratio of Observed to Nutritional Value
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
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 COC at the mouth of and in Swan Island Lagoon, and five other locations (along the east bank at RM 3.7, along the east and west bank near RM 5.7, along the west bank at approximately RM 6.2, and on the west bank at RM 7.4) based on primarily on predicted tissue-residue data. At a number of these locations, however, field-collected tissue data indicated no unacceptable risk to benthic invertebrates. Thus, the predicted tissue-residue concentrations that were based on laboratory studies (relationship shown in Table 6-20) are likely to overestimate actual exposure levels.

The risk to the benthic invertebrate community from BEHP in tissue is also low. BEHP had only one HQ > 1.0 (RM 8.8, west) in the empirical tissue dataset, and the HQ was 2.8. Only two toxicological studies were available from which to derive the TRV, making the TRV uncertain.

Total PCBs is identified as a COC with HQs ranging from 1.0 to20 throughout the Study Area, based on predicted tissue-residue concentrations. Actual concentrations in nine tissue samples had HQs ranging from 1.1 to 7.5. Taken together, the empirical and predicted tissue concentrations suggest that total PCBs might pose localized unacceptable risks to benthic invertebrates at locations including RM 2.3 (east), the International Slip, Willamette Cove, RM 6.9 (west), RM 8.1 (west), RM 8.8 (west) Swan Island Lagoon, RM 9.6 (west), and RM 11.4 (east). The question of whether this rises to the level of unacceptable risks to benthic communities is a matter of judgment not necessarily implied by the data. However,

it may not be necessary to resolve this question because the benthic community risk from exposure to PCBs is almost certainly subsumed by PCB risks to receptors at higher trophic levels.

The risk to the benthic invertebrate community from 4,4′-DDD in tissue is low. 4,4′-DDD had only one HQ > 1.0 (RM 6.9, west), which was for laboratory-exposed worms, and the TRV exceedance (HQ) was 1.2. Only two toxicological studies were available from which to derive the TRV, making the TRV uncertain. Total DDx was identified as a COC primarily along the west bank approximately between RM 7.2 and RM 7.6, based on predicted tissue-residue concentrations. The tissue residues in the two bioaccumulation tests identified total DDx as a COC at two locations (RM 6.9 west, and RM 7.3 west) with HQs of 2.1, 2.2, and 3.2. Total DDx might pose localized unacceptable risks to benthic invertebrates along the west bank between RM 6.9 and RM 7.3. The tissue-residue LOE indicates that there is low to negligible risk to the benthic community from exposure to total DDx, at least beyond that limited area.

6.4.4.3 Evaluation of Non-Study Area Concentrations

Per EPA (2008b), 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[40] were available from the downstream reach (RM 0 to RM 1.9) and two clam and two crayfish composite tissue samples[41] 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; and crayfish and copper; Figures 6-6 through 6-10). One clam sample collected from the downstream reach had a copper concentration slightly greater than the tissue LOAEL; whereas the two samples collected in the downtown reach did not exceed the copper LOAEL. Copper concentrations in crayfish and zinc concentrations in clams collected from the downstream Reach and the downtown reach were greater than the LOAEL. TBT and total PCB concentrations in clams collected from the downstream reach and the downtown reach were less than the LOAELs.

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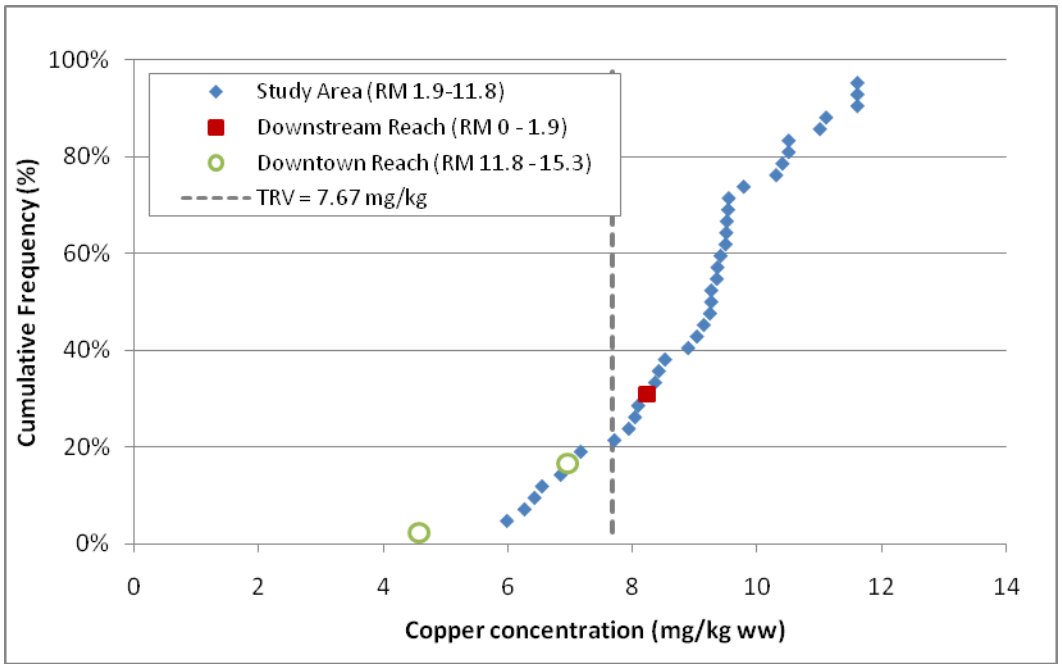
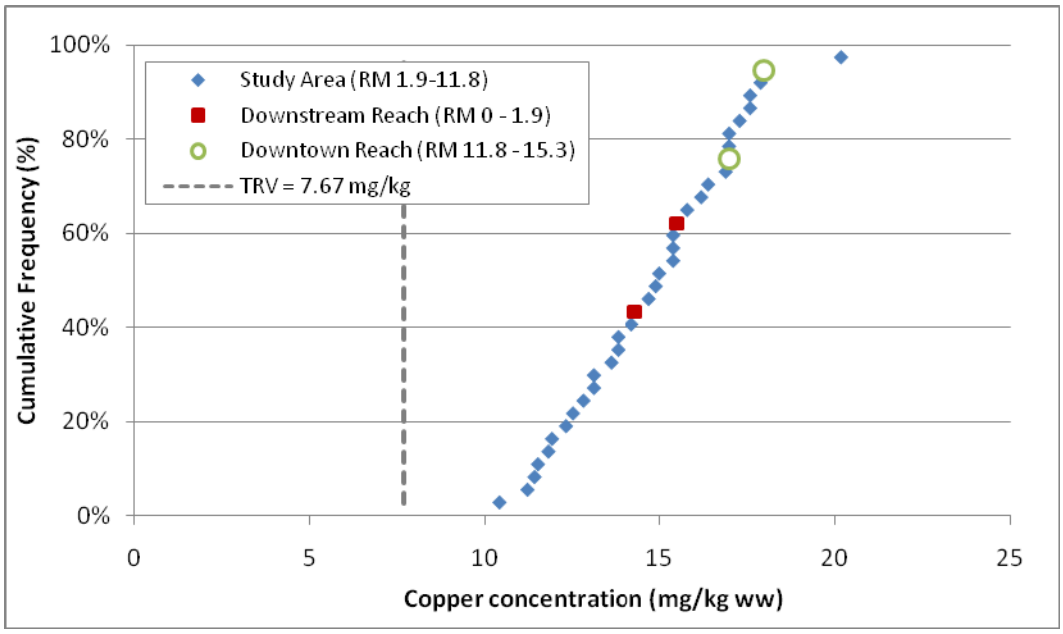
[40] The clam sample was collected from the east bank at RM 1.6. The crayfish samples were collected from the west and east banks at approximately RM 1.0 and RM 1.5. These samples provide limited spatial coverage of the downstream reach (spanning 0.6 mile).

[41] 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).



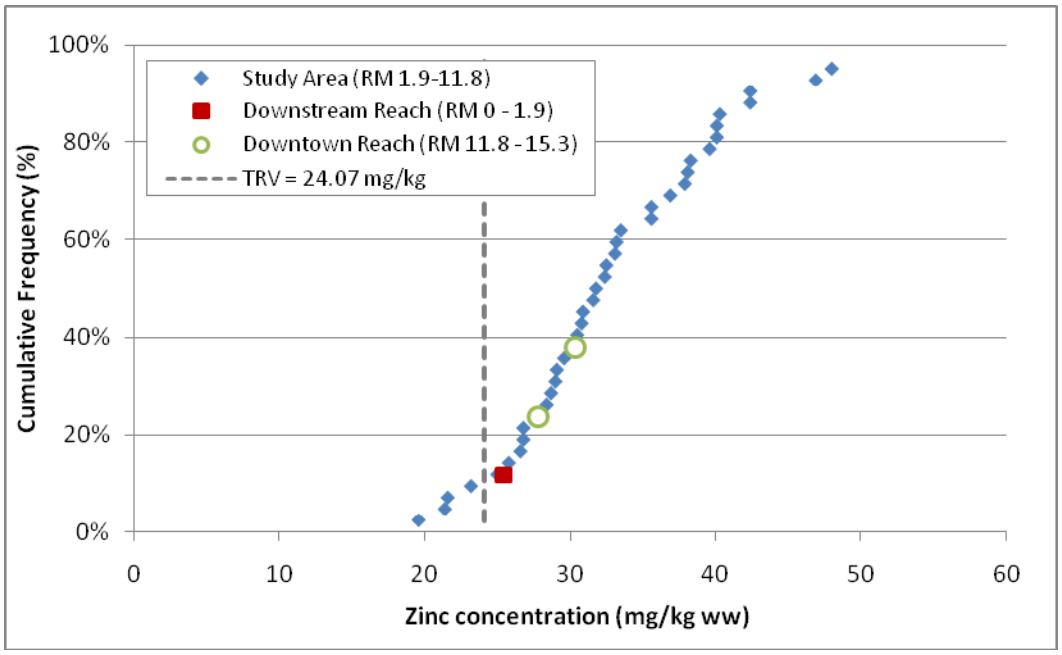
RM – river mile TRV – toxicity reference value

Figure 6-6. CFD of Copper Concentrations in Field-Collected Clam Tissues



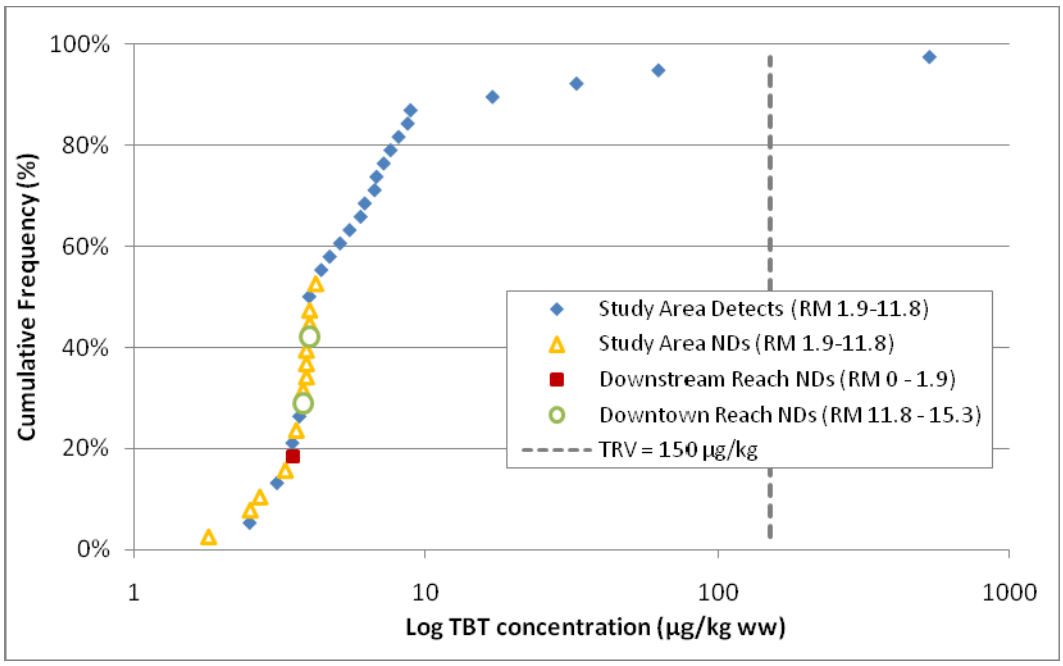
RM – river mile TRV – toxicity reference value

Figure 6-7. CFD of Copper Concentrations in Field-Collected Crayfish Tissues



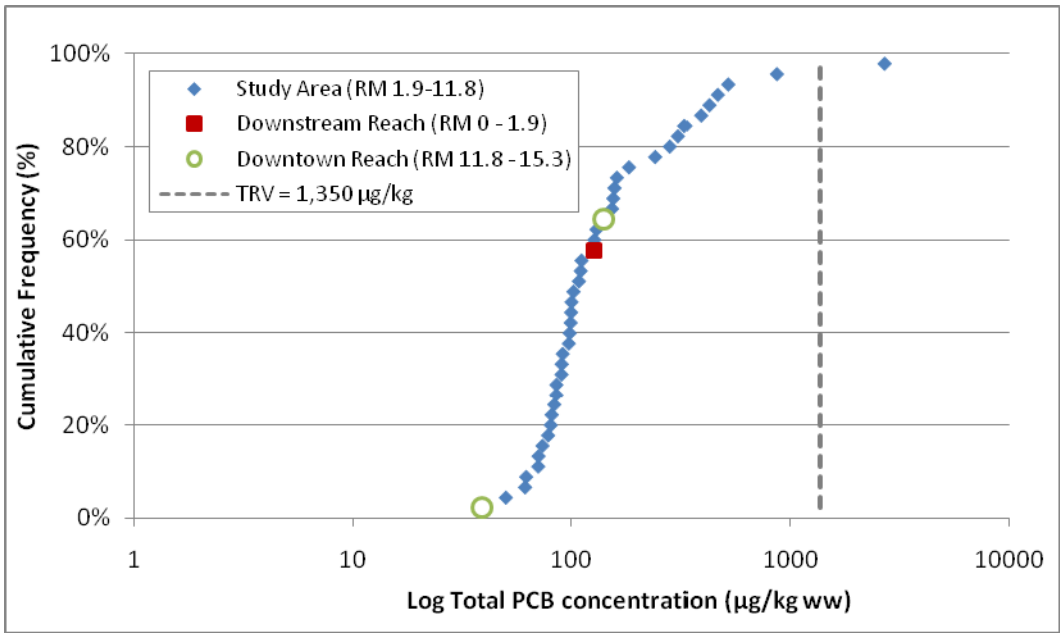
RM – river mile TRV – toxicity reference value

Figure 6-8. CFD of Zinc Concentrations in Field-Collected Clam Tissues



ND – non-detects RM – river mile TRV – toxicity reference value

Figure 6-9. CFD of TBT Concentrations in Field-Collected Clam Tissues



ND – non-detects RM – river mile TRV – toxicity reference value

Figure 6-10. CFD of Total PCBs Concentrations in Field-Collected Clam Tissues

6.4.4.4 Summary of Benthic Invertebrate Tissue COCs

Eight chemicals (arsenic, copper, zinc, TBT, BEHP, total PCBs, 4,4′-DDD, and total DDx) were identified as tissue-residue COCs for the benthic invertebrate community, six chemicals (copper, zinc, TBT, BEHP, total PCBs, and total DDx) were identified as tissue-residue COCs for bivalves, and three chemicals (copper, total PCBs, and total DDx) were identified as tissue-residue COCs for crayfish. Table 6-20 presents the HQs, uncertainties, and risk conclusions for each of the COC-receptor pairs.

In Section 6.7, the results of the benthic invertebrate tissue LOE were compared with the risk characterization for the other LOEs (i.e., sediment toxicity testing, SQGs, and surface water) to identify COCs for which the preponderance of evidence indicates unacceptable risk to benthic invertebrates in the Study Area.

Table 6-20. Summary of Benthic Invertebrate Tissue-Residue COCs

Receptor by COC	Risk Exposure Area	HQ[a]	Frequency of Samples with HQs > 1.0 (%)	Exposure Uncertainty	Effects Uncertainty
Arsenic					
Laboratory-exposed worms	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.	Tissue-residue LOEs for metals are very uncertain. Limited number of toxicological studies (five) was available for derivation of a TRV. Limited spatial extent of TRV exceedances does not represent population-level effect.
Copper					
Field-collected clams	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.	Tissue-residue LOEs for metals are very uncertain, particularly for essential metals. The LOAEL is 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 eight available bivalve studies.
Field-collected crayfish	Site-wide	1.4 – 2.6	100	Crayfish tissue data were collected in the field. No relationship was found between tissue and sediment.	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.

Table 6-20. Summary of Benthic Invertebrate Tissue-Residue COCs

Receptor by COC	Risk Exposure Area	HQ[a]	Frequency of Samples with HQs > 1.0 (%)	Exposure Uncertainty	Effects Uncertainty
Laboratory-exposed worms	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.	Tissue-residue LOEs for metals are very uncertain, particularly for essential metals. Limited spatial extent of TRV exceedances (1 sample) does not represent population-level effect.
Zinc					
Field-collected clams	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.	Tissue-residue LOEs for metals are very uncertain, particularly for essential metals. The LOAEL is 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 eight available bivalve studies.
Field-collected mussels	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.	Tissue-residue LOEs for metals are very uncertain, particularly for essential metals. All tissue residues were below the nutritional requirement for mollusks. The LOAEL is 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 < 1.0 if the predicted tissue-residue concentrations were compared to a LOAEL of 42.89 mg/kg ww derived from eight available bivalve studies.

Table 6-20. Summary of Benthic Invertebrate Tissue-Residue COCs

Receptor by COC	Risk Exposure Area	HQ[a]	Frequency of Samples with HQs > 1.0 (%)	Exposure Uncertainty	Effects Uncertainty
Laboratory-exposed worms	Site-wide	1.0 – 1.3	77	Worm tissue data are from the laboratory bioaccumulation test. No relationship was found between tissue and sediment.	Tissue-residue LOEs for metals are very uncertain, particularly for essential metals.
TBT					
Field-collected clams	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 sample) does not represent population-level effect.
Laboratory-exposed clams	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) does not represent population-level effect.

Table 6-20. Summary of Benthic Invertebrate Tissue-Residue COCs

Receptor by COC	Risk Exposure Area	HQ[a]	Frequency of Samples with HQs > 1.0 (%)	Exposure Uncertainty	Effects Uncertainty
Laboratory-exposed worms	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 6-27)	Limited spatial extent of TRV exceedances (1 sample) does not represent population-level effect.
Predicted worms	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-residue 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) is 20 times lower than the sublethal effect threshold (3 mg/kg) proposed for protection of juvenile salmonid prey ((Meador 2000a), which was based on reduced growth in multiple species (Meador 2000b). Use of this TRV likely overpredicts toxicity.
BEHP					
Laboratory-exposed clams	RM 8.8, east	2.8	2.9	Clam tissue data are from the laboratory bioaccumulation test.	The LOAEL is based on two toxicological studies.

Table 6-20. Summary of Benthic Invertebrate Tissue-Residue COCs

Receptor by COC	Risk Exposure Area	HQ[a]	Frequency of Samples with HQs > 1.0 (%)	Exposure Uncertainty	Effects Uncertainty
Total PCBs					
Field-collected clams	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 sample) does not represent population-level effect.
Laboratory-exposed worms	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: fifteen acceptable toxicity studies were identified. The test organisms included annelids, amphipods, daphnids, decapods, and insects. The PCBs included Aroclors 1016, 1242, 1254, 1268, PCB congeners 153 and 101.
Predicted tissue	RM 2.2, east; RM 3.7, east; Swan Island Lagoon (RM 8.4 to 8.5); RM 11.3, east	1.2-12 (clam) 1.1-20 (crayfish) 1.1-19 (worm)	0.5 (clam) 1.8 (crayfish) 1.4 (worm)	Predicted tissue-residue concentrations were derived using the mechanistic model	Low uncertainty: fifteen acceptable toxicity studies were identified. The test organisms included annelids, amphipods, daphnids, decapods, and insects. The PCBs included Aroclors 1016, 1242, 1254, 1268, PCB congeners 153 and 101.
4,4′-DDD					
Laboratory-exposed worms	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) does not represent population-level effect. The LOAEL is based on two toxicological studies.

Table 6-20. Summary of Benthic Invertebrate Tissue-Residue COCs

Receptor by COC	Risk Exposure Area	HQ[a]	Frequency of Samples with HQs > 1.0 (%)	Exposure Uncertainty	Effects Uncertainty
Total DDx					
Laboratory-exposed clams	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) does not represent population-level effect. Limited number (eight) of toxicological studies was available, based on crustaceans and annelids.
Laboratory-exposed worms	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) does not represent population-level effect. Limited number (eight) of toxicological studies was available, based on crustaceans and annelids.
Predicted tissue	RM 7.2 to RM 7.6, west	1.1 – 6.7 (clam) 1.2-9.1 (crayfish) 1.1-10 (worm)	1.1 (clam) 1.2 (crayfish) 1.3 (worm)	Predicted tissue concentrations were derived using the mechanistic model.	Limited number (eight) of toxicological studies was available, based on crustaceans and annelids.

[a] Only HQs > 1.0 are presented. HQs in all other tissue samples were < 1.0.

BEHP – bis(2-ethylhexyl) phthalate
COC – chemical of concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
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. However, the direct applicability of much of the surface water data to predicting unacceptable risks to benthic invertebrates has considerably more uncertainty associated than do the other LOEs, inasmuch as benthic invertebrates are exposed to surface water only at the sediment-water interface, rather than throughout the water column. 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). In this assessment, the same water TRVs were used to evaluate baseline risks to benthic invertebrates.

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 more appropriate for surface water than for sediment because individual samples of (flowing) surface water, capturing conditions integrated over a wide area, better represent the appropriate spatial scale for a population and community-level analysis than do individual sediment samples.

COPCs with HQs > 1.0 for any individual surface water sample were identified as COCs. For all COCs, the spatial distribution and magnitude of HQs, the seasonal and sampling method 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 COCs and associated uncertainties were further examined to arrive at risk conclusions for the benthic invertebrate assessment endpoints (Section 6.7).

The details of this assessment are presented as follows:

Section 6.5.1 presents a summary of the COPCs evaluated in the surface water LOE.

Section 6.5.2 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 6.5.3 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 6.5.4 presents the risk characterization results, COC-receptor pairs, and associated uncertainties. These COCs are further assessed in the benthic invertebrate risk conclusions (Section 6.7).

Figure 6-11 presents a flowchart showing organization of the benthic invertebrate surface water section.

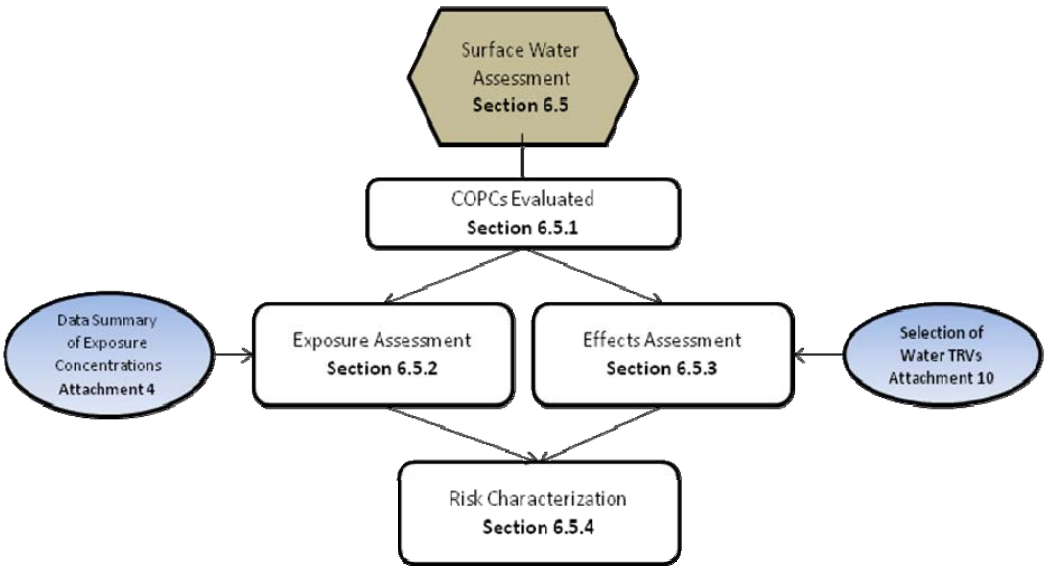


Figure 6-11. Overview of Benthic Invertebrate Surface Water Section Organization

6.5.1 COPCs Evaluated

Twelve[42] surface water COPCs were identified in the SLERA and refined screen (Attachment 5) (Table 6-21). All COPCs were evaluated in this assessment.

Table 6-21. Surface Water COPCs

COPCs	
Metals	
Aluminum[a]	Zinc[b]
Butyltins	
Butyltin	
PAHs	
Benzo(a)anthracene	Naphthalene
Benzo(a)pyrene	
Phthalates	
BEHP	

[42] Three individual DDT metabolites identified in the SLERA (2,4′-DDD, 2,4′-DDT, and 4,4′-DDD) were evaluated as total DDTs 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 6-21. Surface Water COPCs

COPCs	
PCBs	
Total PCBs	
Pesticides	
4,4′-DDT	Total DDx
VOCs	
Ethylbenzene	Trichloroethene

[a] TRV based on total concentration.
[b] TRV based on dissolved concentration.
BEHP – bis(2-ethylhexyl) phthalate
COPC – chemical of potential concern
DDT – dichlorodiphenyltrichloroethane
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
VOC – volatile organic chemical

Aluminum was assessed even though it is a common crustal element. In addition to comparing concentrations of aluminum in surface water within the Study Area to a water TRV, aluminum concentrations in background surface water and sediment were also evaluated (Section 6.5.4). Like aluminum, zinc is naturally occurring in the environment, and background zinc concentrations were also evaluated.

Four surface water COIs could not be evaluated because no toxicological data were available. These COIs (4-chloroaniline, aniline, 2,4-DB, and methylchlorophenoxypropionic acid [MCPP]) were infrequently detected and were detected in isolated areas at different times.

6.5.2 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.

6.5.2.1 Overview of Surface Water Data Collected from 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. 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 RI). In addition, the surface water study was designed to characterize surface water during stormwater-influenced flow (i.e., during active runoff in the Study Area).

6.5.2.1.1 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[43] and vertically or horizontally integrated transect samples[44]) during seven surface water sampling events from November 2004 to January 2007 (Map 4-14). 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 (at 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 4.1.3.

6.5.2.1.2 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 Infiltrex™ 300 system with an XAD-2 resin column (XAD). This XAD method was used to collect samples for the low-level analysis of 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 detection limits 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 column to capture remaining concentrations in the ”dissolved” fraction. The analytical results from the filter and the column were combined to determine total analyte concentrations in the water column. Although PCB, PAH, and organochlorine pesticide analyses were conducted on samples collected using both the peristaltic pump and the XAD methods, different analytical methods were used, resulting in lower sensitivity and detection limits for samples collected using the XAD method. A comprehensive summary of the surface water data is presented in Attachment 4.

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[43] 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.

[44] 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 throughout that cross-section at one point in time.

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 the two metals COPCs (zinc and aluminum), data are available for both dissolved and total fractions. For aluminum, the total concentration was used in this assessment because its TRV is based on a total concentration; for zinc, the dissolved concentration was used to be consistent with its TRV.

Uncertainty Associated with Surface Water Sampling Methods

There are uncertainties associated with the use of multiple sampling types and methods (e.g., duration of sampling time for a single-point grab sample versus an integrated transect sample or volume of water sampled in an XAD versus peristaltic sample) in the evaluation of ecological exposure to surface water. Surface water samples were collected both as single-point samples and as transect (vertical and/or horizontal) 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 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 both transect, single-point, XAD, and peristaltic samples allows for the evaluation of surface water data over a larger spatial scale; however, the relevance of ecological exposure to surface water data collected from the various sampling types is highly uncertain.

6.5.2.1.3 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 RI.

Metals and Butyltins – Aluminum, zinc, and butyltin were analyzed only in peristaltic pump samples. Aluminum (total fraction) was detected in all samples at concentrations ranging from 41 to 1,860 µg/L, and zinc (dissolved) was detected in about half 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 samples and XAD samples, with lower detection limits in the XAD samples. PAHs were detected in less than 15% of the peristaltic samples, and their concentrations ranged from 2.4 to 60,500 ng/L. PAHs were detected more often in the XAD samples (detection frequencies from 22% to 90%) and concentrations ranged from 0.018 to 34.5 ng/L. BEHP detection frequencies in the XAD samples were four 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 samples and XAD samples, with lower detection limits 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 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 1999). The qualification indicates that the analyst believed the result was due to analytical interference from a chemical other than the target analyte.

6.5.2.2 Surface Water EPCs

Surface water EPCs in the assessment were represented by detected concentrations in all individual surface water samples collected using different sampling methods. 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 6.5.4.2), as they might be somewhat more representative of exposure concentrations for benthic invertebrates.

Invertebrate Exposure to Surface Water Uncertainty

Many benthic organisms live and/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.

6.5.3 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. Table 6-27 presents the water TRVs developed for all surface water COPCs, all of which are based on national AWQC, state water quality standards (WQS), or national Tier II criteria. 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 indicated in Table 6-22, often the chosen protective level is significantly lower than the lowest effects concentration for invertebrates.

Table 6-22. Water TRVs for Surface Water COPCs

COPC	Unit	Water TRV	Source	Basis for Water TRV
Metals				
Aluminum	µg/L	87[a]	AWQC	Derived from a number of acute and chronic studies with both invertebrates and fish; the final chronic freshwater value was derived from a study with brook trout which had a no-effect concentration of 87 µg/L; the lowest chronic value for invertebrates (*Daphnia*) was 1,020 µg/L. EPA has acknowledged the conservatism of this TRV by stating that many high-quality waters in the U.S. contain aluminum at concentrations greater than 87 µg/L (EPA 2006e, Footnote L).
Zinc	µg/L	36.5[b]	AWQC	Derived from a number of acute and chronic studies with both invertebrates and fish; the lowest chronic effect concentrations were 46.7 µg/L for invertebrates and 36.4 µg/L for fish.
Butyltins				
Butyltin	ng/L	72[c]	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 value for invertebrates was 0.14 µg/L and the lowest chronic value for fish was 0.26 µg/L.
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 was 619 µg/L (chronic) for fish and 2,194 µg/L (acute) for invertebrates (*Daphnia*). Value was derived by dividing the lowest genus mean acute value (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 were 133 µg/L for invertebrates (*Daphnia*) and 160 µg/L for fish.

Table 6-22. Water TRVs for Surface Water COPCs

COPC	Unit	Water TRV	Source	Basis for Water TRV
VOCs				
Ethylbenzene	µg/L	7.3	Tier II	Derived from two acute studies with fish (fathead minnow and guppy); the lowest effect concentration was 8,450 µg/L.
Trichloroethene	µg/L	47	Tier II	Derived from four acute and one chronic study with fish (fathead minnow and flagfish); the lowest effect concentration was 11,057 µg/L (chronic).
PCBs				
Total PCBs	ng/L	14[d]	AWQC	Protective of dietary exposure of mink.
Pesticides				
4,4′-DDT	ng/L	1[d]	AWQC	Protective of dietary exposure of birds. The lowest acute effect concentrations were 0.18 and 0.6 µg/L for invertebrates and fish, respectively. The lowest chronic effect concentration for fish was 0.74 µg/L (there were no chronic values for invertebrates). The final chronic AWQC of 0.001 µg/L was selected because it was associated with effects on the productivity of the brown pelican.
Total DDx	ng/L	1[d, e]	AWQC	

[a] Chronic TRV is based on total criteria; chronic TRV was compared to total concentration measured in Study Area.
[b] 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
[c] Chronic TRV is based on criteria for tributyltin.
[d] TRVs based only on aquatic organisms were derived for total PCBs and DDx (190 and 11 ng/L, respectively) because the TRVs that were selected per EPA are based on the protection of wildlife via ingestion of contaminated prey which is not appropriate for evaluating direct exposure of aquatic organisms to surface water contaminants. (See text below for details on derivation of alternative water TRVs).
[e] Chronic TRV is based on criteria for 4,4′-DDT.

AWQC – ambient water quality criteria
BEHP – bis(2-ethylhexyl) phthalate
COPC – chemical of potential concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
HCH – hexachlorocyclohexane
LC50 – concentration that is lethal to 50% of an exposed population
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)
TRV – toxicity reference value
VOC – volatile organic compound

Because the selected AWQC for total PCBs and 4,4′-DDT were 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.

Based on the AWQC document for PCBs (EPA 1980c), toxicity data were insufficient to allow derivation of a final acute value (FAV) or a final chronic value (FCV) directly. Therefore, a PCB FAV was calculated in this BERA using additional toxicity data for aquatic organisms reported in EPA's ECOTOX online database published since 1979 (ECOTOX 2009). When these additional data were added to those for aquatic organisms included in the AWQC document for PCBs (EPA 1980c), the data were sufficient to derive an FAV in accordance with the methods used to establish AWQC values. An FAV of 1.6 µg/L was derived and then 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, which is specific to aquatic organisms. This concentration (0.19 µg/L) was evaluated as an alternative water TRV for total PCBs; however, slightly lower chronic values are reported in the AWQC document for fish (lowest chronic value of 0.098 µg/L was reported for fathead minnow) and plants (lowest diatom value is 0.1 µg/L). 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. Although both the selected PCB water TRV (0.014µg/L) and the alternative water TRV (0.19 µg/L) were used to derive water HQs, only the alternative TRV was used to determine risk conclusions.

Based on the AWQC document for DDTs (EPA 1980a), sufficient toxicity data for aquatic organisms were available to derive an FAV directly for 4,4′-DDT FAV (1.1 µg/L), but not an FCV or an ACR. Only a single ACR (65) was identified in the AWQC document for DDT, and Suter and Tsao (1996) recommend using 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) from four studies of chemicals with a DDT-like mode of action. Dividing the FAV (1.1 µg/l) by the median reported ACR of 3.6 results in a chronic value of 0.31µg/L. In accordance with the methods used to establish AWQC s, it is appropriate to use a final tissue-residue value 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 10th percentile fish tissue residue LOAEL (1.6 mg/kg ww) by a bioaccumulation factor (BAF) of 142,960[45] (derived from the DDT AWQC document). This concentration of 0.011 µg/L is lower than the FAV divided by the ACR, and was evaluated as an alternative water TRV for DDx compounds in water. For evaluating direct exposure of aquatic organisms to water, this alternative TRV is considered more appropriate than the AWQC (0.0010 µg/L), which are based on the protection of brown pelican via ingestion of contaminated prey. Although both the selected DDT water TRV (0.001µg/L) and the alternative water TRV (0.011 µg/L) were used to derive water HQs, only the alternative TRV was used to determine risk conclusions.

[45] A BAF of 142,960 was based on the lipid-normalized BAF (17,870) and anchovy lipid percent (8%) as presented in the DDT AWQC document.

Surface Water TRV Uncertainties

TRVs are based on the most sensitive of any aquatic organism and may overestimate effects to benthic invertebrates in cases where benthic invertebrates are less sensitive than another receptor group (e.g., fish). The TRVs established by the AWQC and Tier II sources included toxicity data on a planktonic invertebrate species (*Daphnia sp.*) with multiple endpoints for assessing risks to benthic invertebrates (LC50, EC50 based on mortality, growth, and reproduction)for the following seven COPCs: aluminum, zinc, butyltin, 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,[46] the water TRV that is protective of birds may overestimate risks 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 therefore 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 (and determine COCs), only the alternative TRV was used to determine risk conclusions.

6.5.4 Risk Characterization

This section presents the risk estimates for invertebrates based on the surface water LOE. An HQ calculation was used to quantify surface water risk estimates. HQs were derived for all COPCs using Equation 6-1. The EPC and TRV are represented by surface water chemical concentrations. Section 6.5.4.1 presents the overall approach used to characterize risks via surface water to benthic invertebrate receptors. Section 6.5.4.2 presents the risk characterization results, uncertainty evaluation, and surface water COCs. Section 6.5.4.3 presents an evaluation of background concentrations. Section 6.5.4.4 presents a summary of surface water COCs.

6.5.4.1 Risk Characterization Process

The surface water exposure risk characterization for benthic assessment endpoints was conducted by evaluating individual surface water samples. HQs were determined on a sample-by-sample basis for all surface water samples within the Study Area in 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-

[46] 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).

mixed, flowing medium, whose samples represent exposure areas that are larger than the sampling location.

A COPC with HQ > 1.0 for any individual surface water sample was identified as COC. For all COCs, the spatial distribution and magnitude of HQs, the seasonal and sampling method patterns of HQs, and the associated exposure and effects assumptions were evaluated to provide a more detailed assessment of impacts on benthic invertebrates. COCs and associated uncertainties were further evaluated to arrive at risk conclusions for benthic invertebrates (Section 6.7).

6.5.4.2 Risk Characterization Results and Uncertainty Evaluation

Table 6-23 presents a summary of the individual HQs calculated across all surface water samples for all COPCs. By definition (because the BERA and SLERA TRVs and EPCs were unchanged), all of the COPCs had at least one sample with an HQ > 1.0. For six COPCs, HQs were > 1.0 in less than 2% of samples: zinc, butyltin, benzo(a)anthracene, benzo(a)pyrene, BEHP, and total PCBs. For ethylbenzene and trichloroethene, only one sample had HQs > 1.0 and frequency of exceedance was 3.7%. For aluminum, 92% of the samples had HQs > 1.0, but only 5% exceeded the invertebrate aluminum LOEC of 1,020 µg/L. Frequency of exceedance was higher for naphthalene (3.7%), 4,4′-DDT (11%), and total DDx (21%).

Table 6-23. Number of Surface Water Samples with HQs > 1.0

COPC	Number of Samples with HQs > 1.0 (Maximum HQ)	Percent of Samples with HQs > 1.0
Metals		
Aluminum (total)	132[a]/142 (21)	93%
Zinc (dissolved)	1/167 (1.1)	< 1%
Butyltins		
Butyltin	1/167 (1.2)	< 1%
PAHs		
Benzo(a)anthracene	2/245 (10)	< 1%
Benzo(a)pyrene	3/245 (14)	1.2%
Naphthalene	10/268 (50)	3.7%
Phthalates		
BEHP	2[b]/190 (2.3)	1.1%
PCBs		
Total PCBs	2/160 (1.2)[c]	1.3%
Pesticides		

Table 6-23. Number of Surface Water Samples with HQs > 1.0

COPC	Number of Samples with HQs > 1.0 (Maximum HQ)	Percent of Samples with HQs > 1.0
4,4′-DDT	19[d]/170 (4.7)[e]	11%
Total DDx	35[d]/170 (20)[e]	21%
VOCs		
Ethylbenzene	1/23 (1.6)	3.7%
Trichloroethene	1/23 (4.1)	3.7%

[a] The aluminum TRV is based on a NOEC. Only 5% (7 out of 142) of the surface water samples exceeded the aluminum invertebrate LOEC of 1,020 µg/L.

[b] An additional two samples had DLs that were greater than the TRV. The maximum HQ based on a DL is 1.4 for BEHP.

[c] 0/160 samples had total PCB concentrations greater than the alternative total PCB TRV of 0.19 µg/L, which is specific to aquatic organisms.

[d] An additional four samples had DLs that were greater than the TRV. The maximum HQ based on a DL is 1.6 for both 4,4′-DDT and total DDx.

[e] 1/170 samples had 4,4′-DDT and total DDx concentrations greater than the alternative 4,4′-DDT TRV of 0.011 µg/L, which is specific to aquatic organisms (HQs were 1.7 and 1.8, respectively).

BEHP – bis(2-ethylhexyl) phthalate
COPC – chemical 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

Table 6-24 presents a summary of the individual HQs calculated for near-bottom surface water samples for all COPCs. The percentages of samples with HQs > 1.0 were similar to those presented in Table 6-28 for all surface water samples; aluminum, naphthalene, 4,4-DDT, and total DDx had the greatest number of samples with HQs > 1.0.

Table 6-24. Number of Near-Bottom Surface Water Samples with HQs > 1.0

COPC	Number of Samples with HQs > 1.0 (Maximum HQ)	Percentage of Samples with HQs > 1.0
Metals		
Aluminum (total)	68[a]/72 (21)	94%
Zinc (dissolved)	1/91 (1.1)	1.1%
Butyltins		
Butyltin	0/91	0%

Table 6-24. Number of Near-Bottom Surface Water Samples with HQs > 1.0

COPC	Number of Samples with HQs > 1.0 (Maximum HQ)	Percentage of Samples with HQs > 1.0
PAHs		
Benzo(a)anthracene	2/122 (10)	1.6%
Benzo(a)pyrene	3/122 (14)	2.5%
Naphthalene	10/145(50)	6.9%
VOCs		
Ethylbenzene	1 / 23 (1.6)	4.3%
Trichloroethene	1 / 23 (4.1)	4.3%
Phthalates		
BEHP	2[b]/105 (2.3)	1.9%
PCBs		
Total PCBs	1/86 (1.1)[c]	1.1%
Pesticides		
4,4′-DDT	11[d]/93 (4.7)[e]	12%
Total DDx	21[d]/93 (20)[e]	23%

[a] The aluminum TRV is based on a NOEC. Only 5% (3 out of 72) of the surface water samples exceeded the aluminum invertebrate LOEC of 1,020 µg/L.

[b] An additional sample had a DL that was greater than the TRV. The maximum HQ based on a DL is 1.4.

[c] 0/86 samples had total PCB concentrations greater than the alternative total PCB TRV of 0.19 µg/L, which is specific to aquatic organisms.

[d] An additional three samples had DLs that was greater than the TRV. The maximum HQ based on a DL is 1.6 for both 4,4′-DDT and total DDx.

[e] 1/93 samples had 4,4'-DDT and total DDx concentrations greater than the alternative 4,4'-DDT TRV of 0.011 µg/L, which is specific to aquatic organisms, (HQs were 1.7 and 1.8, respectively).

BEHP – bis(2-ethylhexyl) phthalate
COPC – chemical of potential concern
DL – detection limit
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
HQ – hazard quotient
NA – not analyzed
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)
TRV – toxicity reference value
VOC – volatile organic compound

All COPCs were identified as COCs because HQs were > 1.0 in at least one surface water sample. Aluminum was identified as a COC because the HQ was > 1.0. However, 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 invertebrates and not toxic at

naturally occurring concentrations generally found in surface water, aluminum is not expected to pose unacceptable risks to benthic invertebrates.

Maps 6-30 through 6-34 present the sampling locations with HQ > 1.0. A discussion of these COCs 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.

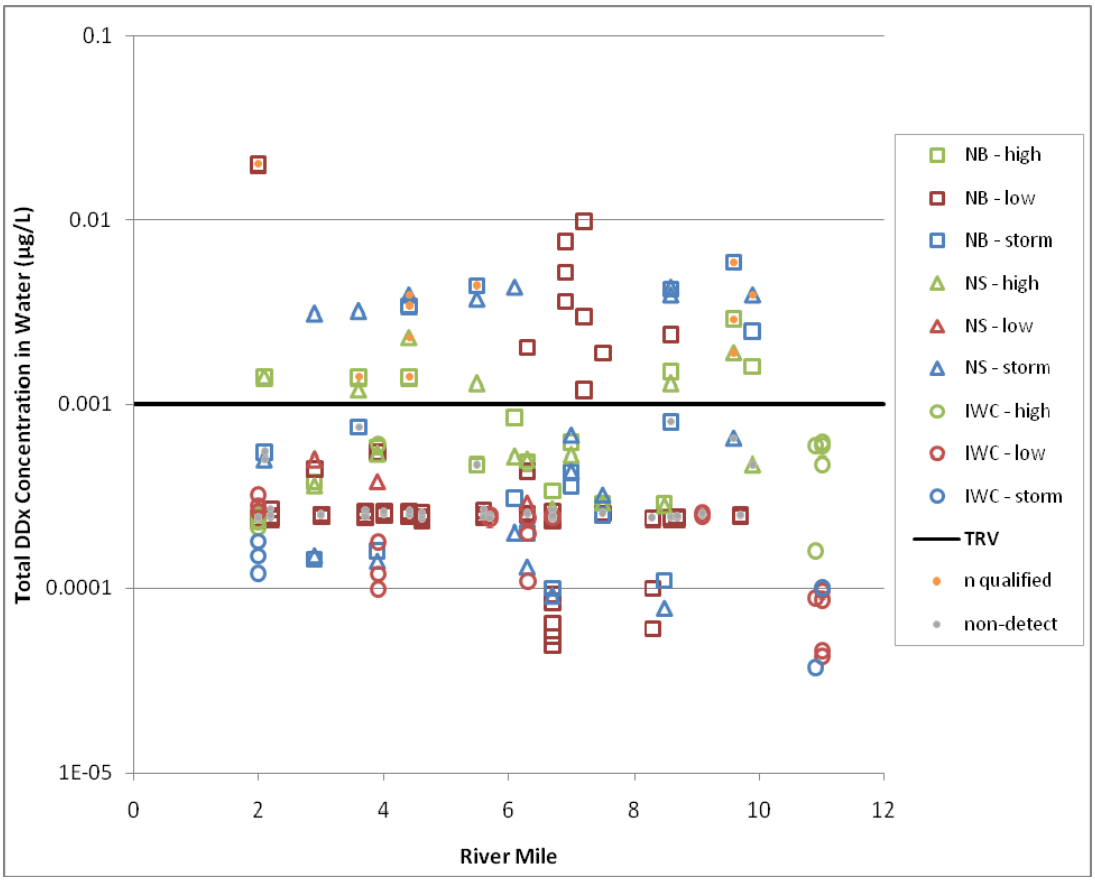
Metals, Butyltins, and VOCs – For four of the COCs (i.e., zinc, butyltin, ethylbenzene, and trichloroethene), calculated HQs were > 1.0 in only one sample. The butyltin evaluation may be somewhat uncertain because the TRV is a surrogate based on the AWQC for TBT. Uncertainty is associated with the evaluation 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 COCs (i.e., benzo[a]anthracene, benzo[a]pyrene, BEHP, total PCBs but excluding two DDT metabolites, discussed below with DDx) had HQs > 1.0 in fewer than five samples. The two benzo(a)anthracene samples and three benzo(a)pyrene samples with HQs > 1.0 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 had HQs > 1.0 in two samples at different locations in the river. Only one near-bottom sample had an HQ > 1.0 for BEHP (HQ = 2.3), and only one near-bottom sample had an HQ > 1.0 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 (which is specific to aquatic organisms) was used, no samples exceeded the TRV.

Naphthalene – Naphthalene did not have any HQs > 1.0 for the peristaltic or XAD samples collected during the LWG surface water sampling events. However, naphthalene HQs were > 1.0 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. Therefore, the naphthalene HQs were > 1.0 only at one localized area.

Aluminum – Concentrations of aluminum were highest in samples collected during the high-flow events, with HQs > 1.0 throughout the river. No pattern with depth could be distinguished. Aluminum is a naturally occurring element in the environment. Aluminum concentrations in background surface water and sediment are discussed in Section 6.5.4.3.

DDx – Total DDx HQs were > 1.0 throughout the Study Area (RM 2.0 to RM 9.9). The highest HQ for total DDx was 20 (RM 2.0 during the March 2005 low-flow event); however, this sample was N-qualified. Nearly one-third of total DDx and 4,4′-DDT exceedances were based on N-qualified data (31% for total DDx and 32% for 4,4′-DDT). The qualification indicates that the analyst believed that the result was due to analytical interference from a chemical other than the target analyte. The highest non-N-qualified HQ was 9.8; this sample was collected at RM 7.2 during a low-flow event (Figure 6-12).



high – high-flow conditions
low – low-flow conditions
NS – near surface
IWC – integrated water column
NB – near bottom
storm – storm flow conditions

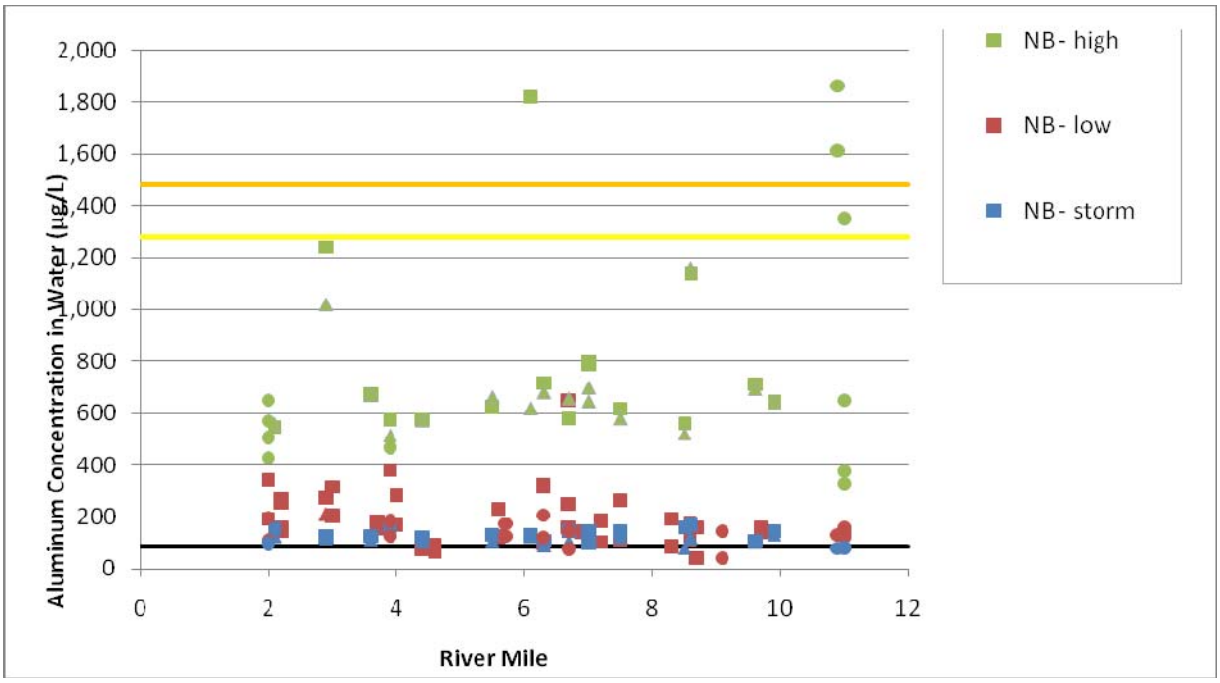
Figure 6-12. Total DDx Concentrations in All Surface Water Samples

The uncertainty associated with the total DDx evaluation is due in part to the use of bird effects data as the basis for the 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 birds are known to be sensitive to DDx compounds,[46] the water TRV that is protective of birds likely overestimates risks to the benthic community. When the alternative DDT TRV of 0.011 µg/L (which is specific to aquatic organisms) was used, total DDx concentration in only one sample (W001 located at RM 2.0) exceeded the TRV (HQ based on alternative TRV = 1.8); however, this result was N-qualified.

6.5.4.3 Evaluation of Background Concentrations

Aluminum was evaluated even though it is a naturally occurring crustal element in the environment. Background concentrations in surface water and sediment were established as part of the RI (Section 7.0 of the draft RI). 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) was 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) was 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 6-13).



high – high-flow conditions
low – low-flow conditions
NB – near bottom
storm – storm flow conditions

Figure 6-13. Aluminum Surface Water Concentrations Compared to Water TRVs and Background Concentrations

Based on these comparisons, it was concluded that potential unacceptable risks 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 risks to benthic invertebrates.

Zinc also occurs naturally as a crustal element in the environment. A background water concentration could not be established because of the number of data points was too limited (see Attachment 11). The Study Area UCL concentration of zinc (2.5 µg/L) was greater than highest zinc concentration detected in background[47] (range of 1.4 to 2.2 µg/L). The Study

[47] Zinc concentrations were detected in only 3 of 22 surface water samples included in the background dataset (see Section 7.0 of the RI).

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 solely be attributed to background.

6.5.4.4 Summary of Surface Water COCs

The following surface water COCs were identified: aluminum, zinc, butyltin, benzo(a)anthracene and benzo(a)pyrene, naphthalene, ethylbenzene, trichloroethene, BEHP, total PCBs, 4,4′-DDT, and total DDx. Table 6-25 summarizes the HQs and exposure and effects uncertainties for each surface water COC. The results of the surface water LOE were compared with other LOEs determine risk conclusions for benthic invertebrates (Section 6.7).

Table 6-25. Summary of Benthic Invertebrate Surface Water COCs

COC	Risk Exposure Area	HQs[a]	Frequency of All Samples with HQs > 1.0	Sampling Event with HQs > 1.0	Key Uncertainties
Metals					
Aluminum	Site-wide	1.0 – 21	93%	All	Aluminum concentrations in background areas indicate aluminum is not attributed to anthropogenic sources in the Study Area.
Zinc	Individual sample with HQ > 1.0 at RM 9.7 west	1.1	0.6%	November 2004 low-flow event	The HQ was > 1.0 based on 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. A single sample with an HQ of 1.1 does not indicate an unacceptable benthic community risk.
Butyltins					
Butyltin	Individual sample with HQ > 1.0 at RM 11 west	1.2	0.6%	Winter 2007 high-flow event	TRV is based on a surrogate TBT TRV; HQ was > 1.0 in one sample only; HQs are < 1.0 for all near-bottom samples; TRV is based on tributyltin effects data for invertebrates and fish. A single sample with an HQ of 1.2 does not indicate an unacceptable benthic community risk.
PAHs					
Benzo(a)anthracene	Individual samples with HQ > 1.0 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 > 1.0 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 > 1.0 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 > 1.0 based on peristaltic samples only at W012 and W031; TRV is based on extrapolated *Daphnia* acute LC50. Findings are consistent with other LOEs.

Table 6-25. Summary of Benthic Invertebrate Surface Water COCs

COC	Risk Exposure Area	HQs[a]	Frequency of All Samples with HQs > 1.0	Sampling Event with HQs > 1.0	Key Uncertainties
Naphthalene	RM 6.4 west	2.9 – 50	4%	May 2005 (non-LWG sampling event)	HQs > 1.0 based on10 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.
Phthalates					
BEHP	Individual samples with HQ > 1.0 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 > 1.0 (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 > 1.0 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 > 1.0 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.

Table 6-25. Summary of Benthic Invertebrate Surface Water COCs

COC	Risk Exposure Area	HQs[a]	Frequency of All Samples with HQs > 1.0	Sampling Event with HQs > 1.0	Key Uncertainties
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 > 1.0 (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 sample 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.
VOCs					
Ethylbenzene	Individual sample with HQ > 1.0 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 > 1.0 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 > 1.0 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 > 1.0 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 > 1.0 are presented. HQs in all other water samples were < 1.0.

BEHP – bis(2-ethylhexyl) phthalate
COC – chemical of 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
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)
VOC – volatile organic compound

6.6 ECOLOGICAL EVALUATION OF TZW

This section presents results of the TZW screening, as well as an evaluation of the ecological relevance of this LOE for estimating risks to benthic invertebrates that may be exposed to TZW. This evaluation is also applicable to other ecological receptors, including benthic fish (i.e., sculpin and lamprey ammocoetes), aquatic plants, and amphibians (see RI Section 5.0 for a description of facilities and RM locations. TZW sampling locations are presented on Map 4-15.

For the purpose of the BERA, TZW is the porewater associated with the sediment matrix within the top 38 cm[48] of the sediment column. TZW is composed of some percentage of both groundwater and surface water. An analysis of available data regarding TZW is provided in Appendix C2 of the draft RI.

Because of the spatial focus of the TZW data collection, the TZW chemical concentrations are not representative of conditions throughout the Study Area, but rather a snapshot of where TZW data were collected (i.e., no Study-Area wide spatial inferences can be made).

The details of this assessment are presented as follows:

- A summary of the water TRVs is presented in Section 6.6.1. 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 6.6.2 presents a summary of the TZW COPCs evaluated in the BERA. Some COPCs were not evaluated because no toxicity thresholds were available.
- A summary of the TZW data is presented in Section 6.6.3. Data in this assessment are represented by COPC concentrations in all TZW samples. .
- Section 6.6.4 summarizes the TZW results and uncertainties associated with the evaluation.
- Section 6.6.5 presents a discussion of the ecological relevance of benthic receptor exposure to TZW.
- Section 6.6.6 presents a summary of the TZW evaluation.

Figure 6-14 presents a flowchart showing organization of the TZW evaluation.

[48] 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.

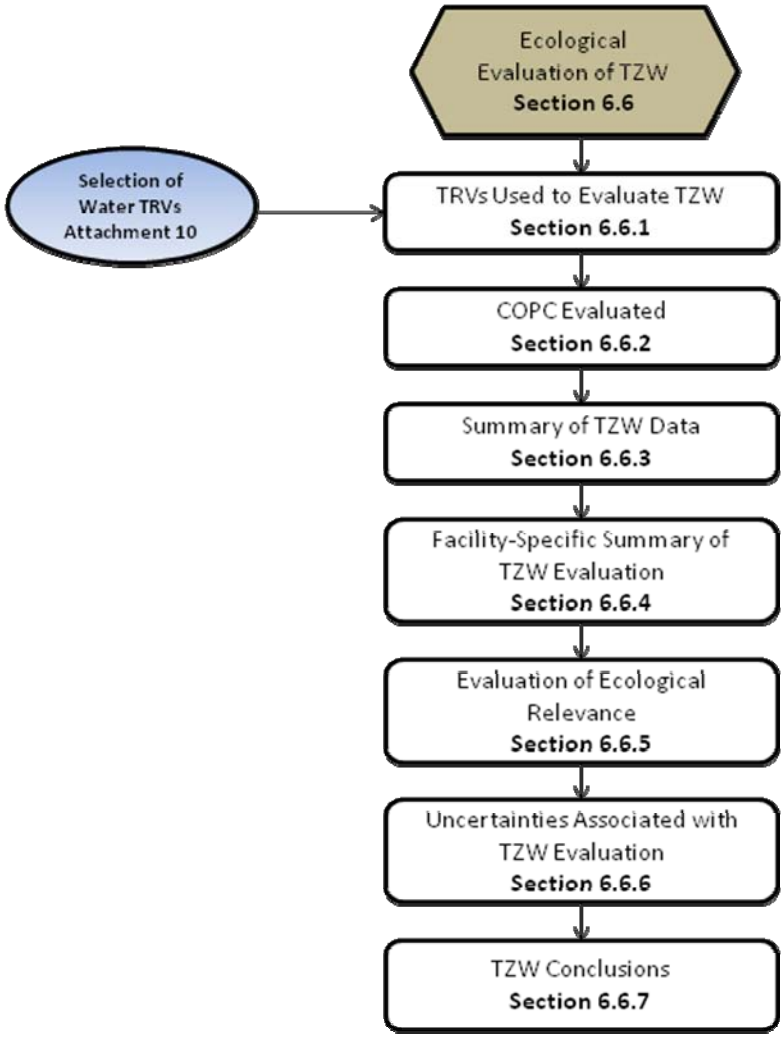


Figure 6-14. Overview of TZW Section Organization

6.6.1 TRVs Used to Evaluate TZW

Chronic water TRVs were developed through a review of water quality standards, criteria, published benchmarks, and toxicity data. TRVs were selected based on the hierarchy detailed in Attachment 10 and were approved by EPA for use in the BERA. Water TRVs were developed for all TZW COPCs except where toxicity data were not available; water TRVs were not developed calcium, titanium, diesel-range hydrocarbons, residual-range hydrocarbons, TPH, and individual dioxin and furan congeners. Metals criteria 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 criteria; otherwise the total concentration for both the sample and criterion were used. Table 6-26 presents the TRVs and their sources.

Table 6-26. TRVs for TZW COPCs

COPC	Unit	TRV	Source
Metals			
Aluminum	µg/L	87[a]	AWQC
Barium	µg/L	4[a]	Tier II
Beryllium	µg/L	0.66[a]	Tier II
Cadmium	µg/L	0.09[b]	AWQC
Cobalt	µg/L	23[a]	Tier II
Copper	µg/L	2.74[b]	AWQC
Iron	µg/L	1,000[a]	AWQC
Lead	µg/L	0.54[b]	AWQC
Magnesium	µg/L	82,000[a]	AWQC
Manganese	µg/L	120[a]	Tier II
Nickel	µg/L	16.1[b]	AWQC
Potassium	µg/L	53,000[a]	Tier II
Sodium	µg/L	680,000[a]	Tier II
Vanadium	µg/L	20[a]	Tier II
Zinc	µg/L	36.5[b]	AWQC
PAHs			
2-Methylnaphthalene	µg/L	2.1	Tier II
Acenaphthene	µg/L	23	Tier II
Anthracene	µg/L	0.73	Tier II
Benzo(a)anthracene	µg/L	0.027	Tier II
Benzo(a)pyrene	µg/L	0.014	Tier II
Benzo(b)fluoranthene	µg/L	0.6774	EPA (2003d)[c]
Benzo(g,h,i)perylene	µg/L	0.4391	EPA (2003d)[c]
Benzo(k)fluoranthene	µg/L	0.6415	EPA (2003d)[c]
Chrysene	µg/L	2.042	EPA (2003d)[c]
Dibenzo(a,h)anthracene	µg/L	0.2825	EPA (2003d)[c]
Fluoranthene	µg/L	6.16	Tier II
Fluorene	µg/L	3.9	Tier II
Indeno(1,2,3-cd)pyrene	µg/L	0.275	EPA (2003d)[c]
Naphthalene	µg/L	12	Tier II
Phenanthrene	µg/L	6.3	Tier II
Pyrene	µg/L	10.11	EPA (2003d)[c]

Table 6-26. TRVs for TZW COPCs

COPC	Unit	TRV	Source
VOCs			
1,1-Dichloroethene	µg/L	25	Tier II
1,2,4-Trimethylbenzene	µg/L	7.3[d]	Tier II
1,3,5-Trimethylbenzene	µg/L	7.3[d]	Tier II
Benzene	µg/L	130	Tier II
Carbon disulfide	µg/L	0.92	Tier II
Chlorobenzene	µg/L	64	ODEQ
Chloroethane	µg/L	47[f]	Tier II
Chloroform	µg/L	28	Tier II
cis-1,2-Dichloroethene	µg/L	590	Tier II
Ethylbenzene	µg/L	7.3	Tier II
Isopropylbenzene	µg/L	7.3[d]	Tier II
m,p-Xylene	µg/L	66.67	EPA (2006d)
o-Xylene	µg/L	13[g]	Tier II
Toluene	µg/L	9.8	Tier II
Total xylenes	µg/L	13[g]	Tier II
Trichloroethene	µg/L	47	Tier II
SVOCs			
1,2-Dichlorobenzene	µg/L	14	Tier II
1,4-Dichlorobenzene	µg/L	15	Tier II
Dibenzofuran	µg/L	3.7	Tier II
Pesticides			
2,4′-DDD	µg/L	0.001[h]	EPA (2006e)
2,4′-DDT	µg/L	0.001[h]	EPA (2006e)
4,4′-DDD	µg/L	0.001[h]	EPA (2006e)
4,4′-DDE	µg/L	0.001[h]	EPA (2006e)
4,4′-DDT	µg/L	0.001	EPA (2006e)
Total DDx	µg/L	0.001[h]	EPA (2006e)
TPH			
Gasoline-range hydrocarbons[i]	µg/L	NV	EPA (2008f)
Aliphatic hydrocarbons C4-C6[k]	µg/L	128[i]	EPA (2008f)
Aliphatic hydrocarbons C6-C8[k]	µg/L	54[i]	EPA (2008f)
Aliphatic hydrocarbons C8-C10[k]	µg/L	9.5[i]	EPA (2008f)
Aliphatic hydrocarbons C10-C12[k]	µg/L	2.6[i]	EPA (2008f)

Table 6-26. TRVs for TZW COPCs

COPC	Unit	TRV	Source
Aromatic hydrocarbons C8-C10[k]	µg/L	212[i]	EPA (2008f)
Other Chemicals			
Perchlorate	µg/L	9,300[e]	Dean et al. (2004) per EPA (2008f)
Cyanide	µg/L	0.0052	AWQC

[a] TRV is based on total criteria; TRV was compared to total concentration detected in Study Area.
[b] TRV is based on dissolved criteria; 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 acute-to-chronic ratio 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] TRV is based on criteria for 4,4′-DDT.
[i] 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 are not individual COPCs. Components of gasoline were used as a surrogate for gasoline-range hydrocarbons. Gasoline-range hydrocarbons were identified as a COPC if any one gasoline component exceeded its TRV.

AWQC – ambient water quality criteria
COPC – chemical of potential concern
DDT – dichlorodiphenyltrichloroethane
EPA – US Environmental Protection Agency
TRV – toxicity reference value
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
SVOC – semivolatile organic compound
TPH – total petroleum hydrocarbons
TRV – toxicity reference value
TZW – transition zone water
VOC – volatile organic compound
NV – no value

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 daphnid (a water column species) toxicity data and may not accurately characterize effects on benthic invertebrates, fish, amphibians, and plants. 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.

6.6.2 COPCs Evaluated

Sixty TZW COPCs were identified after comparing the maximum detected concentrations in any one TZW sample at any location to TRVs, as described in Attachment 5. Table 6-27 presents the detected COPCs by facility.

Some TZW COIs (i.e., calcium, titanium, diesel-range hydrocarbons, residual-range hydrocarbons, TPH, and several individual dioxins and furans other than 2,3,7,8-TCDD) could not be evaluated because no toxicological data were available to allow development of water TRVs. The risks to benthic receptors associated with exposure to these chemicals in TZW are therefore unknown.

Table 6-27. COPCs in TZW by Facility

	Facility									
		Arkema								
COPC	ARCO	Acid Plant Area	Chlorate Plant Area	Mobil Oil	Gasco	Gunderson	Kinder Morgan	Rhône-Poulenc	Siltronic	Willbridge
Metals										
Aluminum (total)[a]	X	X	X	X	X	X	X	X	X	X
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 6-27. COPCs in TZW by Facility

	Facility									
		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	
Pesticides										
2,4′-DDD		X						X		
2,4′-DDT		X						X		

Table 6-27. COPCs in TZW by Facility

	Facility								
		Arkema							
COPC	**ARCO**	**Acid Plant Area**	**Chlorate Plant Area**	**Mobil Oil**	**Gasco**	**Gunderson**	**Kinder Morgan**	**Rhône-Poulenc**	**Siltronic**	**Willbridge**
4,4′-DDD		X						X		
4,4′-DDE		X						X		
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	
Total xylenes					X				X	
Trichloroethene									X	

Table 6-27. COPCs in TZW by Facility

	Facility									
		Arkema								
COPC	**ARCO**	**Acid Plant Area**	**Chlorate Plant Area**	**Mobil Oil**	**Gasco**	**Gunderson**	**Kinder Morgan**	**Rhône-Poulenc**	**Siltronic**	**Willbridge**
Petroleum										
Gasoline-range hydrocarbons	X			X	X		X		X	X
Other Chemicals										
Cyanide					X				X	
Perchlorate		X	X							

[a] Criteria are based on total concentration.
[b] Criteria are based on dissolved concentration.
COPC – chemical of potential concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
PAH – polycyclic aromatic hydrocarbon
SVOC – semivolatile organic compound
TZW – transition zone water
VOC – volatile organic compound

6.6.3 Summary of TZW Data

TZW samples evaluated in this assessment were collected in shallow sediment (≤ 38 cm below the mudline), which included 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. 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. LWG TZW sampling was conducted during the fall of 2005 to obtain samples that represented a relatively higher groundwater discharge to the river. Sampling locations (Map 4-15) were selected along each of the properties of the previously mentioned facilities based on the results of the groundwater discharge mapping field effort (Integral 2006a). All TZW data are presented in Attachment 4.

Four metal COPCs (i.e., barium, iron, magnesium, and manganese) were detected in all samples; other metals were detected less frequently. Cyanide was detected in almost all samples. Acenaphthene and fluorene were the most frequently detected PAH COPCs. Dibenzofuran was the most frequently detected SVOC; benzene, toluene, xylene, and 1,2,4-trimethylbenzene were the most frequently detected VOCs. The most frequently detected pesticides were DDx compounds (2,4′-DDD and total DDx). Diesel-range hydrocarbons were detected in over half the shallow TZW samples.

Uncertainty Associated with Ecological Exposure to TZW

Several uncertainties are associated with the exposure assumptions used to evaluate risks to benthic invertebrate from TZW. The representativeness of the collected TZW samples to characterize exposure of organisms living in or on the sediment to TZW is highly uncertain. TZW samples 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) whereas the biologically active zone is more likely between 10 and 20 cm below the mudline. Deeper TZW is less diluted by the overlying surface water than shallower TZW; thus inclusion of TZW below the biologically active zone likely overestimates the concentrations to which benthic invertebrates are actually exposed. 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). Burrowing organisms that live below the oxygenated zone have developed behavioral strategies (e.g., tube/burrow configurations that alter boundary layer flows, mucus tube linings that decrease porewater infiltration, pumping by various appendages or body movements to irrigate burrows) to introduce overlying water into their tubes or burrows for both respiration and feeding. As a result of these strategies, burrowing organisms have relatively low exposure to porewater compared to surface water. Organisms that live on the sediment surface (or are closely tied to the surface) are even less exposed to porewater because they don't live within the sediment. Thus TZW concentrations compared to TRVs overestimate risks to benthic organisms.

The spatial representativeness of TZW for the Study Area is also uncertain. TZW was sampled in 10 areas (Integral 2006a) and are only a snapshot of the sampling locale. Therefore, the TZW data are not representative of, nor should they be used to infer large-area or site-wide risks to benthic fish, invertebrates, plants, and amphibians from COPCs in TZW.

6.6.4 Summary of TZW Evaluation

Each detected COPC was compared to its respective TRV and the results are reported as a ratio (an HQ) of the sample concentration to the TRV. Specific summaries of where HQs >1.0 follow; the frequencies of individual sample exceedances across all locations are presented in Table 6-28.

ARCO – At the ARCO site, eight COPCs have HQs > 1.0 based on detected concentrations, including five metals (i.e., aluminum, barium, iron, manganese, zinc), two PAHs (i.e., benzo[a]anthracene, benzo[a]pyrene), and gasoline-range hydrocarbons. Of these COPCs, metals generally have the highest HQs, specifically aluminum, iron, and barium. One COPC (cadmium) had HQs > 1.0 based on 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, 15 COPCs have HQs > 1.0 based on detected concentrations, including six metals (barium, aluminum, iron, magnesium, manganese, and sodium), one PAH (naphthalene), DDx compounds (2,4′-DDD, 2,4′-DDT, 4,4′-DDD, 4,4′-dichlorodiphenyldichloroethylene [DDE], 4,4′-DDT, and total DDx), chlorobenzene, and chloroform. DDTs generally have the highest HQs, although they are based on whole water samples that include particulate material.[49] Two COPCs (i.e., carbon disulfide and perchlorate) were not detected at this facility, but each had a DL greater than its TRV.

Arkema Facility's Chlorate Plant Area– Eleven COPCs have HQs > 1.0 at the facility's chlorate plant area based on detected concentrations, including nine metals (aluminum, barium, cadmium, iron, magnesium, manganese, nickel, potassium, and sodium), chloroform, and perchlorate. Barium and manganese have the highest HQs. This is the only site where perchlorate was detected.

Mobil Oil – Fifteen COPCs have HQs > 1.0 at the Mobil Oil site based on detected concentrations, including seven metals (aluminum, 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. Barium, iron, manganese, and gasoline-range hydrocarbons have the highest HQs.

Gasco – At the Gasco site, 31 COPCs have HQs > 1.0 in TZW, including 6 metals (aluminum, 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

[49] Dissolved concentrations tended to be several orders of magnitude lower and were undetected. Detection limits exceeded the TRV.

hydrocarbons. Naphthalene, benzo(a)pyrene, and benzo(a)anthracene have the highest HQs of the PAH COPCs. Ethylbenzene and total xylenes have the highest HQs of the VOC COPCs. Cyanide has the highest HQ of all COPCs at the Gasco site. Gasco is one of two TZW sampling sites where cyanide was detected.

Gunderson – Five COPCs have HQs > 1.0 at the Gunderson site based on detected concentrations, including four metals (aluminum, barium, iron, and manganese), and one VOC (chloroethane). The maximum HQ for the site is for iron.

Kinder Morgan – At Kinder Morgan, six COPCs have HQs > 1.0 in TZW, including four metals (aluminum, barium, iron, and manganese), one PAH (benzo[a]anthracene), and gasoline-range hydrocarbons. Manganese and diesel-range hydrocarbons have the highest maximum HQs at this site.

Rhône-Poulenc – Eighteen COPCs have HQs > 1.0 at the Rhône-Poulenc site, including 10 metals (aluminum, barium, beryllium, cadmium, copper, iron, lead, magnesium, manganese, and nickel), two SVOCs (1,2-dichlorobenzene, 1,4-dichlorobenzene), five pesticides (2,4′-DDD, 2,4′-DDT, 4,4′-DDD, 4,4′-DDE, 4,4′-DDT, and total DDx), and one VOC (chlorobenzene). Aluminum, barium, 2,4′-DDD, and total DDx[50] have the highest maximum HQs at the site. One COPC (4,4′-DDT) had an HQ > 1.0 based on two samples with DLs greater than the TRV, but otherwise was not detected.

Siltronic – Thirty-eight COPCs have HQs > 1.0 at the Siltronic site, including six metals (aluminum, barium, cobalt, iron, manganese, and vanadium), 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), 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. Benzo(a)pyrene, benzo(a)anthracene, naphthalene, and trichloroethene have the highest HQs at this property.

Willbridge – At the Willbridge site, seven COPCs, including six metals (aluminum, barium, beryllium, cadmium, iron, manganese), and gasoline-range hydrocarbons, have HQs > 1.0 based on detected concentrations. Aluminum, 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 its TRV. Otherwise, these two PAHs were not detected at the Willbridge site.

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[50] 2,4-DDD and total DDx exceedances were based on a single sample at Rhône-Poulenc. Dissolved and total concentrations were of similar magnitude.

Table 6-28. TZW COPCs with HQs > 1.0 in Individual Samples by Facility

	Number of Samples with HQs > 1.0 (maximum HQ)									
		Arkema								
COPC	ARCO	Acid Plant Area	Chlorate Plant Area	Mobil Oil	Gasco	Gunderson	Kinder Morgan	Rhône-Poulenc	Siltronic	Willbridge
Metals										
Aluminum (total)	6/7 (130)	3[a]/8 (93)	4[b]/10 (63)	5[c]/11 (390)	6/8 (110)	4/9 (77)	3[a]/8 (35)	8/10 (370)	1[c]/13 (13)	6/9 (470)
Barium (total)	7/7 (73)	8/8 (610)	10/10 (1,100)	11/11 (88)	8/8 (86)	9/9 (68)	8/8 (31)	10/10 (170)	13/13 (57)	9/9 (86)
Beryllium (total)	0/7	0/8	0/10	1/11 (1.8)	0/8	0/9	0/8	1/10 (1.7)	0/13	1/9 (2)
Cadmium (dissolved)	0[d]/5	0/4	3/6 (2.6)	1/12 (1.1)	0/4	0/2	0[d]/3	5[a]/7 (5.8)	0/6	1/6 (1.5)
Cobalt (total)	NA	NA	NA	NA	NA	NA	NA	NA	3/13 (3.6)	NA
Copper (dissolved)	0/5	NA	NA	0/12	0/4	0/2	0/3	1/7 (1.3)	0/6	0/6
Iron (total)	7/7 (75)	7/8 (110)	6/10 (250)	11/11 (110)	8/8 (130)	9/9 (91)	8/8 (49)	10/10 (98)	26/26 (180)	9/9 (120)
Lead (dissolved)	0/5	0/4	0/6	1/12 (3)	2/4 (1.7)	0/2	0/3	1/7 (2.8)	0/6	0/6
Magnesium (total)	0/7	4/8 (7)	1/10 (3.8)	0/11	0/8	0/9	0/8	3/10 (2.2)	0/26	0/9
Manganese (total)	7/7 (52)	8/8 (94)	10/10 (550)	11/11 (150)	8/8 (130)	9/9 (43)	8/8 (72)	10/10 (130)	26/26 (84)	8/9 (110)
Nickel (dissolved)	0/5	0/4	1/6 (1.6)	0/12	1/4 (1.1)	0/2	0/3	1[a]/7 (1.1)	0/6	0/6
Potassium (total)	0/7	0/8	2/10 (3.7)	0/11	0/8	0/9	0/8	0/10	0/13	0/9
Sodium (total)	0/7	1/8 (14)	10/10 (55)	0/11	0/8	0/9	0/8	0/10	0/13	0/9
Vanadium (total)	NA	NA	NA	NA	NA	NA	NA	NA	6/13 (19)	NA
Zinc (dissolved)	1/5 (14)	0/4	0/6	0/12	0/4	0/2	0/3	0/7	0/6	0/6
PAHs										
2-Methylnaphthalene	0/12	NA	NA	0/21	8/12 (40)	NA	0/11	NA	3/19 (17)	0/14
Acenaphthene	0/12	NA	NA	0/21	4/12 (5.2)	NA	0/11	NA	20/32 (17)	0/14
Anthracene	0/12	NA	NA	0/21	10/12 (13)	NA	0/11	NA	18/32 (87)	0/14

Table 6-28. TZW COPCs with HQs > 1.0 in Individual Samples by Facility

	Number of Samples with HQs > 1.0 (maximum HQ)									
		Arkema								
COPC	ARCO	Acid Plant Area	Chlorate Plant Area	Mobil Oil	Gasco	Gunderson	Kinder Morgan	Rhône-Poulenc	Siltronic	Willbridge
Benzo(a)anthracene	1[b]/12 (5.6)	NA	NA	5[b]/ 21 (8.5)	9/12 (120)	NA	2/11 (2.9)	NA	14[c]/32 (1,200)	0[a,d]/14
Benzo(a)pyrene	2[b]/12 (15)	NA	NA	5[b]21 (25)	9/12 (210)	NA	0[b,d]/11	NA	18[a]/ 32 (2,700)	0[a,d]/14
Benzo(b)fluoranthene	0/12	NA	NA	0/21	3/12 (3.1)	NA	0/11	NA	10/32 (49)	0/14
Benzo(g,h,i)perylene	0/12	NA	NA	1/21 (1.1)	3/12 (7.3)	NA	0/11	NA	9/32 (66)	0/14
Benzo(k)fluoranthene	0/12	NA	NA	0/21	3/12 (3.1)	NA	0/11	NA	7/32 (14)	0/14
Chrysene	0/12	NA	NA	0/21	3/12 (2.2)	NA	0/11	NA	7/32 (17)	0/14
Dibenzo(a,h)anthracene	0/12	NA	NA	0/21	1/12 (1.2)	NA	0/11	NA	7/32 (13)	0/14
Fluoranthene	0/12	NA	NA	0/21	3/12 (2.8)	NA	0/11	NA	8/32 (17)	0/14
Fluorene	0/12	NA	NA	3/21 (1.5)	10/12 (7.9)	NA	0/11	NA	23/32 (28)	0/14
Indeno(1,2,3-cd)pyrene	0/12	NA	NA	1/21 (1.2)	3/12 (9.8)	NA	0/11	NA	9/32 (61)	0/14
Naphthalene	0/12	2[a]/9 (2.2)	0/10	0/21	6/12 (260)	0/9	0/12	0/10	23/60 (1,100)	0/14
Phenanthrene	0/12	NA	NA	5/21 (2.4)	10/12 (13)	NA	0/11	NA	21/32 (57)	0/14
Pyrene	0/12	NA	NA	0/21	3/12 (3.2)	NA	0/11	NA	8/32 (15)	0/14
SVOCs										
1,2-Dichlorobenzene	0/7	0/9	0/10	0/11	0/8	0/9	0/9	5/10 (46)	0/54	0/9
1,4-Dichlorobenzene	0/7	0/5	0/6	0/11	0/8	0/9	0/9	2/10 (16)	0/54	0/9
Dibenzofuran	0/12	NA	NA	0/21	1/12 (2.2)	NA	0/11	NA	2/19 (2)	0/14
Pesticides										
2,4′-DDD		8[c]/12 (1,100)	NA	NA	NA	NA	NA	2/2 (170)	NA	NA
2,4′-DDT		1[c]/12 (93)	NA	NA	NA	NA	NA	2/2 (9.8)	NA	NA

Table 6-28. TZW COPCs with HQs > 1.0 in Individual Samples by Facility

	Number of Samples with HQs > 1.0 (maximum HQ)									
		Arkema								
COPC	ARCO	Acid Plant Area	Chlorate Plant Area	Mobil Oil	Gasco	Gunderson	Kinder Morgan	Rhône-Poulenc	Siltronic	Willbridge
4,4′-DDD		5[c]/ 12 (1,300)	NA	NA	NA	NA	NA	1[a]/2 (15)	NA	NA
4,4′-DDE		2[c]/12 (120)	NA	NA	NA	NA	NA	1[a]/2 (15)	NA	NA
4,4′-DDT	NA	3[c]/12 (1,800)	NA	NA	NA	NA	NA	0[b,d]/2	NA	NA
Total DDx	NA	8[c]/12 (3,100)	NA	NA	NA	NA	NA	2/2 (210)	NA	NA
VOCs										
1,1-Dichloroethene	0/7	0/9	0/10	0/11	0/8	0/9	0/9	0/10	2/54 (1.6)	0/9
1,2,4-Trimethylbenzene	NA	NA	NA	NA	NA	NA	NA	NA	11/41 (9.6)	NA
1,3,5-Trimethylbenzene	NA	NA	NA	NA	NA	NA	NA	NA	5/41 (3)	NA
Benzene	0/7	0/9	0/10	0/11	3/8 (4.2)	0/9	0/9	0/10	6/54 (30)	0/9
Carbon disulfide	0/7	0[a,d]/9	0/10	0/11	3/8 (870)	0/9	0/9	0/10	1[c]/54 (1.3)	0/9
Chlorobenzene	0/7	2/9 (190)	0/10	0/11	0/8	0/9	0/9	1/10 (3.3)	0/54	0/9
Chloroethane	0/7	0/9	0/10	0/11	0/8	1/9 (3.4)	0/9	0/10	0/54	0/9
Chloroform	0/7	1/9 (21)	3[b]/10 (7.9)	0/11	0/8	0/9	0/9	0/10	0/54	0/9
cis-1,2-Dichloroethene	0/7	0/9	0/10	0/11	0/8	0/9	0/9	0/10	5/54 (110)	0/9
Ethylbenzene	0/7	0/9	0/10	0/11	3/8 (11)	0/9	0/9	0/10	12/54 (57)	0/9
Isopropylbenzene	0/7	0/9	0/10	0/11	2/8 (1.5)	0/9	0/9	0/10	8/54 (2)	0/9
m,p-Xylene	0/7	0/9	0/10	0/11	0/8	0/9	0/9	0/10	3/54 (4.4)	0/9
o-Xylene	0/7	0/9	0/10	0/11	3/8 (3.6)	0/9	0/9	0/10	9/54 (12)	0/9
Toluene	0/7	0/9	0/10	0/11	4/8 (2.9)	0/9	0/9	0/10	7/54 (18)	0/9
Total xylenes	0/7	0/9	0/10	0/11	3/8 (8.5)	0/9	0/9	0/10	10/54 (34)	0/9

Table 6-28. TZW COPCs with HQs > 1.0 in Individual Samples by Facility

	Number of Samples with HQs > 1.0 (maximum HQ)									
		Arkema								
COPC	**ARCO**	**Acid Plant Area**	**Chlorate Plant Area**	**Mobil Oil**	**Gasco**	**Gunderson**	**Kinder Morgan**	**Rhône-Poulenc**	**Siltronic**	**Willbridge**
Trichloroethene	0/7	0/9	0/10	0/11	0/8	0/9	0/9	0/10	2/54 (1,900)	0/9
Petroleum										
Gasoline-range aliphatic hydrocarbons C4-C6	1/9 (1.1)	NA	NA	3/15 (1.2)	5/10 (7.3)	NA	0/10	NA	6/15 (2.0)	0/9
Gasoline-range aliphatic hydrocarbons C6-C8	0/9	NA	NA	0/9	4/10 (4.3)	NA	0/10	NA	3/15 (1.2)	0/9
Gasoline-range aliphatic hydrocarbons C8-C10	0/9	NA	NA	0/9	0/10	NA	0/10	NA	0/15	0/9
Gasoline-range aliphatic hydrocarbons C10-C12	6/9 (35)	NA	NA	6/15 (85)	9/10 (540)	NA	3/10 (6.9)	NA	9/15 (150)	3/9 (3.8)
Gasoline-range aromatic hydrocarbons C8-C10	0/9	NA	NA	0/15	3/10 (2.7)	NA	0/10	NA	0/15	0/9
Other Chemicals										
Cyanide	NA	NA	NA	NA	8[a]/8 (4,400)	NA	NA	NA	26[a]/26 (130)	NA
Perchlorate	NA	0[a,d]/9	5/10 (19)	NA	NA	0/2	NA	NA	NA	NA

[a] One additional sample had a DL greater than the TRV.
[b] An additional two to three samples had DLs greater than the TRV.
[c] An additional four or more non-detect samples had DLs greater than the TRV.
[d] Only samples with non-detected concentrations had HQs > 1.0.

COPC – chemical 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
TRV – toxicity reference value
TZW – transition zone water
VOC – volatile organic compound
Bold identifies HQs > 1.

6.6.5 Evaluation of Ecological Relevance

TZW samples were collected from nearshore areas of the river that include potential habitat for benthic invertebrates, benthic fish, amphibians, and aquatic plants; thus, TZW is considered a complete and significant pathway for these receptors in the CSM for Portland Harbor. These organisms reside in the sediment column or are in contact with the sediment surface. However, benthic organisms have relatively low exposure to porewater (including TZW). Porewater below the oxygenated zone near the sediment surface is basically toxic to burrowing organisms because the decomposition of organic material that settles to the bottom consumes dissolved oxygen, creates an anoxic environment, and releases products that result in the formation of hydrogen sulfide and ammonia (Forbes et al. 1998).

Through a variety of adaptations, benthic organisms can utilize depths beyond the oxygenated zone (up to about 20 cm below the sediment surface) for both refuge and feeding. Very small or newly settled invertebrates occupy the top few millimeters of the sediment-water interface, reflecting both their size and the depth to which oxidized sediment occurs through the molecular diffusion of oxygen from the overlying water column alone. Larger organisms tend to live deeper in the sediment and help oxygenate the surrounding sediments through their feeding and respiration. Burrowing organisms address their need for oxygen through modifications to the sediment environment or behaviors that increase the interchange of oxygen with overlying water. Such mechanisms and behaviors vary by species. Burrowing and tube-dwelling organisms may 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 sediment adjacent to it. These biofilms may decrease the infiltration of the surrounding porewater into the tube or burrow by decreasing 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.

Feeding strategy also affects exposure to porewater contaminants. Filter-feeding organisms depend on the flow of water, extending specialized appendages or structures into the water column to gather food. As an 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 (e.g., *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)

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).

Respiration and feeding mechanisms, as well as other biological activities, reduce the porewater exposure and chemical concentrations to which benthic organisms are exposed in sediment. Low-level exceedances of COPC TRVs in TZW are unlikely to represent an unacceptable risk to benthic organisms living in the sediment because of the dilution or isolation that occurs.

6.6.6 Uncertainties Associated with TZW Evaluation

TZW chemical concentrations are greater than toxicity thresholds developed for surface water; however, in several cases, the applicability of those TRVs for evaluation of effects on benthic invertebrates is questionable. The EPA-selected TRV for DDTs (0.001 µg/L) is based on the protection of reproduction in birds exposed via ingestion of contaminated prey; exposure via this route is not comparable to direct exposure of aquatic organisms to surface water contaminants. To reduce uncertainty, an alternative TRV based only on toxicity to aquatic organisms was developed for the BERA. In accordance with the methods used to establish AWQC, it is appropriate to use a final residue value for establishing the alternative water TRV, provided that it is likely to be lower than a final chronic value (FCV). A tissue-residue-derived water TRV of 0.011 µg/L DDx compounds was calculated by dividing the PCB 10th percentile fish tissue residue LOAEL (1.6 mg/kg ww) by a BAF of 142,960[51] (derived from the DDT AWQC document). This alternative water TRV (0.011 µg/L) is lower than the final acute value (1.1 µg/L) divided by the ACR (65) and is more appropriate than the AWQC for DDx for evaluating direct exposure of organisms to water. Whole water TZW concentrations also exceed the alternative TRV, albeit at a lower magnitude.

Uncertainty is associated with TRVs for several trace elements and nutrients (sodium, potassium, and manganese), VOCs (e.g., ethylbenzene, trichloroethene), and PAHs because they were derived from a limited number of studies and species (TRVs are all based on Tier II or other non-regulatory values). In the case of sodium and potassium, the Tier II value was cited by Suter and Tsao (1996) as being below commonly occurring ambient

[51] A BAF of 142,960 was based on the lipid-normalized BAF (17,870) and anchovy lipid percent (8%) as presented in the DDT AWQC document.

values and not appropriate for use as a screening value. The AWQC that was the source of the iron TRV was 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).

Although there are many anthropogenic sources of metals, almost all of the metals measured in TZW are also common crustal elements in sediment. Barium, iron, and manganese are among the most common metals associated with sediments and were detected in all TZW samples. Aluminum, also a common crustal element, tends to covary with iron, and was frequently detected in TZW.

The contribution of geochemical processes in sediments to the concentrations of selected metals in TZW was extensively evaluated in Appendix C2 of the draft RI. 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 RI) 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.

The comparison of whole water sample results to TRVs is 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 detection limits) or do not exceed their associated TRVs when evaluated on a dissolved basis.

6.6.7 TZW Conclusions

One or more chemicals were detected in almost all TZW samples from the 10 locations; however, TZW concentrations of these chemicals were equally if not more likely controlled by chemical partitioning from sediment rather than transport from upland groundwater at five of the sites (i.e., Kinder-Morgan Linnton, ARCO, Mobil, Gunderson, and Willbridge). Complete groundwater pathways for chemicals were identified at the Arkema, Gasco, Rhône-Poulenc, and Siltronic facilities (as presented in Appendix C2 of the draft RI). At these locations, TZW likely contributes to the concentrations of some contaminants of concern found in sediment.

At all facilities, one or more TZW chemical concentrations were greater than the corresponding TRVs; however many HQs were low (< 3.0). Areas with higher HQs for TPH, PAHs, VOCs, pesticides, perchlorate, and cyanide were generally associated with

known source areas and have been identified as posing unacceptable risks to benthic invertebrates through other lines of evidence.

The toxicity benchmarks (i.e., AWQC) originally selected for DDx compounds in water are not ecologically relevant for evaluation of risks to benthic invertebrates, fish, or plants; however, use of an alternative TRV for DDx compounds based only on toxicity to aquatic organisms still yields HQs > 1.0 in TZW. In most cases, where metals exceed their respective TRVs, the cause is more likely to be geochemical processes that govern partitioning from sediment rather than contribution from upland groundwater. For other substances, ecological relevance of low-level exceedances is limited because of the ability of benthic organisms to limit their exposure to porewater (Section 6.6.5).

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, and water quality. TZW was evaluated but was not used to identify COCs and is therefore not discussed further in the conclusions. Because sediment, tissue, and surface water represent different routes of exposure, each medium was assessed separately. The results were then superimposed to create an overall portrayal of potential benthic risks. COCs were identified for each medium.

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 the COCs were reached by evaluating the weight of the evidence for population- and community-level risks. A higher level of uncertainty is associated with risk predictions based solely on the sediment toxicity LOE as predicted by chemistry because the historical distribution of chemicals in sediment is limited, and sediment samples do not integrate well over a wide area; it is therefore difficult to use sample-by-sample exposure in a population and community-level analysis.

The WOE framework was simple. First, sediment concentrations were mapped,[52] and the maps used to predict site-specific SQG exceedances.[53] Next, the sediment chemistry maps were used (with bioaccumulation models[54]) to predict where organisms might accumulate COCs to concentrations above tissue TRVs. Areas where empirical or NN-interpolated

[52] Mapping was done using a 10-ft-square grid overlaid on the Study Area. Natural neighbors (de Smith et al. 2008) was used to interpolate sediment concentrations.

[53] The low and high SQGs were used to identify three sediment effect categories: Level 1 (below the low SQG), Level 2 (between the high and low SQGs) and Level 3 (above the high SQG). See Section 6.1.1.1.

[54] The analysis used the same bioaccumulation models built for developing the PRGs to be used in the FS.

sediment concentrations exceeded site-specific SQGs, or empirical or predicted tissue concentrations exceeded TRVs were identified as potential benthic risk areas (PBRAs).[55] The results of the empirical toxicity tests within the PBRAs and the sediment chemistry at these stations were used to evaluate the potential unacceptable risks identified based on site-specific SQG exceedances to arrive at final risk conclusions for the sediment LOE. Tissue residue data were mapped as points to allow a visual assessment of concordance across lines of evidence. Water TRV exceedances were not displayed on maps but were considered along with sediment SQG and tissue TRV exceedances; they were found to co-occur with SQG exceedances.

The WOE for benthic population and community-level risks was assessed within PBRAs. Areas outside the PBRAs, which pose no unacceptable risk to benthic organisms, were assumed to pose no unacceptable risk to the benthic community or populations.

Three types of factors were considered in extrapolating from organism-level effect threshold exceedances to population and community-level risk conclusions:

Spatial extent of PBRAs

Magnitude 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 COCs is provided next, after which the framework just described is applied to draw conclusions about risks to the benthic assessment endpoints.

6.7.1 Summary of Benthic Invertebrate COCs

Table 6-29 presents a summary of the benthic invertebrate COCs. Twenty-eight COPCs (individual chemicals, sums, or totals) were identified as COCs based on the site-specific SQG, tissue TRV, and surface water TRV exceedances. COCs do not necessarily pose unacceptable risk from the standpoint of the assessment endpoints because they are based on point-by-point exceedances of organism-level effect thresholds, not on population or community-level endpoints, and their spatial scale is not relevant for assessing risks to populations or communities.

[55] The qualifier "potential" is used because the measurement endpoints used to delineate the PBRAs are organism-level endpoints, and the assessment endpoints are the benthic community and populations. Population-level processes may compensate for adverse effects on individuals. For example, localized TRV exceedances do not indicate population- or community-level risk.

Table 6-29. Benthic Chemicals of Concern

	Line of Evidence Resulting in COC			
Analyte	**Site-Specific SQGs**	**Tissue Residue TRV – Empirical**	**Tissue Residue TRV – Predicted**	**Surface Water TRV**
Metals				
Aluminum	○[a]	○[b]	◉[c]	●
Arsenic	◉	●	◉[c]	◉
Cadmium	●	◉	◉[c]	◉
Copper	●	●	◉[c]	◉
Mercury	●	◉	◉[c]	◉
Silver	●	◉	○	◉
Zinc	◉	●	◉[c]	●
TBT				
Butyltin	○[a]	◉	◉[c]	●
Tributyltin	○[a]	●	●	◉
PAHs				
Benzo(a)anthracene	○[d]	○	○	●
Benzo(a)pyrene	○[d]	○	○	●
Naphthalene	○[d]	○	○	●
Total benzofluoranthenes	●	○	○	◉
Total HPAHs	●	○	○	◉
Total LPAHs	●	○	○	◉
VOCs				
Ethylbenzene	○[e]	○	○	●
Trichloroethene	○[e]	○	○	●
SVOCs				
Benzyl alcohol	●	◉	○	◉
Carbazole	●	◉	○	◉
Phenols				
Phenol	●	◉	○	◉
Phthalates				
BEHP	○[a]	●	◉[c]	●
PCBs				
Total PCBs	●	●	●	●
Pesticides				
4,4′-DDD	○[d]	●	◉	◉
delta-HCH	●	◉	○	◉

Table 6-29. Benthic Chemicals of Concern

Analyte	Line of Evidence Resulting in COC			
	Site-Specific SQGs	Tissue Residue TRV – Empirical	Tissue Residue TRV – Predicted	Surface Water TRV
Dieldrin	●	⭘	⭘	⭘
Endrin	●	⭘	○	⭘
Endrin ketone	●	⭘	○	⭘
Total DDx	●	●	●	●

○ – not evaluated/insufficient data
● – potential unacceptable risk
⭘ – evaluated; no unacceptable risk
[a] No difference between hit and no-hit concentrations.
[b] Identified as COPC in the SLERA; not evaluated any further in the BERA.
[c] No relationship between sediment concentrations and tissue concentrations.
[d] Chemicals included in the sums.
[e] Low detection frequency (<30).

BEHP – bis(2-ethylhexyl) phthalate
COC – chemical of concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
HCH – hexachlorocyclohexane
HQ – hazard quotient
HPAH – high-molecular-weight polycyclic aromatic hydrocarbon
LPAH – low-molecular-weight polycyclic aromatic hydrocarbon
NA – not applicable (COC not evaluated using this LOE)
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)
TBT – tributyltin

Benthic invertebrate COCs were organized into four general chemical groups to evaluate risk conclusions: metals and TBT, organic compounds (including PAHs, SVOCs, phenols, and phthalates), PCBs, and pesticides. Two VOCs were identified as COCs for one site in the Study Area based on the surface water LOE; these chemicals are not discussed further in the risk conclusions because they were site-specific COCs (for further details see Section 6.5). Aluminum is also not discussed further because aluminum concentrations in background areas indicate aluminum is not attributed to anthropogenic sources in the Study Area (see Section 6.5.4.3).

6.7.2 Weight of Evidence

Map 6-35 presents the PBRAs that were identified using the following WOE approach:

PBRAs were identified based on NN-interpolated sediment concentrations that exceeded site-specific SQGs or sediment thresholds[56] back-calculated from tissue TRVs using receptor-specific BSARs or a bioaccumulation model. The chemical concentrations

[56] See Section 6.4.2.2 for description of the development of sediment thresholds that were used to predict tissue residues for selected benthic receptors and bioaccumulative chemicals.

associated with empirical toxicity data within each PBRA were examined to evaluate the uncertainties associated with potential risk predictions based on sediment toxicity. The evaluation encompassed several steps; outcomes were used to address uncertainties regarding predicted unacceptable benthic risks within a PBRA:

- The paired empirical toxicity test and chemistry results associated with each PBRA were identified. The data were sorted by toxicity status: toxic (i.e., Level 3 exceedance) versus non-toxic (Level 0 or 1).
- For each chemical that exceeded its respective SQG for a given PBRA, characteristic chemical values of the toxic and non-toxic empirical sample subsets were identified:
- The maximum concentration of the non-toxic samples
- The lowest concentration that exceeded the SQG at a toxic station, where all toxic samples exceeded the SQG
- If toxic sample concentrations were distributed both above and below the SQG, the SQG was retained as the best estimate of the minimum toxic sample concentration for a given chemical.
- If all the bioassays within the PBRA were toxic, then the prediction of probable benthic risk was retained.
- If all bioassays within a PBRA were non-toxic, then the maximum concentration associated with the non-toxic samples was used to identify subareas that are unlikely to represent benthic risks.[57] If the maximum concentration within the PBRA occurred at a non-toxic station, the entire PBRA was identified as unlikely to represent an unacceptable benthic risk.
- If all toxic samples exceeded the SQG and the toxic and non-toxic sample chemistry overlapped (i.e., the minimum concentration in the toxic samples was less than the maximum concentration associated with the non-toxic samples), then both the minimum toxic and maximum non-toxic sample concentrations were used to refine risks within the PBRA.
- The area with chemical concentrations below the minimum toxic sample concentration was considered unlikely to represent an unacceptable risk to the benthic community.
- The area where the interpolated chemistry was greater than the maximum non-toxic sample concentration continued to be an area of unacceptable benthic risk.

[57] To identify subareas that may not represent a risk, the maximum concentration was re-applied to the NN-interpolated data within the specific PBRA and recontoured. Sediment chemical concentrations that exceeded the maximum non-toxic (≤ low reference thresholds defined by the reference envelope approach) sample concentration were considered to represent a risk.

- The area where the chemistry was between the minimum toxic sample concentration (or SQG) and the maximum non-toxic sample concentration was considered uncertain.
- PBRAs that did not contain empirical bioassays were retained but were considered uncertain.

Also shown on Map 6-35 are locations where sediment was non-toxic meaning that low generic SQGs with acceptable predictive reliability (i.e., TELs, TECs or ERLs) were not exceeded.[58] Map 6-36 presents exceedances of the sediment thresholds derived from the empirical tissue TRVs and the empirical tissue TRV exceedances. The Study Area was divided into three river segments to discuss the distribution of COCs, risks, and uncertainties: the lower segment (RM 1.9 to RM 5.0), the middle segment (RM 5.0 to RM 9.0 [including Swan Island Lagoon to RM 9.2]), and the upper segment (RM 9.0 to RM 11.8).

6.7.2.1 Lower River Segment

Toxicity tests suggested limited organism-level effects in the lower segment of the Study Area. Level 3 SQG exceedances occurred in International Slip and at three locations along the western shoreline between RM 3.7 and RM 4.8.

Most COCs were identified on the basis of an SQG exceedance. Toxicity was predicted in the lower river segment based on exceedance of the SQGs for cadmium, PAHs, several SVOCs (benzyl alcohol, carbazole, phenol), total PCBs, endrin ketone, and total DDx. The tissue-residue LOE identified arsenic, copper, zinc, TBT, and total PCBs as COCs and the surface water LOE identified BEHP and total PCBs as a COCs based on one sample. Table 6-30 presents the key COCs in the lower river segment and the LOEs identifying these chemicals as COCs.

[58] Specifically, sediment chemistry was compared to the three sets of low-level SQGs. Stations where sediment chemical concentrations were below all the SQGs in any one of the three sets are indicated on the map. This analysis was conducted only at sediment sampling stations, not at interpolated grid points.

Table 6-30. Summary of Key Benthic Invertebrate COCs for the Lower River Segment and Uncertainties Across LOEs

Chemical	LOE	EF or HQ	RM	Key Uncertainties	Risk Conclusions
Metals					
Arsenic	Tissue residue	1.1	RM 3.7, east	Using the tissue-residue LOE for metals is highly uncertain because benthic invertebrates regulate metals, confounding inter-species extrapolation, and the tissue-residue TRV for arsenic cannot reliably predict toxicity. Limited spatial extent of exceedances does not equate to population or community-level risk. HQ was low.	Based on a WOE approach, arsenic poses negligible risks to the benthic community or populations because of a single exceedance of the TRV, low magnitude of the HQ, high uncertainty with respect to metals toxicity based on the tissue LOE, and two LOEs (site-specific SQG and surface water) that identify no unacceptable risk associated with arsenic.
Cadmium	SQGs	1.6 – 2.8	RM 4.6 (Terminal 4)	Limited spatial extent of exceedances does not equate to population or community-level risk. HQs were low.	Based on a WOE approach, cadmium poses negligible risks to the benthic community or populations because of low magnitude, low frequency, and limited spatial extent of EFs > 1.0 and two LOEs (tissue residue and surface water) that identify no unacceptable risk associated with cadmium. No bioassay station was located within the area with SQG exceedances.
Copper	Tissue residue	1.0 – 2.2	RM 1.9 – RM 4.8	Using the tissue-residue LOE for metals is highly uncertain because benthic invertebrates regulate metals, confounding inter-species extrapolation, and the tissue-residue TRV for copper cannot reliably predict toxicity. HQs were low (1.0 to 2.2).	Based on a WOE approach, copper poses negligible risks to the benthic community or populations because of low magnitude of HQs > 1.0, high uncertainty with respect to metals toxicity based on the tissue LOE, and two LOEs (site-specific SQG and surface water) that identify no unacceptable risk associated with copper.
Silver	SQG	1.4, 2.1	RM 4.6 (Terminal 4)	Limited spatial extent of exceedances does not equate to population or community-level risk. HQs were low	Based on a WOE approach, silver poses negligible risks to the benthic community or populations because of low magnitude, low frequency, limited spatial extent of EFs > 1.0, and two LOEs (tissue residue and surface water) that identify no unacceptable risk associated with silver. No bioassay station was located within the area with SQG exceedances.
Zinc	Tissue residue	1.0 – 2.0	RM 1.9 – RM 4.8	Using the tissue-residue LOE for metals is highly uncertain because benthic invertebrates regulate metals, confounding inter-species extrapolation, and the tissue-residue TRV for zinc cannot reliably predict toxicity. HQs were low (1.0 to 2.0).	Based on a WOE approach, zinc poses negligible risks to the benthic community or populations because of the low magnitude of HQs > 1.0, high uncertainty with respect to metals toxicity based on the tissue LOE, and two LOEs (site-specific SQG and surface water) that identify no unacceptable risk associated with zinc.
Butyltins					
Tributyltin	Tissue residue	149	RM 3.7, east	Exceedance based on predicted tissue concentration; sediment concentration range used to develop BSAR was an order of magnitude lower than concentration on which the predicted tissue exceedance was based. TRV likely overpredicts effects to invertebrates.	Based on a WOE approach, TBT poses negligible risks to the benthic community or populations because of the limited spatial extent, high uncertainty associated with predicted tissue concentration, and two LOEs (site-specific SQG and surface water) that identify no unacceptable risk associated with TBT.
Organics					
PAHs	SQGs	1.5 – 7.2	RM 2.8, east; RM 4.6, east; RM 4.9, east;	Only three single-sample locations exceeded an SQG for PAHs (total benzofluoranthenes or total LPAH). Limited spatial extent of exceedances does not equate to population or community-level risk. HQ was low.	Based on a WOE approach, PAHs pose negligible risks to the benthic community or populations in this segment of the river because of the low magnitude, low frequency, or limited spatial extent of EFs/HQs > 1.0. One LOE (surface water) identified no unacceptable risk associated with PAHs. PAHs were not assessed using the tissue-residue LOE. No bioassay station was located within the areas with SQG exceedances.
Benzyl alcohol	SQGs	2.5	RM 3.7	Detected concentration exceeded SQG at only one location, which does not equate to a population or community-level risk. Quantification of benzyl alcohol uncertain.	Based on a WOE approach, benzyl alcohol poses negligible risks to the benthic community or populations because of low magnitude, low frequency, limited spatial extent of EFs > 1.0, and two LOEs (tissue residue and surface water) that identify no unacceptable risk associated with benzyl alcohol. One bioassay station was located in the area with SQG exceedance; the bioassay station was toxic based on all four endpoints. Two other chemicals, phenol and total PCBs, also exceeded their SQGs at this station.

Table 6-30. Summary of Key Benthic Invertebrate COCs for the Lower River Segment and Uncertainties Across LOEs

Chemical	LOE	EF or HQ	RM	Key Uncertainties	Risk Conclusions
Carbazole	SQGs	2.5	RM 4.7	Detected concentrations exceeded SQG at only one location, which does not equate to a population or community-level risk.	Based on a WOE approach, carbazole poses negligible risks to the benthic community or populations because of the low magnitude, low frequency, and limited spatial extent of EFs > 1.0 and two LOEs (tissue residue and surface water) that identify no unacceptable risk associated with carbazole. No bioassay station was located within the area with SQG exceedances.
Phenol	SQGs	3.8, 5.7	RM 3.7	Detected concentrations exceeded SQG at only one location, which does not equate to a population or community-level risk. Quantification of phenol uncertain.	Based on a WOE approach, phenol poses negligible risks to the benthic community or populations because of the low magnitude, low frequency, and limited spatial extent of EFs > 1.0 and two LOEs (tissue residue and surface water) that identify no unacceptable risk associated with phenol. One bioassay station was located in the area with SQG exceedance; the bioassay station was toxic based on all four endpoints. Two other chemicals, benzyl alcohol and total PCBs, exceeded their SQGs at this station.
BEHP	Surface water	2.3	RM 3.9 (transect location)	TRV exceeded during only one sampling event for this chemical. Exceedance based on analysis of a whole water sample. Sediment concentrations of BEHP in this segment of the river lower than those in other segments.	Based on a WOE approach, BEHP poses negligible risks to the benthic community or populations because of the low magnitude, low frequency, and limited spatial extent of HQ > 1.0 and two LOEs (site-specific SQG and surface water) that identify no unacceptable risk associated with BEHP.
PCBs					
Total PCBs	SQGs	1.6 – 7.0	RM 2.1 – 2.5, RM 3.8; RM 4.0; RM 4.4; RM 4.6	Spatial distribution of exceedances does not pose population or community-level risk, but might pose unacceptable risk to local subpopulations given the aggregation of exceedances in three areas. Extrapolating to community-level risk is even more difficult than extrapolating to population-level. Community-level risk would occur if population-level risks were significantly disrupting the performance of specific ecological functions. Whether localized community-level risk might be occurring is unknown.	Possible unacceptable risks to the benthic community along the east/north shoreline between RM 2.1 and RM 2.4 based on bioaccumulation risk. Unacceptable risks were predicted over a greater area based on SQG exceedances; however, the empirical toxicity data did not identify unacceptable risk based on six non-toxic bioassays. Possible unacceptable risks to the benthic community on the south side of in International Slip based on bioaccumulation risk, sediment concentrations greater than the maximum concentration at the non-toxic bioassay stations, and toxic bioassay stations. The slips of Terminal 4 were identified as possible unacceptable risk areas based on SQG exceedances. The risk is uncertain because no bioassays were performed in these areas, and there is no predicted bioaccumulation risk. A definitive assessment of possible unacceptable risks would require better population- and community-level measurement endpoints than those that were stipulated in the BERA Problem Formulation (Attachment 2). Negligible risks to the benthic community at RM 4.0 based on non-toxic bioassay stations in the area with SQG exceedances and no predicted bioaccumulation risk.
	Tissue residue	1.1 – 5.1	RM 2.2, east; RM 3.7, east	Empirical tissue residue HQs > 1.0 were limited spatially to between RM 2.2 east and RM 3.7 east. Laboratory exposures under static conditions resulted in higher HQs than did field-collected samples. Most HQs for both empirical and predicted tissues were less than 2.0.	
	Surface water	1.1	RM 3.7, east	No samples exceed an alternative surface water TRV that is based on direct exposure of aquatic organisms to surface water.	

Table 6-30. Summary of Key Benthic Invertebrate COCs for the Lower River Segment and Uncertainties Across LOEs

Chemical	LOE	EF or HQ	RM	Key Uncertainties	Risk Conclusions
Pesticides					
Endrin ketone	SQG	1.4	RM 3.7	Single exceedance of SQG does not represent a population or community-level risk.	Based on a WOE approach, endrin ketone poses negligible risks to the benthic community or populations in this river segment because of the low magnitude of EF, a single sample with an EF > 1.0, and two LOEs (tissue residue and surface water) that identify no unacceptable risk associated with endrin ketone. No bioassay station was located within the area with SQG exceedance.
DDx compounds[a]	SQG	1.1	RM 3.4	Single exceedance of SQG does not represent a population or community-level risk.	Based on a WOE approach, DDx compounds pose negligible risks to the benthic community or populations in this river segment because of the low magnitude of EF/HQ, a single sample with an EF/HQ > 1.0, and one LOE (tissue residue) that identify no unacceptable risk associated with DDX compounds. No bioassay station was located within the area with SQG exceedance.
	Surface water	1.2-20	RM 2.0 - 4.5	One sample at RM 2.0 exceeds alternative water TRV that is protective of direct exposure of aquatic organisms to surface water (HQ = 1.8); however, this sample is N-qualified. The indication of sample 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.	

[a] Includes the following DDT COCs: 2,4′-DDD, 2,4′-DDT, 4,4′-DDD, 4,4′-DDT, and total DDx.

BEHP – bis(2-ethylhexyl) phthalate
BERA – baseline ecological risk assessment
COC – chemical of concern
DDD – dichlorodiphenyldichloroethane
DDT – dichlorodiphenyltrichloroethane
EF – exceedance factor
HPAH – high-molecular-weight polycyclic aromatic hydrocarbon
HQ – hazard quotient
J – estimated concentration
LOE – line of evidence
LPAH – low-molecular-weight polycyclic aromatic hydrocarbon
N – presumptive evidence of a compound
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
RM – river mile
SQG – sediment quality guideline
T – value calculated or selected from multiple results

Metals exceedances of SQGs had a limited distribution (primarily associated with Terminal 4). TBT and arsenic were greater than their tissue-residue TRVs at one location each based on predicted tissue and empirical tissue, respectively. The TBT TRV (0.15 mg/kg) is 20 times lower than the sublethal effect threshold (3 mg/kg) proposed for protection of juvenile salmonid prey (Meador 2000a), which was based on reduced growth in multiple species (Meador 2000b). Use of EPA's TRV probably caused TBT toxicity to invertebrates to be overpredicted. Copper and zinc frequently exceeded their respective tissue TRV in clams and crayfish, but this LOE is considered weak because of the ability of aquatic organisms to regulate metals. As an example, copper is the basis of the respiratory protein in many crustaceans and molluscs; these animals readily pick up and excrete copper (Kaestner 1980). An evaluation of the studies used to derive the TRVs for copper and zinc indicated that a TRV for clams and crayfish (primarily mollusks and a few shrimp, as crayfish are not well represented in toxicity studies) would likely be higher (the 10th percentile from the SSD would be 12.2 mg/kg for copper [vs. 7.7 mg/kg] and 46.7 mg/kg [vs. 24.1] for zinc) and only a few samples would exceed these alternative TRVs. A single sample exceeded the SQG for PAHs at RM 2.8 (eastern shoreline). Benzyl alcohol and carbazole only exceeded their respective SQGs at one, and two locations, respectively, in this river segment. PCBs exceeded its SQG and tissue TRV in samples associated with the eastern shoreline between RM 2.1 to RM 2.4 and from RM 3.7 to RM 4.6. DDx exceedance of the SQG was limited to a single location.

In summary, five areas below RM 5.0 were identified as posing localized risk to subpopulations of benthic invertebrates and possible unacceptable benthic community risk. Three areas are based on PCB exceedances of the SQG and tissue TRV. One is along the eastern shoreline between RM 2.1 – RM 2.4, another is on the south side of International Slip, and the third is within the Terminal 4 slips. One other possible risk area is associated with LPAHs at RM 2.8. Terminal 4 slips, which are PCB risk areas, are also characterized by PAH exceedances of the SQGs. One other area at RM 4.8 that is defined by empirical toxicity results alone (i.e., no chemical exceedances) is an uncertain benthic risk area.

The BERA is inconclusive about whether these areas pose unacceptable risks to benthic populations or the benthic community. The answer hinges on issues of appropriate population and community-level measurement endpoints that would not be easily resolved. The areas will be identified as posing unacceptable risk to other non-benthic receptors, so it seems prudent to leave this as an unresolved uncertainty.

6.7.2.2 Middle River Segment

The majority of the toxicity tests with Level 3 exceedances used sediment collected from this segment of the Study Area, particularly between RM 6.0 and RM 7.4 along the western shoreline.

Most COCs were identified on the basis of SQG or tissue TRV exceedance. Toxicity was predicted in the middle river segment based on exceedances of the SQGs for metals (cadmium, copper, mercury, and silver), PAHs, several SVOCs (benzyl alcohol, carbazole, phenol), total PCBs, and pesticides (delta-HCH, dieldrin, endrin, endrin ketone, and DDx

compounds). The tissue-residue LOE identified arsenic, copper, zinc, TBT, BEHP, total PCBs, and DDx compounds as COCs. The surface water LOE identified PAHs, BEHP, and total PCBs as COCs. Table 6-31 presents the key COCs in the middle river segment and the LOEs identifying these chemicals as COCs.

Site-specific SQGs for metals were exceeded primarily at the mouth of Swan Island Lagoon. Those for PAHs and carbazole were exceeded adjacent to historical petroleum source areas between RM 5.1 and RM 6.8. Quantification of phenol and benzyl alcohol is uncertain and detected concentrations exceeded their SQGs at one and two locations, respectively. PCBs exceeded the SQG at numerous locations associated with historical industrial facilities between RM 6.8 and RM 7.4 (east and west banks), in Swan Island Lagoon, and between RM 8.8 and 9.3 (western shoreline). DDx exceeded SQGs primarily along the western shoreline between RM 6.5 and RM 7.5.

The arsenic tissue TRV was exceeded at only one location, whereas copper and zinc tissue residues exceeded their respective TRVs throughout this segment of the Study Area (and the rest of the Study Area as well, indicating that the copper and zinc benthic tissue TRVs are probably too low as effect thresholds for clam and crayfish tissue residues). Even organism-level effects of low-level exceedances of tissue TRVs for metals are highly uncertain because of aquatic organisms' ability to regulate metals. TBT tissue exceedances were based almost entirely on predicted tissue concentrations at the mouth of Swan Island Lagoon. The BEHP tissue TRV was exceeded at only one location. PCBs and DDx compounds exceeded their respective TRVs based on both empirical and predicted tissue concentrations. Empirical exceedances were few and were primarily based on laboratory exposures in static renewal bioaccumulation tests.

Benzo(a)anthracene, benzo(a)pyrene, naphthalene, and several VOCs exceeded their respective water TRVs in samples collected during several monitoring events between RM 6.1 and RM 6.4. Detected concentrations of BEHP exceeded its water TRV once, at one location. PCBs were detected above their water TRV once at one location. Total DDx repeatedly exceeded its TRV at multiple locations sampled between RM 6.9 and RM 7.6; however, the TRV was based on protection of bird reproduction, which is not relevant to the benthic risk assessment. When an alternative water TRV based on direct contact of aquatic organisms with surface water was used, there were no TRV exceedances.

In summary, the benthic assessment found evidence of unacceptable risk to benthic populations and the benthic community in areas of the middle river segment from exposure to PAHs, PCBs, and DDx at locations described in Table 6-36 and shown on Map 6-35 based on predicted sediment toxicity and tissue-residue LOEs. Level 3 effects based on empirical toxicity tests corroborated predictions of risk, particularly between RM 6.1 and RM 6.6 and between RM 7.0 and RM 7.5. Risk areas are most often associated with nearshore locations. An exception to this distributional pattern is three mid-channel risk areas between RM 5.1 to RM 5.8 defined by PAH exceedances of SQGs.

Table 6-31. Summary of Key Benthic Invertebrate COCs for the Middle River Segment and Uncertainties Across LOEs

Chemical Group	LOE	EF or HQ	RM	Key Uncertainties	Risk Conclusions
Metals					
Arsenic	Tissue residue	1.5	RM 7.4, east	Using the tissue-residue LOE for metals is highly uncertain because benthic invertebrates regulate metals, confounding inter-species extrapolation, and the tissue-residue TRV for arsenic cannot reliably predict toxicity. Limited spatial extent of exceedances does not equate to population or community-level risk. HQ was low.	Based on a WOE approach, arsenic poses negligible risks to the benthic community or populations because of a single exceedance of the TRV, the low magnitude of the HQ, high uncertainty with respect to metals toxicity based on the tissue LOE, and two LOEs (site-specific SQG and surface water) that identify no unacceptable risk associated with arsenic.
Cadmium	SQGs	1.2-13	RM 8.1, west; RM 8.8, Swan Island Lagoon (at RM 9.1)	Limited spatial extent of exceedances does not equate to population or community-level risk.	Based on a WOE approach, cadmium poses negligible risks to the benthic community or populations in this segment of the river based on the low frequency and limited spatial extent of EFs > 1.0 and two LOEs (tissue residue and surface water) that identify no unacceptable risk associated with cadmium. Only one bioassay station was located within the areas with SQG exceedances; the station at RM 8.8, west bank, was toxic based on all four endpoints.
Copper	SQGs	1.1 – 1.9	RM 5.9 – RM 8.3	Limited spatial extent of exceedances does not equate to population or community-level risk.	Based on a WOE approach, copper poses negligible risks to the benthic community or populations in this segment of the river because of the low magnitude of EFs/HQs > 1.0, high uncertainty with respect to metals toxicity based on the tissue LOE, and one LOE (surface water) that identify no unacceptable risk associated with copper. Only one bioassay station was located within the areas with SQG exceedances; the station was toxic based on the *Hyalella* mortality and growth endpoints.
	Tissue residue	1.0 – 2.6	RM 5.2 – RM 9.1 (including Swan Island Lagoon)	Limited spatial extent of exceedances does not equate to population or community-level risk. Tissue residue LOE highly uncertain because benthic invertebrates regulate metals, confounding inter-species extrapolation, and the tissue-residue TRV for copper cannot reliably predict toxicity. HQs were low (1.0 to 2.6)	
Mercury	SQGs	1.1 – 90	RM 5.7 – RM 9.1 (including Swan Island Lagoon)	SQG exceedances were low (1.1 to 2.3), except at two locations along the east bank at RM 5.7 and RM 6.7, with EFs of 6.7 and 90, respectively.	Based on a WOE approach, mercury poses negligible risks to the benthic community or populations, except in Willamette Cove and at the mouth of Swan Island Lagoon because of the low frequency, limited spatial extent of EFs > 1.0 and two LOEs (tissue residue and surface water) that identify no unacceptable risk associated with mercury. Only one bioassay station was located within the areas with SQG exceedances; the station at RM 8.8, west bank, was toxic based on all four endpoints. Unacceptable risks to the benthic community are likely in Willamette Cove based on a high EF and elevated concentrations in the PBRA as compared with the maximum concentrations in the non-toxic bioassay station and at the mouth of Swan Island Lagoon based on an elevated EF.
Silver	SQGs	1.1 – 8.6	RM 5.6 – RM 9.2 (including Swan Island Lagoon)	Limited spatial extent of exceedances does not equate to population or community-level risk.	Based on a WOE approach, silver poses negligible risks to benthic community or populations, except for potential risk to the benthic community at the southern end of Swan Island Lagoon because of the low magnitude, low frequency, and limited spatial extent of EFs > 1.0 and two LOEs (tissue residue and surface water) that identify no unacceptable risk associated with silver. Only one bioassay station was located within the areas with SQG exceedances; the station at RM 8.8, west bank, was toxic based on all four endpoints.
Zinc	Tissue residue	1.0 – 2.2	RM 5.5 – RM 9.0 (including Swan Island Lagoon)	Limited spatial extent of exceedances does not equate to population or community-level risk. Tissue residue LOE highly uncertain because benthic invertebrates regulate metals, confounding interspecies extrapolation, and the tissue-residue TRV for zinc cannot reliably predict toxicity. HQs were low.	Based on a WOE approach, zinc poses negligible risks to the benthic community or populations because of the low magnitude of HQs > 1.0, high uncertainty with respect to metals toxicity based on the tissue LOE, and two LOEs (site-specific SQG and surface water) that identify no unacceptable risk associated with zinc.
TBT	Tissue residue	1.1 – 46.9	RM 5.4 – RM 8.9 (including Swan Island Lagoon)	HQs ranging from 1.1 to 46.9 based on primarily predicted tissue concentrations (empirical samples from only one location in this segment of the Study Area exceeded the TRV). TRV probably overpredicts effects on invertebrates. Use of alternative TRV (3 mg/kg) would result in only a single exceedances of the TRV.	Based on a WOE approach, TBT poses negligible risks to the benthic community or populations in this segment of the river, except at the mouth of Swan Island Lagoon, because of low magnitude, frequency of HQs > 1.0, and two LOEs (site-specific SQG and surface water) that identify no unacceptable risk associated with TBT. The mouth of Swan Island Lagoon represents an area of potential effects on the benthic community based on elevated HQs, primarily based on predicted tissue concentrations.

Table 6-31. Summary of Key Benthic Invertebrate COCs for the Middle River Segment and Uncertainties Across LOEs

Chemical Group	LOE	EF or HQ	RM	Key Uncertainties	Risk Conclusions
Organics					
PAHs	SQGs	1.1 – 160	RM 5.4 – RM 6.8, west	EFs of 1.1 to 160 were for three PAH sums (total benzofluoranthenes, LPAHs, and HPAHs) at locations along the west bank near known historical sources. LPAHs had majority of EF > 1.0 and the highest EF.	Probable benthic risks along the west bank between RM 5.1 and RM 6.8 based on the magnitude, frequency, and spatial extent of EFs > 1.0. Surface water exceedances are in known source areas and confirm the likelihood of unacceptable benthic risks based on sediment contamination. PAHs were not assessed using the tissue-residue LOE. Fifteen bioassays stations were located within the areas with SQG exceedances; four of these stations were non-toxic, four were toxic based on all four endpoints, and seven were toxic based on 1 to 3 endpoints.
	Surface water	1.4 – 50	RM 6.1, west; RM 6.3, west; RM 6.4, west	EFs of 1.4 to 50 were for three individual PAHs (benzo[a]anthracene, benzo[a]pyrene, and naphthalene) at locations along the west bank near known historical sources.	
Benzyl alcohol	SQGs	2.4, 6.8	RM 5.5 and 5.6	Detected concentration exceeded SQG at only two locations; quantification of benzyl alcohol is uncertain.	Based on a WOE approach, benzyl alcohol poses negligible risks to the benthic community or populations because of low frequency, limited spatial extent of EFs > 1.0, and two LOEs (tissue residue and surface water) that identify no unacceptable risk associated with benzyl alcohol. No bioassay stations at the two locations with SQG exceedances.
Phenol	SQGs	1.2	RM 8.4 (Swan Island Lagoon)	Single exceedance at one location does not represent a population- or community-level effect. Quantification of phenols is uncertain.	Based on a WOE approach, phenol poses negligible risk because of a single low-magnitude EF > 1.0 and two LOEs (tissue residue and surface water) that identify no unacceptable risk associated with phenol. No bioassay was performed at the location with SQG exceedance.
Carbazole	SQGs	1.1 – 29	RM 5.6 – RM 6.5	EFs were low (<3.0), except at seven locations with EFs between 7.5 and 29. Quantification of carbazole is uncertain.	Based on a WOE approach, carbazole poses negligible risks to the benthic community or populations because of low frequency, limited spatial extent of EFs > 1.0 and two LOEs (tissue residue and surface water) that identify no unacceptable risk associated with carbazole. Six bioassay stations were located within the areas with SQG exceedances; three of these stations were toxic based on all four endpoints, and three were toxic based on 1 to 2 endpoints. Areas with higher EFs co-occur with LPAH exceedances.
BEHP	Tissue residue	2.8	RM 8.8, east	One HQ was > 1.0 based on laboratory-exposed clam tissue concentration. The LOAEL TRV is based on two toxicological studies.	Based on a WOE approach, BEHP poses negligible risks to the benthic community or populations in the middle river segment because of low magnitude and frequency of EFs/HQs > 1.0, and one LOE (site-specific SQG) that identify no unacceptable risk associated with BEHP.
	Surface water	1.2	RM 6.7, west	Low HQ (1.2) at one location during high-flow event.	
PCBs					
Total PCBs	SQGs	1.2 – 62	RM 6.7 to 9.2	EFs were low (< 6.5) except at one location with EF of 62. Extrapolating to community-level risk is even more difficult than extrapolating to population-level. Community-level risk would occur if population-level risks were significantly disrupting the performance of specific ecological functions.	Probable benthic risks at the mouth of and in localized areas of Swan Island Lagoon and at RM 8.8 based on predicted bioaccumulation risks and sediment toxicity data. Risks were predicted over a greater area within Swan Island Lagoon based on SQG exceedances; however, the empirical toxicity data did not identify unacceptable risk based on three non-toxic bioassay stations in Swan Island Lagoon.
	Tissue residue	1.1 – 20.4	RM 6.7, east; RM 6.9, west; Swan Island Lagoon; RM 8.8, west	Empirical tissue residue HQs > 1.0 were limited spatially. Laboratory exposures under static conditions resulted in higher HQs than field-collected samples. The predicted tissue HQs exceeded the empirical HQs at similar locations by a factor of 2 to 3. The majority of the HQs for both empirical and predicted tissues were less than 2.0.	Risks from total PCBs between RM 6.9 and RM 7.5 is unlikely based on sediment concentrations less than the maximum concentration at non-toxic bioassay stations. PCB co-occurred with total DDx, which was associated with unacceptable risk in this area.
	Surface water	1.2	RM 6.7	No samples exceeded an alternative water TRV based on direct exposure of aquatic organisms to surface water.	
Pesticides					
delta-HCH	SQGs	1.1 – 2.3	RM 6.2; RM 6.9; RM 7.2 –	Most reported values based on detection limits; only six detected	Based on a WOE approach, delta-HCH poses negligible risks to the benthic community or populations

Table 6-31. Summary of Key Benthic Invertebrate COCs for the Middle River Segment and Uncertainties Across LOEs

Chemical Group	LOE	EF or HQ	RM	Key Uncertainties	Risk Conclusions
			RM 7.3	concentrations. Limited spatial extent of exceedances does not equate to a population or community-level risk.	because of low magnitude, frequency, number of locations with EFs > 1.0 and two LOEs (tissue residue and surface water) that identify no unacceptable risk associated with delta-HCH. Two bioassays were located within the areas with SQG exceedances; one of these stations was non-toxic and the other station was toxic based on all four endpoints. PAHs, total DDx, and carbazole exceeded their SQGs at this station.
Dieldrin	SQGs	1.3, 17	RM 8.3; RM 8.8	Most reported values based on detection limits; only three detected concentrations. Limited spatial extent of exceedances does not equate to a population or community-level risk.	Based on a WOE approach, dieldrin poses negligible risks to the benthic community or populations because of the low magnitude, frequency, and number of locations with EFs > 1.0 and two LOEs (tissue residue and surface water) that identify no unacceptable risk associated with dieldrin. Only one bioassay station was located within the areas with SQG exceedances; the station at RM 8.8, west bank, was toxic based on all four endpoints.
DDx compounds[a]	SQGs	1.1 – 73	RM 5.6 – RM 7.7; RM 8.8	NJ qualified data included in the assessment.	Probable benthic risks at RM 6.8 to RM 7.5 based on the magnitude, frequency, and spatial extent of EFs/HQs > 1.0 and two LOEs (site-specific SQG and tissue-residue).
	Tissue residue	1.1 – 10	RM 6.9 – RM 7.6, west	Empirical tissue concentrations based on laboratory exposures exceeded the TRV, whereas field-collected organisms did not. Predicted tissues concentrations overestimate empirical tissue concentrations in this area by a factor of 2 to 3.	DDx compounds are unlikely to contribute to unacceptable risks between RM 6.2 and RM 6.6 and RM 8.8 based on sediment concentrations less than the maximum concentration at non-toxic bioassay stations. Between RM 6.2 and RM 6.6 DDx compounds in sediments co-occurred with PAHs, which was associated with unacceptable risk in this area.
	Surface water	1.2 - 9.8	RM 5.5 – RM 8.6	No samples exceed an alternative water TRV that is based on direct exposure of aquatic organisms to surface water.	
Endrin	SQGs	1.2 - 1.5	RM 6.1;RM 6.7; RM 7.3	Most reported values based on detection limits; only four detected concentrations exceeded TRV. Limited spatial extent of exceedances does not equate to a population or community-level risk.	Based on a WOE approach, endrin poses negligible risks to the benthic community or populations because of the low magnitude and frequency of EFs > 1.0 and two LOEs (tissue residue and surface water) that identify no unacceptable risk associated with endrin. Two bioassays were located within the areas with SQG exceedances, and both stations were toxic based on the *Chironomus* growth and mortality endpoints. PAHs exceeded their SQGs at both stations, and carbazole exceeded its SQG at one station.
Endrin ketone	SQGs	1.2 – 11	RM 6.1; RM 6.4; RM 7.3; RM 8.8	Most reported values based on detection limits; only four detected concentrations exceeded TRV. Limited spatial extent of exceedances does not equate to a population or community-level risk.	Based on a WOE approach, endrin ketone poses negligible risks to the benthic community or populations because of the low magnitude and frequency of EFs > 1.0 and two LOEs (tissue residue and surface water) that identify no unacceptable risk associated with endrin ketone. Only one bioassay station was located within the areas with SQG exceedances; the station at RM 8.8, west bank, was toxic based on all four endpoints.

[a] Includes the following DDT COCs: 2,4′-DDD, 2,4′-DDT, 4,4′-DDD, 4,4′-DDT, sum DDE, and total DDx.

COC – chemical of concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
EF – exceedance factor
HPAH – high-molecular-weight polycyclic aromatic hydrocarbon
HQ – hazard quotient
J – estimated concentration
LOE – line of evidence
LPAH – low-molecular-weight polycyclic aromatic hydrocarbon
N – presumptive evidence of a compound
PCB – polychlorinated biphenyl
RM – river mile
SQG – sediment quality guideline
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)
TRV – toxicity reference value

6.7.2.3 Upper River Segment

No Level 3 toxicity test exceedances were found in this segment of the Study Area.

Most COCs were identified on the basis of SQG or tissue TRV exceedance. Toxicity was predicted in the upper river segment based on exceedance of the SQGs for metals (copper and mercury), benzyl alcohol, phenol, total PCBs, and DDx compounds. The tissue-residue LOE identified copper, zinc, and total PCBs as COCs. The surface water LOE identified zinc and DDx compounds as COCs. Table 6-32 presents the key COCs in the upper river segment and the LOEs identifying these chemicals as COCs

Exceedance of the metal SQGs occurred in single samples. The benzyl alcohol exceedance was also based on a single sample. PCB exceedances of the SQG were associated with the nearshore sediments between RM 11.1 and RM 11.3. Two samples exceeding the total DDx SQG were co-located with the PCB exceedances at RM 11.2 and RM 11.3.

Copper and zinc tissue residues exceeded their respective TRVs in the majority of the invertebrate tissue samples analyzed from this segment of the Study Area; however, tissue residues of these metals is considered highly uncertain as an indication of unacceptable risk because of aquatic organisms' ability to regulate metals. PCB tissue concentrations (predicted and empirical) exceeded the TRV in this segment of the Study Area.

In summary, one area above RM 9.0 was identified as posing localized risk to benthic communities. The area is a previously unidentified area between RM 11.0 and RM 11.3, along the eastern shore. The area was identified based on PCB exceedances of the SQG and tissue TRV. Another area between RM 9.0 and RM 9.2 was identified as a PBRA based on total PCB SQG exceedances; however, based on three non-toxic bioassay stations that also represented the maximum PCB concentration in that area, unacceptable risk to the benthic community is unlikely.

The BERA is inconclusive about whether these areas pose unacceptable risks to benthic populations or the benthic community. The determination depends on issues related to appropriate population and community-level measurement endpoints that are not easily resolved. Because this area will be identified as posing unacceptable risk to non-benthic receptors, the determination of potential risks to benthic populations and communities will remain as an unresolved uncertainty.

Table 6-32. Summary of Key Benthic Invertebrate COCs for the Upper River Segment and Uncertainties Across LOEs

Chemical Group	LOE	EF or HQ	RM	Key Uncertainties	Risk Conclusions
Metals					
Copper	SQGs	5	RM 11.1, east	A single exceedance does not equate to population or community-level risk.	Based on a WOE approach, copper poses negligible risks to the benthic community or populations because of the low magnitude of EFs/HQs > 1.0, high uncertainty with respect to metals toxicity based on the tissue LOE, and one LOE (surface water) that identify no unacceptable risk associated with copper. No bioassay was performed at the location with the SQG exceedance.
	Tissue residue	1.2 – 2.6	RM 9.3 – RM 11.3	Limited spatial extent of exceedances does not equate to population or community-level risk. Tissue residue LOE highly uncertain because benthic invertebrates regulate metals and the tissue-residue TRV for copper cannot reliably predict toxicity. HQs were low.	
Mercury	SQG	1.5	RM 10.5	A single exceedance does not equate to population or community-level risk.	Based on a WOE approach, mercury poses negligible risks to the benthic community or populations because of the low magnitude and spatial extent of EFs > 1.0 and two LOEs (tissue residue and surface water) that identify no unacceptable risk associated with mercury. No bioassays were performed in the areas with the SQG exceedances.
Zinc	Tissue residue	1.1 – 1.6	RM 9.3 – RM 11.3	Limited spatial extent of exceedances does not equate to population or community-level risk. Tissue residue LOE highly uncertain because benthic invertebrates regulate metals and the tissue-residue TRV for zinc cannot reliably predict toxicity. HQs were low.	Based on a WOE approach, zinc poses negligible risks to benthic invertebrates because of the low magnitude of HQs > 1.0 and high uncertainty with respect to metals toxicity based on the tissue LOE, and one LOE (site-specific SQG) that identify no unacceptable risk associated with zinc.
	Surface water	1.1	RM 9.7, west	A single low exceedance does not represent a population or community-level risk. TRV is based on toxicity to fish and invertebrates; invertebrates may be less sensitive to zinc than are fish.	
Butyltin	Surface water	1.2	RM 11, west	A single low exceedance does not represent a population or community-level risk. Butyltin was identified as a COC using the TBT TRV as a surrogate.	No risks to the benthic invertebrate community are expected from surface water. Two LOEs (tissue residue and site-specific SQG) identify no unacceptable risk associated with butyltin.
Organics					
Benzyl alcohol	SQGs	1.1, 1.6	RM 9.5; RM 10.4	Detected concentration exceeded SQG at only two locations; quantification of benzyl alcohol uncertain.	Based on a WOE approach, benzyl alcohol poses negligible risks to the benthic community or populations because of the low magnitude, low exceedance frequency, and limited spatial extent of EFs > 1.0 and two LOEs (tissue residue and surface water) that identify no unacceptable risk associated with benzyl alcohol. No bioassays were performed in the areas with the SQG exceedances.
Phenol	SQGs	1.1	RM 10.5	Single exceedance at one location does not represent a population- or community-level effect. Quantification of phenols is uncertain.	Based on a WOE approach, phenol poses negligible risk based on a single low magnitude EF > 1.0 and two LOEs (tissue residue and surface water) that identify no unacceptable risk associated with phenol. No bioassay was performed at the location with the SQG exceedance.
PCBs					
Total PCBs	SQGs	1.3 - 12	RM 9.1-9.2; RM 9.7; RM 10.2; RM 11.2 – 11.3	EFs were low (< 7.6) except at one location with EF of 12. Extrapolating to community-level risk is even more difficult than extrapolating to population-level. Community-level risk would occur if population-level risks were significantly disrupting the performance of specific ecological functions.	Possible unacceptable risks to the benthic community along the east/north shoreline near RM 11.3 based on bioaccumulation risk. Unacceptable risks were predicted over a greater area based on SQG exceedances; however, the empirical toxicity data in this area were not toxic. A definitive assessment would require better population and community-level measurement endpoints than those stipulated in the BERA Problem

Table 6-32. Summary of Key Benthic Invertebrate COCs for the Upper River Segment and Uncertainties Across LOEs

Chemical Group	LOE	EF or HQ	RM	Key Uncertainties	Risk Conclusions
	Tissue residue	1.1 – 4.0	RM 9.7, west; RM 11.3, east	Empirical tissue residue HQs > 1.0 were limited spatially. Laboratory exposures under static conditions resulted in higher HQs than field collected samples. The majority of the HQs for both empirical and predicted tissues were less than 2.0.	Formulation. Negligible risks to the benthic community in the area of RM 9.1 to 9.2 based on lower concentrations in the PBRA as compared to the maximum concentration at the non-toxic bioassay stations. Three non-toxic bioassay stations were located within the area with SQG exceedances. Risks are uncertain in two small PBRAs at RM 9.7 and RM 10.2 based on SQG exceedances; no bioassay stations were within these areas.
Pesticides					
DDx compounds[a]	SQG	1.2, 2.1	RM 11.2 – RM 11.3	NJ qualified data included in the assessment.	Based on a WOE approach, DDx compounds pose negligible risks to the benthic community or populations because of low magnitude, frequency, number of EFs/HQs > 1.0, and one LOE (tissue residue) that identify no unacceptable risk associated with DDx compounds.
	Surface water	1.6 – 5.9	RM 9.6, west; RM 9.9, east	No samples exceed an alternative water TRV that is based on direct exposure of aquatic organisms to surface water.	

[a] Includes the following DDT COCs: 2,4′-DDD, 2,4′-DDT, 4,4′-DDD, 4,4′-DDT, and total DDx.

COC – chemical of concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
EF – exceedance factor
HQ – hazard quotient
J – estimated concentration
LOE – line of evidence
N – presumptive evidence of a compound
PCB – polychlorinated biphenyl
RM – river mile
SQG – sediment quality guideline
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)
TRV – toxicity reference value

6.7.2.4 Qualitative Assessment of Benthic Community Health

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 (may be 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 are indicators of communities that are recovering 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., high 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. In environments 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 be the result of 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 6-33.

Table 6-33. Classification of Community Response to Physical Transport Regime

Biological Community	Physical Regime			
	Highly Depositional	Depositional	Erosional/ Transport	Mixed
Early (Stage 1)	Expected	Unexpected; may indicate non-physical factors are affecting community	Expected	Expected
Transitional	Expected	Unexpected; may indicate	Expected	Expected

Table 6-33. Classification of Community Response to Physical Transport Regime

Biological Community	Physical Regime			
	Highly Depositional	**Depositional**	**Erosional/ Transport**	**Mixed**
(Stage 2)		non-physical factors are affecting community		
Mature (Stage 3)	Unexpected, but meets overall goal of community health	Expected	Unexpected, but meets overall goal of community health	Unexpected, but meets overall goal of community health

Once individual locations were reclassified, the evaluation focused on the locations where community response was unexpected and 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 determined59, approximately 48% of the sediment profile image locations within the Study Area were classified as Stage 3 and 49% as Stage 1 (about 3% were 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 a reclassification to "expected" because of a likely instability in the bottom substrate. Notes from the SPI analysis suggested 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 3-36 shows the location of the unexpected community successional stages relative to the benthic risk areas defined by exceedance of Level 3 SQGs. The reach between RM 5.0 and RM 9.0 is also the reach with the greatest area predicted to be toxic to benthic invertebrates based on the sediment toxicity and tissue-residue LOEs, suggesting possible chemical toxicity, among other potential factors, as the reason for the presence of lower successional stages in this reach.

[59] Successional community stage could not be determined where debris or hard substrates prevented camera penetration.

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 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. 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

In summary, the general conclusions for the benthic risk assessment are as follows:

The spatial WOE evaluation identified approximately 5% of the Study Area as posing unacceptable risk to the benthic community.

Qualitative evidence from the SPI analysis suggests that the benthic community structure is largely physically controlled, with limited areas of potential chemical toxicity.

Unacceptable benthic risks are highly associated with shoreline areas, slips, and areas of elevated sediment chemical concentrations, with the exception of an area of PAH contamination in mid-channel between RM 5.1 and RM 5.8.

Most unacceptable benthic risks are contained in the middle section of the Study Area, between RM 5.0 and RM 9.0.

There was no significant ($P < 0.05$) sediment toxicity above RM 9.2 based on the two toxicity tests.

- PCBs are the most widespread bioaccumulative COC associated with unacceptable benthic risks. TBT and DDx compounds are associated with bioaccumulative risks over more localized areas.
- Exceedance of SQGs for metals occurred primarily between RM 5.6 and RM 9.2 of the Study Area, typically on a small scale (e.g., one sample). The metals include cadmium, copper, mercury, and silver. Toxicity associated with metals in sediments in the lower river segment was limited to cadmium, mercury, and silver at RM 3.7 and from RM 4.3 to RM 4.8. Toxicity associated with metals in the upper river segment was limited to copper and mercury at RM 11.1, RM 10.2, and RM 10.5.
- Exceedance of site-specific SQGs for PAHs is associated with known source areas between RM 5.0 and RM 7.0.
- Exceedance of SQGs for pesticides 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 BERA for fish in the Study Area. For fish receptors, multiple exposure routes (i.e., direct contact, ingestion) were evaluated for various media (i.e., sediment, water, and tissue) 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. They are:

- Invertivorous fish – sculpin, peamouth, and juvenile Chinook salmon[60]
- Omnivorous fish – largescale sucker, carp,[61] and pre-breeding white sturgeon
- Piscivorous fish – smallmouth bass and northern pikeminnow
- Detritivorous fish – Pacific lamprey ammocoetes[60]

The three primary quantitative LOEs for characterizing risks to all fish are assessments of exposures via: 1) COPC residues in tissue of the receptor species, 2) COPCs ingested via the diet, and 3) direct exposure to COPCs in surface water. Benthic fish direct exposure to PAHs in sediment was also evaluated as a semi-quantitative LOE per EPA's Problem Formulation (Attachment 2) and included an assessment of the apparent health of pre-breeding sturgeon. Benthic fish exposure to TZW was also evaluated.

Exposure estimates for all fish LOEs were compared to TRVs expressed as effects thresholds. LOAEL TRVs provide a basis for evaluating whether exposure concentrations are at or above a level that may cause an effect on survival, growth, or reproduction of some individual organisms in experimentally exposed populations. This approach follows the conventional practice in ecological risk assessment of using organism-level TRVs defined in this manner to evaluate the potential effects on populations. No explicit population modeling was included in this BERA. Organism-to-population extrapolation is a source of uncertainty and, in many but not all cases, a source of conservatism in risk estimates. Whether the organism-to-population extrapolation is a source of conservatism depends in large part on the level of effect represented by the TRV, which varies because of differences in the toxicological databases for different chemicals and receptor groups (Solomon et al. 2008; Suter 2007; Posthuma et al. 2002; Pastorok et al. 2001).

Risks to fish were assessed by comparing media-specific (i.e., tissue, sediment, surface water, diet) exposure estimates to TRVs. Several factors had to be accounted for to estimate

[60] Juvenile Chinook salmon and Pacific lamprey ammocoetes are special status species (i.e., federally threatened and an Oregon State sensitive species that are of special concern to tribes, respectively) and were evaluated at the individual level. All other fish receptors were evaluated at the population level.

[61] 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.

dietary exposure levels. These included information about feeding rates, foraging areas, prey home ranges, and diets. TRVs were selected from published studies and approved by EPA for use in the BERA.

All of the fish COPCs that were 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 1997a). First, risk estimates were calculated on a sample-by-sample basis for all LOEs. This was done in accordance with the methods described in the EPA's Problem Formulation (Attachment 2). If those estimates indicated a potential unacceptable risk to a fish receptor, 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. The fish risk characterization methods for each 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; and Section 7.4, the assessment of direct exposure to PAHs in sediment. The assessment of direct exposure to PAHs in sediment was used as a semi-quantitative LOE to evaluate risks to benthic fish. The comparison of TZW concentrations to ecological thresholds is presented in Section 7.5. As required by EPA, TZW was screened against surface water screening values. TZW data were limited, and no attempt was made to further characterize TZW exposure; accordingly, the results of the TZW assessment were not incorporated into the risk conclusions section.

Section 7.6 presents the overall risk conclusions for fish, including the identification of fish COCs and the general uncertainty analysis for the fish risk assessment. Specific uncertainties associated with each line of evidence are discussed in the individual LOE sections. Section 7.6 provides the final determination of COCs, a description of the qualitative WOE approach used to determine risk conclusions for fish, the key uncertainties in the exposure assessment and effects characterization, and an interpretation of the risk assessment findings.

Figure 7-1 presents a flowchart of the fish risk assessment section organization.

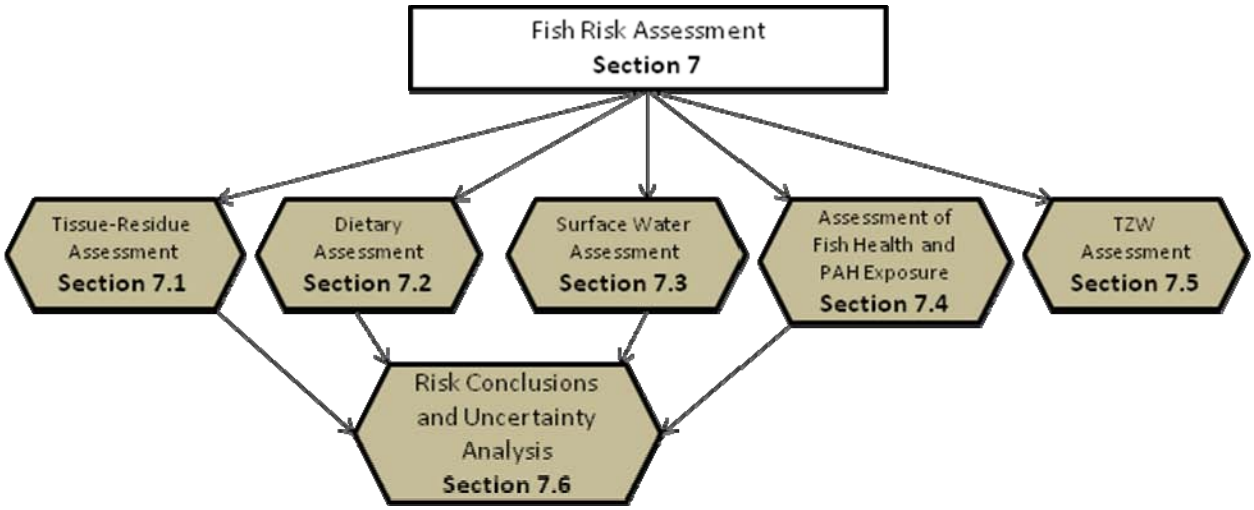

Figure 7-1. Overview of Fish Risk Assessment Section Organization

7.1 TISSUE-RESIDUE ASSESSMENT

The tissue-residue assessment was one LOE for evaluating risks to all fish receptors. This LOE incorporated multiple exposure pathways for fish (i.e., dietary ingestion, direct contact with water and sediment).

Receptor-specific fish tissue-residue COPCs were identified in the SLERA and refined screen using screening-level TRVs (Attachment 5). These COPCs were evaluated by comparing tissue TRVs to whole-body tissue chemical concentrations to derive HQs. HQs were calculated in a step-wise process:

HQs were first calculated on a sample-by-sample basis.

HQs were then derived for relevant exposure areas based on receptor-specific home ranges[62]

HQs in the first step were calculated per EPA (2008j), as outlined in EPA's Problem Formulation (Attachment 2). The HQ results from the first step were used only to narrow the list of COPCs for evaluation in the second step. For each receptor, COPCs with HQs > 1.0 based within a relevant home range (based on the second step) were retained as COCs. COCs, including the magnitude of HQs, the spatial distribution and frequency of HQ > 1.0, the results of other LOEs (e.g., dietary dose, surface water), when applicable, and the associated exposure and effects assumptions, were evaluated to arrive at risk conclusions for fish.

The details of this assessment are presented as follows:

[62] The relevant exposure area for sculpin was considered roughly equivalent to the sampling area of sculpin. Sculpin were evaluated only on a sample-by-sample basis.

- Section 7.1.1 presents a summary of the COPCs identified for all receptors that are evaluated in the tissue-residue LOEs.
- Section 7.1.2 presents an overview of 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 7.1.3 presents a summary of 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 7.1.4 presents the risk characterization results, COC-receptor pairs, and associated uncertainties. These COCs are further assessed in the fish risk conclusions section (Section 7.6). The individual sample-by-sample assessment is presented in Attachment 12.

Figure 7-2 presents a flowchart of the fish tissue-residue assessment section organization.

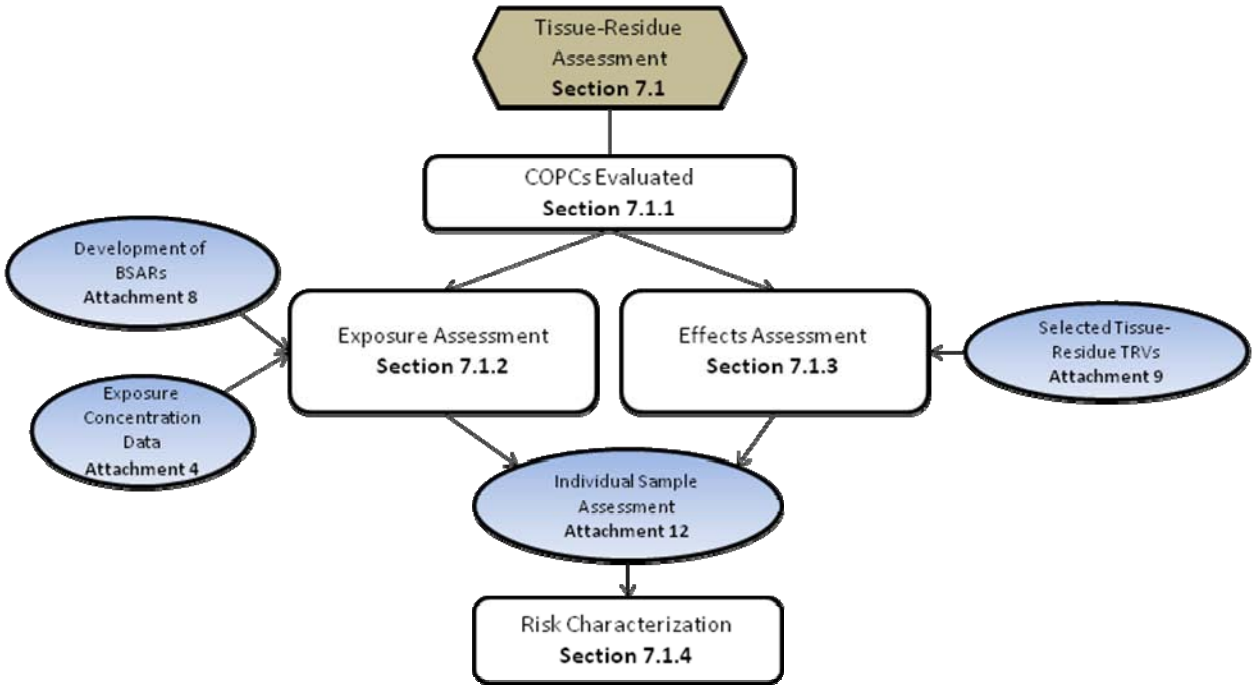

Figure 7-2. Overview of Fish Tissue Assessment Section Organization

7.1.1 COPCs Evaluated

COPC-receptor pairs were identified in the SLERA and refined screen (Attachment 5). Table 7-1 presents the fish tissue COPCs. All of these COPC-receptor pairs were evaluated, with the exception of aluminum. 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.[63] 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. Also, aluminum concentrations in Study Area fish tissue were compared to upriver tissue data (Section 7.1.4.3), and sediment and surface water concentrations were compared to background concentrations. Background aluminum concentrations were similar to or greater than Study Area concentrations in sediment and surface water. The absence of an aluminum tissue-residue effect threshold value is less important if aluminum concentrations in the Study Area are not elevated relative to background or other upriver conditions.

Table 7-1. Fish Tissue-Residue COPCs

	Tissue Type							
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	X				X	X		
PCBs								
Total PCBs	X				X	X	X	
Pesticides								
4,4′-DDD[b]	X		X		X	X		X

[63] The aluminum SL TRV was based on the 5th percentile of LOAELs reported in Dyer et al. (2000).

Table 7-1. Fish Tissue-Residue COPCs

	Tissue Type							
COPC	**Large-scale Sucker**	**Juvenile White Sturgeon**	**Juvenile Chinook Salmon**	**Pea-mouth**	**Sculpin**	**Small-mouth Bass**	**Northern Pike-minnow**	**Pacific Lamprey**
4,4′-DDT[b]					X			
beta-HCH					X			
Total DDx	X				X	X	X	

[a] Aluminum was identified as a COPC based on the SLERA and refined screen; however, aluminum was not further evaluated in the tissue residue LOE, per EPA (EPA 2008e).

[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 – chemical of potential concern
DDD – dichlorodiphenyldichloroethane
DDT – dichlorodiphenyltrichloroethane
HCH – hexachlorocyclohexane
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)

Per EPA (Attachment 2), carp was selected as a surrogate ecological 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.

Six fish tissue COIs could not be evaluated because no toxicological data associated with tissue residues were available (Table 7-2). The risk to fish receptors associated with these chemicals is unknown because no toxicological data for tissue residues are available. Two of the six, benzoic acid and 4-chloro-3-methylphenol, were also identified as COIs in surface water. However, the surface water concentrations of these two chemicals were below TRVs as presented in the SLERA (Attachment 5).

Table 7-2. Fish Tissue COIs with No SL TRVs

COI	
Metals	
Manganese	
SVOCs	
Benzoic acid	Bis(2-chloroethoxy) methane
Benzyl alcohol	
Phenols	
4-Chloro-3-methylphenol	4-Nitrophenol

COI – chemical of interest
SL – screening level
SVOC – semivolatile organic compound
TRV – toxicity reference value

Uncertainties Associated with Metals in the Tissue-Residue Approach

Eight metals were identified as COPCs for fish. The usefulness of applying the tissue-residue approach to metals has been questioned because the uptake, distribution, and disposition of 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). These differences create difficulties when interpreting the toxicological significance of whole-body residues and increase the uncertainty when extrapolating across different exposure routes, durations, and species. The uncertainty of this LOE for metals is considered in the evaluation of COCs across multiple LOEs in Section 7.6.

7.1.2 Exposure Assessment

The whole-body tissue concentration reflects exposure and subsequent accumulation from all pathways (i.e., dietary ingestion and direct contact with water and sediment). EPCs in this assessment were primarily represented by concentrations detected in composite samples collected from the Study Area. EPCs were also represented by predicted concentrations that were derived using surface sediment concentrations and Study Area tissue-sediment relationships.

7.1.2.1 Empirical Tissue EPCs

This section presents a summary of the methods used to derive EPCs in empirical tissue residues.

The fish risk evaluation using the tissue-residue LOE involved two steps, in which the data used to represent EPCs differed:

Step 1 – In the first step, EPCs were based on individual sample-by-sample tissue-residue concentrations from throughout the Study Area. The step was conducted in accordance with EPA's Problem Formulation (Attachment 2). Maps 4-6 through 4-13 present the individual composite sample locations for all of the fish receptors. COPC concentration data for all individual composite samples are presented in Attachment 4.

Step 2 – Because the home range of most of the fish receptors is greater than the areas represented by the composite sampling areas, tissue EPCs based on representative home ranges were also developed. In the second step, EPCs were based on tissue-residue concentrations calculated over a relevant exposure scale (based on species-specific foraging range assumptions). Table 7-3 presents the exposure scale assumptions used to derive tissue EPCs for each fish receptor based on home range and foraging range data. The rationale for these exposure scale assumptions is presented in Attachment 13. Fish risk conclusions for the tissue-residue LOE were based on relevant exposure scale EPCs.

Table 7-3. Summary of Receptor-Specific Exposure Area Assumptions for Tissue EPC Calculations

Receptor(s)	Exposure Scale[a]	Exposure Areas	EPC Basis
Largescale sucker, juvenile white sturgeon, juvenile Chinook salmon, peamouth, Pacific lamprey ammocoetes	Site-wide	RM 1.9 to RM 11.8 (Study Area)	EPCs were calculated as Study Area UCL[b] concentrations.
Sculpin	0.1 mile	Individual sampling locations	EPCs were represented on a sample-by-sample basis (tissue sample compositing area is roughly equal to exposure scale).
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	EPCs were represented as the maximum COPC concentration within a 1-mile segment.[c]

[a] The rationale for the selected exposure areas is presented in Attachment 13.

[b] Where insufficient data were available to calculate a UCL, the maximum concentration was used to represent the EPC.

[c] Insufficient data were available to calculate a UCL within a 1-mile exposure area for smallmouth bass and northern pikeminnow; therefore, the maximum concentration within each 1-mile exposure area was used to represent the EPC.

COPC – chemical of potential concern

EPC – exposure point concentration

RM – river mile

UCL – upper confidence limit on the mean

Tissue EPCs based on an upper confidence limit of the mean 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 was required to derive a UCL (EPA 2007f). In the case where an insufficient number of detected data values was available, the maximum concentration[64] was used to represent the EPC. 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.

Uncertainty is associated with the use of maximum concentrations to represent tissue EPCs. Given the small exposure scales evaluated for smallmouth bass and northern pikeminnow and small composite sample sizes for some fish receptors, only limited data were available

[64] When the maximum concentration was a non-detected value, the full detect limit was used to represent the EPC.

to derive the UCL concentrations for each receptor species. When a limited number of samples are available, the maximum concentration represents a reasonable conservative estimate of the mean, however any estimate based on a few samples is uncertain. The use of maximum concentrations to represent tissue EPCs may result in an over or underestimate of risk.

Assessment Based on Individual Samples

In the tissue-residue assessment, risks to sculpin were determined based on an evaluation of each individual composite fish tissue sample. Individual sampling areas were considered roughly equivalent to the estimated exposure scale of sculpin (0.1 mile). This approach was used as directed by EPA in order to identify locations within the Study Area where adverse effects might occur to fish within the populations that are being assessed. Point-by-point assessment is a conservative and questionable method for evaluating risks to populations. It rests on inferences that have little or no scientific basis because population-level processes compensate for adverse effects on individuals (Pastorok et al. 2001). Sample-level evaluations do not represent population-level effects.

Several methods have been used in an attempt to address potential population-level effects, but no consensus on approach currently exists. In other ERAs, risks to fish populations have been assessed using the 95th UCL concentrations representing the site or some portion of the site (Windward 2007; MacDonald Environmental 2002). In Oregon State, acceptable risk to a population is defined as ≤ 10% chance that > 20% of the total population would receive an adverse exposure (adverse exposure is defined as greater than the LD50 or LC50 for a hazardous substance based on studies with route and exposure duration simulating field exposure conditions of the ecological receptor) (ODEQ 1998).

7.1.2.2 Predicted Tissue EPCs

In addition to using the empirical tissue data to represent exposure concentrations, tissue chemical concentrations were predicted using either a mechanistic bioaccumulation model, or site-specific BSARs to represent EPCs. BSARs were only developed for COPC-receptor pairs, for which a relationship between sediment and tissue could be established. 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). Sculpin tissue chemical concentrations were predicted because, although sculpin were collected from throughout the Study Area, it was not practical or feasible to collect sculpin from every tenth of a mile within the Study Area to represent their small home range (Map 4-10). Sculpin tissue chemical concentrations were predicted using surface sediment concentrations and either the mechanistic bioaccumulation model or species-specific BSARs.

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 (2009).

The mechanistic model was available for predicting total PCB, pesticide, and dioxin and furan concentrations.[65] 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 sculpin 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. Windward (2009) presents the details of the BSAR analysis and the mechanistic bioaccumulation model.

Table 7-4 presents the sculpin COPCs and the selected models used to predict tissue concentrations. Three sculpin tissue COPCs were modeled mechanistically (i.e., total PCBs, total DDx, and beta-HCH). For the other two sculpin COPCs that were not modeled mechanistically (i.e., copper and BEHP), the co-located data indicate no relationship between co-located sediment and sculpin tissue concentrations in the BSAR evaluation. 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 copper and BEHP and no predicted sculpin tissue concentrations could be calculated for these COPCs. 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	Predicted Tissue?	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

[65] Dioxins and furans were not tissue COPCs for sculpin (or any fish receptor), so the mechanistic model was not used to predict dioxin and furan sculpin tissue concentrations.

[a] Site-specific BSARs were not selected for these COPCs because these COPCs did not meet the appropriate BSAR analysis assumptions (Windward 2009), did not have a statistically significant positive slope ($p < 0.05$), or had an $r^2 < 0.30$.

BEHP – bis(2-ethylhexyl) phthalate
BSAR – biota-sediment accumulation regression
COPC – chemical 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)

7.1.3 Effects Assessment

This section presents the selected tissue TRVs used to characterize effects for fish COPC-receptor pairs and the uncertainties associated with these selected values. A NOAEL and a LOAEL were selected for each COPC. The effects data presented in this section are assessed in combination with exposure data (presented in Section 7.1.2) in the risk characterization (Section 7.1.4).

Tissue TRVs were derived in cooperation with EPA based on EPA'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 tissue-residue chemical concentrations in whole-body tissues associated with measured adverse effects related to survival, growth, and reproduction were measured. 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, 2009b). 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 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, a 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. The 5th and 10th percentiles of the SSD were selected as the fish tissue LOAELs. Details on how SSDs were derived are presented in Attachment 9.

Table 7-5 presents the selected TRVs expressed as fish tissue- residues. TRVs are expressed as literature-based LOAELs and NOAELs or as 5th and 10th percentile LOAELs. Only one toxicological study was identified for BEHP; in this study, fathead minnows were exposed to BEHP for 56 days, and no adverse effects on growth or survival associated with a tissue residue of 9.6 µg/g ww were reported (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 BEHP concentrations were compared to only the literature-based NOAEL reported in this study (≥ 9.6 µg/g ww).

Table 7-5. Selected Fish Whole-Body Tissue TRVs

	TRV (mg/kg ww)			
COPC	**5th Percentile LOAEL or Literature-Based NOAEL**	**10th Percentile LOAEL or Literature-Based LOAEL**	**Derivation**	**Key Uncertainties**
Metals				
Antimony	NC	9.0	LOAEL was derived from Doe et al. (1987)	Only one acceptable toxicity study was identifiedand therefore, TRV may over- or underestimate risks.
Cadmium	0.17	0.22	LOAELs are based on 5th and 10th percentile SSD, respectively	TRVs may over- or underestimate risks
Chromium	NC	44.1	LOAEL was derived from Roling et al. (2006).	Only one acceptable toxicity study was identified; concentrations and therefore, 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 of 2.2 mg/kg ww.
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; TRV may over- or underestimate risks.
Mercury	0.37	0.44	LOAELs are based on 5th and 10th percentile SSD, respectively.	
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.

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 LOAEL or Literature-Based LOAEL	Derivation	Key Uncertainties
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.
PCBs				
Total PCBs	0.43	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 have significant associated uncertainties (see Attachment 9); studies reporting adverse effects associated with egg or embryo residues were not included in SSD.
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	0.76	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 and egg-residue studies were generally below the identified 10th percentile TRV and inclusion would have resulted in lower TRVs.

COPC – chemical of potential concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
HCH – hexachlorocyclohexane
LOAEL – lowest-observed-apparent-effect level
NA – not applicable (no acceptable TRV was derived)
NC – not calculated (NOAEL not needed for risk evaluation
NOAEL – no-observed-apparent-effect level
PCB – polychlorinated biphenyl
SSD – species sensitivity distribution
TRV – toxicity reference value
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)
ww – wet weight

Tenth percentile and literature-based LOAELs were generally used to assess risks to the fish populations. Per EPA (2008j), as documented in the Problem Formulation (Attachment 2), fish 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 individual level by comparing tissue EPCs to NOAELs.[66]

Uncertainties Associated with Selected Tissue-Residue TRVs

TRVs based on SSDs were developed for 7 of the 13 fish tissue COPCs. For six COPCs, limited toxicological data were available (i.e., four or fewer toxicological 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 result in either an over- or underestimation of risks associated with a given tissue residue in Study Area fish.

In the derivation of TRVs based on SSDs, an ACR was applied to all tissue-residue LOAELs based on a survival endpoint to account for non-lethal (i.e., reproductive and/or growth) effects. The application of ACRs assumed that concentrations required to elicit acute mortality are generally higher than the concentrations that reduce growth and/or reproduction. The ACRs were chemical-specific and were applied to all tissue-residue concentrations based on mortality and an exposure duration of ≤ 30 days (i.e., an acute exposure). The ACRs applied were those reported in Raimondo et al. (2007) 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 because the ACRs were calculated as the ratio of the LC50/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. Hence, the ACR-adjusted tissue chemical concentrations included in the SSDs, which are distributions of adverse effects, may actually represent no adverse effects and result in an overestimate of risk.

There is high uncertainty associated with the selected 10th percentile LOAEL for total PCBs (0.93 mg/kg ww) and this selected LOAEL may overestimate risks to fish. The total PCB SSD was derived from 19 literature-based LOAELs (Attachment 9). Five of the lowest LOAEL used in the SSD are associated with significant uncertainty, as summarized in Table 7-6. The implications of these uncertainties are discussed in the risk characterization section (Section 7.6).

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	Reduction in live fry body weight of Atlantic salmon	Fish 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 uncertain

[66] For the tissue residue LOE, 5th percentile LOAELs were used to evaluate special-status species.

Table 7-6. Summary of Lowest PCB LOAELs and Associated Uncertainties

Study	LOAEL (mg/kg ww)	Endpoint	Uncertainties
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; measured egg PCB and DDT residues (prior to initiation of experiment) were 7.6 and 3.8 µg/g, respectively; tissue residues and adverse effects weren't 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 wasn't 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 not measured, but 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
Broyles and Noveck (1979)	9.2	51-87% mortality in lake trout	

DDT – dichlorodiphenyltrichloroethane
LOAEL – lowest-observed-adverse-effect level
PCB – polychlorinated biphenyl

7.1.4 Risk Characterization

This section presents the risk characterization process and results for the tissue-residue LOE for fish. An HQ was used to quantify risk estimates. HQs were derived for tissue COPC-receptor pairs using the following equation:

$$HQ = \frac{EPC}{TRV}$$ **Equation 7-1**

Where:

HQ = hazard quotient
EPC = exposure point concentration
TRV = toxicity reference value

The EPC and TRV are represented by tissue-residue concentrations expressed as mg/kg ww.

Section 7.1.4.1 presents the overall approach used to characterize risk to fish receptors. Section 7.1.4.2 presents the risk characterization results, uncertainty evaluation, and COCs for each fish receptor. Tissue-residue COCs, along with COCs from other LOEs were further evaluated in the fish risk conclusions section (Section 7.6). Section 7.1.4.3 presents

an evaluation of non-Study Area concentrations. Section 7.1.4.4 presents the uncertainty evaluation of non-target ecological receptors.

7.1.4.1 Risk Characterization Process

A deterministic risk characterization was conducted to characterize risks to fish receptors via the tissue-residue LOE. A risk characterization of fish COPC-receptor pairs based on the tissue-residue LOE was conducted using an HQ approach that integrated exposure and effects data. Multiple HQs were derived using various assumptions in order to implement the methods presented in EPA's Problem Formulation (Attachment 2).

As described in Section 7.1.2.1, HQs were calculated in a two-step process. HQs in the first step were calculated per EPA (2008j), as outlined in EPA's Problem Formulation (Attachment 2). The HQ results from the first step are presented in Attachment 12 and were used to narrow the list of COPCs for evaluation in the second step. COPCs with HQs > 1.0 in the second step were retained as COCs. The second step better characterizes risks to the established assessment endpoints.

The two-step process is presented below:

Step 1 – Evaluation of Individual Samples – In the first step, HQs were evaluated on a sample-by-sample basis. HQs were derived for individual tissue samples using Equation 7-1. The results of this screening step were used to guide the risk evaluation.

Step 2 – Evaluation of Relevant Exposure Areas – In the second step, HQs were calculated within relevant exposure areas.[67] For those COPC-receptor pairs that resulted in HQs > 1.0 in Step 1, tissue HQs were derived within relevant exposure areas using Equation 7-1 and tissue UCLs. If insufficient data were available to derive a UCL (i.e., fewer than six detected concentrations were available), HQs were based on maximum tissue concentrations. Receptor-specific EPCs for individual exposure areas were calculated based on the assumptions presented in Table 7-3.

COCs were identified based on those COPCs that resulted in HQs > 1.0 based on Step 2.[68] Once COCs were identified, an evaluation was conducted for all COCs to determine the quantitative risk results (i.e., magnitude, spatial distribution, and frequency of HQs), underlying uncertainties of exposure and effects data, agreement of HQs based on other LOEs (where applicable), and comparison of background concentrations. The evaluation of COCs is discussed in the fish risk conclusions section (Section 7.6).

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[67] The relevant exposure area for sculpin was considered roughly equivalent to the sampling area of sculpin samples. Sculpin were evaluated only on a sample-by-sample basis.

[68] For sculpin, COCs were identified as those COPCs with HQs > 1.0 based on a sample-by-sample basis.

Determination of COCs

COCs were identified as those COPCs with HQs > 1.0 based on ecologically relevant exposure scales (as presented in Table 7-3), regardless of the uncertainties associated with the exposure and effects data or assumptions. COPCs were identified as COCs based on individual composite sample HQs for sculpin, smallmouth bass, and northern pikeminnow because fish samples were collected and composited over areas that represent what are conservatively assumed to be ecologically relevant exposure scales. For the other species sampled, the ecologically relevant exposure scale is assumed to be the Study Area, so the EPC was calculated at that scale.

The uncertainties associated with risk estimates for individual COCs, the spatial distribution of COC exceedances, the magnitude of exceedance, and the type of effect threshold all play a role in identifying whether chemicals pose a population-level risk. The likelihood that a COC with a limited spatial distribution of exceedances and a low HQ poses significant risks to populations is low. COCs with a broader distribution and higher magnitude of exceedances have a greater potential for posing population-level risks. The type of effect threshold endpoint (i.e., survival, growth, or reproduction) affects the interpretation of HQs simply because effects on the survival of individuals have different implications for populations than do effects on the growth of individuals. Effects on the reproductive success of individuals have different implications for populations than do effects on either survival or growth.

Even for a particular type of endpoint (e.g., reduced survival), a particular HQ (e.g., HQ = 5) has different implications for each COC-receptor pair, because the exposure-response relationship of one COC-receptor pair differs from that of other COC-receptor pairs. Short of developing structured population models and incorporating the effects of COC-receptor-specific exposure-response data, the demographic (i.e., population-level) consequences of any particular HQ simply cannot be quantified.

Populations are resilient to environmental stressors (Barnthouse et al. 2009). By corollary, it is unreasonable to assume that survival, growth, or reproduction HQs > 1.0 will necessarily have detrimental effects on the population whose individuals are potentially affected.

7.1.4.2 Risk Characterization Results and Uncertainty Evaluation

The HQ results from the first step are presented in Attachment 12 and were used to narrow the list of COPCs for evaluation in the second step. Table 7-7 presents the COPC-receptor pairs with HQs > 1.0 based on individual samples (Step 1) that were retained for further evaluation. The following subsections present the HQs for each fish receptor based on tissue residues over relevant exposure areas. COPCs with HQs > 1.0 based within relevant exposure areas were retained as COCs.

Table 7-7. COPCs with HQs > 1.0 Based on Individual Tissue Samples

	Large-Home-Range Fish			Small-Home-Range Fish		
COPC	**Large-scale Sucker**	**Pea-mouth**	**Pacific Lamprey**	**Sculpin**	**Small-mouth Bass**	**Northern Pike-minnow**
Metals						
Copper			X	X		
Lead		X			X	
Mercury						X

Table 7-7. COPCs with HQs > 1.0 Based on Individual Tissue Samples

	Large-Home-Range Fish			Small-Home-Range Fish		
COPC	**Large-scale Sucker**	**Pea-mouth**	**Pacific Lamprey**	**Sculpin**	**Small-mouth Bass**	**Northern Pike-minnow**
PCBs						
Total PCBs	X			X	X	X
Pesticides						
Total DDx				X		
BEHP				X[a]	X[a]	

[a] No BEHP LOAEL was available; retained as a COPC-receptor pair based on an HQ > 1.0 derived from a NOAEL.
BEHP – bis(2-ethylhexyl) phthalate
COPC – chemical of potential concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
HQ – hazard quotient
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)

It should be noted that juvenile white sturgeon and juvenile Chinook salmon were not evaluated beyond the refined screen because all individual sample COPC concentrations were less than their respective 5th and 10th percentile LOAELs (Attachment 12). Five COPCs (i.e., antimony, cadmium, chromium, zinc, and beta-HCH) were not evaluated for any fish receptor beyond the refined screen because these COPCs never exceeded their respective TRVs in any fish sample (Attachment 12). BEHP was retained as a COPC for largescale sucker, sculpin, and smallmouth bass because tissue concentrations exceeded a NOAEL (no LOAEL was available).

The following subsections present the risk characterization results for each fish receptor.

7.1.4.2.1 Large-Home-Range Fish

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**	**Site-wide EPC**	**Literature-Based LOAEL TRV**	**HQ**	**Site-wide EPC**	**5th Percentile LOAEL TRV**	**HQ**
Metals										
Copper	mg/kg	NA	NA	NA	NA	NA	NA	6.2[a]	2.8	**2.2**
Lead	mg/kg	NA	NA	NA	10.6[a]	4.0	**2.7**	NA	NA	NA
PCBs										
Total PCBs	µg/kg	1,498[b]	930	**1.6**	NA	NA	NA	NA	NA	NA

[a] EPC is based on maximum concentration; insufficient data were available to derive site-wide UCL.
[b] EPC is based on site-wide UCL concentration.
EPC – exposure point concentration
HCH – hexachlorocyclohexane
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
NA – not applicable (not a COPC-receptor pair)
PCB – polychlorinated biphenyl
TRV – toxicity reference value
Bold identifies HQs >1.0.

Largescale Sucker

Five COPCs were identified for largescale sucker in the SLERA and refined screen. One COPC (total PCBs) had an HQ > 1.0 based on individual samples (Table 7-7). The total PCB HQ based on the site-wide EPC was 1.6 (Table 7-8), therefore total PCBs was identified as a COC for largescale sucker.

Peamouth

Lead was the only COPC identified for peamouth in the SLERA and refined screen. Insufficient data were available to calculate a site-wide UCL for lead concentrations in peamouth in order to evaluate risks. Lead was retained as a COC for peamouth based on a maximum HQ of 2.7 (Table 7-8).

Pacific Lamprey Ammocoetes

Two COPCs were identified for Pacific lamprey ammocoetes in the SLERA and refined screen. One COPC (copper) had an HQ > 1.0 based on individual samples and the 5th percentile LOAEL (Table 7-7). Insufficient data were available to calculate a site-wide

UCL for copper concentrations in lamprey. Copper was retained as a COC for lamprey based on a maximum HQ of 2.2 (Table 7-8).

7.1.4.2.2 Small-Home-Range Fish

Sculpin

Six COPCs were identified for sculpin in the SLERA and refined screen. Three COPCs (copper, total PCBs, and total DDx) had 10th percentile LOAEL HQs > 1.0 based on individual samples (Table 7-7).

The relevant exposure area for sculpin was considered roughly equivalent to the sampling area of sculpin. Therefore, sculpin data were evaluated only on a sample-by-sample basis. Table 7-9 presents a summary of the individual sample HQs based on empirical and predicted tissue chemical concentrations. The locations of individual samples with HQs > 1.0 for copper, total PCBs, and total DDx are presented on Maps 7-1 though 7-3.

Table 7-9. Number of Sculpin Samples with 10th Percentile LOAEL HQs > 1.0

	Number of Samples with HQs > 1.0 (HQ range)	
COPC	**Empirical Tissue**	**Predicted Tissue[a]**
Metals		
Copper	3/38 (0.30 – 2.3)	NA
PCBs		
Total PCBs	4/38 (0.098 – 9.4)	90/1,100 (0.042 – 111)
Pesticides		
Total DDx	1/38 (0.0081 – 1.9)	29/1,128 (0.0070 – 21)

[a] HQs are based on predicted sculpin tissue concentrations using the mechanistic model and individual surface sediment samples with detected chemical concentrations.

COPC – chemical of potential concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
HCH – hexachlorocyclohexane
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
NA – not applicable; tissue concentrations were not predicted
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)

Three COCs were identified for sculpin based on HQs > 1.0: copper, total PCBs, and total DDx (including 4,4′-DDT). The spatial distribution of estimated risks is discussed below:

- **Total PCBs** – Four empirical composite tissue samples (11% of all empirical samples) had a total PCB HQ > 1.0 in the following locations: at RM 2.3 and RM 2.4 on the east side of the Study Area, Willamette Cove, and RM 11.3 in the east side of the side of the Study Area (Map 7-1). Ninety predicted tissue samples

(8% of all predicted samples) had chemical concentrations had HQs > 1.0. Map 7-1 presents the NN-interpolated areas where predicted total PCB tissue concentrations result in HQs > 1.0. Predicted HQs were > 1.0 in the same locations where tissue samples had HQs > 1.0 and several additional areas: International Slip, RM 4.1 on the east side, Slip 1, RM 5.8 on the east side, along RM 7.0 to RM 7.7 on the west side, Swan Island Lagoon, and along RM 8.8 to RM 9.7 on the west side. In general, the areas where predicted HQs were > 1.0 were near or at the same locations as detected sculpin tissue samples with chemical concentrations had HQs < 1.0 (Map 7-1). Empirical data are assumed to be more representative of tissue concentrations in Study Area sculpin than are predicted tissue chemical concentrations.

- **Total DDx** – Twenty-nine predicted sample (3% of all predicted samples[69]) and one empirical tissue (3% of all empirical samples) had total DDx HQs > 1.0. The empirical sample with an HQ > 1.0 was located at RM 7.3 on the west side of the Study Area (Map 7-2). Map 7-2 presents the NN interpolated areas[70] where predicted total DDx tissue concentrations result in HQs > 1.0. Predicted HQs were > 1.0 from approximately RM 7.1 to 7.4 on the west side of the Study Area.
- **Copper** – Three tissue samples (8% of all samples) had HQs > 1.0 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 ranged from 1.1 to 2.3 (Map 7-3). Sculpin tissue chemical concentrations could not be predicted from sediment because no relationship between copper concentrations in tissue and sediment were identified (see Section 7.1.2).

One of the thirty-eight 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. However, BEHP was not identified as a COC for sculpin because an HQ > 1.0 based on an unbounded NOAEL does not conclusively indicate unacceptable risks to sculpin populations, even if HQs > 1 were widespread, which they are not. The remaining 37 sculpin samples throughout the Study Area were less than the NOAEL TRV for BEHP. Risk to sculpin from BEHP is unknown and uncertain because of the lack of an acceptable LOAEL TRV. An evaluation of the distribution of BEHP concentrations in sediment is provided in Section 5.0 of the draft RI.

Smallmouth Bass

Six fish tissue COPCs were identified for smallmouth bass in the SLERA and refined screen. Two COPCs (lead and total PCBs) had individual samples resulting in

[69] Eleven of the twenty-nine predicted sediment samples with HQs > 1.0 were based on N-qualified data.

[70] 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 geographic information system 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.

10th percentile LOAEL HQs > 1.0 (Table 7-7). HQs for these COPCs were calculated based on 1-mile exposure areas throughout the Study Area (Table 7-10). Insufficient data were available to calculate exposure area UCLs; therefore, the maximum value was used to assess risks to smallmouth bass.

Table 7-10. Smallmouth Bass 1-Mile Exposure Area-Specific Tissue 10th Percentile LOAEL HQ

	LOAEL HQ[a]	
Exposure Area	**Lead**	**Total PCBs**
RM 1.5 to RM 2.5	0.0025	**1.5**
RM 2.5 to RM 3.5	0.45	0.84
RM 3.5 to RM 4.5	0.014	**1.6**
RM 4.5 to RM 5.5	0.0028	0.42
RM 5.5 to RM 6.5	0.0043	0.73
RM 6.5 to RM 7.5	0.0084	**2.2**
RM 7.5 to RM 8.5	0.0022	0.97
RM 8.5 to RM 9.5	0.0088	1.0
Swan Island Lagoon	0.076	**5.3**
RM 9.5 to RM 10.5	**280**	0.87
RM 10.5 to RM 11.8	0.033	**7.1**

[a] The 10th percentile LOAEL HQ was calculated using the maximum concentration available from within each exposure area.
HQ – hazard quotient
LOAEL – lowest-observed-apparent-effect level
PCB – polychlorinated biphenyl
RM – river mile
UCL – upper concentration limit on the mean
Bold identifies HQs > 1.0.

Two COCs were identified for smallmouth bass: lead and total PCBs. HQs on a composite sample-by-sample basis[71] are presented on Maps 7-4 and 7-5. The spatial distributions of these risk estimates are discussed below:

- **Lead** – Two smallmouth bass samples had HQs > 1.0 between RM 9.5 and RM 10.5; calculated HQs were 280 and 1.7. The high lead HQ of 280 (1,100 mg/kg ww) appears to be an outlier, although, this sample may be an indicator of a

[71] 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.

localized source. The lead tissue concentrations detected in all of the other samples (n = 31) ranged from 0.0048 to 6.8 mg/kg ww, with HQs ranging from < 1.0 to 1.7.

- **Total PCBs** – Total PCB HQs were > 1.0 in smallmouth bass samples from several locations across the Study Area, with HQs ranging from 1.5 to 7.1 in five exposure areas.

Two of the 38 smallmouth bass samples had a BEHP concentration greater than the NOAEL (no LOAEL was available). The NOAEL HQs were 3.3 and 9.1 in these two samples 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. BEHP was not identified as a COC for smallmouth bass because an HQ > 1.0 based on an unbounded NOAEL does not conclusively indicate unacceptable risks to smallmouth bass populations. Risk to smallmouth bass from BEHP is unknown and uncertain because of the lack of an acceptable LOAEL TRV. An evaluation of the distribution of BEHP concentrations in sediment is provided in Section 5.0 of the draft RI.

Northern Pikeminnow

Three fish tissue COPCs were identified for northern pikeminnow in the SLERA and refined screen. Two COPCs (mercury and total PCBs) had individual samples that resulted in 10th percentile LOAEL HQs > 1.0 (Table 7-7). HQs for these COPCs were calculated based on 1-mile exposure areas throughout the Study Area (Table 7-11). Insufficient data were available to calculate exposure area UCLs; therefore the maximum concentration was used to assess risks to northern pikeminnow.

Table 7-11. Northern Pikeminnow 1-Mile Exposure Area-Specific Tissue 10th Percentile LOAEL HQs

	LOAEL HQ[a]	
Exposure Area	**Mercury**	**Total PCBs**
RM 1.5 to RM 2.5	0.39	0.77
RM 2.5 to RM 3.5	0.39	0.77
RM 3.5 to RM 4.5	ND	ND
RM 4.5 to RM 5.5	0.74	0.47
RM 5.5 to RM 6.5	0.74	0.47
RM 6.5 to RM 7.5	**1.1**	**2.0**
RM 7.5 to RM 8.5	0.89	**1.1**
RM 8.5 to RM 9.5	0.89	**1.1**
Swan Island Lagoon	0.35	0.84
RM 9.5 to RM 10.5	ND	ND

Table 7-11. Northern Pikeminnow 1-Mile Exposure Area-Specific Tissue 10th Percentile LOAEL HQs

	LOAEL HQ[a]	
Exposure Area	**Mercury**	**Total PCBs**
RM 10.5 to RM 11.8	ND	ND

[a] The 10th percentile LOAEL HQ was calculated using the maximum concentration available from within each exposure area.

HQ – hazard quotient
LOAEL – lowest-observed-apparent-effect level
ND – no data
PCB – polychlorinated biphenyl
RM – river mile
UCL – upper confidence limit on the mean
Bold identifies HQs > 1.0.

Two COCs for were identified for northern pikeminnow: mercury and total PCBs. HQs on a composite sample-by-sample basis[72] are presented on Maps 7-6 and 7-7. The spatial distribution of these risk estimates are discussed below:

- **Mercury** – The mercury concentration in one sample was slightly elevated over the 10th percentile LOAEL (HQ = 1.1) in one exposure area (RM 6.5 to RM 7.5).
- **Total PCBs** – The total PCBs 10th percentile LOAEL HQ in northern pikeminnow samples ranged from 1.1 to 2.0 in three exposure areas (between RM 6.5 and RM 9.5).

7.1.4.3 Evaluation of Non-Study Area Concentrations

The following 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).

7.1.4.3.1 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 in the BERA. Two sculpin composite tissue samples[73] were available from the downstream reach (RM 0 to RM 1.9), and two sculpin composite tissue samples[74] were

[72] HQs were calculated and are presented on a sample-by-sample basis because the compositing area of northern pikeminnow was generally within a 1-mile area, which is consistent with the species' assumed exposure area.

[73] Samples were collected from the west and east bank at approximately RM 1.5, which provides limited spatial coverage of the downstream reach (spanning 2miles).

[74] Samples were collected from the west and east bank at approximately RM 12, which provides limited spatial coverage of the downtown reach (spanning 3.5 miles).

available from the downtown reach (RM 11.8 to RM 15.3) (Map 4-10). The non-Study Area sediment and tissue chemistry data from these reaches are presented in Attachment 4.

A CFD approach was used to evaluate the relationships between non-Study Area and Study Area data. A CFD of sculpin tissue composite concentrations within the Study Area, the downstream reach, and the downtown reach were plotted for sculpin COPCs (i.e., copper, total PCBs, and total DDx) with tissue 10th percentile LOAELs (Figures 7-3 through 7-5). 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. One sculpin sample collected from the downstream reach had a copper concentration slightly greater than the tissue 10th percentile LOAEL; however, the inclusion of this sample in the fish risk assessment would not change the risk conclusions because three Study Area sculpin tissue samples also have copper concentrations that are greater than the 10th percentile LOAEL.

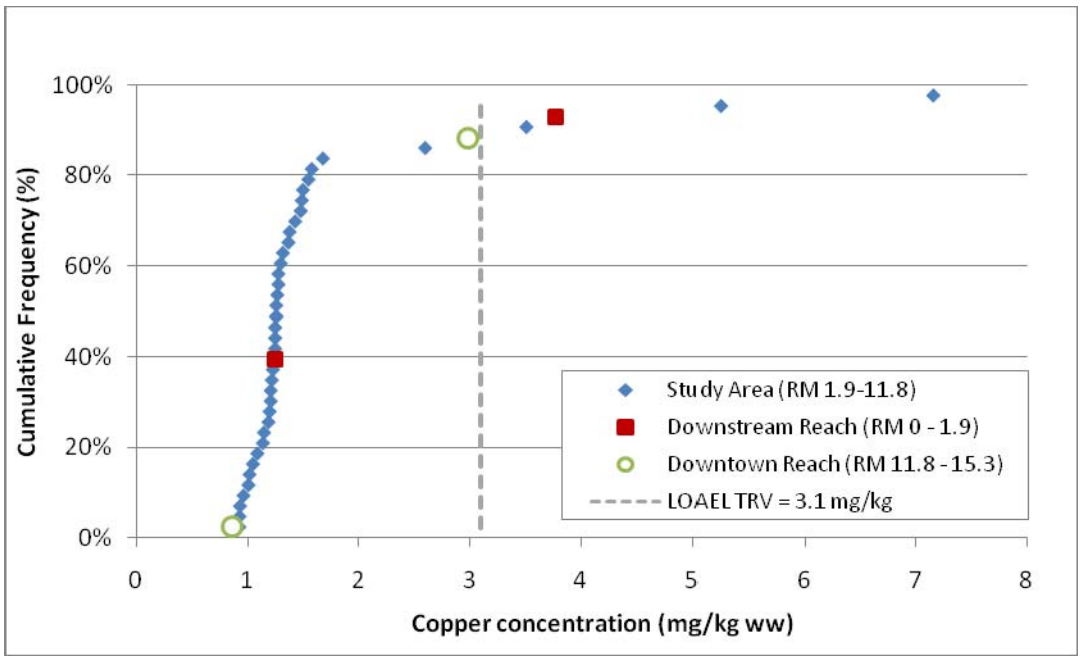

Figure 7-3. CFD of Sculpin Copper Concentrations

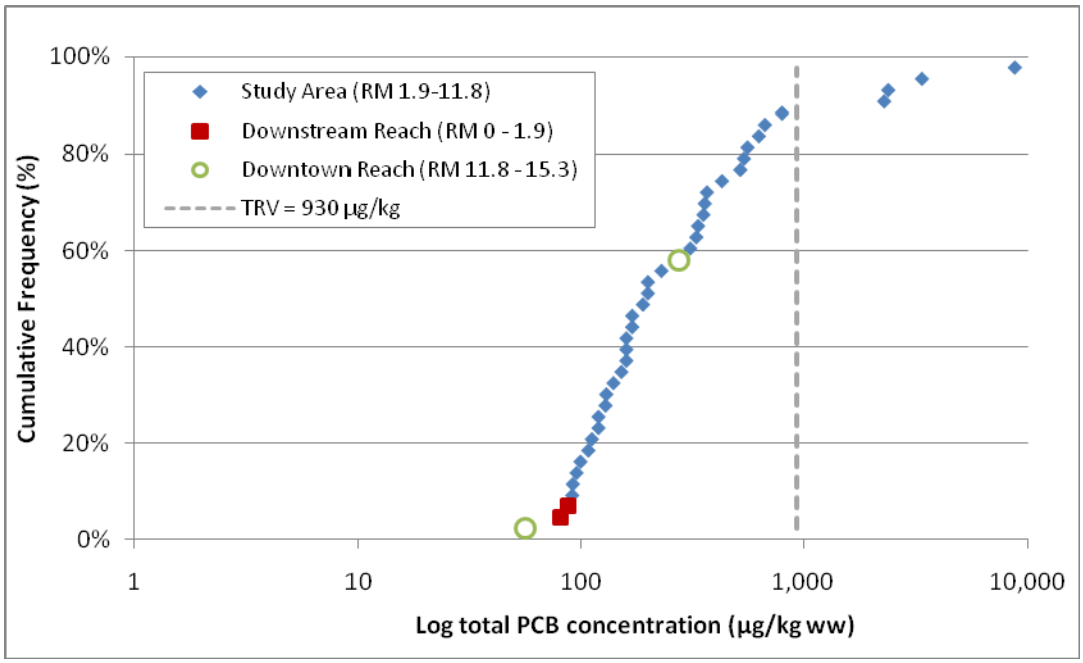

Figure 7-4. CFD of Sculpin Total PCB Concentrations

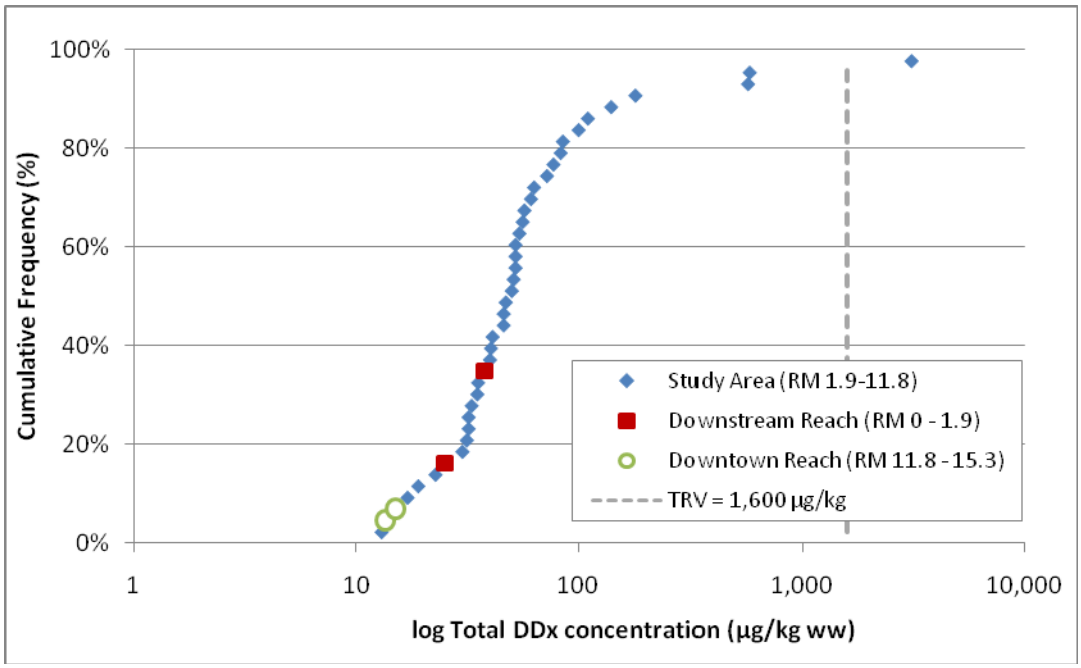

Figure 7-5. CFD of Sculpin Total DDx Concentrations

7.1.4.3.2 Tissue Data from the Upriver Reach

Tissue data for four fish receptor species were available from the upriver reach (from RM 15.3 to RM 28.4). Four juvenile Chinook salmon, three brown bullhead, six smallmouth bass, and four lamprey ammocoete samples were collected by LWG from the upriver reach between June 2002 and September 2006. The non-Study Area sediment and tissue chemistry data from these reaches are presented in Attachment 4.

For those COPCs for which no TRVs were available (i.e., aluminum), Study Area tissue chemical concentrations were compared to upriver tissue chemical concentrations (Figure 7-6). Aluminum concentrations in the Study Area were generally similar to or less than concentrations detected in samples in the upriver reach.

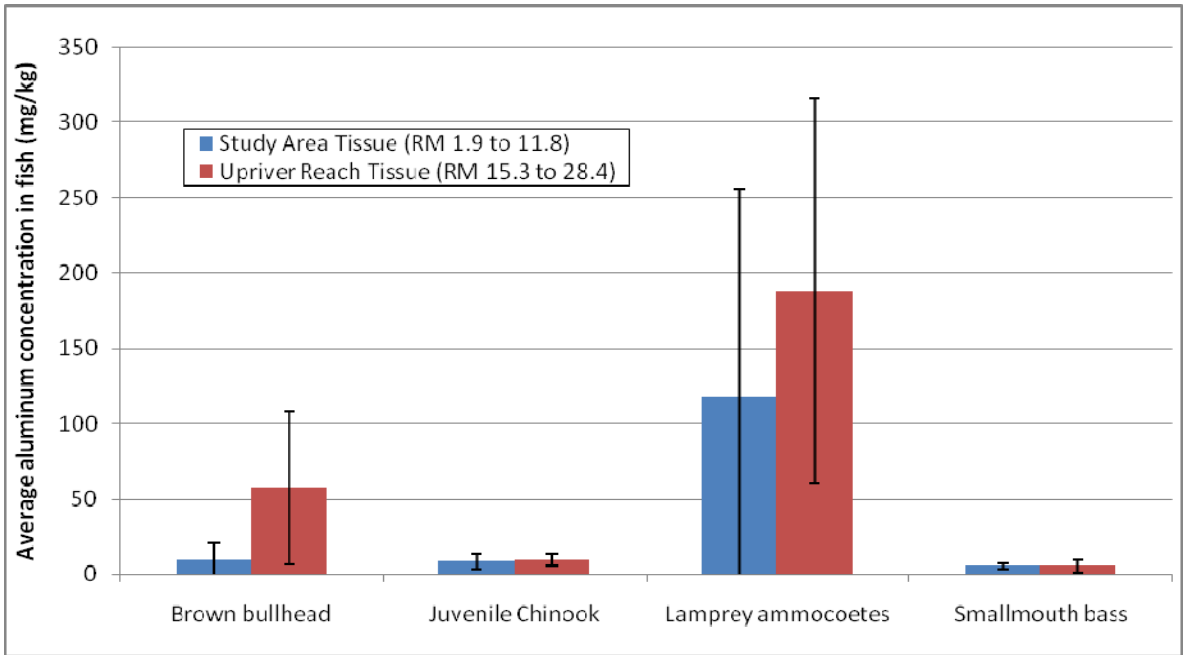

Figure 7-6. Aluminum Concentrations in Fish Tissue

Mercury is a regional issue in the Willamette River Basin (Oregonian 2006) (see Section 7.1.4.3.3). Mercury was identified as a COC for northern pikeminnow because one sample had a mercury concentration slightly elevated over the mercury 10th percentile LOAEL (HQ = 1.1). Study area tissue chemical concentrations were compared to upriver tissue chemical concentrations (Figure 7-7).Average mercury concentrations in brown bullhead, lamprey ammocoetes, and smallmouth bass were greater in tissues collected from upriver reach than in those collected from the Study Area. Upriver reach data were not available for northern pikeminnow and other ecological receptors (i.e., largescale sucker, peamouth, sculpin, and juvenile sturgeon). These upriver reach data indicate that elevated concentrations (i.e., greater than the 10th percentile LOAEL of 0.44 mg/kg ww) of mercury in certain fish species (i.e., smallmouth bass and lamprey) are found in other areas of the Willamette River.

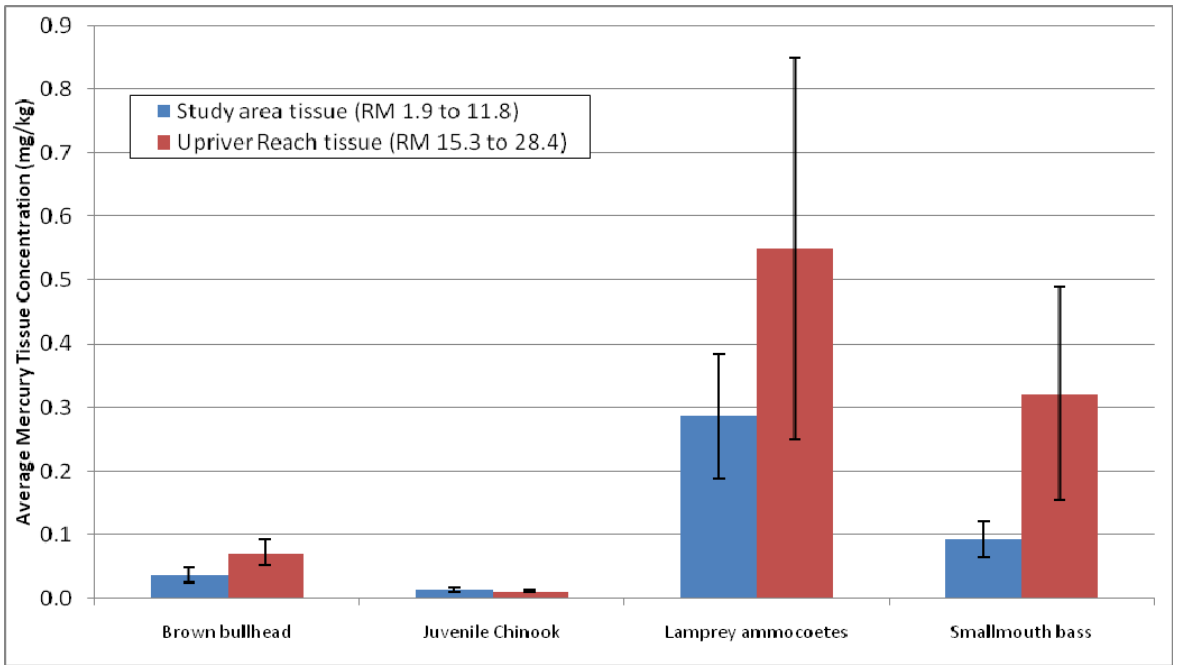

Figure 7-7. Comparision of Study Area and Upriver Reach Mercury Tissue Concentrations in Fish

Copper was identified as a COC for lamprey ammocoetes because the 5th percentile LOAEL HQ was 2.2, and for sculpin, for which the 10th percentile LOAEL HQs ranged from 1.1 to 2.3 in four individual samples. Study Area tissue copper concentrations also were compared to upriver tissue copper concentrations (Figure 7-8). 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. These data indicate that elevated concentrations of copper (i.e., greater than the 5th percentile and 10th percentile LOAEL of 2.8 and 3.1 mg/kg ww, respectively) are also in upriver reach lamprey and greater than those in lamprey in the Study Area. No sculpin data were available from the upriver reach.

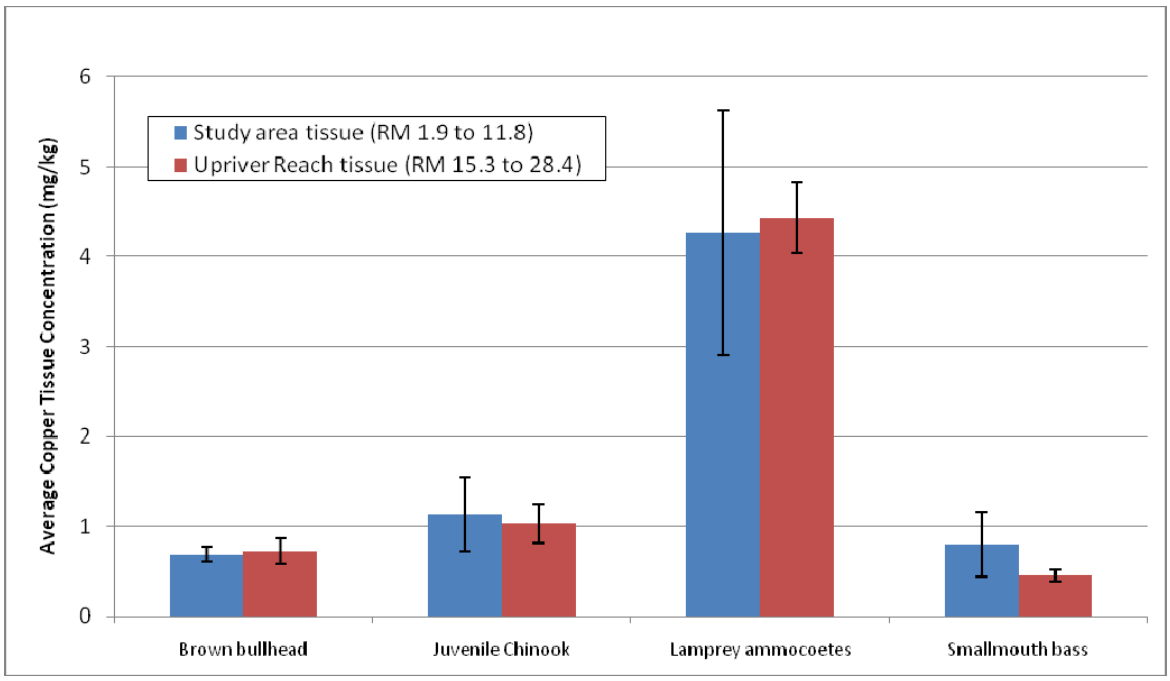

Figure 7-8. Comparision of Study Area and Upriver Reach Copper Tissue Concentrations in Fish

7.1.4.3.3 Additional Regional Data

Mercury is a regional issue in the Willamette River Basin (Oregonian 2006). Willamette-wide average mercury concentrations (as reported in Hope (2003)) are approximately 2 to 6 times as high as the same species' (i.e., carp, largescale sucker, northern pikeminnow and smallmouth bass) average mercury concentrations in the Study Area fish. For example, the average northern pikeminnow mercury tissue concentration in the Study Area was 0.28 mg/kg ww, whereas, the average reported northern pikeminnow concentration in the Willamette River was 0.60 mg/kg ww (Table 7-12).

Table 7-12. Average Fish Tissue Mercury Concentrations

	Fish Collected from the Study Area		Fish Collected from the Entire Willamette Basin[a]	
Species	**n**	**Average Concentration (mg/kg ww)**	**n**	**Average Concentration (mg/kg ww)**
Carp	15	0.045	64	0.28
Largescale sucker	6	0.068	135	0.22
Northern pikeminnow	6	0.28	95	0.60
Smallmouth bass	32	0.093	10	0.28

[a] As reported by Hope (2003).
ww – wet weight

7.1.4.4 Summary of Fish Tissue COCs

Five fish tissue COCs were identified: copper, lead, mercury, total PCBs, and total DDx. Table 7-13 presents the HQs and exposure and effects uncertainties for each of the COC-receptor pairs. In Section 7.6, the results of the fish tissue LOE were compared with the risk characterization results for the other fish LOEs (i.e., water and diet) to determine risk conclusions for fish.

Table 7-13. Summary of Fish Tissue COCs

Receptor by COC	Risk Exposure Area	HQ[a]	Exposure Uncertainty/Considerations	Effects Uncertainty
Copper				
Sculpin	RM 5.5, east; RM 10.3, west; RM 11.3, east	1.1 – 2.3	Similar copper concentrations were detected in the upriver reach and the Study Area for those species with upriver tissue available (none available for sculpin). Identification of COC and the range of HQs are based on individual samples.	TRV is highly uncertain because fish actively regulate tissue copper concentrations.
Pacific Lamprey ammocoetes	Site-wide	2.2	HQ is based on the maximum concentration (insufficient data to calculate side-wide UCL); similar concentrations were detected in tissue from the upriver reach.	TRV is highly uncertain because fish actively regulate tissue copper concentrations.
Lead				
Peamouth	Site-wide	2.7	HQ is based on the maximum concentration (there is insufficient data to calculate side-wide UCL).	TRV is highly uncertain because the 10th percentile LOAEL TRV is based on a single toxicity study.
Smallmouth bass	RM 9.5 to RM 10.5	280	Maximum HQ (280) appears to be a data outlier; the HQ for the other sample from RM 9.7 to RM 10.5 was 1.7. The HQ is based on maximum individual sample.	TRV is highly uncertain because the 10th percentile LOAEL TRV is based on a single toxicity study.
Mercury				
Northern pikeminnow	RM 6.5 to RM 7.5	1.1	Upriver reach tissue chemical concentrations were greater than mercury concentrations in the Study Area for those species with upriver tissue available (none available for pikeminnow). Willamette-wide northern pikeminnow average mercury concentrations (as reported in Hope (2003)) are approximately 2 times as high as average concentrations in northern pikeminnow from the Study Area. Identification of COC and HQ is based on maximum individual sample.	There are no key effects uncertainties.

Table 7-13. Summary of Fish Tissue COCs

Receptor by COC	Risk Exposure Area	HQ[a]	Exposure Uncertainty/Considerations	Effects Uncertainty
Total PCBs				
Largescale sucker	Site-wide	1.6	There are no key exposure uncertainties; HQ is based on a site-wide UCL.	The 10th percentile LOAEL TRV may overestimate toxicity because the lowest study-specific LOAELs included in the SSD have significant uncertainties.
Sculpin	RM 2.3 to RM 2.4, east; Willamette Cove; RM 11.3, east; additional exceedances from predicted locations[b]	2.5 – 9.4 (1.0 – 111[c])	Predicted HQs > 1.0 were more widespread than empirical HQs > 1.0; predicted HQs are based on mechanistic model. In general, the areas where predicted HQs were > 1.0 were near or at the same locations as detected sculpin tissue samples with chemical concentrations had HQs < 1.0. Empirical data are assumed to be more representative of tissue concentrations in Study Area sculpin than are predicted tissue chemical concentrations. The range of HQs is based on individual empirical samples.	The 10th percentile LOAEL TRV may overestimate toxicity because the lowest study-specific LOAELs included in the SSD have significant uncertainties.
Smallmouth bass	RM 1.5 to RM 2.5; RM 3.5 to RM 4.5; RM 6.5 to RM 7.5; Swan Island Lagoon; RM 10.5 to RM 11.5	1.5 – 7.1	The range of HQs is based on individual samples.	The 10th percentile LOAEL TRV may overestimate toxicity because the lowest study-specific LOAELs included in the SSD have significant uncertainties.
Northern pikeminnow	RM 6.5 to RM 9.5	1.1 – 2.0	The range of HQs is based on individual samples.	The 10th percentile LOAEL TRV may overestimate toxicity because the lowest study-specific LOAELs included in the SSD have significant uncertainties.

Table 7-13. Summary of Fish Tissue COCs

Receptor by COC	Risk Exposure Area	HQ[a]	Exposure Uncertainty/Considerations	Effects Uncertainty
Total DDx				
Sculpin	RM 7.1 to RM 7.4, west	1.9 (1.1 – 21[c])	Predicted and empirical HQ > 1.0 are co-located; predicted HQs > 1.0 along the west bank from approximately RM 7.1 to 7.4 and the empirical HQ is > 1.0 at a single composite location at RM 7.3 on the west bank. Predicted HQs are based on mechanistic model. Identification of COC and the range of HQs are based on individual empirical samples.	There are no key effects uncertainties.

[a] HQ is based only on exposure areas where HQs were > 1.0. HQs in all other exposure areas were < 1.0. HQs are based on 10th percentile or literature-based LOAELs for all receptors except for Pacific lamprey ammocoetes. HQs for Pacific lamprey ammocoetes are based on 5th percentile LOAELs or literature-based NOAELs.

[b] HQs were > 1.0 in predicted tissue concentrations in the same locations as the empirical tissue concentrations and in the following locations: International Slip, RM 4.1 on the east side, Slip 1, RM 5.8 on the east side, along RM 7.0 to RM 7.7 on the west side, Swan Island Lagoon, and along RM 8.8 to RM 9.7 on the west side.

[c] HQ is based on predicted tissue chemical concentrations.

BSAR – biota-sediment accumulation regression
COC – chemical of concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
PCB – polychlorinated biphenyl
RM – river mile
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
UCL – upper confidence limit on the mean

7.1.4.5 Evaluation of Non-Target Ecological Receptors

Per EPA (2008j), the fish species not identified as ecological receptors of concern (i.e., brown bullhead and black crappie) were evaluated as part of the fish tissue uncertainty assessment. This assessment is presented in this section.

Following 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 – chemical 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.

Two tissue COPCs (i.e., BEHP and total PCBs) were identified for brown bullhead. Attachment 4 presents a summary of these COPC concentrations in individual brown bullhead tissue samples. The individual composite sample total PCB concentrations were compared to the fish 10th percentile LOAEL. Only one of the six brown bullhead samples had total PCBs higher than the fish 10th percentile LOAEL (HQ = 1.8). The total PCBs site-wide UCL concentration of 1,400 µg/kg was also greater than the 10th percentile LOAEL (HQ = 1.5). No LOAEL was available for BEHP; however, all brown bullhead concentrations were less than the BEHP literature-based NOAEL.

Total PCBs were identified as a COC 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 ecological receptors were determined to be protective of black crappie and brown bullhead.

7.2 DIETARY ASSESSMENT

The dietary-dose assessment was a LOE for evaluating risks from exposure to metabolized and regulated chemicals (i.e., metals[75] 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 sediments. Toxicity thresholds were expressed as concentrations in tissue and sediment that were back-calculated from dietary-dose thresholds using receptor-specific exposure assumptions.

HQs were calculated in a step-wise process:

HQs were calculated on a sample-by-sample basis for each dietary component (assuming exclusive contribution to the diet).

HQs were derived assuming exclusive contribution to the diet for each individual dietary component (including sediment) within relevant exposure areas.[76]

HQs were derived assuming the ingestion of multiple prey (using receptor-specific assumptions) within relevant exposure areas. Receptor diets were based on the prey portions reported in literature.

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 each receptor, COPCs with HQs > 1.0 in the third step were retained as COCs. For each COC, the magnitude of HQs, the spatial distribution and frequency of HQ > 1.0, the results of other LOEs (e.g., tissue residue, surface water) when applicable, and the associated exposure and effects assumptions were evaluated to arrive at risk conclusions for fish.

The details of this assessment are presented as follows:

- Section 7.2.1 presents a summary of the COPCs identified for all receptors that were evaluated in the dietary risk evaluation.

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[75] Per EPA (EPA 2008f), mercury was included as a COPC in the dietary-dose evaluation, although it is not a metabolized or regulated chemical.

[76] The relevant exposure area for sculpin was considered roughly equivalent to the sampling area for prey. Sculpin diets were evaluated only on a sample-by-sample basis.

- Section 7.2.2 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 7.2.3 presents a summary of the effects data. Effects data in this assessment 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 7.2.4 presents the risk characterization results, COC-receptor pairs, and associated uncertainties. These COCs are further assessed in the fish risk conclusions (Section 7.6). The individual sample-by-sample and dietary component assessment is presented in Attachment 12.

Figure 7-9 presents a flowchart of the fish dietary assessment section organization.

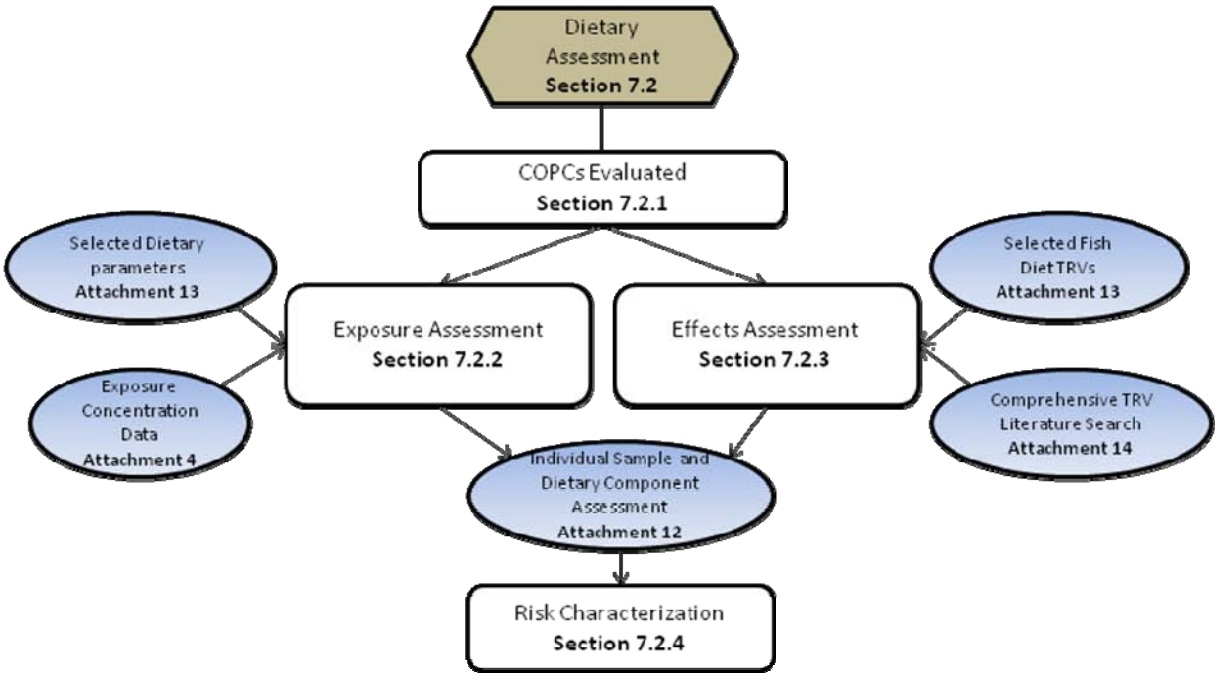

Figure 7-9. Overview of Fish Dietary Assessment Section Organization

7.2.1 COPCs Evaluated

COPC-receptor pairs were identified in the SLERA and refined screen (Attachment 5). Table 7-15 presents the fish dietary COPCs. All of these COPC were evaluated, with the exception of butyltin ion, dibutyltin ion, and tetrabutyltin, which could not be evaluated because no LOAEL was available from the literature for butyltins other than tributyltin.

Table 7-15. Dietary Fish 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							
Butyltin 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 – chemical 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). Thus, the risk associated with these chemicals to fish receptors is unknown.

Table 7-16. Fish Dietary COIs with No Screening-Level Threshold

COI	
Metals	
Antimony	Nickel
Chromium	Thallium
Manganese	
PAHs	
1-Methylnaphthalene	Dibenzothiophene
2-Methylnaphthalene	Perylene
Benzo(e)pyrene	Alkylated PAHs

COI – chemical of interest
PAH – polycyclic aromatic hydrocarbon

7.2.2 Exposure Assessment

This section presents the methods and assumptions that were used to estimate fish dietary exposures to COPCs. Exposure concentrations in this assessment were represented by concentrations in prey tissue and surface sediment. Prey tissue and sediment concentrations were compared to receptor-specific toxicity thresholds for each fish COPC-receptor pair.

7.2.2.1 Exposure Concentrations

This section presents a summary of the methods used to derive EPCs in prey tissue and sediment to calculate fish risk estimates using the dietary-dose approach. Prey tissue EPCs were evaluated to assess dietary risks. Sediment EPCs were also evaluated to address potential exposure via incidentally ingested sediment using the dietary-dose approach. Tissue and sediment EPCs were represented by concentrations detected in composite samples collected from the Study Area or from tissue samples following laboratory bioaccumulation testing.

The fish risk evaluation using the dietary LOE involved multiple steps, in which the data used to represent EPCs varied. Figure 7-10 presents the process used for developing EPCs in the fish assessment.

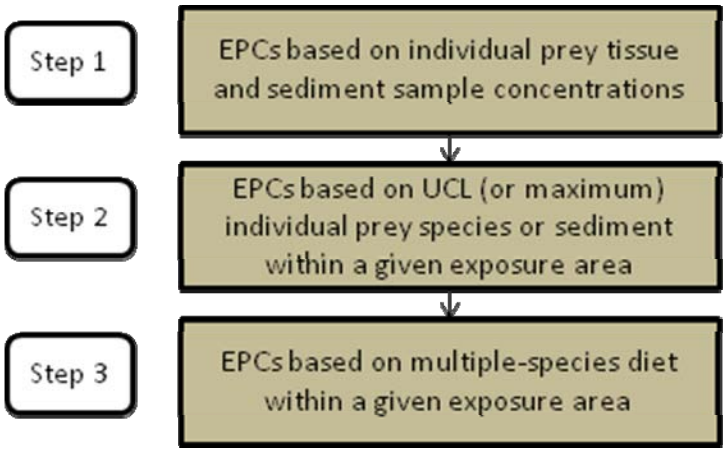

Figure 7-10. EPC Derivation Process for Fish

The general EPC derivation process for evaluating dietary exposure for each of these iterative risk characterization steps is detailed in the BERA analysis plan of EPA's Problem Formulation (Attachment 2). Details of how the derivation process was implemented were vetted and approved by EPA during a meeting on May 12, 2008, prior to initiating the BERA. The process was as follows:

Step 1 – EPCs were first represented by individual prey tissue composite and sediment sample concentrations from throughout the Study Area to evaluate dietary risks on a sample-by-sample basis per EPA's Problem Formulation (Attachment 2). COPC concentration data for all individual samples are presented in the Attachment 4.

EPCs based on individual tissue composite and sediment samples were used to calculate HQs using the following equations, respectively:

$$HQ = \frac{EPC}{TTC} \quad \text{and} \quad HQ = \frac{EPC}{TSC} \qquad \textbf{Equation 7-2}$$

Where:

HQ = hazard quotient
EPC = exposure point concentration
TTC = threshold tissue concentration
TSC = threshold sediment concentration

Step 2 – Prey tissue and sediment EPCs were then calculated within receptor-specific exposure areas. Table 7-17 presents the exposure area assumptions used to derive EPCs for each fish receptor. The rationale for these exposure area assumptions is presented in Attachment 13.

Table 7-17. Receptor-Specific Exposure Area Assumptions for Dietary EPC Calculations

Receptor(s)	Exposure Scale[a]	Exposure Areas	EPC Basis
Largescale sucker, juvenile white sturgeon, juvenile Chinook salmon, peamouth	Site-wide	RM 1.9 to RM 11.8 (Study Area)	EPCs were calculated as Study Area UCL[b] concentrations for all prey and sediment.
Sculpin	0.1 mile	Individual sampling locations	EPCs were represented on a sample-by-sample basis for all prey and sediment (tissue sample compositing area is roughly equal to exposure scale).
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 11	EPCs were calculated within each 1-mile exposure area for each prey species and sediment 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 peamouth).

[a] The rationale for the selected exposure areas is presented in Attachment 13.
[b] Where insufficient data were available to calculate a UCL, the maximum concentration was used to represent the EPC.
[c] Insufficient data were available to calculate a UCL within a 1-mile exposure area for smallmouth bass and northern pikeminnow; therefore, the maximum concentration within a 1-mile exposure area was used to represent the EPC.
COPC – chemical of potential concern
EPC – exposure point concentration
RM – river mile
UCL – upper confidence limit on the mean

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. The ProUCL software used for this analysis allows detected and undetected values to be considered and creates interpolated values for non-detects based on the perceived distribution of the detected concentrations. Once any necessary interpolation is performed, the software conducts an analysis of the data to determine the most appropriate UCL and makes a recommendation. The UCL recommended by ProUCL was used as the EPC for the risk calculations. A minimum of six detected concentrations was required to derive a UCL (EPA 2007f). In the case where an insufficient number of detected data values was available ($n \geq 6$), the maximum concentration[77] was used to represent the 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 Equation 7-2.

Uncertainty is associated with the use of maximum concentrations to represent prey EPCs. Given the small exposure scales evaluated for smallmouth bass and northern pikeminnow, only limited data were available to derive the UCL concentrations for each prey species within 1-mile exposure areas; in most cases, one to four samples were available per prey species in a given 1-mile exposure area. When a limited number of samples are available, the UCL cannot be reliably estimated and the maximum concentration represents a reasonable conservative estimate of the mean. The use of maximum concentrations to represent prey EPCs may result in an over or underestimate of risk.

Uncertainty Associated with Juvenile White Sturgeon Exposure Scale

A site-wide exposure scale of the10-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 uncertainty associated with the site-wide exposure scale of the juvenile white sturgeon may likely overestimate risks to sturgeon, assuming that the contribution of exposure to contaminants to sturgeon outside the Study Area is less than within the Study Area.

[77] When the maximum concentration was a non-detected value, the full detection limit was used to represent the EPC.

Step 3 – In the first two steps of risk characterization, dietary exposure was evaluated for each prey species individually. In order to estimate dietary risks that account for the ingestion of multiple prey, dietary portions were assigned to each prey item for a given receptor. Prey portions were used to derive total HQs using the following equation:

$$HQ_{total} = \left(\sum_{i=1}^{n} HQ_i F_i \right) + (HQ_{sed}) \qquad \textbf{Equation 7-3}$$

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

Prey portions that were selected were based on the diets reported in regional literature studies and are presented in Section 7.2.2.2. Risk conclusions were based on the exposure assumptions of Step 3.

The Evaluation of HQs on a Sample-by-Sample Basis

Dietary risks were evaluated on a prey sample-by- prey sample basis per EPA's Problem Formulation (Attachment 2). 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 potential COCs when no individual prey sample HQs are > 1.0; but if individual sample HQs are > 1.0, the appropriate conclusion is that the exposure assumptions should be refined 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 ultimately 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.

7.2.2.2 Use of Exposure Parameters

Dietary exposure parameters were used to derive receptor-specific toxicity thresholds. Receptor-specific toxicity thresholds were represented by threshold tissue concentrations (TTCs) (expressed as mg/kg ww) in prey and threshold sediment concentrations (TSCs) (expressed as mg/kg dw). This subsection presents how receptor-specific TTCs and TSCs were derived for the dietary evaluation.

EPA's Problem Formulation (Attachment 2) specified that TTCs and TSCs should be back-calculated from dietary-dose TRVs using receptor-specific parameters (i.e., body weight, prey ingestion rate, incidental sediment ingestion rate and site use) compiled from

general and region-specific literature. The following equations were used to develop receptor-specific TTCs and TSCs from dietary-dose TRVs:

$$TTC = \frac{TRV_{diet}}{\left(\frac{FIR}{BW} \times SUF\right)}$$ **Equation 7-4**

Where:

TTC = threshold tissue concentration (mg/kg ww)

TRV_{diet} = dietary-dose toxicity reference value (mg/kg bw/day)

FIR = food ingestion rate (kg ww food/day)

BW = body weight (kg)

SUF = site use factor (unitless); fraction of time that a receptor spends foraging at the site relative to the entire home range and based on consideration of seasonal use

And:

$$TSC = \frac{TRV_{diet}}{\left(\frac{SIR}{BW} \times SUF\right)}$$ **Equation 7-5**

Where:

TSC = threshold sediment concentration (mg/kg dw)[78]

TRV_{diet} = dietary-dose toxicity reference value (mg/kg bw/day)

SIR = incidental sediment ingestion rate (kg dw sediment/day)

BW = body weight (kg)

SUF = site use factor (unitless); fraction of time that a receptor spends foraging at the site relative to the entire home range and based on consideration of seasonal use

Dietary-dose TRVs were derived from the literature using the following equation:

$$TRV_{diet} = \frac{(IR \times C_{diet})}{BW}$$ **Equation 7-6**

[78] Note that the TSC applies only to the incidental ingestion of sediment.

Where:

TRV_{diet} = dietary-dose toxicity reference value (mg/kg bw/day)

IR = ingestion rate (kg food/day)

C_{diet} = chemical concentration in diet (NOAEL or LOAEL) as measured and reported in the toxicity study (mg/kg ww)

BW = body weight (kg)

When using the back-calculation approach, tissue and sediment concentrations can be compared directly to TTCs and TSCs. The alternative would be to forward-calculate dietary-dose estimates, using the same exposure parameters and equations. It should be noted that the two methods are mathematically equivalent (i.e., they result in the same answers).

Dietary exposure parameters are presented in Section 7.2.2.3; selected dietary-dose TRVs and back-calculated TTCs and TSCs are presented in Section 7.2.3.

7.2.2.3 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.

7.2.2.3.1 Exposure Parameters

Body weights, food ingestion rates, sediment ingestion rates, and site use factors vary among fish receptors. Table 7-18 presents the dietary exposure parameters for fish receptors. Details and the rationale for the selected receptor-specific exposure parameters and uncertainties are presented in Attachment 13.

Table 7-18. Exposure Parameters Used for Fish Dietary Risk Calculations

Receptor	BW (kg)[a]	FIR (kg ww/day)[b]	SI (%)[c]	SIR (kg dw/ day)[d]	% Moisture in Prey	SUF
Largescale sucker	0.79	0.040	8%	0.00048	85%[f]	1.0
Juvenile white sturgeon	7.6	0.28	8%; 56%[e]	0.0033; 0.023[e]	85%[f]	1.0
Juvenile Chinook salmon	0.012	0.0011	1%	0.0000024	79%[g]	1.0
Peamouth	0.10	0.0072	5%	0.000075	79%[g]	1.0
Sculpin	0.020	0.0017	5%	0.000018	79%[g]	1.0
Smallmouth bass	0.40	0.022	1%	0.000058	74%[h]	1.0
Northern pikeminnow	0.56	0.030	1%	0.000078	74%[h]	1.0

[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 station near the SP&S Railroad Bridge).

[c] Percent of incidental sediment ingestion.

[d] SIR = FIR x SI. The SIR was calculated as a percent of the FIR on a dw basis. The dw FIR was calculated based on the following equation: FIR (dw) = FIR (ww) x (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
SUF – site use factor
ww – wet weight

Body weights were based on the average body weights measured in individual fish from the Round 1, 2, and 3 sampling efforts. Measured food ingestion rates were not available for the fish receptors and were estimated using the equation presented in Arnot and Gobas (2004):

$$FIR = (0.022 \times BW^{0.85}) \times \exp^{(0.06*T)}$$ **Equation 7-7**

Where:

FIR = food ingestion rate (kg ww/day)
BW = body weight (kg)
exp = 2.71828
T = temperature (degrees Celsius)

Dietary fish ingestion rates were estimated using Equation 7-7 assuming a temperature of 13.4º C, which was based on the average of data collected by ODEQ from a station 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 7.2.4.4.

To determine sediment ingestion rates on a dry weight basis, the food ingestion rate (based on wet weight) was converted to dry weight based on the average percent moisture across relevant prey (Table 7-18).

$$SIR = (FIR \times F_{solids}) \times SI$$ **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. Thus, the portion of 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 12 for details).

Fish habitat areas were not defined for the Study Area and therefore, habitat was not factored into the development of site use factors or exposure areas. All exposure areas throughout the Study Area were assumed to provide some type of fish habitat; however, a lack of habitat (and foraging fish) in some areas could contribute to an overestimation of exposure.

7.2.2.3.2 Dietary Prey Assumptions

Table 7-19 presents the prey species in the BERA dataset that were used to derive prey tissue EPCs. Prey included fish and invertebrate species (i.e., largescale sucker, carp, peamouth, sculpin, northern pikeminnow, crayfish, clams, epibenthic invertebrate tissue, and mussels) collected in the Study Area and invertebrate species (i.e., clams and worms) that underwent laboratory bioaccumulation testing. Juvenile white sturgeon and juvenile Chinook salmon stomach content tissue chemical concentration data were also available. The selected prey species for each receptor were based on information from the literature. Details and the rationale for the assumptions for selected prey species and associated uncertainties are presented in Attachment 13.

Table 7-19. Receptor-Specific Prey Species Used to Derive Risk Estimates Based on Single Prey Consumption

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 first 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.

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 7-10).

Uncertainties Associated with Using Laboratory Bioaccumulation Testing to Represent Prey Chemical Concentrations

There is uncertainty 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 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 affecting bioavailability and uptake. Clam and worm tissue concentrations of neutral organic COPCs were adjusted to reflect 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 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. It is assumed that tissues of field-collected clams are more representative of field conditions than are laboratory-exposed tissues.

For those COPC-receptor pairs that were retained through the third step in the process (Figure 7-10), portions of individual prey were varied to better represent multi-species diets presented in the literature. Table 7-20 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 sucker chemical concentrations). The impact of varying prey portions on the estimated HQs was evaluated as part of the uncertainty analysis (Section 7.2.4.4). It should be noted that stomach content tissues for juvenile Chinook salmon and juvenile white sturgeon were not evaluated beyond the first step of risk characterization, as the maximum concentrations of the stomach tissues did not exceed the TRVs for any COPCs (see Attachment 12).

Table 7-20. 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	Small-mouth 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 invertebrates			0.30[h] and 1.0		0.40[h]		

Table 7-20. 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	Small-mouth Bass	Northern Pike-minnow
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, one based on 100% ingestion of epibenthic invertebrates.
[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 (e.g., juvenile Chinook salmon and peamouth) with significant portions of pelagic (water column) prey species, risk estimates are uncertain due to a lack of chemistry data for some COPCs. The pelagic prey component was represented by benthic invertebrate organisms (i.e., field clams or laboratory-exposed worms) for those COPCs for which no chemistry data were available for epibenthic inverterbrates. No TBT data were available for epibenthic invertebrate tissue; therefore, the TBT risk estimates for juvenile Chinook salmon and peamouth are highly uncertain.

Bioconcentration factors can be used to estimate pelagic prey concentrations; however, bioconcentration factors are generally not available for small pelagic invertebrates. EPA (2003a) reported freshwater TBT bioconcentration factors 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 these species with available BCFs represent appropriate pelagic prey for selected pelagic feeding fish receptors; however, the wide range of reported BCFs demonstrates the variability of TBT uptake from the water column in aquatic organisms.

7.2.2.4 Exposure Concentrations

The exposure assessment was an iterative process. If a particular COPC was found to not pose a potential unacceptable risk to a fish receptor, it was not carried forward any further in the process. If an EPC represented by a single sample led to a no risk conclusion (i.e., all Study Area HQs were < 1.0) for a particular COPC-receptor pair, that EPC was the final exposure estimate derived. If an EPC represented by a single sample led to a potential risk conclusion (i.e., at least one sample had an HQ > 1.0), then the COPC was carried forward to the next step in the evaluation process, at which point an EPC was derived by averaging across individual single-species prey tissue samples (during the second step), or by accounting for prey fractions (during the third step). Attachment 4 provides all EPCs (expressed as tissue and sediment concentrations) for all COPC-receptor pairs for the multiple exposure steps.

7.2.3 Effects Assessment

This section presents the TRVs used to characterize effects for fish COPC-receptor pairs and the uncertainties associated with these selected values. Dietary-dose TRVs (expressed as mg/kg bw/day) were based on LOAELs and NOAELs derived from the toxicological literature using Equation 7-6. Dietary-dose TRVs were used to derive receptor-specific TTCs and TSCs.

Per EPA's Problem Formulation (Attachment 2), LOAELs were used to assess effects on all fish receptors evaluated at the population-level[79] except juvenile Chinook salmon. As directed in EPA's Problem Formulation (Attachment 2), fish receptors that are threatened, endangered, otherwise protected under federal law, or are of particular cultural significance were assessed using a NOAEL. This status applied only to juvenile Chinook salmon; therefore, NOAELs were used to assess COPC effects on juvenile Chinook salmon.

7.2.3.1 Selected Dietary TRVs

EPA (2008f) provided the dietary NOAELs and LOAELs for fish, which were based on an extensive search of the available toxicological literature. The TRVs selected for each fish diet COPC were 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 diet TRVs for all COPCs. The fish dietary TRVs adopted for this BERA, as well as the key uncertainties associated with each of the selected TRVs are presented in Table 7-21.

[79] There is high uncertainty associated with the use of LOAELs to assess effects to populations, as LOAELs are based on organism-level effects. The magnitude and frequency of LOAEL exceedances as well as the endpoints used to derive the LOAEL were evaluated to determine whether the potential for population-level effects may exist. See Section 7.2.4 for further discussion of how LOAELs were used in risk characterization.

Table 7-21. Fish Dietary-Dose TRVs

COPC	TRV (mg/kg bw/day) NOAEL	TRV (mg/kg bw/day) LOAEL	Source	Key Uncertainty
Metals				
Cadmium	0.002[a]	0.01	Kim et al. (2004); Kang et al. (2005)	The selected 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.
Copper	0.24	0.48	Murai et al. (1981)	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).
Mercury	0.005	0.013	Matta et al. (2001)	
Butyltins				
TBT	0.00042[a]	0.0021	Shimasaki et al. (2003)	Highly uncertain; limited toxicity data were available (TRVs are based on the only available study); NOAEL was extrapolated from the LOAEL; ingestion rate was not reported in the study used to derive the dose TRV (generic estimates were used).
PAHs				
Benzo(a)pyrene	0.66	1.4	Rice et al. (2000)	Limited toxicity data were available.
Total PAHs	6.1	18	Meador et. al. (2006)	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) plus seven additional PAHs.

bw – body weight
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 food ingestion rate. There was a limited availability of fish feeding rates and body weights reported in the toxicological studies; when toxicological studies did not report fish body weight or ingestion rate, these assumptions were derived from other literature sources. Wildlife dose-based TRVs are frequently used in ecological risk assessments, and standard ingestion rates and body weights are available; however, for fish, 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.

7.2.3.2 Back-Calculated TTCs and TSCs for Fish

Once dietary TRVs had been selected, receptor-specific TTCs and TSCs were back-calculated using receptor-specific parameters (i.e., body weight, biota [prey] ingestion rate, and incidental sediment ingestion rate). This method of back-calculation was used per EPA (2008j) so that TTCs and TSCs could be compared directly to empirical concentrations of COPCs in prey tissues or sediment concentrations, respectively, (without the need to convert the empirical data into dose units), on a sample-by-sample and medium-specific basis. Equations 7-4 and 7-5 were used to develop receptor-specific TSCs and TTCs, respectively. Tables 7-22 and 7-23 present the receptor-specific TTCs and TSCs, respectively, derived for all fish dietary COPC-receptor pairs.

Table 7-22. Calculated Prey TTCs for Fish COPC-Receptor Pairs

	Prey TTC													
	Largescale Sucker		Juvenile 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 (mg/kg ww)														
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 (µg/kg ww)														
Tributyltin ion	8.3	41.5	11.4	57	4.58	22.9	5.83	29.2	4.94	24.7	7.64	38.2	7.84	39.2
PAHs (µg/kg ww)														
Benzo(a)pyrene	NA	NA	17,900	38,000	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Total PAHs	120,000	356,000	166,000	489,000	NA	NA	84,700	250,000	71,800	212,000	NA	NA	NA	NA

COPC – chemical of potential concern
LOAEL – lowest-observed-adverse-effect level
NA – not applicable (not a COPC-receptor pair)
NOAEL – no-observed-adverse-effect level
PAH – polycyclic aromatic hydrocarbon
TTC – threshold tissue concentration
ww – wet weight

Table 7-23. Calculated TSCs for Fish COPC-Receptor Pairs

	TSC													
	Largescale Sucker		Juvenile 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 (mg/kg dw)														
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	3310	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 (µg/kg dw)														
Tributyltin ion	691	3,460	139 (967)[a]	694 (4,840)[a]	2,100	10,500	560	2,800	467	2,330	2,900	14,500	3,020	15,100
PAHs (µg/kg dw)														
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 (and 8% incidental ingestion in parenthesis).
COPC – chemical of potential concern
LOAEL – lowest-observed-adverse-effect level
NA – not applicable (not a COPC-receptor pair)
NOAEL – no-observed-adverse-effect level
PAH – polycyclic aromatic hydrocarbon
TSC – threshold sediment concentration
ww – wet weight

7.2.4 Risk Characterization

This section presents the risk characterization process and results for the dietary LOE for fish. Section 7.2.4.1 presents the overall approach used to characterize the risk to fish receptors via the dietary LOE. Section 7.2.4.2 presents the risk characterization results, uncertainty evaluation, and COCs for each fish receptor. Section 7.2.4.3 presents a summary of risk conclusions and dietary COCs for all fish receptors. Dietary COCs, along with COCs from other LOEs were further evaluated and are presented in the fish risk conclusions (Section 7.6). Section 7.2.4.4 presents the uncertainty risk evaluation for the fish dietary assessment.

7.2.4.1 Risk Characterization Process

A deterministic risk characterization was conducted to characterize dietary risks to fish receptors. A risk characterization of fish COPC-receptor pairs based on the dietary LOE was conducted using an HQ approach that integrated exposure and effects data. Multiple HQs were derived using various assumptions in order to implement the methods presented in EPA's Problem Formulation (Attachment 2).

As described in Section 7.2.2.1, HQs were calculated using a multi-step process. 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 are presented in Attachment 12 and were used to narrow the list of COPCs for evaluation in the third step. COPCs with HQs > 1.0 in the third step were retained as COCs. The third step was used to define COCs because it represents the most reasonable assumptions appropriate for risk characterization.

The multi-step process is presented below:

Step 1 – Evaluation of Individual Samples – In the first step, HQs were evaluated on a sample-by-sample basis. HQs were derived for individual prey tissue and sediments using Equation 7-2.

Step 2 – Evaluation of Relevant Exposure Areas – In the next step, HQs were calculated within relevant exposure areas.[80] For those COPC-receptor pairs that resulted in HQs > 1.0 in Step 1, prey species and sediment HQs were derived within relevant exposure areas using Equation 7-2 in comparison to tissue and sediment UCLs. If insufficient data were available to derive a UCL (i.e., fewer than six detected concentrations were available), HQs were based on maximum prey species or sediment chemical concentrations. Receptor-specific EPCs for individual exposure areas were calculated based on the assumptions presented in Table 7-15.

Step 3 – Evaluation of Multiple Prey Within Relevant Exposure Areas – In order to account for the ingestion of multiple prey species, diet fractions were assigned to individual

[80] The relevant exposure area for sculpin was considered roughly equivalent to the sampling area of prey. Sculpin diets were evaluated only on a sample-by-sample basis.

prey species within an exposure area to derive total HQs using Equation 7-3. Prey portions were based on the diets reported in regional literature studies (Attachment 13). If no data were available for a given prey species, a surrogate species was used.

COCs were identified based on those COPCs that resulted in HQs > 1.0 based on Step 3.[81] Once COCs were determined, an evaluation was conducted for all COCs to identify the quantitative risk results (i.e., magnitude, spatial distribution, and frequency of HQs), underlying uncertainties of exposure and effects data, agreement of HQs based on other LOEs (where applicable), and comparison of background concentrations. The evaluation of COCs is discussed in the fish risk conclusions (Section 7.6).

Figure 7-11 presents the general risk characterization process used to evaluate dietary COPCs and identify fish COCs.

[81] For sculpin, COCs were identified as those COPCs with HQs > 1.0 based on a sample-by-sample basis.

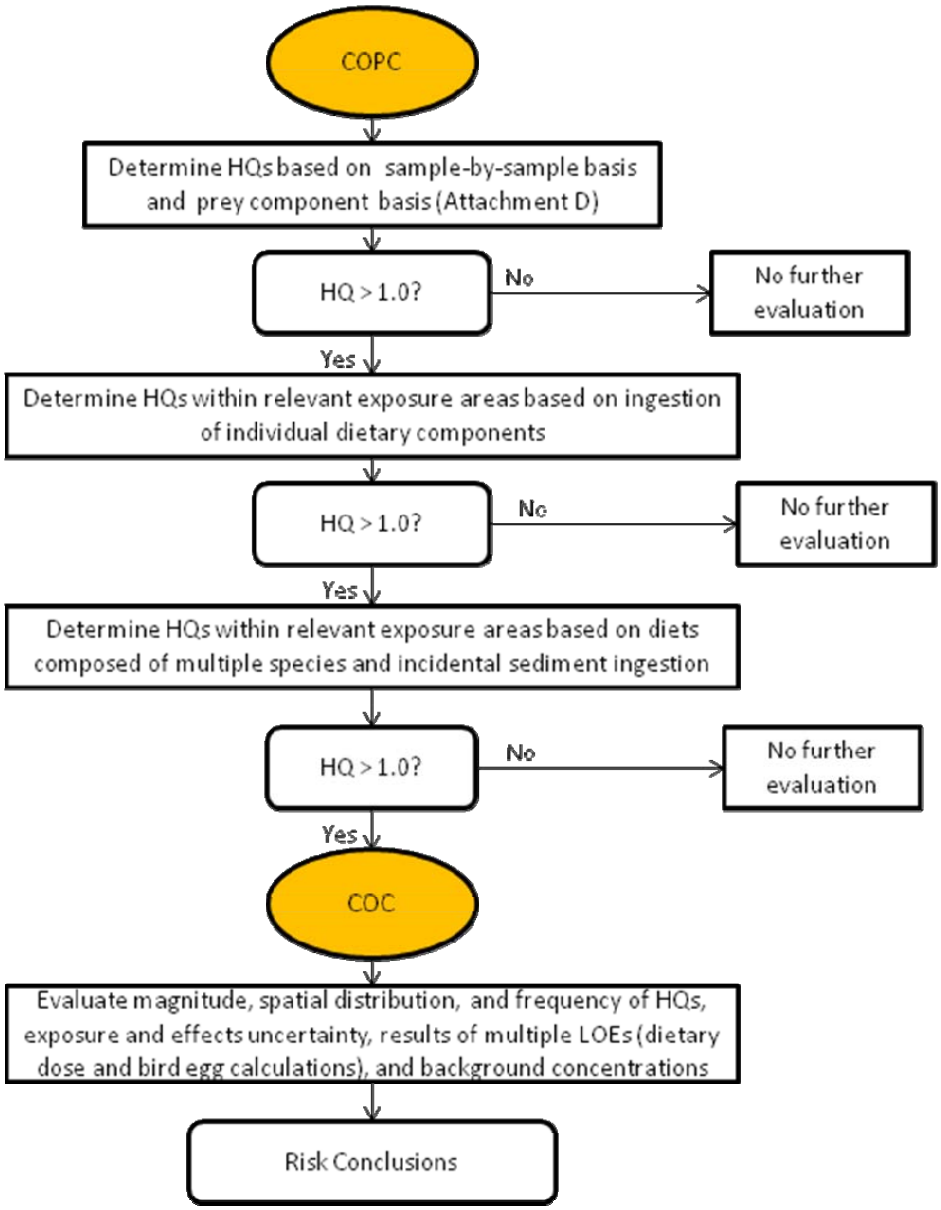

Figure 7-11. General Fish Risk Characterization Process

Determination of COCs

COCs were those COPCs with HQs > 1.0 based on ecologically relevant exposure scales (as presented in Table 7-3), regardless of the uncertainties associated with the exposure and effects data or assumptions. COPCs were not identified as COCs based on individual sample or individual prey component HQs because risks indicated by individual samples are not ecologically relevant.

The uncertainties associated with risk estimates for individual COCs, the spatial distribution of COC exceedances, the magnitude of exceedances, and the type of effect threshold all play a role in identifying whether chemicals pose a population-level risk. The likelihood that a COC with a limited spatial distribution of exceedances and a low HQ poses significant risks to populations is low. COCs with broader distribution and higher magnitude of exceedances generally have a greater potential for posing population-level risks. The type of effect threshold endpoint (i.e., survival, growth or reproduction) affects the interpretation of HQs simply because effects on the survival of individuals have different implications for populations than do effects on the growth of individuals. Effects on the reproductive success of individuals have different implications for populations than do effects on either survival or growth.

Even for a particular endpoint (e.g., reduced survival), a particular HQ (e.g., HQ = 5) has different implications for each COC-receptor pair, because the exposure-response relationship of one COC-receptor pair differs from that of other COC-receptor pairs. Short of developing structured population models and incorporating the effects of COC-receptor-specific exposure-response data, the demographic (i.e., population-level) consequences of any particular HQ simply cannot be quantified.

Populations are resilient to environmental stressors (Barnthouse et al. 2009). By corollary, it is unreasonable to assume that survival, growth, or reproduction HQs > 1.0 will necessarily be detrimental to the populations whose individuals are potentially affected.

7.2.4.2 Risk Characterization Results and Uncertainty Evaluation

The HQ results from the first two steps are presented in Attachment 12 and were used to narrow the list of COPCs for evaluation in the third step. Table 7-24 presents the COPC-receptor pairs with HQs > 1.0 based on individual prey (Step 2) that were retained for further evaluation. The following subsections present the HQs for each fish receptor based on multiple prey species within relevant exposure areas. COPCs with HQs > 1.0 based on multiple prey species were retained as COCs.

Table 7-24. COPCs with HQs > 1.0 Based on Individual Prey Species

COPC	Large-scale Sucker	Juvenile White Sturgeon	Juvenile Chinook Salmon	Peamouth	Sculpin	Small-mouth Bass	Northern Pike-minnow
Metals							
Cadmium			X		X	X	X
Copper			X	X	X	X	X
Mercury							X
Butyltins							
TBT	X	X	X	X	X	X	X

COPC – chemical of potential concern
HQ – hazard quotient
TBT – tributyltin

7.2.4.2.1 Large-Home-Range Fish

HQs for large-home-range fish are presented in Table 7-25 and discussed below.

Table 7-25. Large-Home-Range Fish Site-Wide LOAEL HQs

		Total HQ[a]		
Receptor	**Prey Assumption**	**TBT**	**Cadmium**	**Copper**
Largescale sucker	100% clams	**3.5**	NA	NA
	100% worms	**9.3**	NA	NA
Juvenile white sturgeon	100% clams	**5.1 (2.5)**[b]	NA	NA
	100% worms	**9.3 (6.7)**[b]	NA	NA
Juvenile Chinook Salmon	30% clams, 40% worms, 30% epibenthic tissue	**56**[c]	**3.5**	**2.5**
	100% epibenthic tissue	ND	**1.7**	**2.4**
Peamouth	25% clam, 25% worm, 40% epibenthic tissue, 10% sculpin	**8.0**[d]	NA	1.0

[a] HQs are based on literature-based LOAELs for all receptors except for juvenile Chinook salmon. HQs for juvenile Chinook salmon are based on NOAELs.
[b] HQ is based on 56% incidentally ingested sediment and 8% (in parentheses).
[c] Based on 45% clams and 55% worms; no TBT data available from epibenthic tissue.
[d] Based on 45% clams, 45% worms, and 10% sculpin; no TBT data available from epibenthic tissue.
HQ – hazard quotient
LOAEL – lowest-observed-apparent-effect level
ND – no data
NA – not applicable (not a COPC-receptor pair evaluated)

Largescale Sucker

Five COPCs were identified for largescale sucker in the SLERA and refined screen. One COPC (TBT) had an HQ > 1.0 based on individual prey components (Table 7-24). TBT HQs were calculated across multiple prey species and incidental sediment ingestion using LOAEL-based TTCs and TSCs (Table 7-25).

TBT was identified as a COC for largescale sucker, because site-wide HQs based on ingestion of both clams and worms were > 1.0. TBT concentrations were highest in individual prey located at the mouth of Swan Island Lagoon and at the mouth of the International Slip (see Attachment 12 for evaluation of HQs based on individual samples). Site-wide HQs for TBT appear to be driven by elevated concentrations in a few samples at isolated locations within the site. Sediment concentrations contribute 17% of the estimated risks (i.e., the TBT HQs) for clam/incidental sediment ingested sediment diet and 7% of the estimate risks for the worm/incidental sediment ingested sediment diet.

There are several uncertainties associated with the exposure assumptions used to derive the risk estimates. The largescale sucker diet was modeled based on the available benthic prey items (clams and worms); however, suckers also feed on plant material, and dietary concentrations may be overestimated (assuming COPC concentrations accumulate less in plant tissues than benthic invertebrate tissues). The use of 28-day laboratory bioaccumulation worm tissue chemical concentrations adjusted for steady-state concentrations may not be representative of benthic invertebrate field tissue concentrations in the Study Area. The selected TBT LOAEL is also uncertain; it was derived from the only dietary TBT toxicity study available from the literature. The selected LOAEL was derived using a generic ingestion rate because no ingestion rate was reported in the study.

Juvenile White Sturgeon

Six COPCs were identified for juvenile white sturgeon in the SLERA and refined screen. One COPC (TBT) had an HQ > 1.0 based on individual prey components (Table 7-24). TBT HQs were calculated across multiple prey species and incidental sediment ingestion using LOAEL-based TTCs and TSCs (Table 7-25). Sediment concentrations contribute up to 59% of the estimated risks (i.e., the TBT HQs) for clam/incidental sediment ingested sediment diet and 32% of the estimate risks for the worm/incidental sediment ingested sediment diet.

TBT was identified as a COC for juvenile white sturgeon because site-wide HQs based on ingestion of both clams and worms (assuming 56% or 8% incidental sediment ingestion) were > 1.0. TBT concentrations are highest in individual prey located at the mouth of Swan Island Lagoon and at the mouth of the International Slip (see Attachment 12 for evaluation of HQs based on individual samples). Site-wide HQs for TBT appear to be driven by elevated concentrations in a few samples at isolated locations within the site.

There are several uncertainties associated with the exposure assumptions used to derive the 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 28-day laboratory bioaccumulation worm tissue chemical concentrations adjusted for steady-state concentrations may not be representative of benthic invertebrate field tissue chemical concentrations in the Study Area. The selected TBT LOAEL is also uncertain; it was derived from the only dietary TBT toxicity study available from the literature. The selected LOAEL was derived using a generic ingestion rate because no ingestion rate was reported in the study.

Juvenile Chinook Salmon

Four COPCs were identified for juvenile Chinook salmon in the SLERA and refined screen. Three COPCs (i.e., cadmium, copper, and TBT) had HQs > 1.0 based on individual prey components (Table 7-24). HQs for these COPCs were calculated across multiple prey species and incidental sediment ingestion using NOAEL-based TTCs and TSCs (Table 7-25).

All three COPCs evaluated (cadmium, copper, and TBT) were identified as COCs for juvenile Chinook salmon because HQs were > 1.0 based on either the multiple-prey species diet or the epibenthic invertebrate-only diet. Risk estimates for cadmium and copper were similar regardless of which diet was assumed. A TBT HQ could only be derived using worm and clam data because TBT was not analyzed in epibenthic invertebrate tissue. Juvenile Chinook salmon are primarily pelagic feeders, and the use of benthic organisms (i.e., worms and clams) to represent their diet is unrealistic and probably overestimates their dietary exposures to sediment-associated contaminants. In addition, the use of 28-day laboratory bioaccumulation worm tissue chemical concentrations adjusted to predicted steady-state concentrations might not be representative of benthic invertebrate field tissue chemical concentrations in the Study Area and may over- or underestimate risks.

There are several additional uncertainties associated with the effects data used to derive the risk estimates for each of the COPCs.

- **TBT** – The selected NOAEL used to evaluate dietary risks to juvenile Chinook salmon is uncertain because it was extrapolated from the LOAEL using an uncertainty factor (UF) of 5 because no NOAEL was reported in the single toxicity study available. The selected LOAEL was derived using a generic ingestion rate because no ingestion rate was reported in the study.
- **Cadmium** – The selected NOAEL used to evaluate risks to juvenile Chinook salmon was extrapolated from the LOAEL using a UF of 5. The LOAEL was much lower than other effects thresholds reported in other toxicity studies; it was 2 to 3 orders of magnitude lower than the nine NOAELs identified in 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 indicate that the selected TRVs overpredict cadmium toxicity to juvenile salmon. HQs based on a NOAEL TRV from any other study would result in HQs much lower than 1.0.
- **Copper –** There is high uncertainty associated with the selected TRVs for copper because the sensitivity of channel catfish fingerlings 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) and has been characterized as atypical by another study of copper in fish (Lorentzen et al. 1998) (see Attachment 14 for additional details). These studies confirm that the Murai et al. (1981) study results are anomalous. The next lowest NOAEL from the toxicity studies reviewed was 1.0 mg/kg bw/day, in which no effect on growth was observed in rockfish (Kang et al. 2005). The juvenile Chinook salmon TTC and TSC based on this value would be 10.9 mg/kg ww and 5,000 mg/kg dw, respectively. Table 7-26 presents a comparison of the copper HQs for juvenile Chinook salmon based on the NOAELs presented in Murai et al. (1981) and Kang et al. (2005). No HQs are > 1.0 based on the toxicity data presented in Kang et al. (2005). Furthermore, the selected copper dietary NOAEL and LOAEL

TRVs (0.24 and 0.48 mg/kg bw/day, respectively) are at or near the highest nutritional requirements found in the literature for relevant fish species, which range between 0.2 and 0.3 mg/kg bw/day[82] (Tacon 1992; Lim et al. 2008).

Table 7-26. Comparison of Juvenile Chinook Salmon Copper NOAEL HQs

Prey Assumption	Total NOAEL HQ	
	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

NOAEL – no-observed-adverse-effect level
HQ – hazard quotient
Bold identifies HQs > 1.0.

Peamouth

Five COPCs were identified for peamouth in the SLERA and refined screen. Two COPCs (copper and TBT) had HQs > 1.0 based on individual prey components (Table 7-24). TBT HQs were calculated across multiple prey species and incidental sediment ingestion using LOAEL-based TTCs and TSCs (Table 7-25).

HQs based on multiple prey species for all exposure areas were not > 1.0 for copper, and therefore, this COPC is not expected to pose unacceptable risks to peamouth and was not identified as a COC. TBT was identified as a COC for peamouth, because the site-wide HQ based on a multiple-prey diet was > 1.0. TBT concentrations are highest in individual prey located at the mouth of Swan Island Lagoon and at the mouth of the International Slip (see Attachment 12 for an evaluation of HQs based on individual samples). Site-wide HQs for TBT appear to be driven by elevated concentrations in a few samples at isolated locations within the site.

There are several uncertainties associated with the exposure assumptions used to derive the risk estimates. The peamouth diet for TBT was modeled using clam and lab worm tissue chemical concentrations to represent the invertebrate portion of their diet because no TBT data were available for epibenthic invertebrate tissue. Because peamouth also are pelagic feeders, the use of benthic organisms (i.e., worms and clams) to represent 90% of the peamouth diet is uncertain and may overestimate dietary exposures to sediment-associated contaminants. In addition, the use of 28-day laboratory bioaccumulation worm tissue chemical concentrations adjusted for steady-state concentrations may over or underestimate benthic invertebrate field tissue concentrations in the Study Area. The selected TBT LOAEL is also uncertain; it was derived from the only dietary TBT toxicity study available from the

[82] Nutrition requirement estimates are based on a typical fish consumption level for aquaculture of 5% bw/day (Gatlin and Wilson 1986; Lall and Hines 1987).

literature. The selected LOAEL was derived using a generic ingestion rate because no ingestion rate was reported in the study.

7.2.4.2.2 Small-Home-Range Fish

Sculpin

Five COPCs were identified for sculpin in the SLERA and refined screen. Three COPCs (i.e., cadmium, copper, and TBT) had HQs > 1.0 based on individual prey components (Table 7-24).

The relevant exposure area for sculpin was considered roughly equivalent to the sampling area of prey. Therefore, the sculpin diet was evaluated only on a sample-by-sample and individual prey basis. The prey items evaluated for sculpin were clam, lab worm, and sculpin. Sediment was evaluated as a portion of the diet based on incidental ingestion (5%). Table 7-27 presents a summary of the individual prey sample HQs. A spatial distribution of HQs for each COPC is presented on Maps 7-8 though 7-10 and Figures 7-12 through 7-14.

Table 7-27. Number of Sculpin Prey and Sediment Samples with LOAEL HQs > 1.0

	Number of Samples with HQs > 1.0 (HQ range)			
COPC	**Clam**	**Lab Worm**	**Sculpin**	**Sediment**
Metals				
Cadmium	5[a]/38 (0.34 – 1.8)	2/35 (0.31 – 2.2)	0/38 (0.025 – 0.19)	1/1,348 (0.00014 – 4.2)
Copper	38/38 (1.1 – 2.4)	1/35 (0.32 – 3.6)	1/38 (0.16 – 1.3)	8[b]/1,358 (0.0082 – 5.3)
Butyltins				
Tributyltin ion	3/34 (0.073 – 21)	2/35 (0.018 – 69)	0/12 (0.073 – 0.16)	7/405 (0.00000026 – 20)

[a] HQs were equal to 1.0 in two additional clam samples.
[b] HQs were equal to 1.0 in one additional sediment sample.
COPC – chemical of potential concern
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level

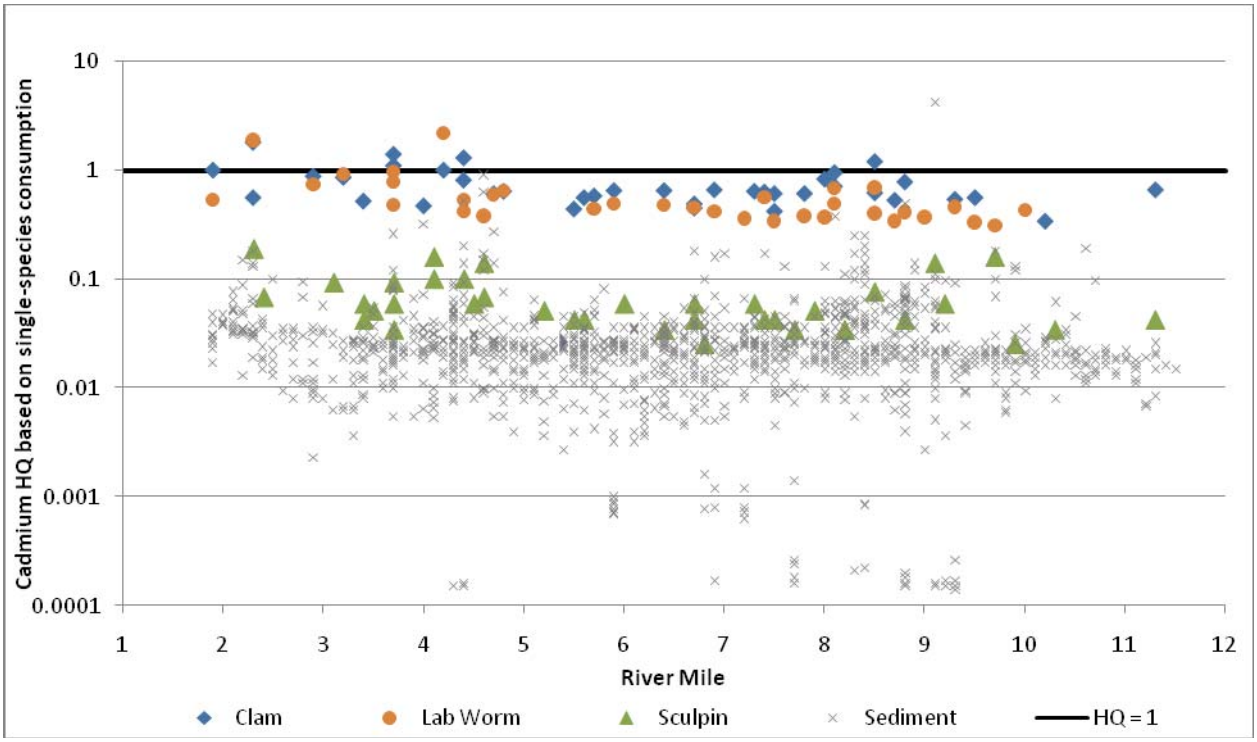

Figure 7-12. Sculpin Prey Tissue and Sediment HQs by RM for Cadmium

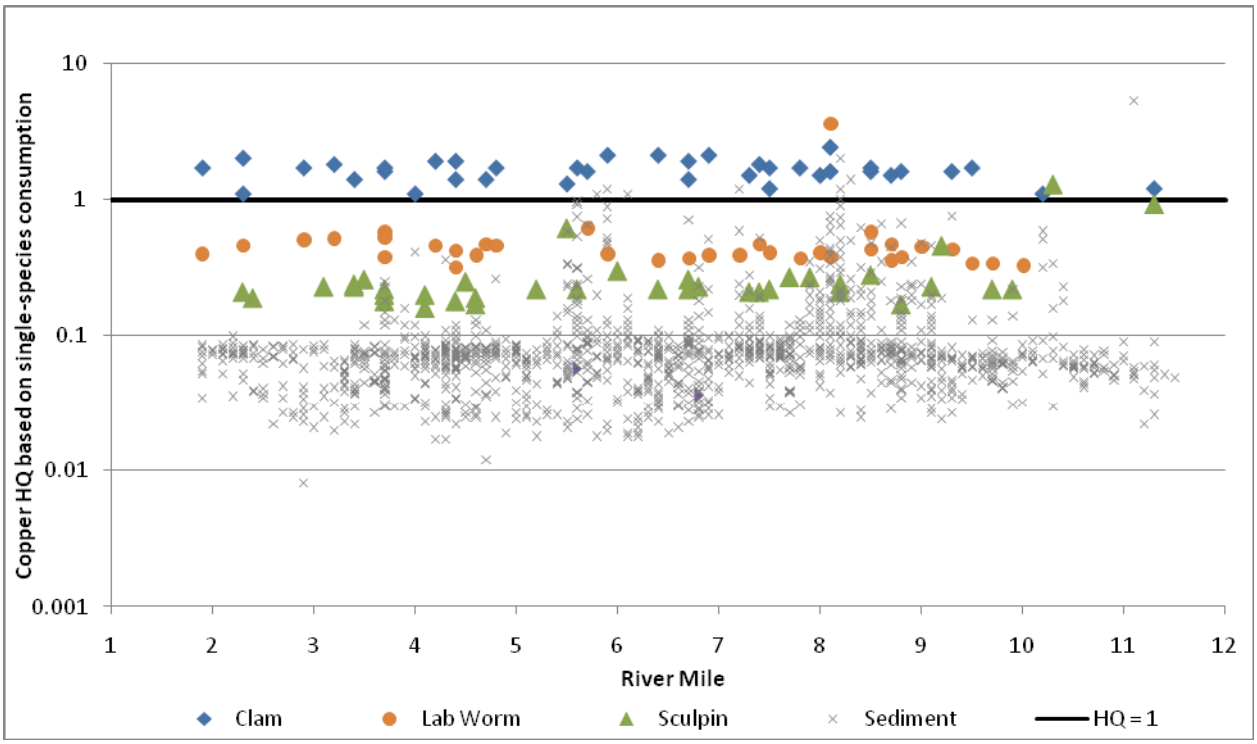

Figure 7-13. Sculpin Prey Tissue and Sediment HQs by RM for Copper

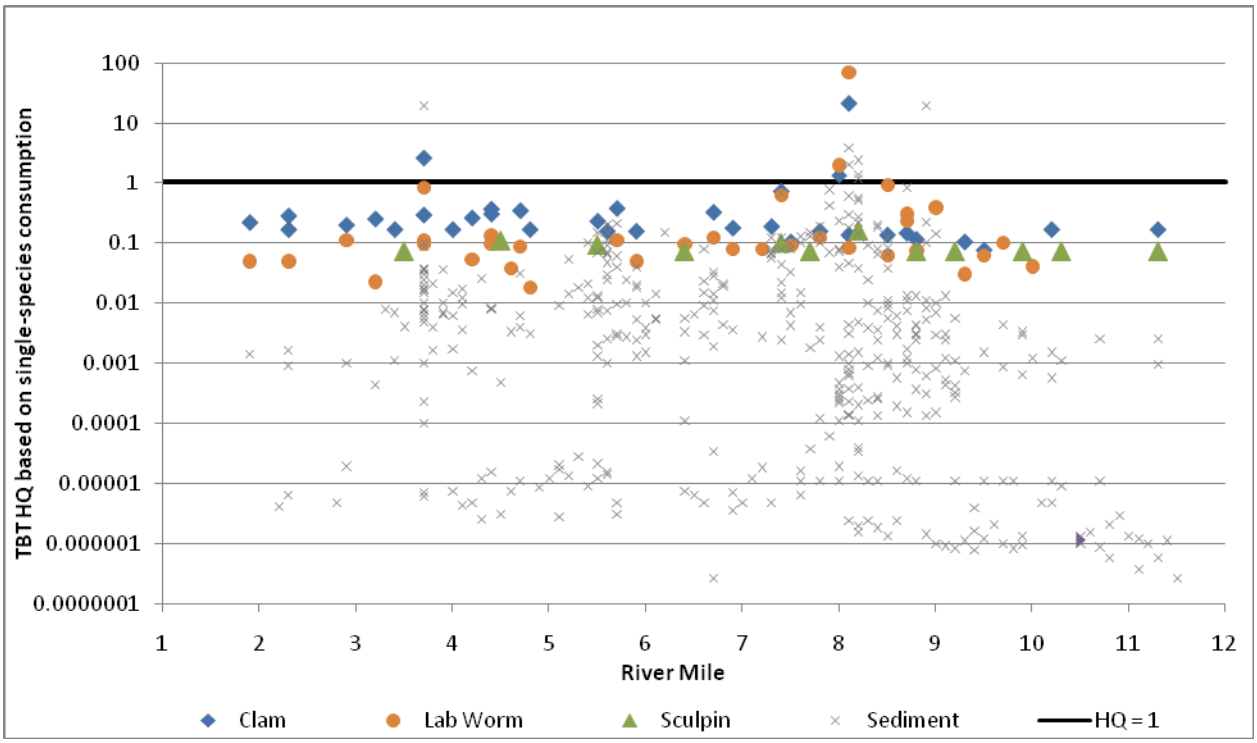

Figure 7-14. Sculpin Prey Tissue and Sediment HQs by RM for TBT

Cadmium, copper and TBT were all identified as COCs for sculpin. The spatial distribution of risk estimates are discussed below:

- **Cadmium** – HQs were > 1.0 in five field clam samples, two worm samples, and one sediment sample. Samples with HQs > 1.0 were located generally within five areas: at approximately RM 2.2 on the east side, and RM 4.2 on the east side, International Slip, Slip 1, and Swan Island Lagoon (Map 7-8 and Figure 7-12). Because of the low magnitude of HQs and the low frequency of exceedances, cadmium might pose dietary risks to sculpin that only 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 were > 1.0 in all field clam samples, one worm sample, one sculpin sample, and eight sediment samples. Copper exceedances in clams are distributed site-wide (Map 7-9 and Figure 7-13); however, risk conclusions for copper are highly uncertain because of uncertainties associated with the TRVs (as discussed in more detail below).
- **TBT** – The samples that exceeded sculpin dietary thresholds for TBT indicate the presence of localized areas of elevated TBT concentrations (Map 7-10 and Figure 7-14). The mouth of Swan Island Lagoon has field clam, lab clam, lab worm, and sediment exceedances, with the HQs in prey tissue ranging from 1.3 to 1.4 on the east side of the mouth to 21 to 69 on the west side of the mouth. A single field clam

and sediment sample at the mouth of the International Slip also had HQs > 1.0 (clam and sediment HQs were 2.6 and 20, respectively).

The primary uncertainty associated with the exposure data is that for all COCs, prey tissue concentrations represented by 28-day laboratory bioaccumulation worm tissue chemical concentrations adjusted for steady-state concentrations may not be representative of benthic invertebrate field tissue chemical concentrations in the Study Area. There are several additional uncertainties associated with the effects data used to derive the risk estimates for each of the COPCs.

- **TBT** – The selected TBT LOAEL is also uncertain; it was derived from the only dietary TBT toxicity study available from the literature. The selected LOAEL was derived using a generic ingestion rate because no ingestion rate was reported in the study.
- **Cadmium** – The selected LOAEL was much lower than other effects thresholds reported in other toxicity studies; it was 2 to 3 orders of magnitude lower than the nine NOAELs identified in the other studies reviewed and 2 to 3 orders of magnitude lower than the four LOAELs reported in other studies reviewed. As such, the selected TRVs are conservative because the majority of the toxicological studies reviewed indicate that the selected TRVs may over-predict cadmium toxicity to fish. HQs based on a LOAEL TRV from any other study would result in HQs much lower than 1.0.
- **Copper** – There is high uncertainty associated with the selected TRVs for copper because the sensitivity of channel catfish fingerlings documented by Murai et al. (1981) 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 has been characterized as atypical by another study of copper in fish (Lorentzen et al. 1998). The next lowest LOAEL from the toxicity studies reviewed was 2.0 mg/kg bw/day, in which no effect on growth was observed in rockfish (Kang et al. 2005). The sculpin TTC and TSC based on this value would be 23.5 mg/kg ww and 2,220 mg/kg dw, respectively. Table 7-28 presents a comparison of the prey-specific copper HQs for sculpin based on the LOAELs presented in Murai et al. (1981) and Kang et al. (2005). Individual prey sample concentrations do not result in HQs > 1.0 based on the toxicity data presented in Kang et al. (2005). One sediment sample results in an HQ of 1.3; however, an individual sediment sample does not represent a realistic dietary exposure concentration. Furthermore, the selected copper dietary NOAEL and LOAEL TRVs (0.24 and 0.48 mg/kg bw/day, respectively) are at or near the highest nutritional requirements found in the literature for relevant fish species, which range between 0.2 and 0.3 mg/kg bw/day[83] (Tacon 1992; Lim et al. 2008).

[83] Nutrition requirement estimates are based on a typical fish consumption level for aquaculture of 5% bw/day (Gatlin and Wilson 1986; Lall and Hines 1987).

Table 7-28. Comparison of Sculpin Copper LOAEL HQs on Individual Prey Basis

Prey Item	Unit	Copper Concentration Range	Risk Estimates Based on Murai et al. (1981)		Risk Estimates Based on Kang et al. (2005)	
			LOAEL TTC/TSC	LOAEL HQ Range	LOAEL TTC/TSC	LOAEL HQ Range
Clam	mg/kg ww	5.99 – 13.5	5.65	**1.1 – 2.4**	23.5	0.25 – 0.57
Lab worm	mg/kg ww	1.83 – 20.2	5.65	0.32 – **3.6**	23.5	0.078 – 0.86
Sculpin	mg/kg ww	0.929 – 7.16	5.65	0.16 – **1.3**	23.5	0.040 – 0.30
Sediment	mg/kg dw	4.37 – 2,830	533	0.0082 – **5.3**	2,220	0.0020 – **1.3**

dw – dry weight
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
NOAEL – no-observed-adverse-effect level
TSC – threshold sediment concentration
TTC – threshold tissue concentration
ww – wet weight
Bold identifies HQs > 1.0.

Smallmouth Bass

Four COPCs were identified for smallmouth bass in the SLERA and refined screen. Three COPCs (i.e., cadmium, copper, and TBT) had HQs > 1.0 based on individual prey components (Table 7-24). HQs for these COPCs were calculated across multiple prey and incidental sediment ingestion using LOAEL-based TTCs and TSCs (Table 7-29).

Table 7-29. Smallmouth Bass 1-Mile Exposure Area LOAEL HQs Across Multiple Prey Items

Exposure Area	Total HQ[a]		
	Cadmium	Copper	TBT
RM 1.5 to RM 2.5	0.19	0.22	0.016[b]
RM 2.5 to RM 3.5	0.10	0.27	0.069[b]
RM 3.5 to RM 4.5	0.14	0.26	0.66
RM 4.5 to RM 5.5	0.12	0.26	0.077
RM 5.5 to RM 6.5	0.064	0.52	0.075
RM 6.5 to RM 7.5	0.062	0.29	0.10
RM 7.5 to RM 8.5	0.056	0.28	0.094
RM 8.5 to RM 9.5	0.069	0.40	0.034
Swan Island Lagoon	0.13	0.41	**6.1**

Table 7-29. Smallmouth Bass 1-Mile Exposure Area LOAEL HQs Across Multiple Prey Items

	Total HQ[a]		
Exposure Area	**Cadmium**	**Copper**	**TBT**
RM 9.5 to RM 10.5	0.11	0.87	0.024
RM 10.5 to RM 11.8	0.042	0.86	0.022

[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%.

[b] TBT data were only available from worms in this exposure area. Total HQ is based on 100% worm ingestion.

COPC – chemical of potential concern
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
RM – river mile
TBT – tributyltin
Bold identifies HQs > 1.0.

HQs based on multiple prey items for all exposure areas were not > 1.0 for cadmium and copper; therefore, these COPCs are not expected to pose unacceptable risks to smallmouth bass and were not identified as COCs.

TBT was identified as a COC for smallmouth bass because the HQ based on a multiple-prey diet was > 1.0 (HQ = 6.1) in one exposure area (i.e., Swan Island Lagoon). There are several uncertainties associated with the exposure assumptions used to derive the risk estimates. The use of 28-day laboratory bioaccumulation worm tissue chemical concentrations adjusted for steady-state concentrations may not be representative of benthic invertebrate field tissue chemical concentrations in the Study Area. Worm made up 10% of the smallmouth bass diet in Swan Island Lagoon because no crayfish data were available from this exposure area. Worms were the only smallmouth bass prey item with elevated TBT concentrations (see Attachment 12 for an evaluation of HQs based on individual samples). These data may not be representative of benthic invertebrate field tissue chemical concentrations in the Study Area or appropriate for estimating concentrations in smallmouth bass prey. The selected TBT LOAEL is uncertain; it was derived from the only dietary TBT toxicity study available from the literature. The selected LOAEL was derived using a generic ingestion rate because no ingestion rate was reported in the study.

Northern Pikeminnow

Four COPCs were identified for northern pikeminnow in the SLERA and refined screen. Four COPCs (i.e., cadmium, copper, mercury, and TBT) had HQs > 1.0 based on individual prey components (Table 7-24). HQs for these COPCs were calculated across multiple prey and incidental sediment ingestion using LOAEL-based TTCs and TSCs (Table 7-30).

Table 7-30. Northern Pikeminnow 1-Mile Exposure Area LOAEL HQs Across Multiple Prey Item

	Total HQ			
Exposure Area	**Cadmium[a]**	**Copper[a]**	**Mercury[a]**	**TBT[b]**
RM 1.5 to RM 2.5	0.43	0.56	0.16	0.090
RM 2.5 to RM 3.5	0.25	0.72	0.16	0.12
RM 3.5 to RM 4.5	0.37	0.70	0.13	0.86
RM 4.5 to RM 5.5	0.21	0.66	0.21	0.073
RM 5.5 to RM 6.5	0.19	0.85	0.23	0.081
RM 6.5 to RM 7.5	0.18	0.76	0.34	0.16
RM 7.5 to RM 8.5	0.16	0.61	0.28	0.10
RM 8.5 to RM 9.5	0.15	0.73	0.25	0.072
Swan Island Lagoon	0.23	**1.3**	0.17	**25**
RM 9.5 to RM 10.5	0.16	0.91	0.19	0.028
RM 10.5 to RM 11.8	0.12	**1.5**	0.16	0.021

[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.

[b] Total HQ was calculated using the following prey portions: 30% crayfish, 25% lab worms, 25% sculpin, and 20% carp. TBT data were not available for northern pikeminnow, largescale sucker, or peamouth. When no crayfish data were available, lab worms were assigned a prey portion of 55%. When no sculpin data were available, carp were assigned a prey portion of 45%. When no lab worm data were available, sculpin were assigned a prey portion of 50%.

COPC – chemical of potential concern

HQ – hazard quotient

LOAEL – lowest-observed-adverse-effect level

TBT – tributyltin

Bold identifies HQs > 1.0.

HQs based on multiple prey for all exposure areas were not > 1.0 for cadmium and mercury; therefore, these COPCs are not expected to pose unacceptable risks to northern pikeminnow and were not identified as COCs.

Copper and TBT were identified as COCs for northern pikeminnow because HQs based on a multiple-prey diet were > 1.0 in at least one exposure area. The TBT HQ was > 1.0 (HQ = 25) in one exposure area (Swan Island Lagoon). The copper HQ was > 1.0 in two

exposure areas (HQs were 1.3 and 1.5 in Swan Island Lagoon and between RM 10.5 and RM 11.8, respectively).

There are several uncertainties associated with the exposure assumptions used to derive the risk estimates. The primary uncertainty associated with the exposure data is that for both COCs (i.e., TBT and copper), prey tissue chemical concentrations were represented by 28-day laboratory bioaccumulation worm tissue chemical concentrations adjusted for steady-state concentrations. For the TBT HQ at Swan Island Lagoon, 55% of the diet was represented by lab worms.[84] Worms were the only northern pikeminnow prey item with elevated TBT concentrations (see Attachment 12 for an evaluation of HQs based on individual samples). These data may not be representative of benthic invertebrate field tissue concentrations in the Study Area or appropriate for estimating concentrations in northern pikeminnow prey.

There are several additional uncertainties associated with the effects data used to derive the risk estimates for each of the COCs.

- **TBT** – The selected TBT LOAEL is uncertain; it was derived from the only dietary TBT toxicity study available from the literature. The selected LOAEL was derived using a generic ingestion rate because no ingestion rate was reported in the study.
- **Copper** – There is high uncertainty associated with the selected TRVs for copper because the sensitivity of channel catfish fingerlings documented by Murai et al. (1981) 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 has been characterized as atypical by another study of copper in fish (Lorentzen et al. 1998). The next lowest LOAEL from the toxicity studies reviewed was 2.0 mg/kg bw/day, in which no effect on growth was observed in rockfish (Kang et al. 2005). The northern pikeminnow TTC and TSC based on this value would be 37.3 mg/kg ww and 14,400 mg/kg dw, respectively. Table 7-31 presents a comparison of the copper HQs for northern pikeminnow based on the LOAELs presented in Murai et al. (1981) and Kang et al. (2005). No HQs in any exposure area result in any HQs > 1.0 based on the toxicity data presented in Kang et al. (2005). Furthermore, the selected copper dietary NOAEL and LOAEL TRVs (0.24 and 0.48 mg/kg bw/day, respectively) are at or near the highest nutritional requirements found in the literature for relevant fish species, which range between 0.2 and 0.3 mg/kg bw/day[85] (Tacon 1992; Lim et al. 2008).

[84] At the Swan Island Lagoon exposure area for estimating TBT diet concentrations, lab worms represented 55% of the diet because no data were available for crayfish which were supposed to represent 30% of the diet.

[85] Nutrition requirement estimates are based on a typical fish consumption level for aquaculture of 5% bw/day (Gatlin and Wilson 1986; Lall and Hines 1987).

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
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 > 1.0.

7.2.4.3 Summary of Fish Diet COCs

Three dietary fish COCs were identified across all fish receptors: cadmium, copper, and TBT. Table 7-32 presents the HQs and exposure and effects uncertainties for each of the COC-receptor pairs. The results of the dietary LOE were compared with other LOEs in Section 7.6 to determine the fish risk conclusions.

Table 7-32. Summary of Fish Dietary COCs

Receptor/COC	Risk Exposure Area	HQ[a]	Exposure Uncertainty	Effects Uncertainty
TBT				
Largescale sucker	Site-wide	3.5 – 9.3	Benthic invertebrate diet was modeled using clams (HQ = 3.5) and lab worms (HQ = 9.3). Site-wide risks are driven by elevated TBT concentrations located at the mouth of Swan Island Lagoon and at the mouth of the International Slip. Because of the limited distribution of elevated TBT exposures, it is unlikely that the population of largescale sucker in the Study Area would be impacted by TBT.	High uncertainty; only one toxicological study was available; LOAEL was derived using a generic ingestion rate (ingestion rate was not reported in the study).
Juvenile white sturgeon	Site-wide	5.1 – 9.3 (2.5 – 6.7)[b]	Home range is likely greater than the Study Area; benthic invertebrate diet was modeled using clams (HQ = 5.1 [2.5]) and lab worms (HQ = 9.3 [6.7]). Site-wide risks are driven by elevated TBT concentrations located at the mouth of Swan Island Lagoon and at the mouth of the International Slip.	High uncertainty; only one toxicological study was available; LOAEL was derived using a generic ingestion rate (ingestion rate was not reported in the study).
Juvenile Chinook salmon	Site-wide	56	Pelagic diet is represented using benthic invertebrates (clams and lab worms) because no TBT tissue data were available from pelagic prey. Chinook salmon are primarily pelagic feeders, and the use of benthic organisms (i.e., worms and clams) to represent their diet is unrealistic and probably overestimates their dietary exposures to sediment-associated contaminants. Site-wide risks are driven by elevated TBT concentrations located at the mouth of Swan Island Lagoon and at the mouth of the International Slip.	High uncertainty; NOAEL was extrapolated from the LOAEL using a UF; Only one toxicological study; LOAEL was derived using a generic ingestion rate (ingestion rate was not reported in the study).

Table 7-32. Summary of Fish Dietary COCs

Receptor/COC	Risk Exposure Area	HQ[a]	Exposure Uncertainty	Effects Uncertainty
Peamouth	Site-wide	8.0	90% of benthopelagic diet is represented using only benthic invertebrates (clams and lab worms). Peamouth are also pelagic feeders, and the use of benthic organisms (i.e., worms and clams) to represent 90% of the peamouth diet is uncertain and may overestimate dietary exposures to sediment-associated contaminants. Site-wide risks are driven by elevated TBT concentrations located at the mouth of Swan Island Lagoon and at the mouth of the International Slip. Because of the limited distribution of elevated TBT exposures, it is unlikely that the population of peamouth in the Study Area would be impacted by TBT.	High uncertainty; only one toxicological study available; LOAEL was derived using a generic ingestion rate (ingestion rate was not reported in the study).
Sculpin	Mouth of Swan Island Lagoon and International Slip	1.3 – 69[c]	HQs are based on a sample-by-sample analysis, including lab worms. Because of the limited distribution of elevated TBT exposures, it is unlikely that the population of sculpin in the Study Area would be impacted by TBT.	High uncertainty; only one toxicological study available; LOAEL was derived using a generic ingestion rate (ingestion rate was not reported in the study).
Smallmouth bass	Swan Island Lagoon	6.1[d]	10% of the diet is based on lab worms because no crayfish data were available from the exposure area; lab worms are the only prey species with elevated TBT concentrations and may not be representative prey. HQ would be < 1.0 if lab worms did not represent any portion of the diet and if incidental sediment ingestion was < 2% (see Section 7.2.4.4 below). Because of the limited distribution of elevated TBT exposures, it is unlikely that the population of smallmouth bass in the Study Area would be impacted by TBT.	High uncertainty; only one toxicological study available; LOAEL was derived using generic ingestion rate (ingestion rate was not reported in the study).

Table 7-32. Summary of Fish Dietary COCs

Receptor/COC	Risk Exposure Area	HQ[a]	Exposure Uncertainty	Effects Uncertainty
Northern pikeminnow	Swan Island Lagoon	25[d]	55% of the diet is based on lab worms because no crayfish data were available from the exposure area; lab worms are the only prey species with elevated TBT concentrations and may not be representative prey. HQ would be < 1.0 if lab worms did not represent any portion of the diet and if incidental sediment ingestion was < 2% (see Section 7.2.4.4 below). Because of the limited distribution of elevated TBT exposures, it is unlikely that the population of northern pikeminnow in the Study Area would be impacted by TBT.	High uncertainty; there was only one toxicological study available; LOAEL was derived using a generic ingestion rate (ingestion rate was not reported in the study).
Copper				
Juvenile Chinook salmon	Site-wide	2.4 – 2.5	Low HQ (2.4) is based on limited epibenthic invertebrate tissue data; high HQ (2.5) is based on diet of only benthic invertebrates (clams and worms), which is not representative of pelagic diet.	High uncertainty; lowest NOAEL is based on a study that could not be replicated in subsequent experiments and is within the range of nutritional requirements found in the literature for relevant fish species; the next highest NOAEL from literature results in HQs < 1.0.
Sculpin	Individual samples with HQs > 1.0 throughout Study Area	1.1 – 3.6[c]	HQs are based on a sample-by-sample analysis.	High uncertainty; lowest LOAEL is based on a study that could not be replicated in subsequent experiments and is just above the range nutritional requirements found in the literature for relevant fish species; the next highest LOAEL from literature results in HQs < 1.0.

Table 7-32. Summary of Fish Dietary COCs

Receptor/COC	Risk Exposure Area	HQ[a]	Exposure Uncertainty	Effects Uncertainty
Northern pikeminnow	Swan Island Lagoon; RM 10.5 to RM 11.8	1.3 – 1.5[e]	HQs would be < 1.0 if crayfish and lab worms collectively made up approximately <20% of the diet (see Section 7.2.4.4 below).	High uncertainty; lowest LOAEL is based on a study that could not be replicated in subsequent experiments and is just above the range nutritional requirements found in the literature for relevant fish species; the next highest LOAEL from literature results in HQs < 1.0.
Cadmium				
Sculpin	Individual samples with HQs > 1.0 at: RM 2.0, east; RM 4.2, east; Slip 1; International Slip; Swan Island Lagoon	1.1 – 2.2[c]	HQs are based on a sample-by-sample analysis.	LOAEL is several orders of magnitude lower than other TRVs reported in literature.
Juvenile Chinook salmon	Site-wide	1.7 – 3.5	Low HQ (1.7) is based on limited epibenthic invertebrate tissue data; high HQ (3.5) is based on diet of only benthic invertebrates (clams and worms), which is not representative of pelagic diet.	NOAEL was extrapolated from the LOAEL using a UF; LOAEL is several orders of magnitude lower than other salmonid TRVs reported in the literature.

[a] HQ is based only on exposure areas where HQs were > 1.0. HQs were < 1.0 in all other exposure areas. HQs are based on LOAEL TRVs for all receptors except juvenile Chinook salmon and Pacific lamprey, for which HQs are based on NOAEL TRVs.
[b] HQ is based on 56% incidentally ingested sediment and 8% (in parentheses).
[c] HQ is based on individual prey samples for which HQ > 1.0.
[d] HQ is based on Swan Island Lagoon exposure area. HQs in all other exposure area are < 1.0.
[e] HQ is based on exposure areas where HQ > 1.0 (i.e., Swan Island Lagoon and RM 10.5 to RM 11.8). HQs in all other exposure area are < 1.0.

HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
NOAEL – no-observed-adverse-effect level
RM – river mile
TBT – tributyltin
TRV – toxicity reference value
UF – uncertainty factor

7.2.4.4 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.

7.2.4.4.1 Evaluation of Varying Prey Portions

Selected prey portions (Table 7-20) were based on dietary information presented in the literature. In the Problem Formulation (Attachment 2), EPA requested that prey portions be varied from 0 to 100%. An evaluation was conducted to determine how varying the prey portions in the diet would change risk conclusions (i.e., whether or not a COPC would be identified as a COC) for all COPC-receptor pairs, for which multiple prey species were evaluated in the diet.[86]

The prey portion uncertainty evaluation determined the range of possible HQs if all prey species could make up 0 to 100% of the diet. The range of HQs was used to determine whether any additional COPCs would have been identified as a COC (i.e., have HQs > 1.0) had different prey portions been used (other than those presented in Table 7-20) or whether any COPCs would have not been identified as COCs. Table 7-33 presents the results of this evaluation

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[86] 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	Selected Prey Portions Used in Risk Characterization	Prey Portion Uncertainty Evaluation	Does Uncertainty Evaluation Change COC Status?
Juvenile Chinook Salmon					
TBT	56	All exposure areas	45% clams and 55% worms	HQs could range from 27 to 80 based on 100% ingestion of clams and lab worms, respectively.	No, would be a COC regardless of varying prey portions.
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, would be a COC regardless of varying 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, would be a COC regardless of varying prey portions.
Peamouth					
TBT	8.0	All exposure areas	45% clams, 45% worms, and 10% sculpin	HQs could range from 0.89 to 13 based on 100% ingestion of sculpin and lab worms, respectively.	No, would have not been identified as a COC if diet was primarily sculpin, which is not an assumption supported by the literature.
Copper	1.0	All exposure areas	25% clams, 25% worms, 40% epibenthic tissue, and 10% sculpin	More than 67% of diet would have to be represented by clams for HQs to be > 1.0 in all exposure areas; clam prey portion of > 67% is not supported by the literature.	No, would have been identified as a COC if > 67% of diet is composed of clams; however, this is not an assumption supported by the literature.

Table 7-33. 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 COC Status?
Smallmouth Bass					
TBT	6.1	Swan Island Lagoon	90% sculpin, 5% crayfish, and 5% lab worms	HQs could range from1.6 to 47 based on 100% ingestion of sculpin and lab worms, respectively.	Yes, would not have been as a COC if lab worms did not represent any portion of the diet and incidental sediment ingestion was < 2%, which are plausible assumptions for a higher trophic level pelagic feeding receptor.
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 > 1.0 in all exposure areas, or more than 42% of diet would have to be represented by lab worms for HQs to be > 1.0 in one exposure area (Swan Island Lagoon).	Yes, would have been identified as a COC if > 71% of diet was composed of crayfish, which is a reasonable assumption.[a]
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 > 1.0 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, would have been identified as a COC if > 77% of diet is composed of worms; however, this is not an assumption supported by the literature.

Table 7-33. 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 COC Status?
Northern Pikeminnow					
TBT	25	Swan Island Lagoon	30% crayfish, 25% lab worms, 25% sculpin, and 20% carp	HQs could range from 1.6 to 45 based on 100% ingestion of sculpin and lab worms, respectively.	Yes, would not have been identified as a COC if lab worms did not represent any portion of the diet and incidental sediment ingestion was < 2%, which are plausible assumptions for a higher trophic level pelagic feeding receptor.
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, would not have been identified as a COC 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 > 1.0 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, would have been identified as a COC 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 > 1.0 in five exposure areas (RM 4.5 to RM 9.5); pikeminnow prey portion of > 77% is not supported by the literature.	No, would have been identified as a COC 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)

COC – chemical of concern
COPC – chemical of potential concern
HQ – hazard quotient
RM – river mile
TBT – tributyltin

For copper-juvenile Chinook salmon, cadmium-juvenile Chinook salmon, and TBT-juvenile Chinook salmon, varying the prey portions would not change which pairs would be identified as COC-receptor pairs (Table 7-33).

For a few COPC-receptor pairs, assuming that any given prey species could make up 0 to 100% of the diet results in additional COPC-receptor pairs that would have HQs > 1.0.

- Copper-peamouth
- Cadmium-smallmouth bass
- Copper-smallmouth bass
- Cadmium-northern pikeminnow
- Mercury-northern pikeminnow

However, the portion of the diet that a given prey species would have to represent as part of the total diet to result in an HQ > 1.0 for four of these receptor pairs (copper-peamouth, cadmium-smallmouth bass, cadmium-northern pikeminnow, and mercury-northern pikeminnow) is not likely based on the diet prey portions presented in the general literature and region-specific studies. Therefore, the likelihood of risk to these fish COPC-receptor pairs is very low.

The portion of the diet that a given prey species would have to represent as part of the total diet to result in an HQ > 1.0 is possible for copper-smallmouth bass. It is possible that more than 71% of the smallmouth bass diet may be composed of crayfish, resulting in HQs > 1.0 but not > 2.4. The evaluation of copper exposure to a piscivorous fish is already covered in the evaluation of northern pikeminnow because northern pikeminnow and copper have already been identified as a COC-receptor pair.

A few COPC-receptor pairs would have not been identified as COCs had different dietary assumptions been used:

- TBT-peamouth
- TBT-smallmouth bass
- TBT-northern pikeminnow
- Copper-northern pikeminnow

For TBT-smallmouth bass, TBT-northern pikeminnow, and copper-northern pikeminnow, the assumptions that would yield HQs < 1.0 are plausible, and the identification of these COPC-receptor pairs as COCs introduces high uncertainty into the estimated risks. For TBT-peamouth, the assumptions that are needed to result in HQs > 1.0 are not likely because they and not supported by the literature.

7.2.4.4.2 Evaluation of Water Temperature on Calculated Fish Ingestion Rates

Dietary fish ingestion rates were estimated using Equation 7-7 and assuming a temperature of 13.4 ºC, which is based on the average data collected by ODEQ from a station within the Study Area (SP&S Railroad Bridge) from 1995 to 2005 (ODEQ 2005). Per EPA's Problem Formulation (Attachment 2), fish food ingestion rates were also calculated using a high-end water temperature provided by EPA (16.2 ºC). Food and sediment ingestion rates increase approximately 17 and 16%, respectively, using the higher water temperature. HQs based on ingestion rates calculated at the higher water temperature of 16.2º C would increase by approximately 16 to 17% compared to 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 COCs would be identified).

7.2.4.4.3 Evaluation of Varying Fish Body Weight on Calculated Fish Ingestion Rates

Per EPA's Problem Formulation (Attachment 2), fish body weights were also varied to determine the effect of using the range of body weights of fish collected from the Study Area (instead of the average body weight) on risk calculations. Assuming minimum and maximum body weights for each fish receptor results in HQs less than 25% different from the HQ calculated assuming the 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 fish populations. 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. The comparison of surface water concentrations to water TRVs was conducted on an individual sample basis per EPA's direction (Attachment 2). A sample-by-sample basis was considered a relevant scale in the evaluation of small-home-range fish. COPCs with HQs > 1.0 for any individual surface water sample were identified as COCs for these receptors. The surface water assessment conducted for small-home-range fish is similar to the assessment conducted for benthic invertebrates (Section 6.5). For large-home-range and migratory fish receptors, a site-wide scale was considered more appropriate for the evaluation of risks. COPCs with HQs > 1.0 based on site-wide UCLs were identified as COCs for large-home-range receptors.

For all COCs (for both small- and large-home-range fish), the spatial distribution and magnitude of HQs, the seasonal and sampling method patterns of HQs, and the associated exposure and effects assumptions were evaluated to provide a more detailed assessment of impacts to fish. The evaluation of COCs and associated uncertainties were further evaluated to arrive at risk conclusions for fish (Section 7.6).

Per EPA's Problem Formulation (Attachment 2), an additional LOE was evaluated for the surface water assessment. Specifically, the exposure of adult Chinook salmon to metals concentrations in water was evaluated to determine if these concentrations might disrupt olfactory function in migrating populations. Adult Chinook salmon are not a selected

ecological receptor of concern; however, this receptor was evaluated per EPA's Problem Formulation (Attachment 2).

The details of this assessment are presented as follows:

- Section 7.3.1 presents a summary of the COPCs evaluated in the surface water LOE.
- Section 7.3.2 presents an overview of how exposure concentrations were derived. Exposure data in this assessment are represented by COPC concentrations in surface water samples. All surface water chemical concentrations and calculated UCLs are presented in Attachment 4.
- Section 7.3.3 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. An evaluation of whether the water TRVs are protective of lamprey is also presented based on the sensitivity study results of the lamprey toxicity tests that were conducted. 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 the results of the lamprey toxicity tests are presented in Attachment 15.
- Section 7.3.4 presents the risk characterization results, COC-receptor pairs, and associated uncertainties. These COCs are further assessed in the fish risk conclusions (Section 7.6).

Figure 7-15 presents a flowchart of the fish surface water assessment section organization.

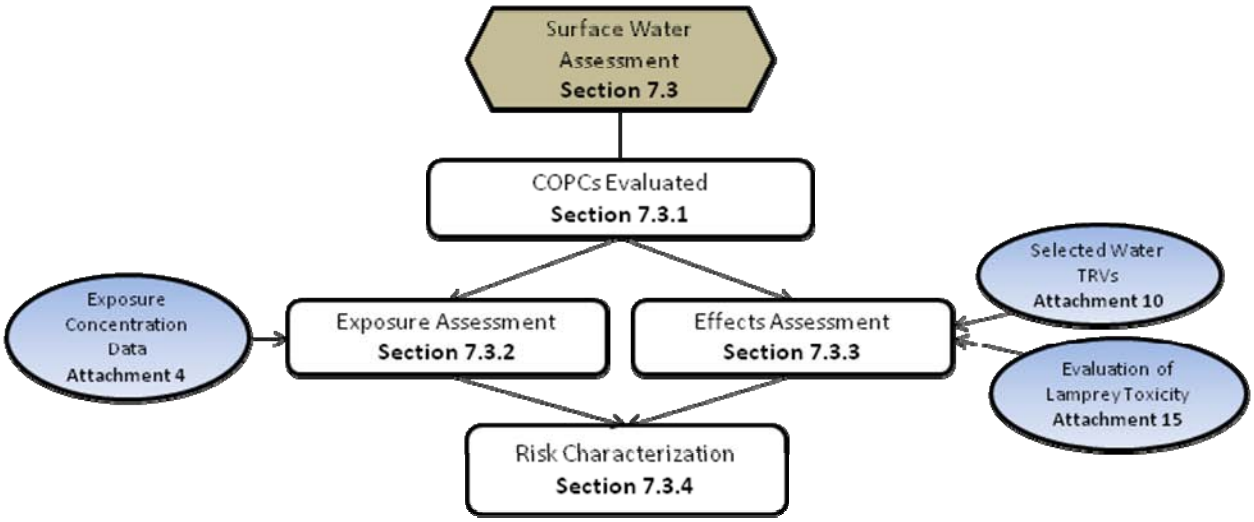

Figure 7-15. Overview of Fish Surface Water Assessment Section Organization

7.3.1 COPCs Evaluated

Twelve[87] surface water COPCs were identified in the SLERA and refined screen (Attachment 5). The surface water COPCs are presented in Table 6-26. Aluminum was assessed even though it is a naturally occurring element in the environment. In addition to comparing concentrations of aluminum in surface water within the Study Area to a water TRV, aluminum concentrations in background surface water and sediment were also evaluated (Section 7.3.4). Like aluminum, zinc is naturally occurring in the environment, and background zinc concentrations were also evaluated.

Four surface water COIs could not be evaluated because no toxicological data were available. These COIs (4-chloroaniline, aniline, 2,4-DB, and MCPP) were infrequently detected and were detected in isolated areas at different times.

7.3.2 Exposure Assessment

Risks to fish from surface water were assessed by comparing all the COPC concentrations in surface water to TRVs. Surface water EPCs are discussed in Section 7.3.2.1, and surface water exposure concentrations used to evaluate olfactory function in migrating populations of adult Chinook salmon are presented in Section 7.3.2.2.

7.3.2.1 Surface Water EPCs

An overview of surface water data (i.e., sampling events and rationale, and sample types) and general trends in COPC concentrations from the Study Area are presented in the benthic risk assessment (Section 6.5.2). For fish, all surface water data collected using different sampling methods were used to develop EPCs. The surface water data were collected with reasonable temporal and spatial coverage throughout the Study Area, except for non-LWG surface water near-bottom peristaltic samples, which 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). Two COPCs (i.e., ethylbenzene and trichloroethene) were analyzed only in samples collected as part of a non-LWG sampling event; therefore, risk estimates to fish from exposure to these COPCs are highly uncertain because only limited data are available.

[87] Three individual DDT metabolites identified in the SLERA (2,4′-DDD, 2,4′-DDT, and 4,4′-DDD) were evaluated as total DDTs 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.

Uncertainty Associated with Surface Water Sampling Methods

There are uncertainties associated with the use of multiple sampling types and methods (e.g., duration of sampling time for a single point grab sample versus an integrated transect sample or volume of water sampled in an XAD versus peristaltic sample) in the evaluation of ecological exposure to surface water. Surface water samples were collected both as single-point samples and as transect (vertical and/or horizontal) 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 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 both transect, single-point, XAD, and peristaltic samples allows for the evaluation of surface water data over a larger spatial scale; however, the relevance of ecological exposure to surface water data collected from the various sampling types is highly uncertain.

For small-home-range fish, EPCs were represented by individual samples or by samples collected from within a very limited spatial scale.[88] A sample-by-sample basis (or a limited spatial scale for VOCs and naphthalene) was considered a relevant scale for surface water exposure in the evaluation of sculpin. Therefore, the surface water assessment conducted for sculpin is similar to the assessment conducted for benthic invertebrates (Section 6.5). For smallmouth bass and northern pikeminnow, EPCs were represented by UCLs (or maximum concentrations 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, and peamouth), 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.[89] COPC surface water concentration data for all individual samples and 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

[88] 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 sacle for sculpin. In addition, naphthlene results from samples collected over this 0.15-mile stretch were also treated as a single EPC.

[89] 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.

are based on high-resolution analyses with lower detection limits and higher accuracy of results.

Fish habitat areas were not defined for the Study Area and therefore, habitat was not factored into the development of site use factors or exposure areas. All exposure areas throughout the Study Area were assumed to provide some type of fish habitat; however, a lack of habitat (and foraging fish) in some areas could contribute to an overestimation of exposure.

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. 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 was required to derive a UCL (EPA 2007f). In the case where an insufficient number of detected data values were available ($n < 6$), the maximum concentration[90] was used to represent the EPC. 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.

Uncertainty is associated with the use of maximum concentrations to represent surface water EPCs. For some surface water COPCs, only limited data were available to derive the surface water UCL concentrations within 1-mile exposure areas for evaluating risks to smallmouth bass and northern pikeminnow. When a limited number of samples are available, the UCL cannot be reliably estimated and the maximum concentration represents a reasonable conservative estimate of 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.

7.3.2.2 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 if these concentrations might disrupt olfactory function in migrating populations. Impaired olfactory function and avoidance response behaviors have been associated with concentrations of metals in water. Olfactory function and avoidance toxicity studies of salmonids exposed to individual metals in water have focused on two chemicals (i.e., copper and cobalt) with the bulk of the work conducted on copper. Since cobalt was not analyzed in Study Area surface water only copper was evaluated in this assessment.

Surface water concentrations of dissolved copper at 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

[90] When the maximum concentration was a non-detected value, the full detect limit was used to represent the EPC.

copper concentrations as they return from the ocean and migrate through the Study Area, making their way upstream to spawn. The migration of adult spring Chinook salmon through the Multnomah Channel and the Willamette 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).

7.3.3 Effects Assessment

This section present the effects thresholds used to evaluate surface water risks to fish receptors. Section 7.3.3.1 presents the water TRVs that were compared to surface water concentrations. Water TRVs were used to evaluate all fish receptors. Section 7.3.3.2 summarizes the results of the lamprey ammocoete toxicity testing. Lamprey toxicity testing was conducted to determine whether selected water TRVs were protective of lamprey. Section 7.3.3.3 presents effects thresholds associated with impaired olfactory function in salmonids.

7.3.3.1 Water TRVs

Surface water concentrations were compared to 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 6.5.3 and Table 6-27 present the water TRVs developed for all surface water COPCs.

Because the selected AWQC-derived values for total PCBs and 4,4′-DDT 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.

Based on the AWQC document for PCBs (EPA 1980c), toxicity data were insufficient to allow derivation of an FAV or an FCV directly. Therefore, a PCB FAV was calculated using additional toxicity data reported in EPA's ECOTOX online database published since 1979 (ECOTOX 2009). When these additional data were added to those included in the AWQC document for PCBs (EPA 1980c), the data were sufficient to derive an FAV in accordance with the methods used to establish AWQC values. An FAV of 1.6 µg/L was derived and then divided by the geometric mean of the ACRs presented in the AWQC document for PCBs (8.39), to yield a FCV of 0.19 µg/L. This concentration (0.19 µg/L) was evaluated as an alternative water TRV for total PCBs; however, slightly lower chronic values are reported in the AWQC document for fish (lowest chronic value of 0.098 µg/L was reported for fathead minnow) and plants (lowest diatom value is 0.1 µg/L). For evaluating direct exposure of organisms to water, this alternative water TRV is considered more appropriate than the selected total PCB criterion (0.014 µg/L), which is based on protection of mink via ingestion of contaminated prey. Although both the selected PCB water TRV (0.014 µg/L) and the

alternative water TRV (0.19 µg/L) were used to derive water HQs, only the alternative TRV was used to determine risk conclusions.

Based on the AWQC document for DDTs (EPA 1980a), sufficient toxicity data for aquatic organisms were available to derive an FAV directly for 4,4′-DDT FAV (1.1 µg/L) but not an FCV or an ACR. Only a single ACR (65) was identified in the AWQC document for DDT, and Suter and Tsao (1996) recommend using 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) from four studies of chemicals with a DDT-like mode of action. Dividing the FAV (1.1 µg/l) by the median reported ACR of 3.6 results in a chronic value of 0.31µg/L. In accordance with the methods used to establish AWQC values, it is appropriate to use a final tissue-residue value when establishing the numerical criterion, provided that it is likely to be lower than an FCV or final plant value. A tissue-residue-derived water criterion of 0.011 µg/L was calculated by dividing the DDT 10th percentile fish tissue residue LOAEL (1.6 mg/kg ww) by a BAF of 142,960[91] (derived from the DDT AWQC document). This water concentration of 0.011 µg/L is lower than the FAV divided by the ACR, and was evaluated as an alternative water TRV for DDTs in water. For evaluating direct exposure of organisms to water, this alternative TRV is considered more appropriate than the selected DDT criterion (0.0010 µg/L), which are based on the protection of brown pelican via ingestion of contaminated prey. Although both the selected DDT water TRV (0.001µg/L) and the alternative water TRV (0.011 µg/L) were used to derive water HQs, only the alternative TRV was used to determine risk conclusions.

Surface Water TRV Uncertainties

The TRVs established by the AWQC or Tier II criteria were based on or included toxicological data for fish for the eight COPCs: aluminum, zinc, butyltin, naphthalene, total PCBs, BEHP, ethylbenzene, and trichloroethene. These TRVs were considered adequately protective for evaluating risks to fish. However, the Tier II-derived TRVs for two COPCs (i.e., benzo[a]anthracene and benzo[a]pyrene) were based on invertebrate species (*Daphnia* sp.) toxicity data. Because the water TRVs for these COPCs are based on the effects data for benthic invertebrates, they may over- or underestimate risks to fish.

In addition, the water TRV for DDTs based on the 4,4′-DDT AWQC is derived from the effects data for only one bird species (brown pelican). Birds are known to be more sensitive to DDTs, and the water TRV that is protective of birds is of questionable relevance and likely overestimates risks to fish. Similarly, the total PCB AWQC is based on the protection of mink via ingestion of contaminated prey. Alternative water TRVs were developed for total PCBs and DDTs following the methods used to develop AWQC values; 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 (and determine COCs), only the alternative TRV was used to determine risk conclusions.

[91] A BAF of 142,960 was based on the percent lipid-normalized BAF (17,870) and anchovy lipid percent (8%) as presented in the DDT AWQC document.

7.3.3.2 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 (2006c) to determine whether aquatic toxicological water thresholds used in this risk evaluation (i.e., water TRVs) are protective of lamprey survival and growth at the organism level.

With the exception of studies with sea lamprey during the development of a lampricide for use in the Great Lakes watershed, lamprey species have not been widely studied by aquatic toxicologists. To reduce this uncertainty, acute toxicity tests of six chemicals (i.e., pentachlorophenol, copper, aniline, diazinon, naphthalene, and lindane) were conducted on field-collected lamprey ammocoetes from four uncontaminated Oregon coastal streams. These six chemicals were selected to represent a range of toxic modes of action to assess if TRVs based on 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 (concentration that is lethal to 50% of an exposed population) for each species on the x-axis and the cumulative probability (estimated fraction of species with LC50s at or below the corresponding LC50 value on the x-axis) on the y-axis. The measured lamprey LC50 concentrations were compared to 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 chemicals, lamprey ammocoetes were most sensitive to pentachlorophenol, which represented the oxidative phosphorylation uncoupler mode of action. However, even in the case of pentachlorophenol, the ammocoetes were not more sensitive than rainbow trout. The sensitivity of lamprey ammocoetes to copper approximated the average for other aquatic species. For the other four chemicals (i.e., aniline, diazinon, naphthalene, and lindane), the sensitivity of lamprey ammocoetes fell within the upper percentiles of their respective SSDs, indicating that lamprey ammocoetes were less sensitive than were most aquatic species tested.

Table 7-34. Summary of Lamprey LC50s Compared to LC50s of other Aquatic Species

Analyte	Lamprey LC50 (µg/L)	LC50 Range of Other Aquatic Species (µg/L)	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 **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.

NA – not applicable

Because lamprey were found to be equally or less sensitive than rainbow trout and other salmonids for all of the six chemicals 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.

7.3.3.3 Olfactory Function and Avoidance Behavior Effects in Migrating Salmonids

Toxicological studies on the disruption of olfactory function and avoidance behavior in salmonids exposed to metals in water were reviewed, and the relevance of these endpoints on migrating salmonids was evaluated.

7.3.3.3.1 Toxicological Studies on Olfactory Function and Avoidance Behavior Response

Chinook salmon olfactory function can be affected by exposure to dissolved metals and other contaminants in water (Klaprat et al. 1992; Baldwin et al. 2003). Olfactory function in fish plays a major role in mediating behaviors important for both survival and reproduction, such as juvenile imprinting on home waters, predator avoidance, and adult migration and homing (Baldwin et al. 2003). Typically, the presence of metals in water produces avoidance behavior in fish (Hansen et al. 1999c; Atchison et al. 1987). Field studies have shown that in cases where distinct spatial gradients of metals occur (e.g., near point source discharges), fish may use their sense of smell to avoid contamination (Saunders and Sprague 1967). Lorz and McPherson (1976) reported that the exposure of 18-month-old coho salmon to 5 µg/L of copper resulted in a 30% reduction in downstream migration. Similarly, Baldwin et al. (2003) reported that the presence of metals in water may disrupt migration patterns or prevent fish from inhabiting areas that would otherwise offer productive habitat.

The exposure of salmonids to individual metals in laboratory studies at levels of contamination below those known to be lethal has been shown to produce avoidance behavior. A literature review was conducted of laboratory studies involving salmonids and exposure to metals in water in which avoidance behavior was observed, and the results are summarized in Table 7-35. Based on the reviewed literature, salmonids displayed avoidance behavior at copper concentrations ranging over several orders of magnitude, from 0.10 to 88 µg/L. Hardness affects copper chemistry; therefore, 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 . It should be noted, however, that Baldwin et al. (2003) demonstrated that hardness does not influence toxicity of copper to olfactory function and similarly may not be an important factor in avoidance behavior. Interestingly, dissolved organic carbon in concentrations ≥6 mg/L was shown to reduce the effects of copper on salmon olfactory function (McIntyre et al. 2008).

Impairment of the olfactory system after exposure to metals is temporary. Partial recovery of olfactory cells after short-term exposure (30 minutes) to low copper concentrations (< 25 µg/L) occurs within 60 minutes (Hansen et al. 1999a). 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.

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)
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[e]	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]
Juvenile coho	13[f]	100	Baldwin et al. (2003)
Rainbow trout	< 22[g]	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]
Juvenile coho	2[h]	120	Sandahl et al. (2007)

[a] Effect level is the lowest reported concentration at which avoidance behavior was observed for a given species in each toxicological study reviewed.

[b] As cited in Brooks (2004).

[e] 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] 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.

[f] Fish experienced a 50% loss of olfactory function when exposed to 13 µg/L copper for 30 minutes.

[g] Copper concentrations were measured as Cu^{2+}.

[h] 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.

7.3.3.3.2 Relevance of Avoidance Response to Migration

Salmonid populations can survive in environments with dissolved copper concentrations that range from 2 to 23 µg/L (e.g., Copper River, Alaska; Sacramento River, California) (Brooks

2004). Adult and juvenile salmonids survive and successfully navigate in both of these complex river systems (i.e., the Copper River and Sacramento River), 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 may not avoid lethal concentrations. A field study conducted by Saunders and Sprague (1967), however, 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 Risk Characterization

This section presents the risk estimates for fish based on the surface water LOE. An HQ calculation was used to quantify surface water risk estimates and characterize surface water risks. HQs were derived for all COPCs using Equation 7-1, in which the EPC and TRV represent surface water concentrations. Section 7.3.4.1 presents the overall approach used to characterize risks via surface water to fish receptors. Section 7.3.4.2 presents the risk characterization results, uncertainty evaluation, and COCs for each fish receptor. Section 7.3.4.3 presents an evaluation of background concentrations, and Section 7.3.4.4 presents a summary of surface water COCs.

7.3.4.1 Risk Characterization Process

The risk characterization process for assessing risks to small-home-range fish (i.e., sculpin, smallmouth bass, and northern pikeminnow) via surface water was conducted by evaluating individual surface water samples. An HQ was calculated for each individual surface water sample within the Study Area in accordance with the EPA's Problem Formulation (Attachment 2). A sample-by-sample basis was considered a relevant scale in the evaluation of sculpin. COPCs with HQs > 1.0 for any individual surface water sample were identified as COCs for sculpin.

For smallmouth bass and northern pikeminnow, exposure areas of 1 mile were considered a relevant scale. COPCs with HQs > 1.0 based on 1-mile-exposure-area UCLs (or maximum concentrations) were identified as COCs for smallmouth bass and northern pikeminnow. For large-home-range and migratory fish receptors (i.e., largescale sucker, juvenile white sturgeon, juvenile Chinook salmon, and peamouth), the Study Area was considered a relevant scale. COPCs with HQs > 1.0 based on site-wide UCLs were identified as COCs for large-home-range fish.

7.3.4.2 Risk Characterization Results and Uncertainty Evaluation

HQs calculated for all surface water COPCs on a sample-by-sample basis are discussed and presented in Section 6.5.4 and Table 6-28. Sculpin HQs were > 1.0 for all COPCs except ethylbenzene. Sculpin VOC HQs were derived from a comparison of a single EPC (based on samples collected as part of a non-LWG sampling event from approximately RM 6.4 to

RM 6.5) to the TRV. The sculpin HQ for ethylbenzene was 0.32 based on the UCL concentration, and the sculpin HQ for trichloroethene was 4.1 based on the maximum concentration collected from approximately RM 6.4 to RM 6.5. The trichloroethene HQ for sculpin based on the maximum concentration may overestimate risks to sculpin because trichloroethene was detected in only 2 of 23 samples collected from RM 6.4 to RM 6.5; the trichloroethene concentrations were 0.61 µg/L and 194 µg/L in the two detected samples, and trichloroethene was not detected above the detection limit (0.2 µg/L) in all other samples. Trichloroethene HQs were < 1.0 in all other samples, except for the sample in which the maximum detected concentration was measured. Sculpin HQs for naphthalene were < 1.0 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 was 14.

HQs calculated based on 1-mile-exposure-area and site-wide UCLs are presented in Tables 7-36 and 7-37, respectively. HQs in 1-mile exposure areas were > 1.0 for all COPCs except total PCBs. Site-wide HQs were > 1.0 for three COPCs (i.e., aluminum, naphthalene, and total DDx). No site-wide or 1-mile exposure area UCLs were derived for VOCs (ethylbenzene and trichloroethene) because of the limited spatial extent of the data.

Table 7-36. Surface Water 1-Mile Exposure Area HQs

Exposure Area	Aluminum	Zinc	Butyltin ion	Benzo(a) anthracene	Benzo(a) pyrene	Naphthalene	BEHP	Total PCBs	4,4'-DDT	Total DDx
RM 1.5 to RM 2.5	**3.7**	0.052	*0.35*[a]	0.020	0.028	*0.0037*[a]	*0.53*	0.23	0.061	**7.6**
RM 2.5 to RM 3.5	**10**	*0.056*[a]	*0.24*[a]	0.072	0.13	*0.0043*[a]	*0.43*	0.039	0.085	0.43
RM 3.5 to RM 4.5	**6.9**	0.058	*0.25*	0.11	0.12	*0.0041*[a]	***2.3***	0.42	0.53	**1.2**
RM 4.5 to RM 5.5	*1.0*	*0.13*	*0.28*[a]	*0.29*	*0.54*	*0.00050*[a]	*0.073*[a]	*0.089*[a]	*0.26*[a]	*0.26*[a]
RM 5.5 to RM 6.5	**8.2**	0.062	*0.28*	**4.7**	**7.6**	**2.8**[b]	*0.73*	0.045	**1.7**	**2.0**
RM 6.5 to RM 7.5	**6.7**	0.049	*0.24*[a]	0.063	0.18	0.00052	0.51	0.55	**2.1**	**6.5**
RM 7.5 to RM 8.5	**5.7**	*0.18*	*0.039*[a]	*0.14*[a]	*0.31*[a]	*0.0013*[a]	*0.68*[a]	*0.091*[a]	*0.69*	***1.9***
RM 8.5 to RM 9.5	**30**[c]	*0.13*	*0.076*[a]	*0.14*[a]	*0.31*[a]	*0.013*	*0.93*	*0.094*[a]	***2.9***	**3.4**
Swan Island Lagoon	**4.0**	0.089	*0.26*	0.010	*0.43*	*0.0027*[a]	*0.32*[a]	0.078	*0.058*	0.21
RM 9.5 to RM 10.5	**8.5**[c]	***1.2***	*0.21*	*0.14*[a]	*0.31*[a]	*0.003*	*0.33*	0.29	***4.7***	**3.4**
RM 10.5 to RM 11.8	**23**[c]	*0.060*	***1.2***	0.015	0.035	*0.0033*[a]	*0.37*	0.032	0.19	0.84[c]

[a] Maximum concentration is based on one-half DL (where one-half DL > maximum detected concentration or where COPC is not detected).

[b] 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.

[c] Calculated UCL concentration is greater than the maximum concentration.

[c] HQ is based on 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.

DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
HQ – hazard quotient
NA – not applicable
ND – no data
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)

Bold identifies HQs > 1.0.

Italics identify HQs derived from maximum concentration; UCL concentration could not be derived (n detects < 6).

Table 7-37. Summary of Site-Wide Surface Water UCL HQs

COPC	Unit	UCL	Water TRV	HQ
Metals				
Aluminum	µg/L	460	87	**5.3**
Zinc	µg/L	2.5	36.5	0.068
Butyltins				
Butyltin ion	ng/L	4.3	72	0.059
PAHs				
Benzo(a)anthracene	ng/L	6.9	27	0.26
Benzo(a)pyrene	ng/L	9.5	14	0.68
Naphthalene	ng/L	3.18 [a]	12,000	0.00027
Phthalates				
Bis(2-ethylhexyl) phthalate	ng/L	540	3,000	0.18
PCBs				
Total PCBs	ng/L	1.8	14	0.13[b]
Pesticides				
4,4′-DDT	ng/L	0.67	1	0.67[c]
Total DDx	ng/L	1.6	1[d]	**1.6**[c]

[a] Naphthalene UCL is based on results of all surface water samples 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.

[b] UCL did not exceed the alternative total PCB TRV of 190 ng/L either.

[c] UCLs for 4,4′-DDT and total DDx did not exceed the alternative 4,4′-DDT TRV of 11 ng/L (HQs were 0.063 and 0.15, respectively).

[d] Criterion for 4,4′-DDT used to evaluate total DDx.

COPC – chemical of potential concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
HQ – hazard quotient
PAH – polycyclic aromatic hydrocarbon
Bold identifies HQs > 1.0.
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
VOC – volatile organic compound

All 12 COPCs were identified as COCs for sculpin because HQs were > 1.0 in at least one surface water sample. A detailed discussion of HQs on a sample-by-sample basis is presented in Section 6.5.4, and samples with HQs > 1.0 are presented on Maps 6-14 through 6-18. Ten COPCs (i.e., aluminum, zinc, butyltin, benzo[a]anthracene, benzo[a]pyrene, naphthalene, BEHP, 4,4’-DDT, total DDx, and trichloroethene) were identified as COCs for smallmouth bass and northern pikeminnow, for which HQs were > 1.0 in at least one 1-mile exposure

area. Three COPCs (i.e., aluminum, naphthalene, and total DDx) were identified as COCs for large-home-range fish, for which site-wide HQs were > 1.0. There is high uncertainty with the identification of total PCBs as a COC for all fish and of total DDx as a COC for smallmouth bass, northern pikeminnow, and large-home-range fish because HQs were < 1.0 when calculated using the alternative TRV, which is based on direct exposure to surface water.

Aluminum was identified as a COC because the HQ > 1.0. However, aluminum and other trace elements are major constituents of the mineral fraction of sediment but contribute to the analytical chemical results as a result of the acid extraction step during analysis. Because aluminum is not biologically available to fish and not toxic at naturally occurring concentrations generally found in surface water, aluminum is not expected to pose unacceptable risks to fish.

7.3.4.3 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 disrupt olfactory function in migrating populations. Copper concentrations associated with salmonid avoidance response behavior were compared to 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 (0.10 to 88 µg/L). 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 that it is unlikely there is an adverse effect from copper concentrations in surface water in the Study Area. Furthermore, these copper concentrations are typical of regional levels; based on summary statistics from the US Geological Survey (USGS) National Stream Quality Accounting Network program, water copper concentrations at six sites along the Columbia River Basin from 1996 to 2003 ranged from < 1.0 µg/L to 6.7 µg/L (USGS 2006). Therefore, the pathway associated with reproductive impairment is incomplete and the potential risk from olfactory copper avoidance is insignificant.

7.3.4.4 Evaluation of Background Concentrations

Aluminum was evaluated even though it is a naturally occurring crustal element in the environment. Background concentrations were established as part of the RI (see Section 7.0 of the draft RI). 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 risks 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 unacceptable risks to fish from aluminum are occurring.

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[92] (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 solely be attributed to background.

7.3.4.5 Summary of Surface Water COCs

Eleven surface water COCs were identified for small-home-range fish (i.e., aluminum, zinc, butyltin, benzo[a]anthracene, benzo[a]pyrene, naphthalene, BEHP, total PCBs, 4,4′-DDT, total DDx, and trichloroethene). Three COCs were identified for large-home-range fish (i.e., aluminum, naphthalene, and total DDx). Table 7-38 summarizes the HQs, key uncertainties, and risk conclusions for each of the COCs. The results of the surface water LOE were compared with other LOEs to determine risk conclusions for fish and are presented in Section 7.6.

[92] Zinc concentrations were detected in only 3 of 22 surface water samples included in the background dataset (see Section 7.0 of the RI).

Table 7-38. Summary of Fish Surface Water COCs

Receptor by COC	Risk Exposure Area	HQs[a]	Frequency of All Samples with HQs > 1.0	Sampling Event with HQs > 1.0	Key Uncertainties
Aluminum					
Sculpin	Site-wide	1.0 – 21	93%	All	Aluminum concentrations in background areas indicate aluminum is not attributed to anthropogenic sources in the Study Area and not expected to be toxic to fish at naturally occurring concentrations; TRV is based on toxicity to fish and invertebrates; fish may be less sensitive to aluminum than are invertebrates.
Smallmouth bass and northern pikeminnow	Site-wide	1.0 – 30[b]	NA	NA	
Large-home-range fish receptors	Site-wide	5.3	NA	NA	
Zinc					
Sculpin	Individual sample with HQ > 1.0 at RM 9.7,west	1.1	0.6%	November 2004 low-flow event	HQ was > 1.0 based on one sample only; TRV is based on toxicity to fish and invertebrates; fish may be more sensitive to zinc than are invertebrates; zinc concentrations are elevated in Study Area as compared to background.
Smallmouth bass and northern pikeminnow	RM 9.5 to 10.5	1.1	NA	November 2004 low-flow event	
Butyltin					
Sculpin	Individual sample with HQ > 1.0 at RM 11, west	1.2	0.6%	Winter 2007 high-flow event	TRV is based on surrogate TBT TRV; HQ was > 1.0 based on one sample only; TRV is based on toxicity to fish and invertebrates; fish may be less sensitive to butyltin than are invertebrates.
Smallmouth bass and northern pikeminnow	RM 10.5 to 11.8	1.2	NA	Winter 2007 high-flow event	

Table 7-38. Summary of Fish Surface Water COCs

Receptor by COC	Risk Exposure Area	HQs[a]	Frequency of All Samples with HQs > 1.0	Sampling Event with HQs > 1.0	Key Uncertainties
Benzo(a)anthracene					
Sculpin	Individual samples with HQs > 1.0 at RM 6.1, west; and RM 6.3, west	4.1 – 10	0.8%	July 2005 low-flow event and winter 2007 high-flow event	HQ > 1.0 based on peristaltic samples only at W012 and W031; TRV is based on extrapolated *Daphnia* acute LC50.
Smallmouth bass and northern pikeminnow	RM 5.5 to RM 6.5	4.7	NA	November 2004 and July 2005 low-flow events and winter 2007 high-flow event	
Benzo(a)pyrene					
Sculpin	Individual samples with HQs > 1.0 at RM 6.1, west; and RM 6.3, west	1.4 – 14	1%	November 2004 and July 2005 low-flow events and winter 2007 high-flow event	HQ > 1.0 based on peristaltic samples only at W012 and W031; TRV is based on extrapolated *Daphnia* acute LC50.
Smallmouth bass and northern pikeminnow	RM 5.5 to RM 6.5	7.6	NA	November 2004 and July 2005 low-flow events and winter 2007 high-flow event	

Table 7-38. Summary of Fish Surface Water COCs

Receptor by COC	Risk Exposure Area	HQs[a]	Frequency of All Samples with HQs > 1.0	Sampling Event with HQs > 1.0	Key Uncertainties
Naphthalene					
Sculpin	RM 6.4 to RM 6.5, west	14	4%	May 2005 (non-LWG sampling event)	HQs > 1.0 are based on a UCL of samples over a 0.15-mile stretch from RM 6.4 to RM 6.5; HQs for all other samples were < 1.0; TRV is based on extrapolated TRV from fish LC50. No samples exceeded the only chronic value for fish (619 µg/L) reported in Suter and Tsao (1996).
Smallmouth bass and northern pikeminnow	RM 5.5 to RM 6.5	2.8	NA	May 2005 (non-LWG sampling event)	HQs > 1.0 are based on elevated concentrations from samples over a 0.15-mile stretch from RM 6.4 to RM 6.5; TRV is based on risk to fish and invertebrates.
BEHP					
Sculpin	Individual samples with HQs > 1.0 at RM 3.9 (transect location) and RM 6.7 (Willamette Cove)	1.2 – 2.3	2%	November 2006 storm runoff event and winter 2007 high-flow event	HQs > 1.0 (n = 2 samples) in peristaltic samples only; two additional exceedances are based on DLs; TRV is based on risk to fish and invertebrates.
Smallmouth bass and northern pikeminnow	RM 2.5 to RM 3.5	2.3	NA	November 2006 storm runoff event	HQs based on maximum concentration from exposure area (peristaltic sample); TRV is based on risk to fish and invertebrates.

Table 7-38. Summary of Fish Surface Water COCs

Receptor by COC	Risk Exposure Area	HQs[a]	Frequency of All Samples with HQs > 1.0	Sampling Event with HQs > 1.0	Key Uncertainties
Total PCBs					
Sculpin	Individual samples with HQs > 1.0 at RM 3.7 (International Slip) and RM 6.9 (Willamette Cove)	1.1 – 1.2	1%	November 2004 low-flow event and March 2005 low-flow event	HQs > 1.0 in peristaltic samples only at W014 and W004; TRV is based on risks to mink. No samples exceed the alternative water TRV that is protective of direct exposure to water.
Total DDx					
Sculpin	Site-wide; the highest HQs that were based on non-N-qualified data were located at RM 7.2 and RM 6.9	1.1 – 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 > 1.0 (n = 11 samples) 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 (0.001 µg /L) is based on risk to birds. One individual sample exceeds alternative water TRV (0.011 µg/L) that is protective of direct exposure to surface water (HQ = 1.8); however, this sample (W001 at RM 2.1 on the east bank) is N-qualified. The lowest chronic effect value for fish in the AWQC document is 0.74 µg/L.
Smallmouth bass and northern pikeminnow	RM 1.5 to RM 2.5; RM 3.5 to RM 4.5; RM 5.5 to RM 6.5; RM 6.5 to RM 7.5; RM 7.5 to RM 8.5; RM 9.5 to RM 10.5	1.2 – 7.6	NA	NA	
Large-home-range fish receptors	Site-wide	1.6	NA	NA	

Table 7-38. Summary of Fish Surface Water COCs

Receptor by COC	Risk Exposure Area	HQs[a]	Frequency of All Samples with HQs > 1.0	Sampling Event with HQs > 1.0	Key Uncertainties
Trichloroethene					
Sculpin	RM 6.4 to RM 6.5, west	4.1	4%	May 2005 (non-LWG sampling event)	Available VOC data are limited to single samples over a 0.15-mile stretch at RM 6.4 to RM 6.5 on the west bank; HQ based on maximum concentration and risks may be overestimated because trichloroethene was detected in only 2 of 23 samples collected from RM 6.4 to RM 6.5; HQs were < 1.0 in all other samples except for the sample in which the maximum detected concentration was measured; TRV is based on risk to fish.

[a] HQs only > 1.0 are presented. HQs in all other water samples are < 1.0.

[b] HQs based on UCLs from 1-mile exposure areas that are > the maximum concentration measured in the exposure area.

BEHP – bis(2-ethylhexyl) phthalate
COC – chemical of concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
HQ – hazard quotient
J – estimated concentration
LC50 – concentration that is lethal to at least 50% of an exposed population
LWG – Lower Willamette Group
N – tentative identification
NA – not applicable
PCB – polychlorinated biphenyl
RM – river mile
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)
TRV – toxicity reference value
VOC – volatile organic compound

7.4 ASSESSMENT OF BENTHIC FISH HEALTH AND PAH EXPOSURE

Per the 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 as follows:

Section 7.4.1 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 7.4.2 presents a comparison of the Study Area sediment PAH concentrations to literature-based thresholds associated with lesion occurrence.

Section 7.4.3 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 7.4.4 presents the conclusions for this LOE.

Figure 7-16 presents a flowchart of the fish health and PAH exposure LOE section organization.

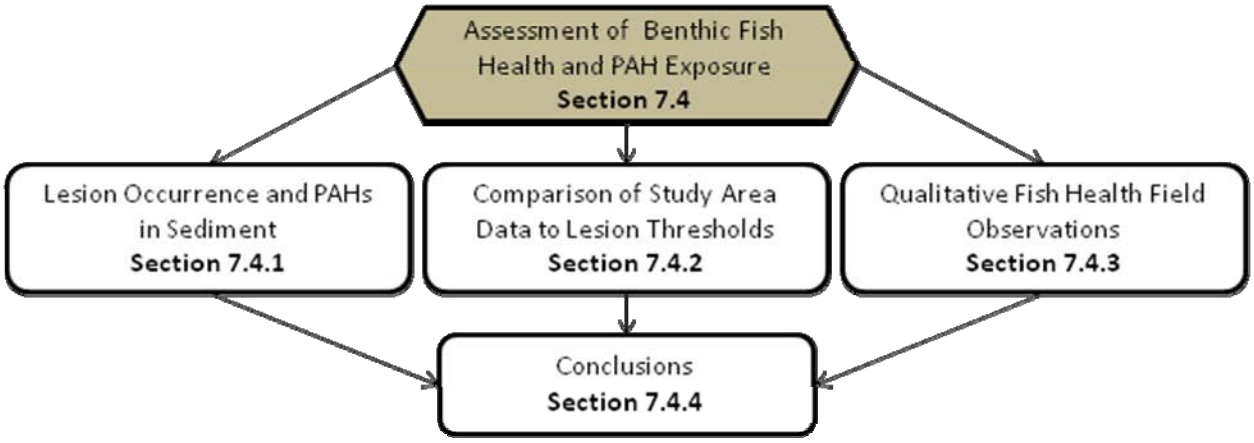

Figure 7-16. Overview of Assessment of Benthic Fish Health and PAH Exposure Section Organization

7.4.1 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 7.4.1.1 summarizes studies that involved external lesions and other deformities; Section 7.4.1.2 summarizes studies that reported the incidence of hepatic lesions. The implications of lesion prevalence for population-level effects are discussed in Section 7.4.1.3.

7.4.1.1 Toxicological Studies on External Lesions and Other Deformities

External lesions observed in fish exposed to PAHs in sediment included epidermal papillomas, mucoid plaques, and lip papillomas, as well as epidermal ulcerations/abrasions and fin lesions (Baumann et al. 1996; Mezin and Hale 2000). In one toxicological study, 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 (Mezin and Hale 2000). 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.[93] Tumor prevalence in Anacostia fish were compared with tumor

[93] Sediment PCBs and chlordane concentrations, which were also elevated within the Study Area, were not reported.

prevalence in fish from an uncontaminated control site with an average total PAH[94] 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 the 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 elevated to within the range of 6,480 to 6,750 µg/kg.

7.4.1.2 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 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[95] 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)[96] 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

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[94] 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.

[95] A hockey stick regression is a dose-response type model that assumes a threshold must be reached before initiation of a response.

[96] Total PAHs were the sum of 10 LPAHs and 8 HPAHs.

average concentration (SWAC)[97] 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).

7.4.1.3 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, 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 in a study of estimated annual mortality rates at both highly contaminated (e.g., Eagle Harbor) and uncontaminated sites throughout Puget Sound. English sole mortality rates from contaminated sites associated with high liver lesion prevalence were not found to be significantly greater than mortality rates for English sole from Puget Sound as a whole. Johnson and Landahl also examined the English sole population structure and found no evidence of increased age-related mortality in fish with lesions or in populations associated with areas of high levels of PAHs and PCBs. The authors concluded that fish populations that have high incidence of lesions do not

[97] Based on the sum of 10 HPAHs.

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.

7.4.1.4 Other Factors Contributing to Lesion Prevalence

Several factors other than sediment PAH concentrations (e.g., age, sex, and other non-chemical stressors) have been shown to be significant risk factors for lesion prevalence in benthic fish (Baumann et al. 1987; Myers et al. 1994; Pinkney et al. 2000, 2004a). Age has been identified as an important risk factor for lesion development in brown bullhead, with the incidence of skin tumors increasing 2.5 times per year and the incidence of liver hepatocarcinoma increasing 3.5 times per year as fish age (Pinkney et al. 2000). Additional studies support the finding that age plays a role in lesion development in benthic fish (Baumann et al. 1987; Myers et al. 1994). The odds of developing hepatocarcinoma in brown bullhead has also been linked to sex, with a higher incidence (by a factor of 4.5) of these lesions observed in female fish over male fish (Pinkney et al. 2000). The accumulation of bile PAH metabolites has been identified as a risk factor for lesions as well; 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).

7.4.1.5 Summary of Toxicological Studies

Several studies have proposed hepatic lesion sediment effects thresholds for PAH ranging from 230 to 4,000 µg/kg and spanning over an order of magnitude (Horness et al. 1998; Johnson et al. 2002; Stern et al. 2003). Johnson et al. (2002) suggested a 1,000-µg/kg PAH threshold as being be both protective of the majority of fish species and practical for making management decisions based on the “any lesions” category. Johnson et al. also reported a threshold of 2,800 µg/kg for neoplasms (benign or cancerous). Stern et al. (2003) reported an effects threshold of 2,700 µg/kg for the “any lesions” category and stated that a

hockey stick regression using methods that account for the size of English sole's foraging range would likely result in an even higher threshold for neoplasms. Pre- and post-dredge data from Eagle Harbor indicate that English sole hepatic lesion incidence was approximately equal to background levels at a SWAC in sediment of 4,000 µg/kg (Stern et al. 2003). Pre- and post-dredge data from the Black River indicate that brown bullhead lesion incidence and population structure were approximately equal to background levels at an area-weighted average sediment concentration of 15,000 µg/kg (Baumann 2000).

7.4.2 Comparison of Study Area Data to Lesion Thresholds

Study area PAH concentrations in sediment were compared to the PAHs concentrations associated with lesion occurrence in field studies. Total PAH[98] 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 level is 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.

There is high uncertainty associated with literature-based PAH thresholds for fish lesion incidence because: 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 link between survival, growth, or reproduction (and effects on the population) and the occurrence of lesions in fish, and effect-level PAH concentrations in sediments has not been demonstrated in field studies. Furthermore, the PAHs in sediment from areas associated with certain industrial activities have lower bioavailability in sediment 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 relative to those in the sediments of other studies is not known, the comparison of Study Area sediment concentrations to literature-based PAH thresholds is

[98] 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.

uncertain. Due to the high uncertainty of this LOE described above, fish exposure to PAHs in sediment was not further quantitatively evaluated.

7.4.3 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 7.4.2), 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[99] 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 the US Fish and Wildlife Service (USFWS) (2007) and the USGS Biomonitoring of Environmental Status and Trends (BEST) protocol (USGS 2002). The USGS BEST protocol for examining fish provides generic identifications 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 et al. 2000), gross external abnormality data were collected as part of the Round 3 juvenile white sturgeon sampling event for 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 to 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 stations in the Columbia River Basin from September 1997 to April 1998 (Hinck et al. 2004). Two of the sampling stations were located in the Willamette River, one station was located at RM 10 (in the Study Area), and one station was located 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.[100]

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[99] 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.

[100] 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.

The incidence of non-fin fish 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 old),[101] 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.

In conclusion, while these data are qualitative and may support exposure to PAH-contaminated sediment, these data are not conclusive of population-level effects to benthic fish as a result of exposure to PAH-contaminated sediment from the Study Area. Comparison of Study Area juvenile white sturgeon health observations are not directly comparable to the health observations made for eight other fish species collected from the Columbia River Basin; however, qualitative comparisons can be made. The incidence of abnormalities on fish is nearly impossible to attribute to a single factor and is likely the result of confounding factors, including variation in species type and 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 the interpretation of these data, there can be no conclusive link between the field observations made on Study Area fish and overall status of the population.

7.4.4 Conclusions

The evaluation of benthic fish health, through the incidence of lesions and abnormalities from excessive exposure to PAHs is uncertain and largely inconclusive. The sediment concentrations established in the literature for lesion occurrence are highly variable and span over several orders 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 conclusively establishing the link between population-level effects of

[101] 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).

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 collected from the Study Area indicate some incidence of abnormalities that is somewhat higher than the incidence of abnormalities reported for other fish species in the region; however the differences in these data may be attributed to such factors as the difference of species, age of fish collected, fish handling methods, and the differences in field researchers' qualitative assessments of abnormalities.

Therefore, though the presence of abnormalities in benthic fish may be an indication of exposure to PAH-contaminated sediment, these data are not conclusive of excessive PAH exposure or of population-level effects to benthic fish.

7.5 TZW ASSESSMENT

The TZW assessment is presented in detail of Section 6.6. A summary of the TZW assessment is provided in this section.

A TZW[102] exposure pathway was evaluated for benthic fish (i.e., sculpin and lamprey ammocoetes) per EPA's Problem Formulation (Attachment 2). TZW was screened against surface water screening values. The TZW data are limited, and results of the TZW assessment were not incorporated into the risk conclusions section (Section 7.6).

TZW was sampled along nine facilities within the Study Area. The TZW exposure pathway for specific aquatic receptors (i.e., benthic invertebrates, benthic fish, amphibians, and aquatic plants) was evaluated by comparing TZW concentrations to water TRVs. Sixty TZW COPCs were identified in the SLERA (Attachment 5) based on a comparison of the maximum detected concentrations in any one TZW sample to water TRVs. A sample-by-sample summary of TRV exceedances of these COPCs along the nine facilities is presented in Table 6-33. At all facilities, one or more TZW chemical concentrations exceeded its corresponding TRV; however, many HQs were low (< 3.0). The complete results of this evaluation are presented in Section 6.6.

There is high uncertainty associated with the TZW evaluation. Because of the spatial focus of the TZW data collection, the TZW chemical concentrations are not representative of conditions throughout the Study Area. The TZW collected from the Study Area is a snapshot of only the specific areas where the data were collected (i.e., no Study Area- wide spatial inferences can be made). Furthermore, there are uncertainties regarding the relevance of certain TRVs for effects on receptors because of the number of studies and species used to derive those values. Uncertainties regarding water TRVs used to evaluate TZW are discussed in Section 6.6.6. Consistent with the agreement between the LWG and EPA managers, COCs were not identified on the basis of the TZW evaluation.

[102] 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.

7.6 RISK CONCLUSIONS AND UNCERTAINTY ANALYSIS

This section presents a summary of the overall risk conclusions of the fish risk assessment. Fish COCs were identified based on the results of the tissue-residue, dietary, and surface water LOEs. For fish, a qualitative WOE was used to evaluate the results of each LOE, when applicable, to determine risk conclusions. Risk conclusions were also determined by evaluating the magnitude of HQs, spatial distribution and frequency of HQ > 1.0, 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.

Several COPCs have, through the BERA's risk characterization, been identified as COCs for one or more fish receptors. These are identified and discussed below. Not all of the COCs are equal potential contributors to unacceptable ecological risk in the Study Area. Most of them are unlikely to pose unacceptable risks to fish populations at the levels of exposure occurring in the Study Area, even though the BERA has estimated exposures greater than TRVs. This is because most COCs only exceed TRVs to a limited extent, and TRVs are largely based on organism-level (versus population-level) effects. For all receptors that were evaluated at the population level, LOAELs were used to derive HQs. However, LOAELs are based on organism-level effects (e.g., reduced growth, reduced fecundity, increased mortality). Because of the uncertainties and limitations of using LOAELs to extrapolate to population-level risks, the magnitude and frequency of LOAEL exceedances, as well as the endpoints used to derive the LOAEL were evaluated to examine the potential for population-level effects. It is unlikely that a COC whose TRV exceedance was of limited spatial distribution and whose HQ was low poses significant risks to populations. COCs with broader distribution and higher concentrations relative to TRVs generally have greater a higher potential for posing population-level risks.

The type of effect threshold endpoint (i.e., survival, growth or reproduction) affects the interpretation of HQs simply because effects on the survival of individuals have different implications for populations than do effects on the growth of individuals. Effects on the reproductive success of individuals have different implications for populations than do effects on either survival or growth.

Even for a particular type of effect threshold endpoint (e.g., reduced survival), a particular HQ (e.g., HQ = 5) has different implications for each COC-receptor pair, because the exposure-response relationship of one COC-receptor pair differs from that of other COC-receptor pairs. Short of developing structured population models and incorporating the effects of COC-receptor-specific exposure-response data, the demographic (i.e., population-level) consequences of any particular HQ simply cannot be quantified.

Populations are resilient to environmental stressors (Barnthouse et al. 2009). By corollary, it is unreasonable to assume that survival, growth, or reproduction HQs > 1.0 will necessarily be detrimental to the population whose individuals are potentially affected.

Section 7.6.1 presents a summary of the fish COCs. Section 7.6.2 presents the qualitative WOE considerations for specific chemical groups. Section 7.6.3 presents the risk conclusions for all fish COCs. In Section 11.0, the fish risk conclusions are combined with the risk conclusions for other ecological receptor groups to provide a holistic view of ecological risks; COCs and receptors that that indicate unacceptable ecological risks are highlighted, in the sense that a remediation based on those COC-receptor pairs would address potential unacceptable risks posed by other COCs and to other ecological receptors.

7.6.1 Fish COCs

Table 7-39 presents a summary the fish COCs identified though each LOE. Sixteen COPCs were identified as COCs for at least one fish receptor based on the tissue, dietary, and surface water LOEs. Fish COCs were grouped into three general chemical groups to evaluate risk conclusions: organics and non-regulated metals, metabolically regulated metals, and PAHs.

Table 7-39. Summary of Fish COCs For Each LOE

COC	Line of Evidence Resulting in COC[a]					Receptor(s)[b]
	Surface Water	Tissue – Empirical	Tissue – Predicted	Diet	Fish Condition	
Organics and Non-Regulated Metal COCs						
Mercury	Not a COC	X	NA	Not a COC	NA	Northern pikeminnow
Tributyltin	Not a COC	Not a COC	NA	X	NA	All fish receptors, except lamprey
Butyltin	X	NA	NA	NA[c]	NA	Sculpin, smallmouth bass, northern pikeminnow
BEHP	X	NA[d]	NA	NA	NA	Sculpin, smallmouth bass, northern pikeminnow
Total PCBs	X	X	X	NA	NA	Largescale sucker, sculpin, smallmouth bass, northern pikeminnow
Total DDx	X[e]	X	X	NA	NA	All fish receptors
Trichloroethene	X	NA	NA	NA	NA	Sculpin
Regulated Metal COCs						
Aluminum	X	NA	NA	Not a COC	NA	All fish receptors
Cadmium	Not a COC	Not a COC	NA	X	NA	Sculpin, juvenile Chinook salmon
Copper	Not a COC	X	NA	X	NA	Sculpin, juvenile Chinook salmon, northern pikeminnow, lamprey ammocoetes
Lead	Not a COC	X	NA	Not a COC	NA	Peamouth, smallmouth bass
Zinc	X	Not a COC	NA	Not a COC	NA	Sculpin, smallmouth bass, northern pikeminnow
PAHs			NA			
Benzo(a)pyrene	X	NA	NA	Not a COC	NA	Sculpin, smallmouth bass, northern pikeminnow
Benzo(a)anthracene	X	NA	NA	NA[f]	NA	Sculpin, smallmouth bass, northern pikeminnow
Naphthalene	X	NA	NA	NA[f]	NA	All fish receptors

Table 7-39. Summary of Fish COCs For Each LOE

COC	Line of Evidence Resulting in COC[a]					Receptor(s)[b]
	Surface Water	Tissue – Empirical	Tissue – Predicted	Diet	Fish Condition	
Total PAHs	NA[g]	NA	NA	Not a COC	Inconclusive	Benthic fish

[a] COCs were identified as those COPCs for which HQ were > 1.0 based on a relevant exposure scale.
[b] These are the receptors for which a COC was identified.
[c] Butyltin was not evaluated using the diet LOE because no toxicity threshold was available.
[d] BEHP could not be evaluated because no acceptable LOAEL was available. A comparison of tissue chemical concentration is presented in Section 7.1.4.
[e] For the surface water LOE, total DDx is a COC based on the 4,4′-DDT criterion. There is high uncertainty with this criterion as it is protective of brown pelican via ingestion of contaminated prey. The alternative water TRV that is considered more appropriate for evaluating direct exposure of organisms to water is exceeded in one sample with N-qualified DDT concentrations.
[f] Individual PAHs other than benzo(a)pyrene were evaluated as total PAHs for the dietary LOE.
[g] Individual PAHs (not total PAHs) were evaluated for the surface water LOE.

BEHP – bis(2-ethylhexyl) phthalate
COC – chemical of concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
HQ – hazard quotient
NA – not applicable (COPC not evaluated using this LOE)
LOAEL – 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)

7.6.2 WOE Evaluation

For fish, various LOEs were used to evaluate risks (Table 7-39). For organic, non-regulated metal, and regulated metal COCs, up to three LOEs were used to derive HQs (i.e., surface water LOE, tissue LOE, or dietary LOE). Only one LOE (surface water) was used to evaluate VOCs. For PAHs, two LOEs (i.e., dietary LOE and surface water LOE) were evaluated. Fish condition and direct contact with PAHs in sediment were evaluated as a separate semi-quantitative LOE; however, this LOE was inconclusive (Section 7.4.4).

Per the Problem Formulation (Attachment 2), a WOE approach is needed to integrate the results of each LOE. Per EPA's Problem Formulation (Attachment 2), the ultimate goal of a WOE approach 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." A qualitative WOE was considered to arrive at risk conclusions for all fish COCs.

A WOE was necessary only if: 1) there were multiple LOEs evaluated for a given COC, and 2) the results of multiple LOEs did not agree. When only one LOE was used to evaluate a COC or when HQs based on multiple LOEs agreed, no WOE was necessary to arrive at risk conclusions.[103] When the results of multiple LOEs agreed, risk conclusions are based on the concordance of LOEs, taking into account the magnitude of HQs, spatial extent of HQs > 1.0, uncertainties of exposure and effects assumptions, and the likelihood of population-level 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 need for a WOE evaluation was determined on a COC-specific basis, depending on the results of multiple LOEs. The rationale for arriving at risk conclusions, including the need to evaluate multiple LOEs in a qualitative WOE approach, is discussed in the following section.

7.6.3 Evaluation of Fish COCs

Several factors affect interpretation of the quantitative risk analysis

- Results of multiple LOEs
- Magnitude of HQs
- Spatial extent of HQs > 1.0

[103] 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 population-level effects based on the TRV endpoint.

- Implications of TRV exceedances based on COC-specific toxicological endpoints
- Uncertainty of exposure and effects assumptions

Overall, it was concluded there are negligible risks to fish in the Study Area from COCs identified in the BERA using three primary LOEs (i.e., tissue-residue, dietary, and surface water LOEs). Risk conclusions for each fish COC were determined by evaluating the risk estimates and reliability of each LOE. A qualitative WOE was conducted on a COC-specific basis because uncertainties associated with effects and exposure assumptions for each LOE vary across all COCs. For some chemical groups (i.e., metals), one LOE was always considered less reliable than other LOEs. The tissue-residue LOE was not used to determine risk conclusions for regulated metals (all metals other than mercury) because tissue-residue TRVs for metals do not reliably predict toxicity (EPA 2007e).

HQs >1 occurred for total DDx and total PCBs based on the surface water and tissue-residue LOEs; however, no unacceptable risks to fish populations are expected from these COCs. No surface water samples had total PCB concentrations that exceeded the water TRV based on direct exposure, and the only sample with a total DDx concentration that exceeded water TRVs based on direct exposure was based on an N-qualified result. While tissue-residue exceedances of the total PCB 10th percentile LOAEL were calculated, the 10th percentile LOAEL was expected to overpredict risks to fish based on the inclusion of unacceptable toxicity studies in the derivation of the LOAEL TRV. Based on the tissue-residue LOE, sculpin was the only fish with tissue concentrations greater than the total DDx 10th percentile LOAEL, but because of the low magnitude and limited spatial extent of exceedances (only one sample exceeded with an HQ of 1.9), no unacceptable risks are expected at the population-level.

The conclusion that there are no unacceptable risks to fish from PCBs is also supported by the findings of Barnthouse et al. (2009), which state that considerations (other than the comparison of tissue residues to TRVs) are needed to assess risks to fish populations from PCBs, including physiological and genetic adaptation, biological compensation, and other ecological processes. The BERA for the Hudson River predicted risks to fish populations based on exceedence of tissue residue benchmarks. However, Barnthouse et al. (2009) analyzed fish population risks in the Hudson River based on a consistent 30-year time series of life stage class abundance and maternal PCB concentration data, and concluded that fish populations in the Hudson are not at risk from PCB exposure. This provides a well documented example of conservatism in using PCB TRV exceedances to predict fish population risks.

The surface water and dietary LOEs were used to determine risks to regulated metal COCs (i.e., aluminum, cadmium, copper, lead, and zinc) and all three LOEs (tissue-residue, dietary, and surface water LOEs) were used to determine risk conclusions for mercury and TBT. Risk estimates for lead based on the surface water and dietary LOEs indicated no unacceptable risk to fish receptors (HQs were < 1.0). Cadmium and copper were determined to pose no unacceptable risk to fish populations based on the surface water LOE because, although dietary HQs were > 1.0, HQs were of low magnitude. Furthermore, the

conservative dietary TRVs used to calculate the HQs were several orders of magnitude lower than other TRVs from similar toxicological studies and were similar to nutritional requirements (for copper). Zinc was also determined to pose no unacceptable risks to fish populations based on the dietary LOE and because < 1% of surface water sample was greater than the zinc AWQC (HQ = 1.1). Risk estimates for mercury for all LOEs indicated no unacceptable risk to fish except for northern pikeminnow using the tissue-residue LOE; however, the magnitude and spatial extent of TRV exceedances for northern pikeminnow did not indicate population-level risks. No unacceptable risks to any fish receptors from butyltin or TBT are expected based on the surface water or tissue LOE because of the low magnitude and frequency of samples with HQs > 1.0. There are uncertainties associated with the dietary TRV; however, dietary risks from exposure to TBT are driven by localized areas associated with TBT sources; and therefore, unacceptable risks to fish populations are not likely. Though aluminum HQs were > 1.0 based on the surface water LOE, aluminum concentrations in background sediment and surface water indicate that aluminum is not attributed to anthropogenic sources in the Study Area and is not expected to be biologically available or toxic to fish at naturally occurring concentrations. It should also be noted that concentrations of copper and mercury in fish tissue from upriver areas are similar to or greater than fish tissue concentrations in the Study Area.

No unacceptable risks to fish populations from VOCs or BEHP are expected based on the limited spatial extent of COC surface water exceedances. The surface water LOE was the only LOE evaluated for VOCs. Similarly, no unacceptable risks to fish populations from PAHs are expected because dietary risk estimates indicated no risks (HQs were < 1.0) and < 5% of surface water samples had concentrations greater than their respective surface water TRVs.

Fish COCs, HQs, uncertainties associated with exposure and effects, and risk conclusions are summarized in Table 7-40. Fish COCs were combined with COCs for all other ecological receptor groups to determine the overall ecological risk conclusions (Section 11.0).

Table 7-40. Summary of Fish COCs and Risk Conclusions Across LOEs Summary of Fish COCs and Risk Conclusions Across LOEs

Receptor by COC	LOE	HQ[a]	Exposure Considerations	Effects Considerations	Risk Conclusions
Aluminum					
Sculpin	Surface water	1.0 – 21 (site-wide)	For sculpin fish, HQs were > 1.0 over all seasonal events and in 93% of samples; HQs based on UCLs that were greater than maximum concentrations measured in some 1-mile exposure areas; aluminum concentrations were similar to or less than sediment and surface water background chemical concentrations.	The water TRV is based on AWQC, which is protective of fish.	Aluminum concentrations in background areas indicate that aluminum is not attributed to anthropogenic sources in the study area and is not expected to be toxic to fish at naturally occurring concentrations; therefore, negligible risks to fish from aluminum are expected.
Smallmouth bass and northern pikeminnow		1.0 – 30 (all exposure areas)			
Large-home-range fish receptors		5.3 (site-wide)			
All fish	Dietary dose	< 1.0	Not identified as a COPC per this LOE in the SLERA (Attachment 5).		
Cadmium					
All fish	Surface water	< 1.0	Not identified as a COPC per this LOE in the SLERA (Attachment 5).		See cadmium risk conclusions for individual fish receptors (below).
All fish	Tissue residue	< 1.0	Not identified as a COPC per this LOE in the SLERA (Attachment 5).		See cadmium risk conclusions for individual fish receptors (below).
All fish except sculpin and juvenile Chinook salmon	Dietary dose	< 1.0	Modeled cadmium concentrations in the diet based on dietary assumptions derived from the literature were less than TRVs in all exposure areas for these fish receptors.	The selected LOAEL (0.01 mg/kg bw/day) may overpredict risk because it is 2 to 3 orders of magnitude lower than the nine NOAELs (2.5 to 29 mg/kg bw/day) identified in the other studies reviewed and 2 to 3 orders of magnitude lower than the four LOAELs (4.6 to 68 mg/kg bw/day) reported in the other studies reviewed (Attachment 14).	No unacceptable risks to these fish receptors are expected based on the dietary-dose and surface water LOEs. The risk conclusions from the tissue-residue LOE are uncertain because fish regulate metals and the tissue-residue TRV for cadmium cannot reliably predict toxicity.
Sculpin	Dietary dose	1.1 – 2.2[b] (RM 2.0, east; RM 4.2, east; Slip 1; Swan Island Lagoon; International Slip)	Exceedances were limited to five exposure areas based on sample-by-sample evaluation of sculpin prey, which is a conservative estimate of exposure.	The selected LOAEL (0.01 mg/kg bw/day) may overpredict risk because it is 2 to 3 orders of magnitude lower than the nine NOAELs (2.5 to 29 mg/kg bw/day) identified in the other studies reviewed and 2 to 3 orders of magnitude lower than the four LOAELs (4.6 to 68 mg/kg bw/day) reported in the other studies reviewed (Attachment 14).	Although cadmium was identified as a COC under the assumptions required for the BERA based on sculpin diet, it is more reasonable to conclude that cadmium does not pose unacceptable risk to sculpin populations based on the low magnitude of exceedance and the uncertainty associated with a conservative LOAEL, and because the surface water LOE does not identify cadmium as a COC.
Juvenile Chinook salmon	Dietary dose	1.7 – 3.5 (site-wide)	An HQ of 3.5 resulted from the assumption that the juvenile Chinook salmon diet is composed solely of benthic invertebrates, which is not representative of the juvenile Chinook salmon pelagic diet. The HQ of 1.7 resulted from the assumption that the diet is composed solely of epibenthic invertebrate tissue.	The selected NOAEL (0.002 mg/kg bw/day) was extrapolated from the selected LOAEL using a UF of 5. The selected LOAEL is 2 to 3 orders of magnitude lower than the nine NOAELs (2.5 to 29 mg/kg bw/day) identified in the other studies reviewed (including four salmonid NOAELs) and 2 to 3 orders of magnitude lower than the four LOAELs (4.6 to 68 mg/kg bw/day) reported in the other studies reviewed (including two salmonid LOAELs) (Attachment 14).	Although cadmium was identified as a COC under the assumptions required for the BERA based on juvenile Chinook salmon diet, it is more reasonable to conclude that cadmium does not pose unacceptable risk to individual juvenile Chinook salmon based on the uncertainty in the dietary assumption and the uncertainty associated with a conservative NOAEL, and because the surface water LOE does not identify cadmium as a COC.

Table 7-40. Summary of Fish COCs and Risk Conclusions Across LOEs Summary of Fish COCs and Risk Conclusions Across LOEs

Receptor by COC	LOE	HQ[a]	Exposure Considerations	Effects Considerations	Risk Conclusions
Copper					
All fish	Surface water	< 1.0	Not identified as a COPC per this LOE in the SLERA (Attachment 5).		See copper risk conclusions for individual fish receptors (below).
All fish except Pacific lamprey ammocoetes and sculpin	Tissue residue	< 1.0	Modeled copper concentrations in the diet based on dietary assumptions derived from the literature were less than TRVs in all exposure areas for these fish receptors.	The selected TRVs are highly uncertain because fish actively regulate tissue metals concentrations, and effects thresholds in tissues for regulated metals cannot reliably predict toxicity.	The risk conclusions from the tissue-residue LOE are uncertain because fish regulate copper concentrations in tissue and the tissue-residue TRV for copper cannot reliably predict toxicity. See copper risk conclusions for individual fish receptors (below).
Juvenile Chinook salmon	Dietary dose	2.4 – 2.5 (site-wide)	An HQ of 2.5 resulted from the assumption that the juvenile Chinook salmon diet is composed solely of benthic invertebrates (clams and worms), which is not a representative of the juvenile Chinook salmon pelagic diet. An HQ of 2.4 resulted from the assumption that the diet is composed solely of epibenthic invertebrate tissue.	The selected NOAEL may overestimate risks to juvenile Chinook salmon. The NOAEL (0.24 mg/kg bw/day), which is based on Murai et al. (1981), could not be replicated in subsequent experiments (Erickson et al. 2003; Gatlin and Wilson 1986) and has been characterized as atypical by another study of copper in fish (Lorentzen et al. 1998). The selected NOAEL is also within the range of nutritional requirements found in the literature for relevant fish species. The next lowest literature-based NOAEL (1.0 mg/kg bw/day) based on Kang et al. (2005) results in all HQs < 1.0.	No unacceptable risks are expected to juvenile Chinook salmon from copper based on the surface water LOE. The risk conclusions from the dietary LOE are uncertain because HQs are based on a NOAEL that has not been replicated and is within the nutritional range requirement for fish, and therefore dietary risks to juvenile Chinook salmon are assumed to be overpredicted.
Sculpin	Tissue residue	1.1 – 2.3[b] (RM 5.5, east; RM 10.3, west; RM 11.3, east)	Fish tissue copper concentrations are similar in the upriver reach and study area for those species for which upriver reach data are available (Section 7.1.4.3). No sculpin tissue was available from the upriver reach. Identification of COC and the range of HQs are based on individual samples.	The selected LOAEL is highly uncertain because fish actively regulate tissue metals concentrations, and effects thresholds in tissues for regulated metals cannot reliably predict toxicity.	No unacceptable risks are expected to sculpin from copper based on the surface water LOE. The risk conclusions from the tissue-residue LOE are uncertain because fish regulate copper concentrations in tissue and the tissue-residue TRV for copper cannot reliably predict toxicity. The risk conclusions from the dietary LOE are also uncertain because the HQs are based on a LOAEL that has not been replicated and is just above the nutritional range requirement for fish, and therefore dietary risks to sculpin are assumed to be over-predicted. It should be noted that upriver fish tissue copper concentrations are similar to those of study area fish.
	Dietary dose	1.1 – 3.6 (site-wide)	Site-wide exceedances are based on a sample-by-sample evaluation of sculpin prey.	The selected LOAEL may overestimate risks to sculpin. The LOAEL (0.48 mg/kg bw/day) , which is based on Murai et al. (1981), could not be replicated in subsequent experiments (Erickson et al. 2003; Gatlin and Wilson 1986) and has been characterized as atypical by another study of copper in fish (Lorentzen et al. 1998). The selected LOAEL is also just above the range of nutritional requirements found in the literature for relevant fish species. The next lowest literature-based LOAEL (2.0 mg/kg bw/day) based on Kang et al. (2005) results in HQs < 1.0 for all individual prey.	
Northern pikeminnow	Dietary dose	1.3 – 1.5 (Swan Island Lagoon; RM 10.5 to RM 11.8)	The spatial extent of exceedances is limited to two exposure areas (RM 10.5 to RM 11.8 and Swan Island Lagoon).	The selected LOAEL may overestimate risks to northern pikeminnow. The LOAEL (0.48 mg/kg bw/day), which is based on Murai et al. (1981), could not be replicated in subsequent experiments (Erickson et al. 2003; Gatlin and Wilson 1986) and has been characterized as atypical by another study of copper in fish (Lorentzen et al. 1998). The selected LOAEL is also just above the range of nutritional requirements found in the literature for relevant fish species. The next lowest literature-based LOAEL (2.0 mg/kg bw/day) based on Kang et al. (2005) results in HQs < 1.0.	Although copper is identified as a COC under the assumptions required for the BERA based on pikeminnow diet, it is more reasonable to conclude that copper does not pose unacceptable risk to northern pikeminnow populations based on the limited spatial extent of exceedances and the uncertainty associated with the LOAEL, and because the surface water LOE does not identify copper as a COC.

Table 7-40. Summary of Fish COCs and Risk Conclusions Across LOEs Summary of Fish COCs and Risk Conclusions Across LOEs

Receptor by COC	LOE	HQ[a]	Exposure Considerations	Effects Considerations	Risk Conclusions
Pacific lamprey Ammocoetes	Tissue residue	2.2 (site-wide)	Lamprey tissue copper concentrations are similar in the upriver reach and study area (Section 7.1.4.3). The site-wide EPC is based on the maximum copper concentration in lamprey (insufficient data were available to calculate an EPC based on a side-wide UCL).	The selected NOAEL is highly uncertain because fish actively regulate tissue metals concentrations, and effects thresholds in tissues for regulated metals cannot reliably predict toxicity.	No unacceptable risks to lamprey are expected from copper based on the surface water LOE. The risk conclusions from the tissue-residue LOE are uncertain because fish regulate metals and the tissue-residue TRV for copper cannot reliably predict toxicity. It should be noted that upriver lamprey tissue copper concentrations are similar to those of study area lamprey.
Lead					
All fish	Dietary dose	< 1.0	Not identified as a COPC per this LOE in the SLERA (Attachment 5).		See lead risk conclusions for individual fish receptors (below).
All fish	Surface water	< 1.0	Not identified as a COPC per this LOE in the SLERA (Attachment 5).		See lead risk conclusions for individual fish receptors (below).
All fish except peamouth and smallmouth bass	Tissue residue	< 1.0	Not identified as a COPC per this LOE in the SLERA (Attachment 5).		No unacceptable risks to these fish receptors are expected based on dietary-dose and surface water LOEs. The risk conclusions from the tissue-residue LOE are uncertain because a limited number of toxicity studies were available.
Peamouth	Tissue residue	2.7 (site-wide)	The site-wide EPC is based on the maximum lead concentration in peamouth (insufficient data were available to calculate an EPC based on a side-wide UCL).	The selected LOAEL is highly uncertain because fish actively regulate tissue metals concentrations, and effects thresholds in tissues for regulated metals cannot reliably predict toxicity.	No unacceptable risks to peamouth are expected from lead based on the surface water and dietary LOEs. The risk conclusions from the tissue-residue LOE are uncertain because a limited number of toxicity studies were available.
Smallmouth Bass	Tissue residue	280 (RM 9.5 to RM 10.5)	The spatial extent of potential risk is limited to RM 9.5 to RM 10.5, and in this exposure area, the exposure estimate is primarily driven by a single outlier, indicating that most study area fish are not exposed to lead levels that would pose a risk. The maximum smallmouth bass concentration (1,100 mg/kg ww) detected in this exposure area is over 100 times greater than 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 all other smallmouth bass samples (0.0048 to 1.8 mg/kg ww). The HQ for the other sample collected from RM 9.5 to RM 10.5 is 1.7. The HQ is based on maximum individual sample.	The selected LOAEL is highly uncertain because fish actively regulate tissue metals concentrations, and effects thresholds in tissues for regulated metals cannot reliably predict toxicity.	No unacceptable risks to smallmouth bass are expected from lead based on the surface water and dietary LOEs. The risk conclusions from the tissue-residue LOE are uncertain because fish regulate metals and the tissue-residue TRV for lead cannot reliably predict toxicity. Even if lead does pose unacceptable risk, the limited spatial extent indicates that the risk would not rise to the population-level. Because the tissue chemical concentration in the single outlier is so much higher than all other sample concentrations, it is plausible to speculate that the source of the lead is discrete (e.g., possibly resulting from a lead sinker) rather than dispersed in sediment or water.
Mercury					
All fish	Dietary dose	< 1.0	Modeled mercury concentrations in the diet based on dietary assumptions derived from the literature were less than TRVs in all exposure areas for all fish receptors.	TRVs based on lowest LOAELs from acceptable dietary studies from the literature.	See mercury risk conclusions for individual fish receptors (below).

Table 7-40. Summary of Fish COCs and Risk Conclusions Across LOEs Summary of Fish COCs and Risk Conclusions Across LOEs

Receptor by COC	LOE	HQ[a]	Exposure Considerations	Effects Considerations	Risk Conclusions
All fish	Surface water	< 1.0	Not identified as a COPC per this LOE in the SLERA (Attachment 5).		See mercury risk conclusions for individual fish receptors (below).
All fish except northern pikeminnow	Tissue residue	< 1.0	Not identified as a COPC per this LOE in the SLERA (Attachment 5).		No unacceptable risks to these fish receptors are expected based on dietary-dose, surface water, and tissue residue LOEs.
Northern Pikeminnow	Tissue residue	1.1 (RM 6.5 to RM 7.5)	Fish tissue mercury concentrations are higher in the upriver reach than in the study area for species for which upriver reach data are available (Section 7.1.4.3). No northern pikeminnow tissue was available from the upriver reach. Willamette-wide northern pikeminnow average mercury concentrations (as reported in Hope (2003)) are approximately 2 times as great as average concentrations in northern pikeminnow from the Study Area. Identification of COC and the HQ is based on maximum individual sample.	The LOAEL is the 10th percentile of the fish SSD (Attachment 9).	Although mercury was identified as a COC under the assumptions required for the BERA based on tissue concentrations in northern pikeminnow, it is more reasonable to conclude that mercury does not pose unacceptable risk to northern pikeminnow populations based on the low magnitude of exceedance and limited spatial extent of exceedances, and because the surface water and dietary-dose LOEs do not identify mercury as a COC. It should be noted that upriver fish tissue mercury concentrations are similar to those of study area fish and that average Willamette River-wide northern pikeminnow tissue mercury concentrations are greater than average Study Area northern pikeminnow tissue concentrations.
Zinc					
All fish	Dietary dose	< 1.0	Not identified as a COPC per this LOE in the SLERA (Attachment 5).		No unacceptable risks to small-home-range fish are expected based on the dietary LOE. The tissue LOE is to uncertain because fish regulate metals and the tissue-residue TRV for zinc cannot reliably predict toxicity. Although zinc is identified as a COC under the assumptions required for the BERA in surface water, it is more reasonable to concluded that zinc does not pose unacceptable risk to small-home-range fish (i.e., sculpin, smallmouth bass, and northern pikeminnow) populations based on the low magnitude of exceedance and limited spatial extent of exceedances, and because the dietary LOE does not identify zinc as a COC.
All fish	Tissue residue	< 1.0	Exposure concentrations were less than TRVs in all exposure areas for all fish receptors.	TRVs based on only one of two acceptable toxicity studies that were identified. The selected TRVs are highly uncertain because fish actively regulate tissue metals concentrations, and effects thresholds in tissues for regulated metals cannot reliably predict toxicity.	
Sculpin	Surface water	1.1 (RM 9.7, west)	The spatial extent of potential risk is limited to one sample located at RM 9.7 collected during the November 2004 low-flow sampling event.	The water TRV is based on AWQC, which is protective of fish.	
Smallmouth bass and northern pikeminnow		1.1 (RM 9.5 to 10.5)		The water TRV is based on AWQC, which is protective of fish.	
Large-home-range fish		< 1.0	Exposure concentrations were less than TRVs in all exposure areas for all fish receptors.	The water TRV is based on AWQC, which is protective of fish.	No unacceptable risks to these fish receptors are expected based on dietary-dose and surface water, LOEs. The tissue LOE is to uncertain because fish regulate metals and the tissue-residue TRV for zinc cannot reliably predict toxicity.

Table 7-40. Summary of Fish COCs and Risk Conclusions Across LOEs Summary of Fish COCs and Risk Conclusions Across LOEs

Receptor by COC	LOE	HQ[a]	Exposure Considerations	Effects Considerations	Risk Conclusions
Butyltin					
Sculpin	Surface water	1.2 (RM 11, west)	The spatial extent of potential risk is limited to one sample located at RM 11 collected during the winter 2007 high-flow sampling event.	The selected TRV is based on an AWQC value for TBT and is uncertain for estimating risks from butyltin.	Although butyltin was identified as a COC in surface water under the assumptions required for the BERA, it is more reasonable to conclude that butyltin does not pose unacceptable risk to small-home-range fish based on the low magnitude and frequency of surface water samples with HQs > 1.0, and uncertainty associated with use of TBT effects data as a surrogate for butyltin.
Smallmouth bass and northern pikeminnow		1.2 (RM 10.5 to 11.5)		The selected TRV is based on an AWQC value for TBT and is uncertain for estimating risks from butyltin.	
TBT					
All fish	Tissue residue	< 1.0	Not identified as a COPC in the SLERA (Attachment 5).		See TBT risk conclusions for individual fish receptors (below).
	Surface water	< 1.0	Not identified as a COPC in the SLERA (Attachment 5).		See TBT risk conclusions for individual fish receptors (below).
Largescale sucker	Dietary dose	3.5 – 9.3 (site-wide)	The HQ of 3.5 resulted from the assumption that the sucker diet is composed solely of clams; the HQ of 9.3 resulted from the assumption that the sucker diet is composed solely of lab worms.	The selected LOAEL is uncertain because it is based on the only toxicological study available (Shimasaki et al. 2003), and the feeding rate used to derive the LOAEL dose was not reported in the study; instead, the dietary dose was calculated using the average feeding rate used in other dietary toxicity studies. The results of Shimasaki et al. (2003) were based on pseudoreplication in that all fish in a given treatment level were exposed in a single tank, and all treatment levels (including controls) had high rates of mortality. Because of the uncertainties associated with the dietary TRV, the dietary HQs are not conclusive.	No unacceptable risks to any fish receptors from TBT are expected based on the surface water or tissue LOE. There are uncertainties associated with the dietary TRV; however, dietary risks from exposure to TBT are driven by localized areas and therefore, unacceptable risks to fish populations are not likely.
Juvenile white sturgeon	Dietary dose	5.1 – 9.3 (2.5 – 6.7)[c] (site-wide)	The HQ resulted from the assumption that the juvenile white sturgeon home range is limited to the study area; however, it is more plausible that their home range is greater than the study area. HQs of 2.5 and 5.1 resulted from the assumption that the sturgeon diet is composed solely of clams, and HQs of 6.7 and 9.3 resulted from the assumption that the sturgeon diet is composed solely of lab worms.	The dietary effect considerations for juvenile white sturgeon mirror those for largescale sucker.	The risk conclusions for juvenile white sturgeon are the same as those for largescale sucker.

Table 7-40. Summary of Fish COCs and Risk Conclusions Across LOEs Summary of Fish COCs and Risk Conclusions Across LOEs

Receptor by COC	LOE	HQ[a]	Exposure Considerations	Effects Considerations	Risk Conclusions
Juvenile Chinook salmon	Dietary dose	56 (site-wide)	The HQ resulted from the assumption that the juvenile Chinook salmon diet is composed of solely of benthic invertebrates (clams and worms), which is not representative of the juvenile Chinook salmon pelagic diet.	The selected NOAEL was extrapolated from the LOAEL using a UF of 5. The selected LOAEL is uncertain because it is based on the only toxicological study available (Shimasaki et al. 2003), and the feeding rate used to derive the LOAEL dose was not reported in the study; instead, the dietary dose was calculated using the average feeding rate used in other dietary toxicity studies. The results of Shimasaki et al. (2003) were based on pseudoreplication in that all fish in a given treatment level were exposed in a single tank, and all treatment levels (including controls) had high rates of mortality. Because of the uncertainties associated with the dietary TRV, the dietary HQs are not conclusive.	The risk conclusions for juvenile Chinook salmon are the same as those for largescale sucker.
Peamouth	Dietary dose	8.0 (site-wide)	The HQ resulted from the assumption that the peamouth diet is composed of 90% benthic invertebrates (clams and worms), which is not representative of the peamouth benthopelagic diet.	The dietary effect considerations for peamouth mirror those for largescale sucker.	The risk conclusions for peamouth are the same as those for largescale sucker.
Sculpin	Dietary dose	1.3 – 69[b] (Mouth of Swan Island Lagoon, International Slip)	Site-wide exceedances are based on a sample-by-sample evaluation of sculpin prey, which is a conservative estimate of exposure.	The dietary effect considerations for sculpin mirror those for largescale sucker.	The risk conclusions for sculpin are the same as those for largescale sucker.
Smallmouth bass	Dietary dose	6.1 (Swan Island Lagoon)	The spatial extent of potential risk is limited to the Swan Island Lagoon exposure area. The HQ assumes that 10% the smallmouth bass diet is composed of lab worms because TBT data for crayfish were not available from Swan Island Lagoon; however, lab worms are the only prey with elevated TBT concentrations and are not representative prey for smallmouth bass.	The dietary effect considerations for smallmouth bass mirror those for largescale sucker.	The risk conclusions for smallmouth bass are the same as those for largescale sucker.
Northern pikeminnow	Dietary dose	25 (Swan Island Lagoon)	The spatial extent of potential risk is limited to the Swan Island Lagoon exposure area. The HQ assumes that 55% the northern pikeminnow diet is composed of lab worms because TBT data for crayfish were not available from Swan Island Lagoon; however, lab worms are the only prey with elevated TBT concentrations and are not representative prey for northern pikeminnow.	The dietary effect considerations for northern pikeminnow mirror those for largescale sucker.	The risk conclusions for northern pikeminnow are the same as those for largescale sucker.
Benzo(a)anthracene					
Sculpin	Surface water	4.1 – 10 (RM 6.1, west; RM 6.3, west)	The spatial extent of potential risk is limited to two samples located at RM 6.1 and RM 6.3 collected during the July 2005 low-flow and winter 2007 high-flow sampling events.	The water TRV is based on a Tier II value that was extrapolated from a *Daphnia* acute LC50. Water TRV may overestimate or underestimate risks to fish.	No unacceptable risks to fish from PAHs are expected using the dietary LOE based on the benzo(a)pyrene and PAH mixture dietary TRVs associated with growth effects (Attachment 14). Although benzo(a)anthracene was identified as a COC under the assumptions required for the BERA based on surface water, it is more plausible that benzo(a)anthracene does not pose unacceptable risk to small-home-range fish (i.e., sculpin, smallmouth bass, and northern pikeminnow) populations based on the limited spatial extent of exceedances.
Smallmouth bass and northern pikeminnow		4.7 (RM 5.5 to RM 6.5)			

Table 7-40. Summary of Fish COCs and Risk Conclusions Across LOEs Summary of Fish COCs and Risk Conclusions Across LOEs

Receptor by COC	LOE	HQ[a]	Exposure Considerations	Effects Considerations	Risk Conclusions
Benzo(a)pyrene					
Sculpin	Surface water	1.4 – 14 (RM 6.1, west; RM 6.3, west)	The spatial extent of potential risk is limited to three samples located at RM 6.1 and RM 6.3 collected during the November 2004 low-flow, July 2005 low-flow, and winter 2007 high-flow sampling events.	The water TRV is based on a Tier II value that was extrapolated from a *Daphnia* acute LC50. Water TRV may overestimate or underestimate risks to fish.	No unacceptable risks to fish from PAHs area expected using the dietary LOE based on the benzo[a]pyrene and PAH mixture dietary TRVs associated with growth effects (Attachment 14). Although benzo(a)pyrene was identified as a COC under the assumptions required for the BERA based on surface water, it is more reasonable to conclude that benzo(a)pyrene does not pose a unacceptable risk to small-home-range fish (i.e., sculpin, smallmouth bass, and northern pikeminnow) populations based on the limited spatial extent of exceedances.
Smallmouth bass and northern pikeminnow		7.6 (RM 5.5 to RM 6.5)			
All fish	Dietary dose	< 1.0	Exposure concentrations were less than TRVs in all exposure areas for all fish receptors.	TRVs based on limited toxicological data.	
Naphthalene					
Sculpin	Surface water	14(RM 6.4 to RM 6.5, west)	The spatial extent of potential risk is limited, HQs > 1.0 based on a UCL of samples over a 0.15-mile stretch from RM 6.4 to RM 6.5 during the May 2005 non-LWG sampling event; HQs for all other samples were < 1.0.	The water TRV is based on a Tier II value that is protective of fish.	No unacceptable risks to fish are expected from PAHs via the dietary exposure route based on the benzo[a]pyrene and PAH mixture dietary TRVs associated with growth effects (Attachment 14). Although naphthalene was identified as a COC under the assumptions required for the BERA based on surface water, it is more reasonable to conclude that naphthalene does not pose unacceptable risk to fish populations based on the low frequency and limited spatial extent of exceedances (samples with HQs > 1.0 were limited to a single localized area (from RM 6.4 to RM 6.5). No samples exceeded the only chronic value for fish (619 µg/L) reported in Suter and Tsao (1996).
Smallmouth bass and northern pikeminnow		2.8 (RM 5.5 to RM 6.5)			
BEHP					
Sculpin	Surface water	1.2 and 2.3 (RM 3.9 [transect sample] and RM 6.7 [Willamette Cove])	The spatial extent of potential risk is limited to two samples located at RM 3.9 and RM 6.7 collected during the November 2006 storm runoff and winter 2007 high-flow sampling events.	The water TRV is based on a Tier II value that is protective of fish.	BEHP concentrations in one sculpin tissue sample and two smallmouth bass samples were greater than the NOAEL (no LOAEL was available); however, an HQ > 1.0 based on an unbounded NOAEL does not conclusively indicate unacceptable risks. Although BEHP was identified as a COC under the assumptions required for the BERA based on surface water, it is more reasonable to conclude that BEHP does not pose unacceptable risk to small-home-range fish (i.e., sculpin, smallmouth bass, and northern pikeminnow) populations based on the low magnitude of exceedance and limited spatial extent of exceedances.
Smallmouth bass and northern pikeminnow		2.3 (RM 6.5 to RM 7.5)	HQs based on maximum concentration from exposure area (peristaltic sample); TRV is based on risk to fish and invertebrates.		

Table 7-40. Summary of Fish COCs and Risk Conclusions Across LOEs Summary of Fish COCs and Risk Conclusions Across LOEs

Receptor by COC	LOE	HQ[a]	Exposure Considerations	Effects Considerations	Risk Conclusions
Total PCBs					
Largescale sucker	Tissue residue	1.6 (site-wide)	The HQ was derived using an EPC based on a site-wide UCL.	There is high uncertainty associated with the selected LOAEL for total PCBs (0.93 mg/kg ww); this LOAEL is expected to over-predict risks to largescale sucker (see effect considerations for sculpin and smallmouth bass).	No unacceptable risks to largescale sucker from total PCBs are expected via the surface water exposure route. Although total PCBs were identified as a COC under the assumptions required for the BERA based on tissue residues, it is more reasonable to conclude that total PCBs do not pose unacceptable risk to largescale sucker populations based on the low magnitude of exceedances of a highly uncertain LOAEL, which is expected to overestimate risks to fish.
Sculpin	Tissue residue	2.5 – 9.4 (1.0 – 111[d]) (RM 2.3 to RM 2.4, east; Willamette Cove; RM 11.3, east; additional predicted locations[e])	The spatial extent of potential risk based on predicted tissue concentrations is more wide-spread than the HQs based on empirical tissue concentrations. The exposure concentrations associated with the predicted HQs < 1.0 were derived using mechanistic model. In general, the areas where predicted HQs were > 1.0 were near or at the same locations as detected sculpin tissue samples with total PCB HQs < 1.0. Empirical data are assumed to be more representative of tissue concentrations in Study Area sculpin than are predicted tissue chemical concentrations. The range of HQs is based on individual empirical samples.	There is high uncertainty associated with the selected LOAEL for total PCBs (0.93 mg/kg ww); this LOAEL is expected to over-predict risks to fish. The selected LOAEL is based on the 10th percentile of the SSD, which 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. Four of these five LOAELs (ranging from 0.52 to 3.6 mg/kg ww) were less than the highest detected PCB concentration in sculpin (8.77 mg/kg ww). One of these five LOAELs (9.2 mg/kg ww) was less than the highest predicted sculpin concentration (12 mg/kg ww).Two additional LOAELs (derived from Hansen et al. (1971)) that were included in the data used to derive the 10th percentile LOAEL were below study area sculpin or smallmouth bass maximum tissue chemical concentrations; however, these LOAELs were extrapolated from short-term (defined as < 30 days) LOAELs using an acute- to-chronic ratio, and specific adverse effects cannot be determined using these extrapolated LOAELs. Furthermore, subsequent studies report higher chronic LOAELs for spot, one of the fish species associated with these extrapolated LOAELs.	Although total PCBs were identified as a COC under the assumptions required for the BERA based on tissue residues, it is more reasonable to conclude that total PCBs do not pose unacceptable risk to sculpin populations based on the high uncertainty associated with the tissue residue LOAEL, which is expected to overestimate risks to fish, and based on the fact that no surface water samples exceeded the alternative water TRV that is based on direct exposure to surface water.

Table 7-40. Summary of Fish COCs and Risk Conclusions Across LOEs Summary of Fish COCs and Risk Conclusions Across LOEs

Receptor by COC	LOE	HQ[a]	Exposure Considerations	Effects Considerations	Risk Conclusions
Smallmouth Bass	Tissue residue	1.5 – 7.1 (RM 1.5 to 2.5; RM 3.5 to RM 4.5; RM 6.5 to RM 7.5; Swan Island Lagoon; RM 10.5 to RM 11.5)	The spatial extent of potential risk is limited to five exposure areas. The range of HQs is based on individual samples.	There is high uncertainty associated with the selected LOAEL for total PCBs (0.93 mg/kg ww), and this LOAEL is expected to over-predict risks to fish. The selected LOAEL is based on the 10th percentile of the SSD, which 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. Four of these five LOAELs (ranging from 0.52 to 3.6 mg/kg ww) were less than the highest detected PCB concentrations in smallmouth bass (6.6 mg/kg ww). Two additional LOAELs (derived from Hansen et al. (1971)) that were included in the data used to derive the 10th percentile LOAEL were below study area sculpin or smallmouth bass maximum tissue chemical concentrations; however, these LOAELs were extrapolated from short-term (defined as < 30 days) LOAELs using an acute-to-chronic ratio, and specific adverse effects cannot be determined using these extrapolated LOAELs. Furthermore, subsequent studies report higher chronic LOAELs for spot, one of the fish species associated with these extrapolated LOAELs.	No unacceptable risks to smallmouth bass from total PCBs are expected via the surface water exposure route. Although total PCBs were identified as a COC under the assumptions required for the BERA based on tissue residues, it is more reasonable to conclude that total PCBs do not pose unacceptable risk to smallmouth bass populations based on the high uncertainty associated with the tissue residue LOAEL, which is expected to overestimate risks to fish.
Northern Pikeminnow	Tissue residue	1.1 – 2.0 (RM 6.5 to RM 7.5; RM 7.5 to RM 8.5; RM 8.5 to RM 9.5)	The spatial extent of potential risk is limited to three (of eleven) exposure areas (from RM 6.5 to RM 9.5). The range of HQs is based on individual samples.	There is high uncertainty associated with the selected LOAEL for total PCBs (0.93 mg/kg ww), and this LOAEL is expected to over-predict risks to northern pikeminnow (see effect considerations for sculpin and smallmouth bass).	No unacceptable risks to northern pikeminnow from total PCBs are expected via the surface water exposure route. Although total PCBs were identified as a COC under the assumptions required for the BERA based on tissue residues, it is more reasonable to conclude that total PCBs do not pose unacceptable risk to northern pikeminnow populations based on the high uncertainty associated with the tissue residue LOAEL, which is expected to overestimate risks to fish.
Sculpin	Surface water	1.1 – 1.2 (RM 3.7 [International Slip] and RM 6.9 [Willamette Cove])	The spatial extent of potential risk is limited to two samples located at RM 3.7 and RM 6.9 collected during the November 2004 and March 2005 low-flow sampling events.	The AWQC (0.014 µg/L) was derived on the basis that it was protective of mink. No surface water samples exceed the alternative water TRV (0.19 µg/L) that is based on direct exposure to surface water.	See the total PCB risk conclusions for individual fish receptors above.
Total DDx					
Sculpin	Tissue residue	1.9 (1.0 - 21[d]) (RM 7.1 to 7.4, west)	Predicted and empirical HQ > 1.0 are co-located; predicted HQs > 1.0 along the west bank from approximately RM 7.1 to 7.4 and the empirical HQ is > 1.0 at a single composite location at RM 7.3 on the west bank. Predicted HQs are based on mechanistic model. Identification of COC and the range of HQs are based on individual empirical samples.	The LOAEL is the 10th percentile of the fish SSD (Attachment 9).	Because of the low magnitude and frequency of sculpin tissue HQs > 1.0 in a single localized area (approximately from RM 7.1 to RM 7.4), no unacceptable risks are expected to sculpin; no unacceptable risks are expected to all other fish receptors based on the tissue LOE. Although total DDx were identified as a COC under the assumptions required for the BERA based on the surface water LOE, it is more reasonable to conclude that total DDx do not pose unacceptable risk to all other fish receptors (other than sculpin) because of the indication of sample interference in the only sample that exceeded a threshold intended to be protective of organisms directly exposed to surface water (the alternative TRV).
Sculpin	Surface water	1.2 – 20 (site-wide)	Total DDx HQs were > 1.0 in 21% of all surface water samples, and individual sample HQs ranged from 1.2 to 20. Samples with HQs > 1.0 included samples with DDT concentrations that were N-qualified by the lab. One sample exceeds alternative water TRV that is based on direct exposure to surface water;	The water TRV is highly uncertain for determining risks to fish. The TRV for DDTs was based on the AWQC for 4,4′-DDT. The AWQC (0.001 µg/L) was derived on the basis that it was protective of individual reproductive productivity in brown pelicans). Using the alternative water TRV (0.011 µg/L) that is	

Table 7-40. Summary of Fish COCs and Risk Conclusions Across LOEs Summary of Fish COCs and Risk Conclusions Across LOEs

Receptor by COC	LOE	HQ[a]	Exposure Considerations	Effects Considerations	Risk Conclusions
Smallmouth bass and northern pikeminnow		1.2 – 7.6 (RM 1.5 to RM 2.5; RM 3.5 to RM 4.5; RM 5.5 to RM 6.5; RM 6.5 to RM 7.5; RM 7.5 to RM 8.5; RM 9.5 to RM 10.5)	however, this sample (W001 at RM 2.0) is N-qualified. Data with an N-qualifier indicates "the presence of an analyte that has been 'tentatively identified,' and the associated numerical value represents its approximate concentration" (EPA 1999). The qualifier indicates that the analyst believed that the result was due to analytical interference from a chemical other than the target analyte. Therefore, HQs based on N-qualified data are not conclusive. Surface water samples with HQs > 1.0 based on non-N-qualified data had HQs that ranged from 1.2 to 9.8.	protective of direct exposure to surface water, only one sample has a total DDx HQ > 1.0; however, this sample (W001 at RM 2.0) is N-qualified (see exposure considerations). The lowest chronic effect value for fish in the AWQC document is 0.74 µg/L, indicating that the alternative TRV is protective of fish.	
Large-home-range fish receptors		1.6 (site-wide)			
Trichloroethene					
Sculpin	Surface water	4.1 (RM 6.4 to RM 6.5, west)	The spatial extent of potential risk is limited to one sample located at RM 6.4 on the west bank collected in during the May 2005 non-LWG sampling event. Surface water analyzed for VOCs were only available from one area (at an approximate 0.15-mile stretch between RM 6.4 and RM 6.5 on the west bank of the study area). The EPC for sculpin was based on the maximum concentration because there were insufficient data to derive an EPC based on a UCL.	The water TRV is based on a Tier II value that is protective of fish.	Although trichloroethene was identified as a COC under the assumptions required for the BERA based on surface water, it is more reasonable to conclude that trichloroethene does not pose unacceptable risk to sculpin populations based on the limited spatial extent of exceedances.

[a] HQ based only on exposure areas where HQs > 1.0. HQs in all other exposure areas < 1.0.
[b] HQ based individual prey samples where HQ > 1.0. HQs in all other prey samples are < 1.0.
[c] HQ based on 56% incidentally ingested sediment and 8% (in parentheses).
[d] HQ based on predicted tissue concentrations. Eleven of the twenty-nine predicted sediment samples with HQs > 1.0 were based on N-qualified data.
[e] HQs were >1.0 in predicted tissue concentrations in the same locations as the empirical tissue concentrations and in the following locations: International Slip, RM 4.1 on the east side, Slip 1, RM 5.8 on the east side, along RM 7.0 to RM 7.7 on the west side, Swan Island Lagoon, and along RM 8.8 to RM 9.7 on the west side.

AWQC – ambient water quality criteria
BEHP – bis(2-ethylhexyl) phthalate
BERA – baseline ecological risk assessment
BSAR – biota-sediment accumulation regression
COC – chemical of concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
HQ – hazard quotient
LOE – line of evidence
LOAEL – lowest-observed-adverse-effect level
LWG – Lower Willamette Group
N – tentative identification
NOAEL – no-observed-adverse-effect level
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
UCL – upper confidence limit on the mean
UF – uncertainty factor
VOC – volatile organic compound

8.0 WILDLIFE RISK ASSESSMENT

This section presents the draft BERA for wildlife (birds and mammals) in the Study Area. Dietary exposure[104] is the main pathway[105] by which wildlife receptors are exposed to sediment contaminants (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. They are:

- Sediment-probing invertivorous birds – spotted sandpiper
- Omnivorous birds – hooded merganser
- Piscivorous birds[106] – bald eagle, osprey
- Aquatic-dependent carnivore – mink, river otter

Dietary exposure risks to wildlife receptors were evaluated using two LOEs. Dietary dose was used as an LOE for all six wildlife receptors. Both prey tissue ingestion and incidental sediment ingestion were accounted for in dietary-dose estimates. Egg tissue residue, estimated from maternal dietary dose, was used as a second LOE for bald eagle and osprey.

Exposure estimates for the two wildlife LOEs were compared to TRVs expressed as dietary effect thresholds. The TRVs provide a basis for evaluating whether exposure concentrations are at or above a level that may cause an effect on survival, growth, or reproduction of most, but not all, individual organisms in experimentally exposed populations. This follows the conventional practice in ecological risk assessment of using organism-level TRVs defined in this manner to evaluate the potential effects on populations; however, organism-to-population extrapolation is a source of uncertainty and, in many but not all cases, a source of conservatism in risk estimates. Whether the organism-to-population extrapolation is a source of conservatism depends in large part on the level of effect represented by the TRV, which varies because of differences in the toxicological databases for different chemicals and receptor groups (Solomon et al. 2008; Suter 2007; Posthuma et al. 2002; Pastorok et al. 2001).

[104] Sediment is considered a minor component of many wildlife receptor diets, through incidental ingestion during feeding.

[105] Ingestion of water or dermal contact with air and water are generally classified as complete but insignificant pathways in EPA's CSM.

[106] Belted kingfisher was evaluated in the uncertainty assessment of the dietary-dose risk characterization section (Section 8.1.4.4).

Uncertainty of Extrapolation of Organism-Level to Population-Level Effects

Ecological risk assessments often evaluate risks to populations. Understanding the application of organism-level to population-level effects is one of the difficult challenges that confront ecological risk assessors and risk managers. At the most basic level, the fundamentals of population ecology indicate that in general, species that reproduce more slowly, and invest more energy in rearing their young, are more likely to be affected by relatively "low-level" exposure to chemical stressors. By "low-level" we mean levels close to where effects on individual organisms start to be observed (known in risk assessment jargon as "lowest observed adverse effect levels" or LOAELs). Depending on the species concerned, exposure to concentrations that are slightly to somewhat above the LOAEL could have important population-level effects or no discernable effect on the population at all. All else being equal (e.g., similar magnitudes of organism-level responses between species), a mink population would experience a more dramatic impact than crayfish or forage fish populations because of life history differences between species. Another factor that ecological risk assessors and risk managers have to keep in mind is that the dose-response relationship of individual organisms (e.g., individual mink) to a chemical might differ from one chemical to the next. For one chemical, exposure to twice the LOAEL might affect half of the exposed mink, whereas for another chemical, exposure to twice the LOAEL might only affect 10% of the exposed mink. Finally, the effect of one chemical might have more severe implications for the population than the effects of another. A chemical that causes a 10% decrease in reproductive success at twice the LOAEL might affect the mink population more than a chemical that at twice the LOAEL reduces growth among exposed organisms by 5 to 15%.

Wildlife risks were assessed by comparing exposure estimates to TRVs. Several factors had to be accounted for to estimate exposure levels. These included information about feeding rates, foraging areas, prey home ranges, and diets. TRVs were selected from published studies and approved by EPA for use in the BERA.

All of the wildlife COPCs that were identified through the SLERA and refined screening process were evaluated. 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 1997a).

The wildlife risk assessment process involved a tiered three-step process for evaluating the COPCs identified in the SLERA. In Step 1, HQs were calculated on a sample-by-sample basis for each individual prey component. This was done in accordance with the methods described in the EPA's Problem Formulation (Attachment 2). If individual samples within a single component of a wildlife receptor's diet (sediment or tissue) had an HQ > 1.0, then the exposure assumptions were developed in greater detail in Step 2. Step 2 accounted for the ingestion of multiple samples of a single prey species collected from within an appropriate foraging area for the receptor (this analysis was repeated for each type of prey in the wildlife receptor's diet). If, and only if, the evaluation of any of the single prey in Step 2 yielded an HQ > 1.0, the exposure was assessed for an appropriate multi-species diet within an appropriate foraging area. If the exposure estimates for a COPC based on the multi-species diet evaluation yielded an HQ > 1.0 to the wildlife receptor, then the COPC was designated a COC.

Section 8.1 presents the wildlife risk evaluation process for both LOEs, a summary of the TRVs and exposure assumptions that were used, the risk evaluation results, and a discussion of specific uncertainties associated with each LOE. Section 8.2 presents the

overall conclusions about wildlife risk, including a synoptic analysis of uncertainty. The final determination of COCs, the qualitative WOE approach used to determine risk conclusions for piscivorous birds, the key uncertainties in the exposure assessment and effects characterization, and interpretation of the risk assessment findings are presented in Section 8.2.

8.1 DIETARY ASSESSMENT

Dietary dose was the primary method 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 predicted bird egg concentrations to literature-derived bird egg TRVs.

Receptor-specific dietary and bird egg 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 the chemical concentrations in prey tissue and sediments that are incidentally ingested while feeding. Toxicity thresholds were expressed as concentrations in tissue and sediment that were back-calculated from dietary-dose and egg concentration thresholds using receptor-specific exposure assumptions.

For each receptor, COPCs with HQs > 1.0 at the end of the three-step risk evaluation process were retained as COCs. COCs, including the magnitude of HQs, the spatial distribution and frequency of HQs > 1.0, the results of multiple LOEs (when applicable), and the associated exposure and effects assumptions, were evaluated to arrive at risk conclusions for wildlife.

The details of this dietary risk assessment for wildlife are presented as follows:

- Section 8.1.1 presents a summary of the COPCs evaluated in the dietary risk evaluation. COPCs were identified for all receptors using the dietary-dose approach. COPCs were also identified for piscivorous bird receptors (bald eagle and osprey) using the bird egg approach.
- Section 8.1.2 presents an overview of the assumptions used to derive exposure concentrations, including how UCLs were derived. Exposure data in this assessment are represented by COPC concentrations in composite prey tissue and sediment samples. 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 8.1.3 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 8.1.4 presents the risk characterization results, COC-receptor pairs, and associated uncertainties. These COCs are further assessed in the wildlife risk conclusions section (Section 8.2). 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 organization.

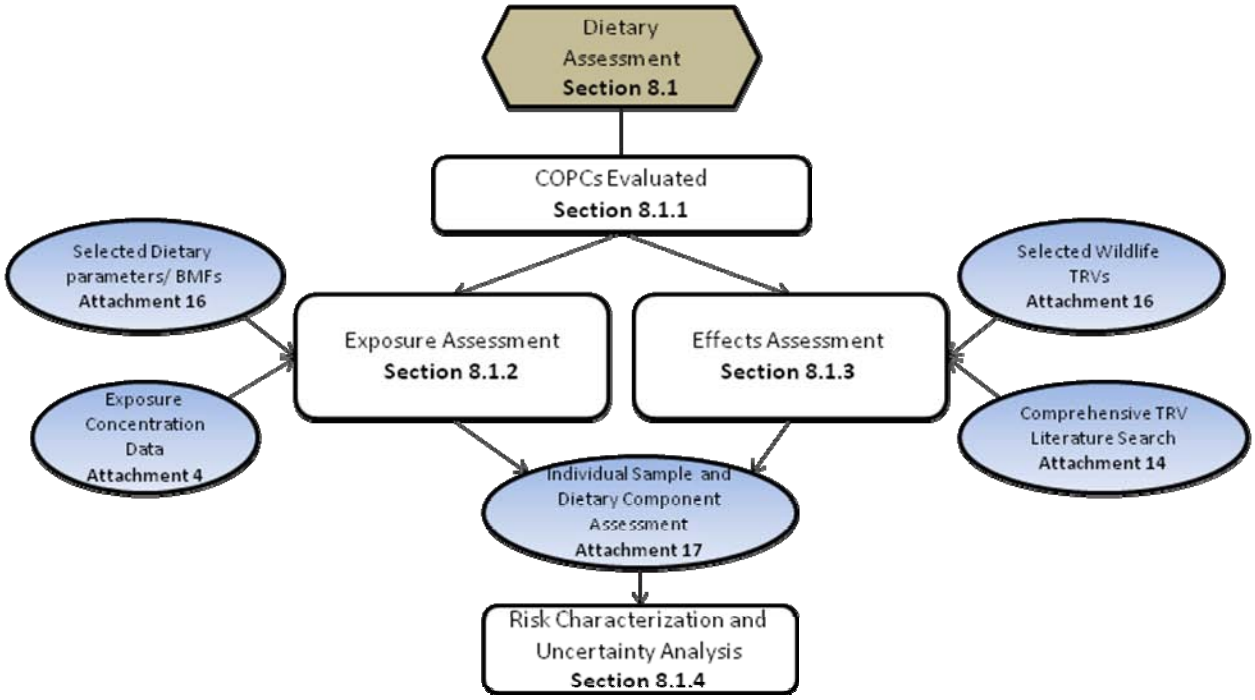

Figure 8-1. Overview of Wildlife Dietary Assessment Section Organization

8.1.1 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 the toxicity of those COIs to wildlife.

COPC-receptor pairs were identified in the SLERA and refined screen (Attachment 5) and the detailed methods and results can be found there. Table 8-1 presents the wildlife COPCs. All of these COPC-receptor pairs were evaluated in a three-step wildlife risk evaluation process, with the exception of aluminum for birds,[107] which could not be evaluated because no BERA-level (LOAEL) TRV could be derived from the literature. Therefore, the wildlife

[107] Aluminum was retained as a COPC because of exceedances of screening levels based on the spotted sandpiper and hooded merganser diets.

risk evaluation for aluminum was only a screening-level assessment. Aluminum is revisited in the risk characterization, where information about background concentrations is introduced to provide perspective on potential risks to wildlife from aluminum exposure.

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[a]	X[a]	X	X
Selenium	X				X	
Thallium	X					
Zinc	X					
PAHs						
Benzo(a)pyrene	X	X		X		
Total HPAHs	X				X	X
Total LPAHs	X					
Total PAHs	X					
Phthalates						
BEHP	X	X	X	X		

Table 8-1. Wildlife Dietary COPCs

COPC	Birds				Mammals	
	Spotted Sandpiper	Hooded Merganser	Bald Eagle	Osprey	Mink	River Otter
Dibutyl phthalate	X	X				
PCBs						
Total PCBs	X	X	X[a]	X[a]	X	X
PCB TEQ[b]	X	X	X[a]	X[a]	X	X
Dioxins/Furans						
Dioxin/furan TEQ[b]	X	X	X[a]	X[a]	X	X
Total TEQ[b]	X	X	X[a]	X[a]	X	X
Pesticides						
Aldrin	X					
4,4′-DDE			X	X		
Sum DDE	X	X	X			
Total DDx	X	X		X	X	X

[a] This COPC was evaluated in both the dietary-dose and bird egg evaluation.
[b] Per EPA (Attachment 2), TEQ was evaluated as PCB TEQ, dioxin/furan TEQ, and total TEQ.

BEHP – bis(2-ethylhexyl) phthalate
COPC – chemical of potential concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
HPAH – high-molecular-weight polycyclic aromatic hydrocarbon
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)

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 were among the COPCs for birds and mammals. A TEQ is the sum of the constituents (i.e., toxicity equivalence concentrations), representing a

total dioxin concentration, modified by factors reflecting the toxicity of each constituent and normalized to the most toxic constituent (2,3,7,8-TCDD). Toxicity equivalency concentrations were calculated consistent with EPA guidance (EPA 2008i). TEQ concentrations for birds and mammals were calculated using toxic equivalency factors (TEFs) presented in Van den Berg et al. (1998) and Van den Berg et al. (2006), respectively. TEFs are used based on evidence that there is a common mechanism of toxicity for certain dioxins, furans, and PCB congeners that involves binding to the aryl hydrocarbon (Ah) receptor as an initial step. Data on the relative binding affinity of particular PCB congeners compared to 2,3,7,8-TCDD are available from *in vivo* and *in vitro* studies. These data have been used to derive TEFs for PCB congeners that show structural similarity to dioxins 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, five studies have been conducted, resulting in a TEF range of < 0.0003 to 0.15 for the various bird species tested for ethoxyresorufin-O-deethylase (EROD) induction or *in ovo* effects. For PCB 81, two identified studies tested several species for EROD induction, with TEFs highly variable, ranging from 0.001 to 0.5. For PCB 126 and 169, data are available from only one study (*in ovo* with chickens). The relevance of TEFs derived by EROD induction or *in ovo* studies to risk assessment based on dietary exposure is also uncertain. It is not known if the uncertainties in the bird TEFs 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.

Eleven COIs could not be screened or otherwise evaluated because toxicological data are not available for birds or mammals (Table 8-2). These COIs were still evaluated in other parts of the ERA (i.e., for fish, amphibian, plant and/or benthic invertebrate receptors). An additional eight COIs lacked avian toxicological data; these COIs were evaluated for mammals but not birds.

Table 8-2. COIs Not Evaluated for Birds and/or Mammals

COI	
Metals	
Antimony[a]	Silver
Manganese	
PAHs	
1-Methylnaphthalene[a]	Dibenzothiophene[a]

Table 8-2. COIs Not Evaluated for Birds and/or Mammals

COI	
2-Methylnaphthalene[a]	Perylene
Benzo(e)pyrene	Alkylated PAHs
SVOCs	
Benzoic acid	n-Nitrosodiphenylamine
Benzyl alcohol[a]	2-Methylphenol
Carbazole	4-Chloro-3-methylphenol
Dibenzofuran	4-Methylphenol[a]
Hexachloroethane[a]	Phenol[a]

[a] No bird dietary screening-level threshold was available; however, a mammal dietary threshold was available.
COI – chemical of interest
PAH –polycyclic aromatic hydrocarbon
SVOC –semivolatile organic compound

8.1.2 Exposure Assessment

This section presents the methods and assumptions that were used to estimate wildlife exposures to COPCs. The exposure media in the wildlife exposure assessment are prey tissue and surface sediment, so the methods and assumptions are presented for estimating both prey tissue and surface sediment EPCs. 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 COPC-receptor pair for each exposure area within the Study Area.

The exposure assessment was an iterative three-step process as described above. If a particular COPC was found to not pose a potential unacceptable risk to a wildlife receptor, it was not carried forward any further in the process. If the EPCs represented by single samples resulted in all Study Area HQs < 1.0 for a particular COPC-receptor pair, that EPC was the final exposure estimate derived. If an EPC represented by a single sample resulted in at least one sample with an HQ > 1.0, then the COPC was carried forward to Step 2 in the evaluation process, at which point an EPC was derived by averaging across individual single-species prey tissue samples (during the second step), or by averaging across all prey tissue samples accounting for prey fractions (during the third step). Attachment 4 provides all EPCs (expressed as tissue and sediment concentrations) for all COPC-receptor pairs for the multiple exposure steps.

8.1.2.1 Exposure Concentrations

This section presents a summary of the methods used to derive EPCs in prey tissue and sediment to derive wildlife risk estimates using both the dietary-dose and bird egg approach. Prey tissue EPCs were evaluated using both the dietary-dose and bird egg approach to assess dietary risks. Sediment EPCs were also evaluated to address potential exposure via incidentally ingested sediment using the dietary-dose approach. Tissue and sediment EPCs were represented by measured concentrations detected in composite samples collected from the Study Area or from tissue samples following laboratory bioaccumulation testing.[108]

The wildlife risk evaluation using the dietary-dose and bird egg approach involved multiple steps, in which the data used to represent EPCs varied. Figure 8-2 presents the process used for developing EPCs in the wildlife assessment.

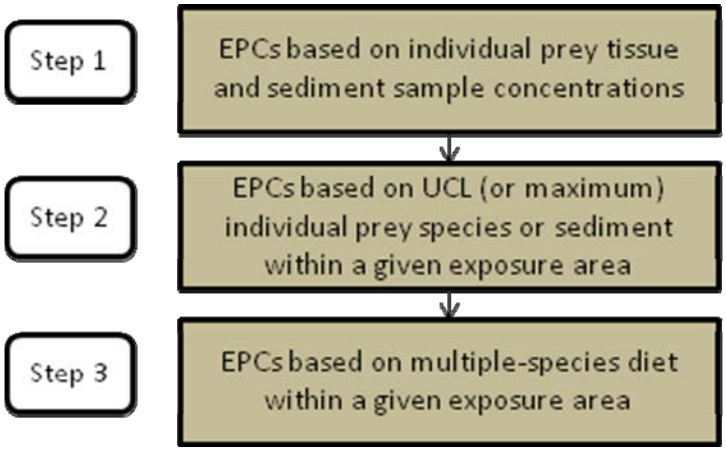

Figure 8-2. EPC Derivation Process for Wildlife

The EPC derivation process for evaluating dietary risks for each of these iterative risk characterization steps was detailed in EPA's Problem Formulation (Attachment 2). Details of how the derivation process was implemented were vetted and approved by EPA during a meeting on May 12, 2008, prior to initiating the BERA. The process was as follows:

Step 1 – EPCs were first represented by individual prey composite tissue and sediment sample concentrations from throughout the Study Area to evaluate dietary risks on a sample-by-sample basis per EPA's Problem Formulation (Attachment 2). COPC concentration data for all individual samples are presented in the Attachment 4.

EPCs based on individual composite tissue and sediment samples were used to calculate HQs using the following equation.

[108] Chemical concentrations of neutral organic COPCs in tissues from bioaccumulation testing were adjusted for steady-state. See Attachment 3 for details on how steady-state concentrations were derived.

$$HQ_{tiss} = \frac{EPC}{TTC} \text{ and } HQ_{sed} = \frac{EPC}{TSC}$$ **Equation 8-1**

Where:

HQ = hazard quotient
EPC = exposure point concentration[109]
TTC = threshold tissue concentration (see Section 8.1.2.2)
TSC = threshold sediment concentration (see Section 8.1.2.2)

Step 2 – Prey tissue and sediment EPCs were then calculated within receptor-specific exposure areas based on foraging assumptions. Table 8-3 presents the exposure area assumptions used to derive EPCs for each wildlife receptor. The rationale for these exposure area assumptions is presented in Attachment 16. Maps 8-1 through 8-3 present the exposure areas for wildlife receptors.

Table 8-3. Summary of Receptor-Specific Exposure Area Assumptions

Receptor	Exposure Areas[a]	Exposure Areas	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 were based on UCL[b] COPC concentrations.
Hooded merganser, bald eagle, osprey, mink	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	Map 8-2	EPCs calculated within each 1-mile-increment exposure area for each prey species and sediment were 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 peamouth).
River otter	3-mile increments of the Study Area	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 were based on UCL[c] COPC concentration; 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).

[109] In Step 1, EPCs were based on individual sample concentrations.

[c] When insufficient data were available to calculate a UCL, the maximum concentration was used to represent the EPC.

COPC – chemical of potential concern

EPC – exposure point concentration

RM – river mile

UCL – upper confidence limit on the mean

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. The ProUCL software used for this analysis allows detected and undetected values to be considered 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 was required to derive a UCL (EPA 2007f). In the case where an insufficient number of detected data values was available, the maximum concentration[110] was used to represent the 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 Equation 8-1.

Step 3 – In the first two steps of risk characterization, dietary exposure was evaluated for each prey species individually. In order to estimate dietary risks that account for the ingestion of multiple prey, dietary portions were assigned to each prey item for a given receptor. Prey portions were used to derive total HQs using the following equation:

$$HQ_{total} = \left(\sum_{i=1}^{n} HQ_i F_i \right) + (HQ_{sed}) \qquad \textbf{Equation 8-2}$$

[110] When the maximum concentration was a non-detected value, the full detect limit was used to represent the EPC.

Where:

HQ_{total} = hazard quotient within a relevant exposure area based on multiple prey and sediment ingestion

HQ_i = hazard quotient[111] 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

Selected prey portions are based on the diets reported in regional literature studies and are presented in Section 8.1.2.3.2. Risk conclusions are based on the exposure assumptions of Step 3.

The Evaluation of HQs on a Sample-by-Sample Basis

In Step 1 of the risk characterization, dietary risks were evaluated on a prey sample-by- prey sample basis, per EPA's Problem Formulation (Attachment 2). These results are presented in Attachment 17. This evaluation is consistent with the screening and iterative refinement procedures. A sample-by-sample analysis can be used to screen out potential COCs when no individual prey sample HQs are > 1.0; but if individual sample HQs are > 1.0, the appropriate conclusion is that the exposure assumptions should be refined before drawing risk conclusions because wildlife 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 wildlife is ultimately based on risk estimates where diets are composed of multiple prey species within a relevant exposure scale. Risk conclusions for wildlife receptors are ultimately based on these more realistic exposure assumptions.

8.1.2.2 Use of Exposure Parameters

This subsection presents the equations that were used to derive receptor-specific toxicity thresholds from dietary-dose and bird egg exposure parameters. The exposure parameter data themselves (i.e., the data that were used in these equations) are presented in Sections 8.1.2.3 and 8.1.2.4. Receptor-specific toxicity thresholds were represented by threshold tissue concentrations (TTCs; expressed as mg/kg ww) in prey and threshold sediment concentrations (TSCs; expressed as mg/kg dw). The following subsections present how receptor-specific TTCs and TSCs were derived for the dietary-dose and bird egg evaluations.

8.1.2.2.1 Dietary-Dose Approach

EPA's Problem Formulation (Attachment 2) specified that TTCs and TSCs should be back-calculated from dietary-dose TRVs using receptor-specific parameters (i.e., body weight, prey ingestion rate, incidental sediment ingestion rate and site use) compiled from general and region-specific literature. The following equations were used to develop receptor-specific TTCs and TSCs from dietary-dose TRVs:

[111] Note that these HQs (for each prey species and for sediment) are the same as HQs calculated in Step 2.

$$TTC = \frac{TRV_{diet}}{\left(\frac{FIR}{BW} \times SUF\right)}$$ **Equation 8-3**

Where:

TTC = threshold tissue concentration (mg/kg ww)

TRV_{diet} = dietary-dose toxicity reference value (mg/kg bw/day)

FIR = food ingestion rate (kg ww food/day)

BW = body weight (kg)

SUF = site use factor (unitless); fraction of time that a receptor spends foraging at the site relative to the entire home range and based on consideration of seasonal use

And:

$$TSC = \frac{TRV_{diet}}{\left(\frac{SIR}{BW} \times SUF\right)}$$ **Equation 8-4**

Where:

TSC = threshold sediment concentration (mg/kg dw)[112]

TRV_{diet} = dietary-dose toxicity reference value (mg/kg bw/day)

SIR = incidental sediment ingestion rate (kg dw sediment/day)

BW = body weight (kg)

SUF = site use factor (unitless); fraction of time that a receptor spends foraging at the site relative to the entire home range and based on consideration of seasonal use.

[112] Note that the TSC applies only to the incidental ingestion of sediment.

Dietary-dose TRVs were derived from the literature using the following equation:

$$TRV_{diet} = \frac{(IR \times C_{diet})}{BW} \qquad \textbf{Equation 8-5}$$

Where:

TRV_{diet} = dietary-dose toxicity reference value (mg/kg bw/day)

IR = ingestion rate (kg food/day)

C_{diet} = chemical concentration in diet (NOAEL or LOAEL) as measured and reported in the toxicity study (mg/kg ww)

BW = body weight (kg)

When using the back-calculation approach, tissue and sediment concentrations can be compared directly to TTCs and TSCs. The alternative would be to forward-calculate dietary-dose estimates, using the same exposure parameters and equations. It should be noted that the two methods are mathematically equivalent (i.e., they result in the same answers).

Dietary-dose exposure parameters are presented in Section 8.1.2.3. Selected dietary-dose TRVs and back-calculated TTCs and TSCs are presented in Section 8.1.3.

8.1.2.2.2 Bird Egg Approach

Specific COPCs were evaluated by comparing predicted bird egg chemical concentrations to literature-derived bird egg TRVs as a second LOE for evaluating risks to piscivorous birds (osprey and bald eagle). No osprey or bald eagle egg chemical concentration data were collected for the Portland Harbor RI/FS. Bird egg concentrations were predicted by multiplying estimated prey tissue concentrations by estimated prey tissue-to-egg tissue BMFs from the literature.

BMFs are sufficiently uncertain as to raise concerns about the reliability of this LOE, which is addressed in Section 8.1.2.3.2. The LOE was evaluated in accordance with EPA's comments on the Ecological PRE (EPA 2006a), which indicated that the bird egg LOE was considered important to be protective of developing embryos. It should be noted that the dietary-dose LOE also evaluates risk to developing embryos (because the TRVs for the specific COPCs for which the bird egg LOE is used are based on reproductive effects) and that it does so without adding BMF uncertainty.

COPC concentrations in osprey or bald eagle eggs were not available from the Study Area at the time of the preparation of this report. Osprey egg data were collected from three nest sites in the Study Area in 2001 (Henny et al. 2008). These data have not been published. The LWG requested these data from the author in December 2008 and January 2009 but they were not provided and were not available for inclusion in this report (Bergquist 2009). Additional egg data collected in 1998 and 1999 between RM 0 and RM 26 (Henny et al. 2008) might include data collected from the Study Area. Osprey eggs were also collected from the Study Area in May 2008. Five eggs were collected from each of the following

areas: the Study Area, Multnomah Channel, and upriver (between RM 69 and RM 73). The data from these 15 eggs were not yet available when this report was prepared.

In the absence of bird egg tissue residues, egg chemical concentrations can be estimated using BMFs applied to fish tissue consumed as prey. BMFs are calculated using the following equation:

$$BMF = \frac{C_{egg}}{C_{fish}}$$ **Equation 8-6**

Where:

BMF = fish-to-egg biomagnification factor (unitless)
C_{egg} = chemical concentration in bird egg (mg/kg ww)
C_{fish} = chemical concentration in fish prey (mg/kg ww)

Using region-specific BMFs and bird egg TRVs, TTCs in fish were calculated for piscivorous birds using the following equation:

$$TTC = \frac{TRV_{egg}}{BMF}$$ **Equation 8-7**

Where:

TTC = threshold tissue concentration (mg/kg ww)
TRV_{egg} = bird egg toxicity reference value (mg/kg ww)
BMF = fish-to-egg biomagnification factor (unitless)

The TTCs were used so that fish tissue chemical concentrations could be compared directly to fish tissue thresholds. BMFs are presented in Section 8.1.2.3. Selected bird egg TRVs and back-calculated TTCs and TSCs are presented in Section 8.1.3.

8.1.2.3 Exposure Parameters and Dietary Prey Assumptions

The following subsections present the exposure parameters used in TTC and TSC equations that were presented in the previous section. The methodology for estimating the sediment ingestion rate also is presented.

8.1.2.3.1 Dietary-Dose Exposure Parameters

Body weights, food ingestion rates, sediment ingestion rates, and site use factors vary among bird and mammal receptors. Table 8-4 presents the dietary parameters for bird and mammal receptors. Details and the rationale for selected receptor-specific exposure parameters and uncertainties are presented 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]	SUF
Spotted sandpiper	0.047	Maxson and Oring (1980)[c]	0.055	Nagy (2001)	18[d]	NA	0.0015	1.0
Hooded merganser	0.54	Dunning (1993)	0.20	Nagy (2001)	2[e]	NA	0.0011	1.0
Bald eagle	4.5	Wiemeyer (1991)[c]	0.54	Stalmaster and Gessaman (1984)[c]	2[e]	74%[f]	0.0028[g]	1.0
Osprey	1.9	Poole (1983)[c]	0.40	Poole (1983)[c]	2[e]	74%[f]	0.0021[g]	1.0
Mink	0.97	Hornshaw et al. (1983)[c]	0.16	Bleavins and Aulerich (1981)[c]	9.4[dh]	74%[h]	0.0038[g]	1.0
River otter	7.7	USGS (2004)	0.76	Nagy (2001)	2[e]	NA	0.0047[i]	1.0

[a] Percent of incidental sediment ingestion.
[b] SIR = FIR x 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. The dw FIR was calculated based on the following equation: FIR (dw) = FIR (ww) x (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
SUF – site use factor
NA – not applicable
USGS – US Geological Survey
ww – wet weight

Uncertainties Associated with Exposure Parameters

A site use factor of 1.0 was assumed for all wildlife receptors, per EPA's Problem Formulation (Attachment 2). Some receptors (particularly hooded merganser and bald eagle) may forage in nearby aquatic and terrestrial environments and may therefore use the site less than 100% of the time. Assuming a site use factor of 1.0 for osprey is conservative because osprey 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. However, Elliott et al. (2007) reported that wintering in different sites had no significant effect on organochlorine contaminant concentration in sample eggs, including those from ospreys from the Lower Columbia River. This uncertainty is further evaluated in Section 8.1.4 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 conservative incidental sediment assumptions do not impact risk estimates.

Dietary doses for all wildlife receptors were primarily based on female exposure parameters. Body weights and food ingestion rates were primarily based on the literature presented in EPA's *Wildlife Exposure Factors Handbook* (EPA 1993). When species-specific data were not available in EPA (1993), food ingestion rates were based on the allometric equations presented in Nagy (2001). Site use factors were based on species-specific data from regional studies. 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:

$$SIR = (FIR \times F_{solids}) \times SI \qquad \textbf{Equation 8-8}$$

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

To determine sediment ingestion rates on a dry-weight basis, food ingestion rates 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).

8.1.2.3.2 Bird Egg Exposure Parameters

This section presents an overview of the literature-based BMFs and associated uncertainties that were used to back calculate bird egg tissue TRVs into TTCs for fish prey tissue.

BMFs were calculated using Equation 8-6. BMFs were compiled for each of the COPCs based on piscivorous bird species from the Willamette River, the Lower Columbia River, the Great Lakes region, and South Carolina. Attachment 16 presents a summary of the literature-based BMFs. In accordance with EPA's comments on the Ecological PRE (EPA 2006a), BMFs were selected from the reviewed literature to be most representative of the Study Area region (from the Willamette River) and the selected piscivorous bird receptor species (i.e., bald eagle and osprey).

Table 8-5 presents the selected BMFs for all bird egg COPCs. The selected BMFs were used for both bald eagle and osprey. BMFs for all COPCs (except mercury) were based on osprey data from the Willamette River reported by Henny et al. (2003; 2008). The application of BMFs based on osprey dietary assumptions (e.g., exclusive consumption of fish) to bald eagles is uncertain. For mercury, no Willamette-specific data were available, and the selected BMF was based on bald eagle data from the Lower Columbia River (Buck 2004). The selected BMF for mercury based on Columbia River bald eagles (BMF = 3.0) is a conservative estimate for predicting osprey egg concentrations, inasmuch as Henny et al. (2008) reported that mercury concentrations did not biomagnify in Willamette River osprey. Because Henny et al. (2003) did not report a BMF for TEQs, the selected BMF for PCB TEQ, dioxin/furan TEQ, and total TEQs was based on the 2,3,7,8-TCDD BMF.

Table 8-5. Selected Literature-Based BMFs

COPC	BMF	Source
Metals		
Mercury	3.0	Buck (2004)
PCBs		
Total PCBs	8.4	Henny et al. (2008)
PCB TEQ	10[a]	Henny et al. (2003)
Dioxins/Furans		
Dioxin/Furan TEQ	10[a]	Henny et al. (2003)
Total TEQ	10[a]	Henny et al. (2003)
Pesticides		
4,4′-DDE	79	Henny et al. (2008)

[a] Based on the reported BMF for 2,3,7,8-TCDD.
BMF – biomagnifications factor
COPC – chemical of potential concern
DDE – dichlorodiphenyldichloroethylene
PCB – polychlorinated biphenyl
TEQ – toxic equivalent

Uncertainties Associated with BMFs

There are uncertainties associated with the selected literature-based BMFs used to estimate bird egg tissue concentrations. The studies from which all BMFs except mercury were derived used egg and fish data collected from throughout the Willamette River, encompassing an area of over 150 river miles (Henny et al. 2003; 2008). Because osprey are opportunist feeders that tend to forage close to the nest and because there was variability in the egg concentrations collected through the river, site-specific BMFs were calculated for each nest (i.e., each egg concentration) using co-located data provided in Henny et al. (2003).[113] Fish tissue and bird egg tissue within approximately 1 mile of each other were considered co-located. The range of these calculated site-specific BMFs at each co-located area is presented in the table below and indicates the uncertainty associated with the river-wide BMFs. For example, though the river-wide BMF for total PCBs was calculated by Henny et al. (2003) as 11, the site-specific BMFs ranged from 4.5 to 31.4.

River-Wide and Site-Specific Osprey BMFs

COPC	River-Wide BMFs[a]	Site-Specific BMFs[b] Range	Geomean
2,3,7,8-TCDD	10	NC	NC
PCB TEQ	ND[c]	3.1 – 7.4	5.1
Dioxin/furan TEQ	ND[c]	2.3 – 24.9	5.0
Total TEQ	ND[c]	3.4 – 10.9	5.1
Total PCBs[d]	11	4.5 – 31.4	11
4,4′-DDE	87	8.1 – 234.9	64.2

[a] Reported by Henny et al. (2003) using geomeans of 10 egg concentrations and 25 fish composite concentrations.

[b] Calculated using seven co-located fish tissue composite samples and bird egg tissues based on data reported by Henny et al. (2003). Co-located samples were defined as composite fish tissue sampling locations and nest locations where egg tissues were collected that were approximately within 1 mile of one another.

[c] No data were available in Henny et al. (2003).

[d] Total PCBs are based on reported sum of PCB congeners.

BMF – biomagnification factor

COPC – chemical of potential concern

DDE – dichlorodiphenyldichloroethylene

NC – not calculated (2,3,7,8-TCDD was frequently undetected in largescale sucker)

ND – no data

PCB – polychlorinated biphenyl

TCDD – tetrachlorodibenzo-*p*-dioxin

TEQ – toxic equivalent

BMFs were not calculated by Henny et al. (2003) for TEQs. As a result, the 2,3,7,8-TCDD BMF was used to estimate bird egg TTCs. There is uncertainty associated with this method because different dioxin and furan congeners have different BMFs, ranging from 0.42 for 2,3,7,8-TCDF to 174 for OCDD (Henny et al. 2003). Data presented in the above table indicate that site-specific BMFs for PCB TEQs, dioxin/furan TEQs, and total TEQs may be lower than the river-wide BMFs (i.e., 5.0 to 5.1 for geomeans of seven site-specific BMFs compared to 10 for the river-wide BMF calculated using geomean data based on 2,3,7,8-TCDD). The use of a TEQ-based BMF (5.1) reduces predicted egg tissue concentrations by about half compared with the predicted tissue chemical concentrations based on the 2,3,7,8-TCDD BMF (10).

[113] Data presented in Henny et al. (2008) from 2001 were not sufficient for calculating site-specific BMFs. Using data reported by Henny et al. (2003), egg tissue data were paired with the closest fish composite data. Data were not used for three eggs, which did not have fish data collected from within 5 miles of the nest.

Willamette River site-specific BMFs that were used to estimate risks based on the bird egg approach per EPA's comments on the Ecological PRE (EPA 2006a) were evaluated to determine if there was a statistically significant relationship between fish tissue concentrations and egg concentrations based on the seven co-located fish composite tissue locations and nest locations where eggs were collected (Henny et al. 2003). This evaluation was undertaken to determine the uncertainty of using BMFs to estimate risks and the uncertainty of this LOE. 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 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. Attachment 8 presents a discussion of methods and results of the BMR data evaluation of co-located fish tissue and egg tissue concentrations for the five bird egg COPCs. No significant relationship (i.e., no BMR) could be found for any bird egg COPC except total TEQ. The implication is that the available dataset is insufficient to estimate a reliable BMF, so this LOE is potentially unreliable, particularly as compared with the dietary-dose LOE. A comparison of the dietary-dose and bird egg LOEs and the reliability of the results of both LOEs is presented in Section 8.2

For total TEQs, a linear relationship between egg and fish tissue is significant ($r^2 = 0.58$) based on the following BMR:

$$C_{egg} = (4.372 \times C_{fish}) + 3.987 \qquad \textbf{Equation 8-9}$$

Where:

C_{egg} = estimated bird egg tissue COPC concentration (mg/kg ww)

C_{fish} = fish tissue COPC concentration (mg/kg ww)

Using this BMR, predicted total TEQ concentrations in eggs are about 48% lower than if the geomean BMF of 10 is used.[114] Use of the BMR provides a better method of estimating egg from prey chemical concentrations than the geomean BMF because it better accounts for the pattern of the relationship between the paired prey and egg tissue data It should be noted that the range of fish prey EPCs for total PCBs, dioxin/furan TEQs, PCB TEQs, total TEQs, and 4,4′-DDE from the Study Area is much higher than the fish chemical concentrations from the Willamette River used to calculate BMFs in Henny et al. (2003) (Table 8-6). The need to extrapolate beyond the range of the empirical fish tissue data used to develop the BMF and BMR is another source of uncertainty that affects the reliability of the bird egg LOE. It is uncertain whether the levels of biomagnification estimated from the Henny et al. (2003) dataset (whether by a geomean or a regression model) apply at the higher prey tissue concentrations detected in the Study Area.

[114] Predicted total TEQ egg concentrations based on Equation 8-9 range from 244 to 338 ng/kg ww, and predicted egg concentrations based on the geomean 2,3,7,8-TCDD BMF (10) range from 606 to 800 ng/kg ww.

Table 8-6. Comparison of Fish Tissue Concentrations Used to Derive BMFs to Study Area Fish Prey Tissue Concentrations

COPC	Unit (ww)	Range of Fish Tissue Concentrations	
		Henny et al. (2003)	Study Area EPCs[a]
Total PCBs[b]	µg/kg	29.95 – 254.77	130 – 19,213
Dioxin/furan TEQ	ng/kg	0.11 – 17.88	2.33 – 232
PCB TEQ	ng/kg	1.78 – 12.36	9.43 – 196
Total TEQ	ng/kg	2.88 – 30.24	11.8 – 262
4,4′-DDE	µg/kg	6.43 – 318.36	53 – 600

[a] Study area EPCs are the range of osprey and bald eagle prey tissue EPCs over all exposure areas used to calculate HQs.

[b] Total PCBs are based on the reported sum of PCB congeners.

BMF – biomagnification factor
COPC – chemical of potential concern
DDE – dichlorodiphenyldichloroethylene
EPC – exposure point concentration
PCB – polychlorinated biphenyl
TEQ – toxic equivalent
ww – wet weight

Summary of Uncertainties Associated with Selected BMFs

BMFs are calculated as the ratio of fish prey and piscivorous bird egg concentrations. The following uncertainties associated with the selected BMFs based on Willamette River data were identified:

BMFs were calculated using prey and egg geometric mean concentrations across a 150-mile area; whereas, the home range of osprey is much more localized. BMFs based on 1-mile co-located prey and egg concentrations are variable (ranging from 1 to 2 orders of magnitude).

Statistical evaluation of co-located data for the development of BMRs shows no significant relationship between fish tissue and bird egg tissue concentrations for all chemicals except total TEQ.

BMFs for PCB TEQ, dioxin/furan TEQ, and total TEQ are based on the BMF reported for 2,3,7,8-TCDD.

BMFs were derived from a range of fish tissue concentrations that were much lower than Study Area fish tissue exposure concentrations.

8.1.2.3.3 Dietary Prey Assumptions

Table 8-7 presents the prey species in the BERA dataset that were used to derive prey tissue EPCs. Prey species included fish and invertebrate species (i.e., largescale sucker, carp, juvenile Chinook salmon, peamouth, sculpin, smallmouth bass, northern pikeminnow, black crappie, brown bullhead, crayfish, clams, epibenthic invertebrate tissue, and mussels) collected in the Study Area, and invertebrate species (i.e., clams and worms) that underwent laboratory bioaccumulation testing. The selected diets for each receptor were based on

information from the literature. Details and the rationale for the assumptions for selected prey species and associated uncertainties are presented in Attachment 16.

Table 8-7. 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] Both laboratory clam tissue and field-collected tissue concentrations were evaluated. Chemical concentrations of neutral organic COPCs in laboratory clam tissues were adjusted to steady-state concentrations.

[b] Worm tissues were 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, so the use of this prey item to represent hooded merganser dietary concentrations is uncertain.

Prey species were evaluated individually (i.e., assuming the consumption of only one prey type) in the first two steps of the wildlife risk evaluation process that was presented in Figure 8-2.

Uncertainties Associated with Using Laboratory Bioaccumulation Testing to Represent Prey Chemical Concentrations

There is uncertainty 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 COPCs with high K_{OW} values may not be able to reach their steady-state concentrations in tissues within the 28-day duration of the tests. In addition, field and steady-state conditions may not be represented because of the physical manipulation of sediments and possible changes in the chemical form that affect bioavailability and uptake. Clam and worm tissue concentrations of neutral organic COPCs were adjusted to reflect 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 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. Tissues in field-collected clams are expected to be more representative of field conditions than tissues in laboratory-exposed clams.

For those COPC-receptor pairs that were retained through the third step in the process (Figure 8-2), portions of individual prey were varied to better represent multi-species diets presented in the literature. Table 8-8 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 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 impact of varying prey portions on the estimated HQs was evaluated as part of the uncertainty analysis (Section 8.1.4.4).

Table 8-8. 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]
Carp			0.45	0.06	0.20	0.40
Peamouth		0.05[f]	0.05			

Table 8-8. 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
Sculpin		0.65			0.20[g]	0.40
Smallmouth bass				0.02	0.20	0.04
Northern pikeminnow			0.05[h]	0.07[i]		
Black crappie						
Brown bullhead				0.02[j]		

[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 were used as a surrogate species when no largescale sucker data were available for the mink and river otter diet.
[f] Sculpin were used as a surrogate species when no peamouth data were available for the hooded merganser diet.
[g] Smallmouth bass were used as a surrogate species when no sculpin data were available for the mink diet.
[h] Peamouth was used as a surrogate species when no northern pikeminnow data were available for the bald eagle diet.
[i] Smallmouth bass were used as a surrogate species when no northern pikeminnow data were available for the osprey diet.
[j] Smallmouth bass were used as a surrogate species when no brown bullhead data were available for the osprey diet.

8.1.2.4 Data Used to Derive Spotted Sandpiper Exposure Concentrations

The spotted sandpiper exposure assessment was somewhat different than those for the other wildlife receptors because the sandpiper only forages on beaches, so the discussion of the exposure parameters for spotted sandpiper has been separated from those of the other wildlife receptors.

All sediment and tissue data from the Study Area were used to develop EPCs and evaluate risks to all the wildlife receptors except sandpiper. In Steps 2 and 3 of the risk evaluation process (Figure 8-2), 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, memo from LWG to EPA (Saban and Andersen 2004). These beach areas were used to represent potential exposure areas for shorebirds. Map 8-1 presents the shorebird beach locations (labeled B1 through B28).

Table 8-9 presents 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, field-collected clams (rather than laboratory-exposed clams) were used to represent clam concentrations because field-collected clams are more representative of exposure concentrations in the Study Area. However, for those COPCs that were not analyzed in all field-collected clams at a given beach (i.e., B10, B11, B19, B21, B24, B27, and B28), laboratory clams were used to represent clam concentrations. Chemical concentrations of neutral organic COPCs were steady-state adjusted in laboratory clam and worm tissues.

Table 8-9. Sediment and Tissue Data Used to Derive Risk Estimates for Spotted Sandpiper at Individual Beach Locations

Beach Area	Transect Surface Sediment Sample(s)	Clam Tissue Sample(s)	Laboratory Worm Tissue Sample(s)
B1[a]	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
B21	07B023	FC022[c]	LW022
B22	B024	FC028	LW028

Table 8-9. Sediment and Tissue Data Used to Derive Risk Estimates for Spotted Sandpiper at Individual Beach Locations

Beach Area	Transect Surface Sediment Sample(s)	Clam Tissue Sample(s)	Laboratory Worm Tissue Sample(s)
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 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 – chemical 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 models (i.e., the mechanistic model or BSARs) 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 (2009). BSARs were only were only developed for those COPCs for which a relationship could be established between co-located prey tissue (i.e., worm or field clam) and sediment chemical concentrations.

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 weren't modeled mechanistically. Windward (2009) presents the details of the BSAR analysis and the mechanistic bioaccumulation model.

Table 8-10 presents the shorebird COPC and the selected models used to estimate worm and clam tissue concentrations. Seven COPCs were modeled mechanistically (i.e., total PCBs, PCB TEQ, 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 met the BSAR acceptability criteria noted above (Table 8-10). For benzo(a)pyrene in laboratory-exposed worms, and the other two COPCs (i.e., copper and dibutyl phthalate),

the co-located data indicate no relationship between co-located sediment and tissue concentrations in the BSAR evaluation. 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 could be calculated for these COPCs.

Table 8-10. Shorebird COPCs and Selected Models Used to Predict Prey Tissue Concentrations

	Field Clam		Lab Worm	
COPC	**Predicted Tissue?**	**Selected Model**	**Predicted Tissue?**	**Selected Model**
Metals				
Copper	No[a]	NA	No[a]	NA
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 for these COPCs because these COPCs did not meet the appropriate BSAR analysis assumptions (Windward 2009), 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 sediment and tissue detected concentration data pairs were available (n=5).

BSAR—biota-sediment accumulation regression
COPC – contaminant of potential concern
C_{tiss} – tissue concentration
C_{sed} – sediment concentrations
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
NA – not analyzed
OC – organic carbon
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)

8.1.3 Effects Assessment

This section presents the selected TRVs used to characterize effects for wildlife COPC-receptor pairs and the uncertainties associated with these selected values. Two types of TRVs were evaluated: dietary-dose TRVs (expressed as mg/kg bw/day) and bird egg TRVs (expressed as mg/kg ww in bird egg tissues). Dietary-dose TRVs were based on LOAELs and NOAELs derived from the toxicological literature using Equation 8-5. Bird egg TRV thresholds were based on LOAELs and NOAELs derived from the toxicological literature. Dietary-dose and bird egg TRVs were used to derive receptor-specific TTCs and TSCs.

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 8.1.2) in the risk characterization (Section 8.1.4).

Per EPA's Problem Formulation (Attachment 2), LOAELs were used to assess effects to all receptors evaluated at the population level,[115] (i.e., all the wildlife receptors 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 was at the organism rather than population level. This variation applied only to bald eagles. Per EPA's Problem Formulation (Attachment 2), NOAELs were used to assess COPC effects on bald eagles.

8.1.3.1 Dietary-Dose TRVs

The following subsection presents the selected dietary-dose TRVs and the TTCs and TSCs that were back-calculated based on the selected TRVs.

8.1.3.1.1 Selected TRVs

EPA provided bird and mammal TRVs for the BERA (EPA 2008f), which were adopted for assessing risks to wildlife. Wildlife TRVs selected by EPA were based on the following hierarchy:

When available, NOAEL and LOAEL TRVs were based on EPA's Ecological Soil Screening Level (Eco SSL) documents.

If no Eco SSLs were available, NOAEL and LOAEL TRVs were based on calculated dose values derived from the toxicological literature.

When TRVs were derived from the toxicological literature, EPA relied on LWG's comprehensive TRV review presented in Attachment 14 or on the TRVs reported in Sample

[115] There is high uncertainty associated with using LOAELs to assess effects to populations, as LOAELs are based on organism-level effects. The magnitude and frequency of LOAEL exceedances as well as the endpoints used to derive the LOAEL were evaluated to determine whether the potential for population-level effects may exist. See Section 8.1.4 for further discussion of how LOAELs were used in risk characterization.

et al. (1996). Attachment 16 presents the details, sources, and uncertainties associated with the selected TRVs. Attachment 14 also presents the results of a parallel effort conducted by the LWG to derive TRVs based on available toxicological studies.

Generally, the EPA TRVs used in this BERA are similar (i.e., within the same order of magnitude) to the TRVs recommended by LWG. This is because, in general, the same types of toxicological studies were reviewed in all three sources (i.e., Eco SSLs, Sample et al. (1996), and LWG's review); however, 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 were not appropriate as LOAEL TRVs because Eco SSLs are based on NOAELs and are applicable only in screening-level assessments to ensure that all potential chemicals contributing to risk are identified early in the process. Therefore, the NOAELs used to derive Eco SSLs were adopted for use as NOAEL TRVs for this BERA. 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.

There is uncertainty associated with the use of the toxicological data presented in the Eco SSLs to derive TRVs for this BERA because Eco SSLs 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 most relevant pathway (dietary) that is being evaluated. Drinking water ingestion involves a method of uptake and absorption that is different than the selected dietary pathway for wildlife receptors being assessed. 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 8-11 and 8-12, respectively. The key uncertainties associated with each of the selected TRVs are also presented in the tables.

Table 8-11. Bird Dietary-Dose TRVs

COPC	TRV (mg/kg bw/day) NOAEL	TRV (mg/kg bw/day) LOAEL	Source	Key Uncertainties
Aluminum	157	NA	Carriere et al. (1986)	TRVs are based on ionic form of aluminum, which is not directly comparable to the form present in the environment.
Arsenic	2.24[a]	4.5[b]	Eco SSL (EPA 2005b)	Lowest acceptable toxicological study reviewed (Attachment 14) presents toxicity at an order of magnitude higher than TRVs based on Eco SSLs.
Cadmium	1.47[a]	6.34[b]	Eco SSL (EPA 2005c)	
Chromium	2.66[a]	15.6[b]	Eco SSL (EPA 2005d)	
Copper	4.05[a]	12.1[c]	Eco SSL (EPA 2007a)	
Lead	1.63[a]	3.26[c]	Eco SSL (EPA 2005e)	Lowest acceptable toxicological study reviewed (Attachment 14) presents toxicity at an order of magnitude higher than TRVs based on Eco SSLs.
Mercury	0.0064	0.064	Heinz (1975, 1979)	NOAEL was extrapolated from the LOAEL using a factor of 10; toxicity studies do not consistently report adverse effects at the selected LOAEL; no effects on reproduction were reported by Heinz (1974, 1976) in birds fed the same dose level as the selected LOAEL.
Selenium	0.29[a]	0.579[c]	Eco SSL (EPA 2007c)	
Thallium	0.48	24	Hudson et al. (1984)	Limited number of toxicological studies available; NOAEL was extrapolated from the LOAEL using a factor of 50.
Zinc	66.1[a]	171[b]	Eco SSL (EPA 2007d)	
Tributyltin	6.8	16.9	Schlatterer et al. (1993)	Limited number of toxicological studies available.
Benzo(a)pyrene	0.28	1.4	Hough et al. (1993)	Limited number of toxicological studies available; TRV were based on weekly intramuscular injection; NOAEL was extrapolated from the LOAEL using a factor of 5.

Table 8-11. Bird Dietary-Dose TRVs

COPC	TRV (mg/kg bw/day) NOAEL	TRV (mg/kg bw/day) LOAEL	Source	Key Uncertainties
LPAHs	40	NA	Patton and Dieter (1980)	NOAEL was based on exposure to petroleum hydrocarbon mixture that contained some individual PAHs not included in the LPAH total; limited toxicological data are available.
HPAHs	40	NA	Patton and Dieter (1980)	NOAEL was based on exposure to petroleum hydrocarbon mixture that contained some individual PAHs not included in the HPAH total; limited toxicological data are available.
Total PAHs	40	NA	Patton and Dieter (1980)	NOAEL was based on exposure to petroleum hydrocarbon mixture that contained some individual PAHs not included as part of total PAHs; limited toxicological data are available.
BEHP	1.1	11	Peakall (1974)	LOAEL was extrapolated from the NOAEL using a factor of 10 based on SREL (1999), as cited by EPA (2008f); lowest acceptable toxicological study reviewed (Attachment 14) presents toxicity at an order of magnitude higher than selected LOAEL (extrapolated from NOAEL).
Di-n-butyl phthalate	0.11	1.1	Peakall (1974)	Limited number of toxicological studies available; NOAEL was extrapolated from the LOAEL using a factor of 10.
Total PCBs	0.29	0.58	Britton and Huston (1973)	TRVs are based on chickens, which have high sensitivity to PCB toxicity compared with other species tested.
PCB TEQ	1.4×10^{-5}	1.4×10^{-4}	Nosek et al. (1992)	TRVs are based on weekly intraperitoneal injection exposure to 2,3,7,8-TCDD.
Dioxin/furan TEQ	1.4×10^{-5}	1.4×10^{-4}	Nosek et al. (1992)	TRVs are based on weekly intraperitoneal injection exposure to 2,3,7,8-TCDD.
Total TEQ	1.4×10^{-5}	1.4×10^{-4}	Nosek et al. (1992)	TRVs are based on weekly intraperitoneal injection exposure to 2,3,7,8-TCDD.
Total DDx	0.227[a]	2.27[c]	Eco SSL (EPA 2007b)	Low uncertainty; Eco SSL similar to literature-based TRVs (Attachment 14)
Sum DDE	0.032	0.32	Mendenhall et al. (1983)	Per EPA (2008f), NOAEL was extrapolated from the LOAEL using a factor of 10; literature-based NOAEL (Attachment 14) is an order of magnitude higher than NOAEL that was extrapolated from the LOAEL.

Table 8-11. Bird Dietary-Dose TRVs

COPC	TRV (mg/kg bw/day) NOAEL	TRV (mg/kg bw/day) LOAEL	Source	Key Uncertainties
Aldrin	0.008	0.04	DeWitt (1956)	Limited number of toxicological studies 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 used to calculate the Eco SSL.
[c] LOAEL was derived from on the same study as the NOAEL, which was used as the basis for the Eco SSL.

BEHP – bis(2-ethylhexyl) phthalate
bw – body weight
COPC – chemical 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
LPAH – low-molecular-weight polycyclic aromatic hydrocarbon
NA – not available (no TRV selected)
NOAEL – no-observed-adverse-effect level
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
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
UF – uncertainty factor

Table 8-12. 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 based on ionic form of aluminum, which is not directly comparable to the form present in the environment.
Antimony	0.059[a]	0.59[b]	Eco SSL (EPA 2005a)	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 2005e)	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 study in which mink were fed field-collected fish (making up 40% of the prepared diet) that may have contained other, uncharacterized chemicals.
Selenium	0.143[a]	0.215[b]	Eco SSL (EPA 2007c)	
Benzo(a)pyrene	2.0[c]	10	MacKenzie and Angevine (1981)	
HPAHs	0.615[a]	3.07[b]	Eco SSL (EPA 2007f)	TRVs are based on exposure only to benzo(a)pyrene.
Total PCBs	0.0074[c]	0.037	Restum et al. (1998)	TRVs are based on study in which mink were fed field-collected fish that contained other chemicals that were analyzed and detected (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)	NOAEL was extrapolated from the LOAEL using a factor of 10.
Dioxin/furan TEQ	2.2×10^{-7}	2.2×10^{-6}	Tillitt et al. (1996)	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)	NOAEL was extrapolated from the LOAEL using a factor of 10.
Total DDx	0.147[a]	0.735[b]	Eco SSL (EPA 2007b)	Lowest acceptable toxicological study reviewed (Attachment 14) presents toxicity level at an order of magnitude higher than 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.

BEHP – bis(2-ethylhexyl) phthalate
COPC – chemical of potential concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
Eco SSL – ecological soil screening level
HPAH – high-molecular-weight polycyclic aromatic hydrocarbon
LOAEL – lowest-observed-adverse-effect level
LPAH – low-molecular-weight polycyclic aromatic hydrocarbon
NA – not available (no TRV selected)
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)
TRV – toxicity reference value
UF – uncertainty factor

8.1.3.1.2 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, biota [prey] ingestion rate, incidental sediment ingestion rate. This method of back-calculation was used so that TTCs and TSCs could be compared directly to empirical concentrations of COPCs in prey tissues or sediment concentrations, respectively (without the need to convert the empirical data into dose units), on a sample-by-sample basis. Equations 8-3 and 8-4 were used to develop receptor-specific TSCs and TTCs, respectively. Tables 8-13 through 8-16 present the receptor-specific TTCs and TSCs derived for all bird and mammal COPC-receptor pairs.

Table 8-13. Calculated TTCs for Bird COPC-Receptor Pairs

		TTC (Prey Tissue)							
		Spotted Sandpiper		Hooded Merganser		Bald Eagle		Osprey	
COPC	Unit (ww)	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 HPAHs	µg/kg	34,200	NA	NA	NA	NA	NA	NA	NA
Total LPAHs	µg/kg	34,200	NA	NA	NA	NA	NA	NA	NA
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

Table 8-13. Calculated TTCs for Bird COPC-Receptor Pairs

COPC	Unit (ww)	TTC (Prey Tissue)							
		Spotted Sandpiper		Hooded Merganser		Bald Eagle		Osprey	
		NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL
PCBs									
Total PCBs	µg/kg	248	496	783	1570	2420	4,830	1,380	2,760
PCB TEQ	ng/kg	12	120	37.8	378	117	1,170	66.5	665
Dioxins/Furans									
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 was available for the derivation of an aluminum LOAEL TTC.

BEHP – bis(2-ethylhexyl) phthalate
COPC – chemical of potential concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
HPAH – high-molecular-weight polycyclic aromatic hydrocarbon
LOAEL – lowest-observed-adverse-effect level
LPAH – low-molecular-weight polycyclic aromatic hydrocarbon
NA – not applicable (not a COPC-receptor 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 8-14. Calculated TSC for Bird COPC-Receptor 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 HPAHs	µg/kg	1,250,000	NA	NA	NA	NA	NA	NA	NA
Total LPAHs	µg/kg	1,250,000	NA	NA	NA	NA	NA	NA	NA
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

Table 8-14. Calculated TSC for Bird COPC-Receptor Pairs

COPC	Unit (dw)	TSC							
		Spotted Sandpiper		Hooded Merganser		Bald Eagle		Osprey	
		NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL	NOAEL	LOAEL
Dibutyl phthalate	µg/kg	3,450	34,500	54,000	540,000	NA	NA	NA	NA
PCBs									
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									
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 – chemical 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
LPAH – low-molecular-weight polycyclic aromatic hydrocarbon
NA – not applicable (not a COPC-receptor 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 8-15. Calculated TTCs for Mammal COPC-Receptor Pairs

COPC	Unit (ww)	TTC (Prey Tissue)			
		Mink		River Otter	
		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
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					
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

BEHP – bis(2-ethylhexyl) phthalate
COPC – chemical of potential concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
HPAH – high-molecular-weight polycyclic aromatic hydrocarbon
LOAEL – lowest-observed-adverse-effect level
LPAH – low-molecular-weight polycyclic aromatic hydrocarbon
NA – not applicable (not a COPC-receptor 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 8-16. Calculated TSCs for Mammal COPC-Receptor 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
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					
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

BEHP – bis(2-ethylhexyl) phthalate
COPC – chemical 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
LPAH – low-molecular-weight polycyclic aromatic hydrocarbon
NA – not applicable (not a COPC-receptor 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

8.1.3.2 Bird Egg TRVs

Section 8.1.3.2.1 presents the selected bird egg TRVs; Section 8.1.3.2.2 presents the TTCs that were back-calculated based on the selected TRVs.

8.1.3.2.1 Selected TRVs

Per EPA (2008j), bird egg TRVs were based on Willamette-specific species field data, when data were available. Field-based TRVs were derived based on detected chemical concentrations in eggs associated with adverse effects (LOAEL) or no adverse effects (NOAEL). Attachment 16 presents 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 presented in Table 8-17. The key uncertainties associated with each of the selected TRVs are also noted.

Table 8-17. Bird Egg Tissue-Residue TRVs

	TRV (mg/kg bw/day)			
COPC	**NOAEL**	**LOAEL**	**Source**	**Key Uncertainty**
Metals				
Mercury	0.5	0.74	NOAEL – Wiemeyer et al. (1984); LOAEL – Heinz and Hoffman (2003)	NOAEL is based on national bald eagle field data associated with productivity; LOAEL is based on lowest reproductive toxicity in the laboratory.
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.
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)	
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.

bw – body weight
COPC – chemical 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 (Section 8.1.3.1), bird egg TRVs were based primarily on field-collected data. Per EPA (2008j), bird egg TRVs were based on thresholds reported in field studies, when data were available. Using field-collected data allows for the selection of toxicological data based on Willamette-specific receptors (i.e., osprey and bald eagles) to ensure that these receptors are protected. However, there are high uncertainties 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. There is likely a large range of concentrations for which adverse effects (e.g., productivity) would not be observed within a population, and it is unknown where a NOAEL based on a field study falls within that range of concentrations.

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.

8.1.3.2.2 Back-Calculated TTCs

Once bird egg TRVs were selected, TTCs were calculated using BMFs (Table 8-5) and Equation 8-7. Table 8-18 presents the TTCs derived for osprey and bald eagles based on bird egg TRVs.

Table 8-18. Calculated TTCs for Piscivorous Bird Egg COPCs

	Egg Tissue TRV (mg/kg ww)			TTC (mg/kg ww)[a]	
COPC	**NOAEL**	**LOAEL**	**BMF**	**NOAEL**	**LOAEL**
Metals					
Mercury	0.5	0.74	3	0.167	0.247
PCBs					
Total PCBs	3.0	4.5	8.4	0.357	0.536
PCB TEQ	2.3×10^{-6}	3.2×10^{-5}	10	2.3×10^{-7}	3.2×10^{-6}
Dioxins/Furans					
Dioxin/furan TEQ	2.3×10^{-6}	3.2×10^{-5}	10	2.3×10^{-7}	3.2×10^{-6}
Total TEQ	2.3×10^{-6}	3.2×10^{-5}	10	2.3×10^{-7}	3.2×10^{-6}
Pesticides					
4,4′-DDE	1.3	3.5	79	0.0165	0.0443

[a] TTCs are the same for both piscivorous bird receptors (i.e., osprey and bald eagle) because the same BMFs were used for both receptors.

BMF – biomagnification factor
COPC – chemical 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
TSC – threshold sediment concentration
TTC – threshold tissue concentration
ww – wet weight

8.1.4 Risk Characterization and Uncertainty Analysis

The following section presents the risk characterization process and results for birds and mammals based on the dietary-dose and bird egg exposure. Section 8.1.4.1 presents the overall approach used to characterize risk to wildlife receptors. Section 8.1.4.2 presents the risk characterization results, uncertainty evaluation, and COCs for each wildlife receptor. Section 8.1.4.3 presents a summary of risk conclusions and dietary COCs for all bird and mammal receptors. A dietary-dose assessment was conducted for belted kingfisher, as part of an uncertainty evaluation to characterize risks to piscivorous birds. The results of this evaluation, including the identification of COCs, are presented in Section 8.1.4.4. Dietary COCs were further evaluated in the wildlife risk conclusions section (Section 8.2).

8.1.4.1 Risk Characterization Process

A deterministic risk characterization was conducted to characterize risks to wildlife receptors. No additional risk modeling was conducted because the deterministic approach following the methods of EPA's Problem Formulation (Attachment 2) was deemed by EPA to be sufficient to identify COCs.[116] A risk characterization of wildlife COPC-receptor pairs based on the dietary-dose and bird egg approach was conducted using an HQ approach that integrated exposure and effects data. Multiple HQs were derived using various assumptions in order to implement the methods presented in EPA's Problem Formulation (Attachment 2).

As described in Section 8.1.2.2, HQs were calculated using a multi-step process. 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 are presented in Attachment 17 and were used to narrow the list of COPCs for evaluation in the third step. COPCs with HQs > 1.0 in the third step were retained as COCs. The third step was used to define COCs because it represents the most reasonable assumptions appropriate for risk characterization.

The multi-step process is presented below:

Step 1 – Evaluation of Individual Samples – In the first step, HQs were evaluated on a sample-by-sample basis. HQs were derived for individual prey tissue and sediments using Equation 8-1.

[116] Probabilistic risk characterization was discussed with EPA at the beginning of the process.

Step 2 – Evaluation of Relevant Exposure Areas – In the next step, HQs were calculated within relevant exposure areas. For those COPC-receptor pairs that resulted in HQs > 1.0 in Step 1, prey species and sediment HQs were derived within relevant exposure areas using Equation 8-1 in comparison to tissue and sediment UCLs. If insufficient data were available to derive a UCL (i.e., fewer than six detected concentrations were available), HQs were based on maximum prey species or sediment chemical concentration. Receptor-specific EPCs for individual exposure areas (Maps 8-1 through 8-3) were calculated based on the assumptions presented in Table 8-4.

Step 3 – Evaluation of Multiple Prey Within Relevant Exposure Areas – In order to account for the ingestion of multiple prey species, HQs were calculated by assigning diet fractions to individual prey species within an exposure area to derive total HQs using Equation 8-2. Prey portions were based on the diets reported in regional studies (Attachment 16). If no data were available for a given prey species, a surrogate species was used.[117]

COCs were identified based on those COPCs that resulted in HQs > 1.0 based on Step 3. Once COCs were determined, an evaluation was conducted for all COCs to discuss the quantitative risk results (i.e., magnitude, spatial distribution, and frequency of HQs), underlying uncertainties of exposure and effects data, agreement of dietary-dose and bird egg HQs (where applicable), and comparison of background concentrations. This qualitative assessment was conducted to focus the analysis on those COCs that are expected to be the primary risk contributors for the Study Area. The evaluation of COCs is discussed in the wildlife risk conclusions section (Section 8.2).

Figure 8-3 presents the general risk characterization process used to evaluate COPCs and identify wildlife COCs.

[117] Prey portions (including a surrogate prey portions) are presented as part of the risk characterization results.

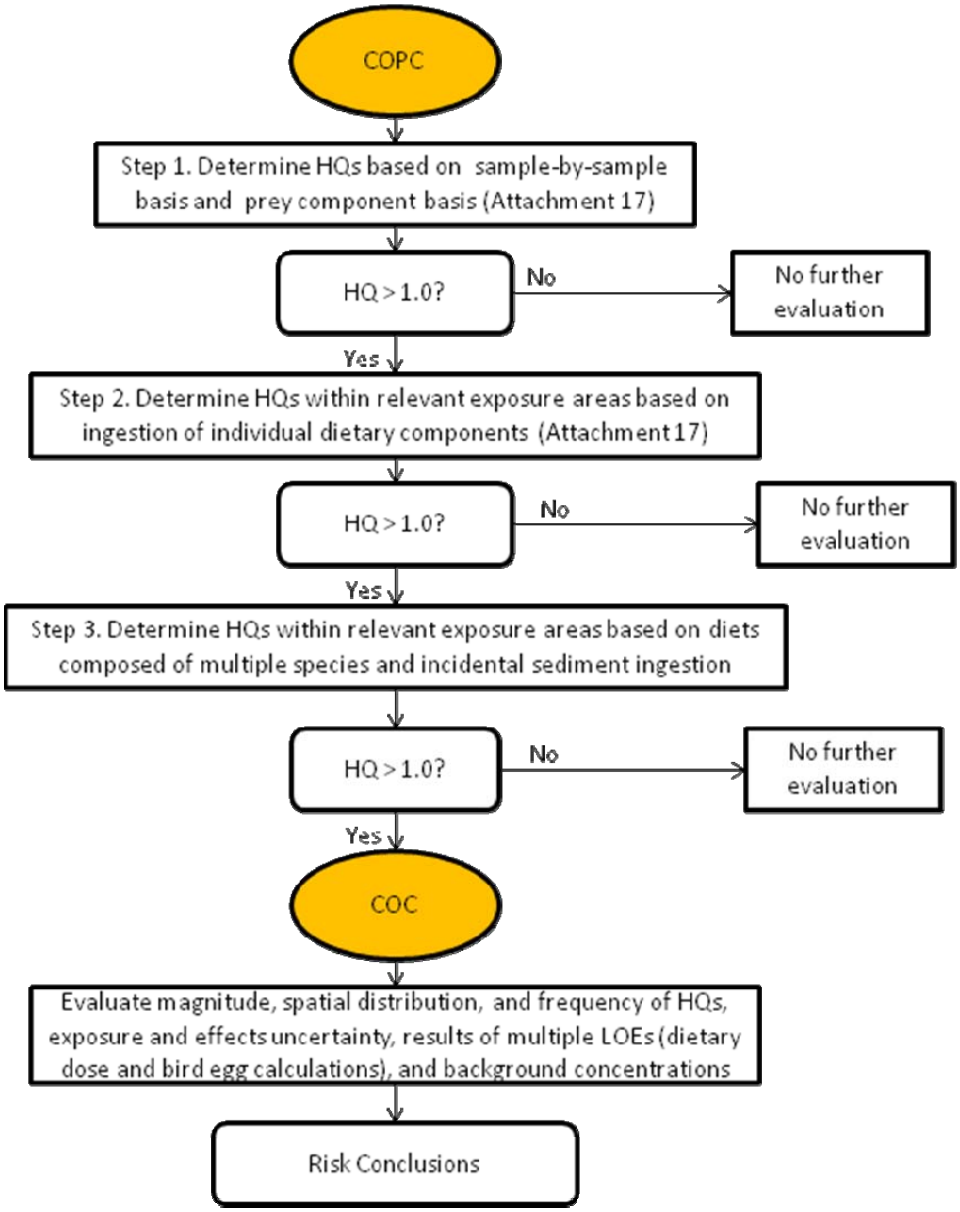

Figure 8-3. General Wildlife Risk Characterization Process

Determination of COCs

COCs were those COPCs with HQs > 1.0 based on ecologically relevant exposure scales (as presented in Table 8-3), regardless of the uncertainties associated with the exposure and effects data or assumptions. COPCs were not identified as COCs based on individual sample or individual prey component HQs because risks suggested by individual samples are not ecologically relevant.

The uncertainties associated with risk estimates for individual COCs, the spatial distribution of COC exceedances, the magnitude of exceedance, and the type of effect threshold all play a role in identifying whether chemicals pose a population-level risk. The likelihood that a COC with a limited spatial distribution and a low HQ poses significant risks to populations is low. COCs with a broader distribution and higher magnitude of exceedances generally have greater potential for posing population-level risks. The type of effect threshold endpoint (i.e., survival, growth or reproduction) affects the interpretation of HQs simply because effects on the survival of individuals have different implications for populations than do effects on the growth of individuals. Effects on the reproductive success of individuals have different implications for populations than do effects on either survival or growth.

Even for a particular type of endpoint (e.g., reduced survival), a particular HQ (e.g., HQ = 5) has different implications for each COC-receptor pair, because the exposure-response relationship of once COC-receptor pair differs from that of other pairs. Short of developing structured population models and incorporating the effects of COC-receptor-specific exposure-response data, the demographic (i.e., population-level) consequences of any particular HQ simply cannot be quantified.

Populations are resilient to environmental stressors (Barnthouse et al. 2009). By corollary, it is unreasonable to assume that survival, growth, or reproduction HQs > 1.0 will necessarily be detrimental to the populations whose individuals are potentially affected.

8.1.4.2 Risk Characterization Results and Uncertainty Evaluation

The risk characterization process is presented in Section 8.1.4.1. The HQ results from the first two steps are presented in Attachment 17 and were used to narrow the list of COPCs for evaluation in the third step. Table 8-19 presents the COPC-receptor pairs with HQs > 1.0 based on individual prey (Step 2) that were retained for further evaluation. The following subsections present the HQs for each bird and mammal receptor based on multiple prey within relevant exposure areas. COPCs with HQs > 1.0 based on multiple prey were retained as COCs.

Table 8-19. COPCs with HQs > 1.0 Based on Individual Prey Species

			Bald Eagle		Osprey			
COPC	**Spotted Sandpiper**	**Hooded Merganser**	**Dietary Dose**	**Bird Egg**	**Dietary Dose**	**Bird Egg**	**Mink**	**River Otter**
Metals								
Antimony							X	
Copper	X							
Lead		X			X		X	X
Mercury			X	X	X	X	X	
PAHs								
Benzo(a)pyrene	X							

Table 8-19. COPCs with HQs > 1.0 Based on Individual Prey Species

			Bald Eagle		Osprey			
COPC	**Spotted Sandpiper**	**Hooded Merganser**	**Dietary Dose**	**Bird Egg**	**Dietary Dose**	**Bird Egg**	**Mink**	**River Otter**
Phthalates								
BEHP		X			X			
Dibutyl phthalate	X							
PCBs								
Total PCBs	X	X	X	X	X	X	X	X
PCB TEQ	X			X		X	X	X
Dioxins/Furans								
Dioxin/furan TEQ	X			X		X	X	X
Total TEQ	X			X		X	X	X
Pesticides								
Aldrin	X							
4,4′-DDE				X		X		
Sum DDE	X		X					
Total DDx	X							

BEHP – bis(2-ethylhexyl) phthalate
COPC – chemical 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)

8.1.4.2.1 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. Ten COPCs had HQs > 1.0 based on individual prey components (Table 8-19): copper, benzo(a)pyrene, dibutyl phthalate, total PCBs, PCB TEQ, dioxin/furan TEQ, total TEQ, aldrin, sum DDE, and total DDx. HQs were calculated for these COPCs across multiple prey and incidental sediment ingestion using LOAEL-based TTCs and TSCs.

A home range of 2 miles was assumed for spotted sandpiper based on the literature, and the spotted sandpiper diet was represented by clams,[118] lab worms, and incidental ingestion of sediment. Individual prey portions were not assigned to each of the representative prey for spotted sandpiper because both clam and worms are meant to be representative of benthic invertebrates prey that sandpiper may ingest; therefore, HQs were based on: 1) ingestion of only clams and incidental sediment, and 2) ingestion of only lab worms and incidental sediment. The HQ results across each 2-mile exposure area based on UCLs are presented in Table 8-20.

Table 8-20. Spotted Sandpiper LOAEL HQs Within 2-Mile Beach Exposure Areas

Exposure Areas (beach areas within 2 miles)	Approximate RM	Total HQ	
		Based on Clam Diet	Based on Lab Worm Diet
Copper			
B1-B6	RM 1.9 – RM 3.9	1.1	0.34
B7-13	RM 4.0 – RM 6.0	1.3	0.67
B14-B24	RM 7.0 – RM 9.0	1.1	0.41
B25-28	RM 9.0 – RM 10.0	0.99	0.31
Benzo(a)pyrene			
B1-B6	RM 1.9 – RM 3.9.0	0.013	0.067
B7-13	RM 4.0 – RM 6.0	0.044	1.6
B14-B24	RM 7.0 – RM 9.0	0.059	0.13
B25-28	RM 9.0 – RM 10.0	0.0073	0.034
Dibutyl phthalate			
B1-B6	RM 1.9 – RM 3.9	0.0089	0.0089
B7-13	RM 4.0 – RM 6.0	0.024	0.15
B14-B24	RM 7.0 – RM 9.0	1.4	0.51
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
B7-13	RM 4.0 – RM 6.0	0.24	1.7
B14-B24	RM 7.0 – RM 9.0	2.2	12
B25-28	RM 9.0 – RM 10.0	0.95	7.7

[118] Clam data were represented by field clams; when no field clam data were available, lab clams were used.

Table 8-20. Spotted Sandpiper LOAEL HQs Within 2-Mile Beach Exposure Areas

Exposure Areas (beach areas within 2 miles)	Approximate RM	Total HQ Based on Clam Diet	Total HQ Based on Lab Worm Diet
Dioxin/Furan TEQ			
B1-B6	RM 1.9 – RM 3.9	0.018	0.13
B7-13	RM 4.0 – RM 6.0	0.061	0.96
B14-B24	RM 7.0 – RM 9.0	0.20	17
B25-28	RM 9.0 – RM 10.0	0.029	0.12
PCB TEQ			
B1-B6	RM 1.9 – RM 3.9	0.38	**10**
B7-13	RM 4.0 – RM 6.0	0.073[a]	0.58[a]
B14-B24	RM 7.0 – RM 9.0	0.56	**11**
B25-28	RM 9.0 – RM 10.0	0.15	0.88
Total TEQ			
B1-B6	RM 1.9 – RM 3.9	0.40	**11**
B7-13	RM 4.0 – RM 6.0	0.12[a]	**1.3[a]**
B14-B24	RM 7.0 – RM 9.0	0.40	**20**
B25-28	RM 9.0 – RM 10.0	0.18	1.0
Aldrin			
B1-B6	RM 1.9 – RM 3.9	0.0080	0.037
B7-13	RM 4.0 – RM 6.0	0.0088	0.056
B14-B24	RM 7.0 – RM 9.0	0.039	**1.7**
B25-28	RM 9.0 – RM 10.0	0.011	0.044
Sum DDE			
B1-B6	RM 1.9 – RM 3.9	0.054	0.20
B7-13	RM 4.0 – RM 6.0	0.073	0.44
B14-B24	RM 7.0 – RM 9.0	0.17	**1.3**
B25-28	RM 9.0 – RM 10.0	0.047	0.15

Table 8-20. Spotted Sandpiper LOAEL HQs Within 2-Mile Beach Exposure Areas

Exposure Areas (beach areas within 2 miles)	Approximate RM	Total HQ	
		Based on Clam Diet	Based on Lab Worm Diet
Total DDx			
B1-B6	RM 1.9 – RM 3.9	0.016	0.044
B7-13	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	0.037

[a] PCB TEQ and total TEQ were not available in sediment at exposure area B14-B24. Total HQ was based only on prey tissue.

COPC – chemical of potential concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
NA – not analyzed
Bold identifies HQs > 1.0.
PAH – polycyclic aromatic hydrocarbon
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)
TEQ – toxic equivalent
UCL – upper confidence limit on the mean

Eight COCs were identified for spotted sandpiper when HQs were > 1.0 within a 2-mile exposure area: copper, benzo(a)pyrene, dibutyl phthalate, total PCBs, total TEQ,[119] aldrin, sum DDE, and total DDx.

Several uncertainties are associated with the exposure and effects assumptions used to derive the risk estimates. The greatest uncertainty with the sandpiper exposure assumptions is whether the prey tissue concentrations used are representative of the sandpiper diet. 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. Prey tissue concentrations may be over- or underestimated using clam and worm tissue concentrations. 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. The use of the mechanistic model to estimate clam and worm tissue concentrations when no data are available is also uncertain. Uncertainties associated with the effects assumptions are discussed in Section 8.1.3.

Table 8-21 presents a summary of the HQs and uncertainties for all sandpiper COCs.

[119] Dioxin/furan TEQ and PCB TEQ also had HQs > 1.0, but was not identified as a COC because total TEQ was identified as a COC.

Table 8-21. Summary of Spotted Sandpiper COCs

COC	Exposure Area(s) with HQ > 1.0	Prey Item(s) Resulting in HQ > 1.0	HQ(s)	TRV Uncertainty	Exposure Uncertainty
Copper	RM 1.9 – RM 3.9; RM 4.0 – RM 6.0; RM 7.0 – RM 9.0	Clam	1.1 – 1.3	Low uncertainty; TRV is supported by the toxicity literature (see Attachment 14).	Low uncertainty; prey tissue chemical concentration is based only on field-collected data.
Benzo(a) pyrene	RM 4.0 – RM 6.0	Worm	1.6	Based on weekly intramuscular injection exposure.	Prey tissue chemical concentration is represented by 28-day bioaccumulation testing of lab worms expressed as steady state; worm EPC is maximum concentration (all other tissue samples concentrations result in HQs < 1.0).
Dibutyl phthalate	RM 7.0 – RM 9.0	Clam	1.4	LOAEL was extrapolated from the NOAEL per EPA (2008h); there were few toxicity data.	Clam EPC is maximum concentration (all other tissue samples concentrations result in HQs < 1.0).
Total PCBs	All exposure areas[a]	Worm	1.7 – 12	Based on chicken study; chickens appear to have a higher sensitivity to PCBs than do other avian species in the laboratory.	Prey is represented by 28-day bioaccumulation testing of lab worms and may not represent field conditions; expressed as steady-state; includes predicted chemical concentrations.
	RM 7.0 – RM 9.0	Clam	2.2		Clam EPC includes one predicted chemical concentration; modeled concentrations may over- or underestimate risks.

Table 8-21. Summary of Spotted Sandpiper COCs

COC	Exposure Area(s) with HQ > 1.0	Prey Item(s) Resulting in HQ > 1.0	HQ(s)	TRV Uncertainty	Exposure Uncertainty
Total TEQ[b]	RM 1.9 – RM 3.9; RM 4.0 – RM 6.0; RM 7.0 – RM 9.0	Worm	11; 1.3; 20	Based on weekly injection exposure to 2,3,7,8-TCDD.	Prey is represented by 28-day bioaccumulation testing of lab worms and may not represent field conditions; expressed as steady-state; worm UCLs include predicted chemical concentrations; modeled concentrations may over- or underestimate risks.
Aldrin	RM 7.0 – RM 9.0	Worm	1.7	Few toxicity data (n = 2 toxicity studies).	Prey is represented by 28-day bioaccumulation testing of lab worms and may not represent field conditions; expressed as steady state.
Sum DDE	RM 7.0 – RM 9.0	Worm	1.3	Low uncertainty; TRV is based on the toxicity literature	Prey is represented by 28-day bioaccumulation testing tissue expressed as steady state; worms and may not represent field conditions; worm UCL includes one predicted chemical concentration; modeled concentrations may over- or underestimate risks.
Total DDx	RM 7.0 – RM 9.0	Worm	1.4	Low uncertainty; TRV supported by the toxicity literature (see Attachment 14).	

[a] The highest HQs were located in exposure areas between RM 1.9 and RM 3.8 and between RM 7.0 and RM 9.0.

[b] The dioxin/furan TEQ HQ was 17 in one exposure area (RM 7.0 to RM 9.0) based on worm prey. PCB TEQ HQs were 10 and 11 in the exposure area between RM 1.9 and RM 3.9 and RM 7.0 and RM 9.0, respectively, based on worm prey.

BSAR – biota-sediment accumulation regression
COC – chemical of concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
EPA – US Environmental Protection Agency
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
NOAEL – no-observed-adverse-effect level
PCB – polychlorinated biphenyl
RM – river mile
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
UCL – upper confidence limit on the mean

8.1.4.2.2 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 had HQs > 1.0 based on individual prey components: lead, BEHP, and total PCBs (Table 8-19). HQs were calculated for these COPCs across multiple prey and incidental sediment ingestion using LOAEL-based TTCs and TSCs.

HQs based on multiple prey and incidental sediment ingestion are presented in Table 8-22.

Table 8-22. 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
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. BEHP was not analyzed in peamouth tissue.

BEHP – bis(2-ethylhexyl) phthalate
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
PCB – polychlorinated biphenyl
RM – river mile
Bold identifies HQs > 1.0.

Total PCBs were identified as a COC for hooded merganser because HQs based on a multiple-prey diet were > 1.0 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 was not > 1.0 for lead and BEHP; therefore, these COPCs are not expected to pose unacceptable risks to hooded merganser and were not identified as COCs.

There are several uncertainties associated with the exposure assumptions used to derive the risk estimates. Uncertainty is associated with the total PCB HQs, as they were based on maximum concentrations in fish and invertebrate prey species within the exposure areas; insufficient data were available (because of small sample size) to derive UCL concentrations for prey species within these 1-mile exposure areas. The assumption that hooded mergansers forage from only within a 1-mile area year-round is highly 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 assumption of a site use factor of 1.0 may be overly conservative. Assuming a site use factor of 0.75 reduces the HQs by 25%; however, HQs are still > 1.0 in the three exposure areas (ranging from 1.1 to 2.9). Uncertainties associated with the effects assumptions are discussed in Section 8.1.3.

Table 8-23 presents a summary of the HQs and uncertainties for total PCBs, the only COC identified for hooded merganser.

Table 8-23. Summary of Hooded Merganser COCs

COC	Exposure Area(s) of HQ > 1.0	HQ Range	TRV Uncertainty	Exposure Uncertainty
Total PCBs	RM 1.5 – RM 2.5; RM 6.5 – RM 7.5; RM 10.5 – RM 11.8	1.5 – 3.8	Based on a chicken study; chickens appear to have a higher sensitivity to PCBs than do other avian species in the laboratory.	Conservative site use factor and foraging range used; maximum concentration in fish and invertebrate prey species used to estimate exposure.

COC – chemical of concern
HQ – hazard quotient
PCB – polychlorinated biphenyl
RM – river mile
TRV – toxicity reference value

8.1.4.2.3 Bald Eagle

Risks to bald eagle were evaluated using the dietary-dose and bird egg assessment. Eight COPCs were identified for bald eagle in the SLERA and refined screen for the dietary-dose and/or bird egg assessment. Three COPCs had dietary-dose HQs > 1.0 based on individual prey components (Table 8-19): mercury, total PCBs and sum DDE. Dietary-dose HQs were calculated for these COPCs across multiple prey and incidental sediment ingestion using NOAEL-based TTCs and TSCs. Six COPCs had bird egg HQs > 1.0 based on individual prey components (Table 8-19): mercury, total PCBs, PCB TEQ, dioxin/furan TEQ, total TEQ, and 4,4′-DDE. Bird egg HQs were calculated for these COPCs across multiple prey using NOAEL-based TTCs.

Dietary-dose HQs based on consumption of multiple prey species and incidental sediment ingestion are presented in Table 8-24. Bird egg HQs based on multiple prey species are presented in Table 8-25.

Table 8-24. 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

[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 > 1.0.

Table 8-25. Bald Eagle Bird Egg NOAEL HQs Within 1-mile Exposure Areas

	Total HQ					
Exposure Area	**Mercury[a]**	**Total PCBs[a]**	**PCB TEQ[b]**	**Dioxin/furan TEQ[b]**	**Total TEQ[b]**	**4,4′-DDE[a]**
RM 1.5 to RM 2.5	0.42	**26**	**240**	**59**	**280**	**10**
RM 2.5 to RM 3.5	0.42	**26**	**240**	**59**	**280**	**10**
RM 3.5 to RM 4.5	0.37	**26**	**240**	**59**	**280**	**9.8**
RM 4.5 to RM 5.5	0.47	**26**	**240**	**59**	**280**	**10**
RM 5.5 to RM 6.5	0.47	**26**	**240**	**59**	**280**	**10**
RM 6.5 to RM 7.5	0.52	**26**	**240**	**59**	**280**	**11**
RM 7.5 to RM 8.5	0.48	**26**	**240**	**59**	**280**	**9.7**
RM 8.5 to RM 9.5	0.48	**26**	**240**	**59**	**280**	**9.7**

Table 8-25. Bald Eagle Bird Egg NOAEL HQs Within 1-mile Exposure Areas

	Total HQ					
Exposure Area	**Mercury[a]**	**Total PCBs[a]**	**PCB TEQ[b]**	**Dioxin/furan TEQ[b]**	**Total TEQ[b]**	4,4′-**DDE[a]**
Swan Island Lagoon	0.41	**26**	**240**	**59**	**280**	**9.5**
RM 9.5 to RM 10.5	0.37	**26**	**240**	**59**	**280**	**9.8**
RM 10.5 to RM 11.8	0.37	**26**	**240**	**59**	**280**	**9.8**

[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%.

[b] Total HQ is based on 100% ingestion of carp (TEQ data were not available for any other prey species). Total HQ was derived using the site-wide 95 UCL carp concentrations.

DDE – dichlorodiphenyldichloroethylene

PCB – polychlorinated biphenyl

RM – river mile

TEQ – toxic equivalent

UCL – upper confidence limit on the mean

Bold identifies HQs > 1.0.

Mercury, total PCBs, total TEQ,[120] and 4,4′-DDE were all identified as COCs for bald eagle because dietary-dose and/or bird egg HQs based on the consumption of multiple prey were > 1.0 in all exposure areas. It should be noted that the two LOEs for assessing dietary risk (dietary-dose and bird egg approach) were not consistent. Mercury HQs were < 1.0 based on the bird egg approach but were > 1.0 based on the dietary-dose approach. Conversely, sum DDE HQs were < 1.0 based on the dietary-dose approach but 4,4′-DDE HQs were > 1.0 using the bird egg approach. TEQ sum HQs (for PCB TEQ, dioxin/furan TEQ, and total TEQ) were all < 1.0 based on individual prey components using the dietary-dose approach (Attachment 17) but were > 1.0 based on multiple prey using the bird egg approach. The differences in the results of these two LOEs are further discussed in Section 8.2.

There are several uncertainties associated with the exposure and effects assumptions used to derive the risk estimates using both the dietary-dose and bird egg approaches. Uncertainties associated with the effects thresholds are discussed in Section 8.1.3. Exposure assumption uncertainties include the following:

- Bald eagles were assumed to ingest 100% fish; however, eagles' diets vary, and they are known to feed on aquatic and terrestrial birds and mammals and carrion and will scavenge through garbage as well. Mercury, PCBs, dioxins, and DDE tend to bioaccumulate in tissues of higher-trophic-level organisms. Therefore, exposure estimates may underestimate risks to bald eagles if bird and mammal prey are

[120] Dioxin/furan TEQ and PCB TEQ also had HQs > 1.0 but were not identified as COCs because total TEQ was identified as a COC.

aquatic and bioaccumulate contaminants in the aquatic food web. Or, more likely, exposure estimates may overestimate risks if a proportion of their diet is terrestrial prey and/or their prey have not accumulated aquatic contaminants.

- 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), increasing the exposure scale area is unlikely to change the risk estimates for bald eagle.

- The assumption that bald eagles forage only within the Study Area year-round is highly conservative. Because bald eagles may forage in both aquatic and terrestrial environments near the Study Area, the assumption of a site use factor of 1.0 may be overly conservative. Because bald eagles forage in the Study Area for only part of the year and forage in terrestrial environment, a site use factor of 0.5 may be more a more realistic assumption. Assuming a site use factor of 0.5 reduces the HQs by 50%; however all HQs are still > 1.0.

- There are limited data available for the assessment of dioxins/furans and dioxin-like PCBs. TEQ data were available only for carp; thus, the HQs for bald eagle are based on the UCL for carp. Dioxins/furans and dioxin-like PCB congener concentrations in other fish prey may be lower or higher. The variability of bird TEFs among species and relevance of TEFs based on EROD induction to evaluate individual and population risks from dioxins/furans and dioxin-like PCBs is uncertain.

The greatest uncertainties associated with the exposure and effects assumptions are specific to the bird egg approach. Per EPA (2008j), BMFs were used to calculate risk estimates; however, no significant relationship between co-located fish prey and bird egg concentrations (i.e., no BMR) could be found for any of the COPCs, except total TEQs (Section 8.1.2.3.2). Using the total TEQ BMR reduced estimated total TEQ in eggs by about 48% relative to using the geometric mean BMF of 10.

There is additional uncertainty with the bird egg approach because TRVs used to derive risk estimates are based on NOAELs derived from field studies. Avian reproductive effects have been associated with exposure to PCBs (including dioxin-like PCBs), dioxins/furans, and DDTs, and it is not possible in field studies to associate effects with exposure to a single contaminant because birds in the field are typically exposed to mixtures of these three contaminant groups. Non-contaminant stressors may also contribute to declines in reproduction. NOAELs derived from field studies may be useful to rule out the possibility of risk (if NOAELs were not exceeded); however, the exceedance of field-based NOAELs does not necessarily indicate risk to populations or individuals. While field-based LOAELs have high uncertainty (due to the fact that observed effects cannot be attributed to a single contaminant or factor), LOAELs may represent more appropriate thresholds for

determining the possibility of risk to receptors, at least in the case when a LOAEL is exceeded.[121] Bald eagle HQs across multiple prey were recalculated using field-based LOAELs. These HQs are presented in Table 8-26. Although the magnitude of these HQs is lower, these HQs still indicate risks to bald eagle. Bald eagle dietary-dose HQs were also recalculated using laboratory-based LOAELs (Table 8-27). Although the magnitude of the total PCB HQ is lower, the total PCB LOAEL HQs still indicate risks to bald eagle. The dietary-dose HQ for mercury is < 1.0 based on the LOAEL (whereas the HQ is > 1.0 based on the NOAEL).

Table 8-26. Comparison of Bald Eagle Bird Egg NOAEL HQs and LOAEL HQs

COPC	NOAEL HQ	LOAEL HQ
PCBs		
Total PCBs	**26**	**17 – 18**
PCB TEQ	**240**	**17**
Dioxins/Furans		
Dioxin/furan TEQ	**59**	**4.2**
Total TEQ	**280**	**20**
Pesticides		
4,4′-DDE	**9.5 – 11**	**3.6 – 4.1**

COPC – chemical of potential concern
DDE – dichlorodiphenyldichloroethylene
LOAEL – lowest-observed-adverse-effect level
NOAEL – no-observed-adverse-effect level
PCB – polychlorinated biphenyl
TEQ – toxic equivalent

Table 8-27. Comparison of Bald Eagle Dietary-Dose NOAEL HQs and LOAEL HQs

COPC	NOAEL HQ	LOAEL HQ
Metals		
Mercury	**1.2 – 1.7**	0.12 – 0.17
PCBs		
Total PCBs	**3.8 – 3.9**	**1.9 – 2.0**
Pesticides		
Sum DDE	0.63 – 0.71	0.063 – 0.071

COPC – chemical of potential concern
DDE – dichlorodiphenyldichloroethylene
LOAEL – lowest-observed-adverse-effect level
NOAEL – no-observed-adverse-effect level
PCB – polychlorinated biphenyl

Table 8-28 presents a summary of the HQs and uncertainties for all bald eagle COCs.

[121] A field-based LOAEL is typically an unbounded LOAEL and hence could be lower than the observed value.

Table 8-28. Summary of Bald Eagle COCs

COC	LOE	Exposure Area(s) of HQ > 1.0	HQ(s)	TRV Uncertainty	Exposure Uncertainty
Mercury	Dietary dose	All exposure areas	1.2 to 1.7	Per EPA (2008f), NOAEL was extrapolated from the LOAEL using a factor of 10, although EPA Region 10 guidance (EPA 1997b) suggests the use of an uncertainty factor of 5 when extrapolating a chronic NOAEL from a chronic LOAEL; using an uncertainty factor of 5 would result in HQs < 1.0.	Diet includes ingestion of fish only; SUF of 1.0 was assumed.
Total PCBs	Dietary dose	All exposure areas	3.8 to 3.9	Based on a chicken study; chickens appear to have a higher sensitivity to PCBs than do other avian species in the laboratory.	Diet includes ingestion of fish only; SUF of 1.0 was assumed.
	Bird egg	All exposure areas	26	Based on field data in which no effect on bald eagle productivity was observed; exceedance of field-based no-effect TRV cannot conclusively indicate risks because of confounding factors in the field.	Assumed a SUF of 1.0 and ingestion of fish only; highly uncertain BMF was used.
Total TEQ[a]	Bird egg	All exposure areas	280	Based on field data in which no effect on osprey productivity was observed; exceedance of field-based no-effect TRV cannot conclusively indicate risks because of confounding factors in the field.	Prey data were only available for carp, which makes up < 50% of diet; variability of TEFs is uncertain; highly uncertain 2,3,7,8-TCDD BMF was used.
4,4′-DDE	Bird egg	All exposure areas	9.5 to 11	Based on field data in which no effect on bald eagle productivity was observed; exceedance of field-based no-effect TRV cannot conclusively indicate risks because of confounding factors in the field.	Assumed a SUF of 1.0 and ingestion of only fish; highly uncertain BMF was used.

[a] Dioxin/furan TEQ and PCB TEQ HQs were 59 and 240, respectively, in all exposure areas.

BMF – biomagnification factor
COC – chemical of concern
DDE – dichlorodiphenyldichloroethylene
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
NOAEL – no-observed-adverse-effect level
PCB – polychlorinated biphenyl
SUF – site use factor
TCDD – tetrachlorodibenzo-*p*-dioxin
TEF – toxic equivalency factor
TEQ – toxic equivalent
TRV – toxicity reference value
UF – uncertainty factor

8.1.4.2.4 Osprey

Risks to osprey were evaluated using the dietary-dose and bird egg assessment. Ten COPCs were identified for osprey in the SLERA and refined screen for the dietary-dose and/or bird egg assessment. Four COPCs had dietary-dose HQs > 1.0 based on individual prey components (Table 8-20): lead, mercury, BEHP, and total PCBs. Dietary-dose HQs were calculated for these COPCs across multiple prey and incidental sediment ingestion using LOAEL-based TTCs and TSCs. Six COPCs had bird egg HQs > 1.0 based on individual prey components (Table 8-20): mercury, total PCBs, PCB TEQ, dioxin/furan TEQ, total TEQ, and 4,4′-DDE. Bird egg HQs were calculated for these COPCs across multiple prey using LOAEL-based TTCs.

Dietary-dose HQs based on the consumption of multiple prey species and incidental sediment ingestion are presented in Table 8-29. Bird egg HQs based on multiple prey species are presented in Table 8-30.

Table 8-29. 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 > 1.0.

Table 8-30. Osprey Bird Egg LOAEL HQs Within 1-Mile Exposure Areas

	Total HQ					
Exposure Area	**Mercury[a]**	**Total PCBs[a]**	**PCB TEQ[b]**	**Dioxin/ Furan TEQ[b]**	**Total TEQ[b]**	4,4′-**DDE[a]**
RM 1.5 to RM 2.5	0.31	**4.7**	**22**	**4.0**	**25**	**3.4**
RM 2.5 to RM 3.5	0.35	**4.5**	**18**	**3.9**	**20**	**3.6**
RM 3.5 to RM 4.5	0.34	**4.7**	**17**	**4.1**	**20**	**3.7**
RM 4.5 to RM 5.5	0.39	**4.5**	**16**	**4.0**	**19**	**3.7**
RM 5.5 to RM 6.5	0.39	**4.5**	**16**	**4.1**	**19**	**3.7**
RM 6.5 to RM 7.5	0.44	**4.8**	**16**	**10**	**25**	**4.2**
RM 7.5 to RM 8.5	0.41	**4.7**	**16**	**4.5**	**19**	**3.5**
RM 8.5 to RM 9.5	0.42	**4.7**	**18**	**4.0**	**21**	**3.5**
Swan Island Lagoon	0.34	**4.8**	**16**	**4.0**	**19**	**3.4**
RM 9.5 to RM 10.5	0.34	**4.6**	**16**	**4.1**	**19**	**3.4**
RM 10.5 to RM 11.8	0.34	**5.7**	**17**	**4.0**	**20**	**3.3**

[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%.

[b] Total HQ was calculated using the following prey portions**:** 89% carp, 9% smallmouth bass, and 2% brown bullhead. When no brown bullhead data were available, smallmouth bass were assigned a prey portion of 11%. No TEQ data were available for largescale sucker and northern pikeminnow.

DDE – dichlorodiphenyldichloroethylene
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
NA – not analyzed
ND – no data
PCB – polychlorinated biphenyl
RM – river mile
TEQ – toxic equivalent
Bold identifies HQs > 1.0.

Lead, total PCBs, total TEQ,[122] and 4,4′-DDE were all identified as COCs for osprey because dietary-dose and/or bird egg HQs based on the consumption of multiple prey were > 1.0 in all exposure areas. HQs based on multiple prey for all exposure areas was not > 1.0 for mercury and BEHP; therefore, these COPCs are not expected to pose risks to osprey and were not identified as COCs. It should be noted that the two LOEs for

[122] Dioxin/furan TEQ and PCB TEQ also had HQs > 1.0, but was not identified as a COC because total TEQ was identified as a COC.

assessing dietary risk (dietary-dose and bird egg approach) were not consistent. Sum DDE HQs were > 1.0 based on the bird egg approach but 4,4′-DDE HQs were < 1.0 using the dietary-dose approach. TEQ sum HQs (for PCB TEQ, dioxin/furan TEQ, and total TEQ) were all < 1.0 based on individual prey components using the dietary-dose approach (Attachment 17) but > 1.0 based on multiple prey using the bird egg approach. The differences in the results of these two LOEs are further discussed in Section 8.2.

Lead is an uncertain COC. The lead HQ for osprey was > 1.0 in the exposure area between RM 9.5 and RM 10.5. Although the HQ was calculated assuming that smallmouth bass comprised 11% of the osprey diet, smallmouth bass contributed 100% to the risk estimate (i.e., HQ). 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 greater than 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 (0.0048 to 1.8 mg/kg ww) from all other areas.

There are several uncertainties associated with the exposure and effects assumptions used to derive the risk estimates using both the dietary-dose and bird egg approaches. There are limited data available for the assessment of dioxins/furans and dioxin-like PCBs. TEQ data are not available for largescale sucker tissue, which comprises 83% of the osprey diet. The use of carp tissue as representative of largescale sucker TEQ concentrations may overestimate or underestimate exposure concentrations to osprey. The variability of bird TEFs among species and relevance of TEFs based on EROD induction to evaluate individual and population risks from dioxins/furans and dioxin-like PCBs is uncertain.

The exposure assumptions (e.g., dietary prey portions, exposure scale area) for osprey are based on regional literature. Assuming a site use factor of 1.0 for osprey is conservative because osprey 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. Elliott et al. (2007) reported that wintering in different sites had no significant effect on organochlorine contaminant concentration in sample eggs, including those of ospreys from the Lower Columbia River; however, the sampling design limitations restricted the power of these statistical conclusions. Osprey that breed along the Willamette River likely winter in Mexico or Central America (Henny et al. 2003), which, though less industrialized than the United States, are within the malarial zone where DDT use is ongoing or has been only recently banned. Moreover, DDE has a half-life in birds of approximately 400 days, which means that breeding along the Willamette River may carry body burdens of DDE from previous breeding and wintering grounds (Clark et al. 1987). Elliott et al. (2007) reported that the ratio of DDE to DDT concentrations in eggs was significantly higher for osprey that wintered in Central or South America compared to those that wintered in Mexico and the United States; thus, pre-breeding exposure at wintering locations has been found to affect the relative concentrations of DDE and DDT in eggs.

The greatest uncertainties associated with the exposure and effects assumptions are specific to the bird egg approach. Per EPA (2008j), BMFs were used to calculate risk estimates; however, no significant relationship between co-located fish prey and bird egg concentrations (i.e., no BMR) could be found for any of the COPCs, except total TEQs (Section 8.1.2.3.2). Using the total TEQ BMR reduced estimated total TEQ in eggs by about 48% relative to using the geometric mean BMF of 10. There is additional uncertainty with the bird egg approach because TRVs used to derive risk estimates are based on LOAELs derived from field studies. 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 associate risk with a single contaminant because birds in the field are typically exposed to mixtures of these three contaminant groups. Non-contaminant stressors may also contribute to declines in reproduction. The use of field-collected data, do, however, allow for the selection of piscivorous-specific (i.e., bald eagle and osprey) TRVs. Table 8-31 presents a summary of the HQs and uncertainties for all osprey COCs.

Table 8-31. Summary of Osprey COCs

COC	LOE	Exposure Area(s) of HQ > 1.0	HQ(s)	TRV Uncertainty	Exposure Uncertainty
Lead	Dietary dose	RM 9.5 – RM 10.5	7.8	LOAEL is based on Eco SSL that is an order of magnitude lower than the lowest acceptable literature-based LOAEL.	Smallmouth bass prey contributed 100% of risk based on maximum concentration (which is over 100 times greater than the other smallmouth bass concentration available from this exposure area and 2 to 5 orders of magnitude greater than lead concentrations detected in all other bass samples).
Total PCBs	Dietary dose	RM 10.5 – RM 11.8	1.1	Based on chicken study; chickens appear to have a higher sensitivity to PCBs than do other avian species in the laboratory.	Exposure assumptions are based on regional literature values.
	Bird egg	All exposure areas	4.5 – 5.7	Based on bald eagle field data associated with productivity; exceedance of field-based TRV cannot conclusively indicate risks because of confounding factors in the field.	Exposure assumptions are based on regional literature values; highly uncertain BMFs were used.
Total TEQ[a]	Bird egg	All exposure areas	19 – 25	Based on bald eagle field data associated with productivity and eggshell thinning; exceedance of field-based TRV cannot conclusively	Data not available for largescale sucker (making up 83% of the diet); variability of TEFs is uncertain; highly uncertain

Table 8-31. Summary of Osprey COCs

COC	LOE	Exposure Area(s) of HQ > 1.0	HQ(s)	TRV Uncertainty	Exposure Uncertainty
				indicate risks because of confounding factors in the field.	2,3,7.8-TCDD BMF was used.
4,4′-DDE	Bird egg	All exposure areas	3.3 – 4.2	Based on bald eagle field data associated with productivity; exceedance of field-based TRV cannot conclusively indicate risks because of confounding factors in the field.	Exposure assumptions are based on regional literature values; highly uncertain BMFs were used.

[a] Dioxin/furan TEQ HQs ranged 3.9 to 4.5 in all exposure areas. PCB TEQ HQs ranged from 16 to 22 in all exposure areas.

BMF – biomagnification factor
COC – chemical of concern
DDE – dichlorodiphenyldichloroethylene
HQ – hazard quotient
PCB – polychlorinated biphenyl
RM – river mile
TCDD – tetrachlorodibenzo-*p*-dioxin
TEF – toxic equivalency factor
TEQ – toxic equivalent
TRV – toxicity reference value

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 is inclusive of the Study Area. Nesting success of osprey was monitored along the Willamette River system between 1993 and 2001 (Henny et al. 2008). 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 ten[123] 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. (2008) as 1.75 young per all types of nest (occupied, active, and successful). This rate of 1.75 is similar to the productivity of osprey that Henny et al. (2008) 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 productive 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 (Wiemeyer et al. 1984; Henny et al. 2008). 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 which is necessary to maintain a stable population.

[123] Three of the ten 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 RM26. The locations of the other six nests within the LWR segment were not reported (Henny et al. 2008).

In addition, residue egg data from a small sample of osprey eggs collected between RM 0 and RM 26 in 1997 to 1999 (five eggs) and again in 2001 (four eggs) indicate that concentrations of mercury, DDE, and total PCBs increased, although not significantly. Dioxins/furans show a decreasing trend similar to decreasing concentrations observed in eggs from the Upper River (Henny et al. 2008). Henny et al. (2003) also reported that the geometric mean residues in osprey eggs collected in 1993 were generally well below known threshold values for adverse effects on productivity. However, the conclusions of Henny et al. (2003) cannot be directly compared to the HQ results of the bird egg LOE because the threshold values used by Henny et al. were not reported, and the bird egg concentrations reported by Henny et al. were based on empirical geometric mean concentrations from eggs collected between RM 0 to RM 26, whereas Study Area egg concentrations in this BERA were estimated using UCL prey concentrations from RM 1.9 to RM 11.8 and BMFs.

8.1.4.2.5 Mink

Twelve COPCs were identified for mink in the SLERA and refined screen. Seven COPCs had HQs > 1.0 based on individual prey components (Table 8-20): antimony, lead, mercury, total PCBs, PCB TEQ, dioxin/furan TEQ, and total TEQ. HQs were calculated for these COPCs across multiple prey and incidental sediment ingestion using LOAEL-based TTCs and TSCs.

HQs based on multiple prey and incidental sediment ingestion are presented in Table 8-32.

Table 8-32. Mink LOAEL HQs Within 1-Mile Exposure Areas

	Total HQ						
Exposure Area	**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	0.0036	0.018	0.14	**23**	**2.1**	0.19	**2.3**
RM 2.5 to RM 3.5	0.011	0.015	0.15	**19**	**1.7**	0.22	**1.9**
RM 3.5 to RM 4.5	0.018	0.030	0.16	**20**	**1.7**	0.23	**1.9**
RM 4.5 to RM 5.5	0.0077	0.055	0.16	**19**	**1.4**	0.21	**1.6**
RM 5.5 to RM 6.5	0.0051	0.025	0.17	**19**	**1.4**	0.24	**1.6**
RM 6.5 to RM 7.5	0.014	0.12	0.28	**22**	**1.5**	**2.0**	**12[c]**
RM 7.5 to RM 8.5	0.0079	0.018	0.19	**19**	**1.4**	0.27	**1.7**
RM 8.5 to RM 9.5	0.026	0.041	0.20	**20**	**1.7**	0.24	**1.8**
Swan Island Lagoon	0.0063	0.025	0.15	**23**	**1.6**	0.27	**1.8**
RM 9.5 to RM 10.5	0.33	**4.0**	0.18	**20**	**1.5**	0.25	**1.8**
RM 10.5 to RM 11.8	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 that is based on a sediment UCL that is 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 > 1.0.

Lead, total PCBs, and total TEQ[124] were all identified as COCs for mink because HQs based on consumption of multiple prey were > 1.0 in at least one exposure area. HQs based on multiple prey for all exposure areas was not > 1.0 for antimony and mercury; therefore, these COPCs are not expected to pose unacceptable risks to mink and were not identified as COCs. Lead is an uncertain COC. The lead HQ for mink was > 1.0 in the exposure area between RM 9.5 and RM 10.5. Although the HQ was calculated assuming that smallmouth bass comprised 20% of the mink diet, smallmouth bass contributed essentially 100% to the risk estimate (i.e., HQ). 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 greater than 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 (0.0048 to 1.8 mg/kg ww) from all other areas.

There are several uncertainties associated with the exposure and effects assumptions used to derive the risk estimates. There is some uncertainty associated with the prey portions assigned to each of the mink prey species. Assigned prey portions were based on those presented in the literature and on the relative abundance of potential prey items in the Study Area (Attachment 16). The effect of varying these deterministic prey portions on the HQs is presented in Section 8.1.4.4; however, in general, varying prey portions has little impact on overall risk conclusions for wildlife receptors. The mink diet was estimated assuming 100% ingestion of fish and crayfish. However, in the wild, mink diets vary greatly by season and availability, and mink feed on other prey such as birds, mammals, and amphibians; in some cases, these other taxa are mink's most important food source (Eagle and Whitman 1987). In addition, mink are not particularly agile in water, and tend to catch a higher proportion of small and slow-moving fish than large or swift ones such as salmonids, though spawning salmonids may be more vulnerable (Melquist et al. 1981; Dunstone and Birks 1987). Some of the mink COCs (i.e., PCBs, and dioxins/furans) tend to bioaccumulate in the tissues of higher-trophic-

[124] Dioxin/furan TEQ and PCB TEQ also had HQs > 1.0 but were not identified as COCs because total TEQ was identified as a COC.

level organisms; it is therefore possible that exposure estimates may underestimate risks to mink if the proportion of salmon or piscivorous birds in their diet is high, but they are more likely to overestimate risks to mink because much of the mink's common prey is made up of smaller fish and herbivorous waterbirds and mammals. Further uncertainties and potential overestimation of risk are associated with the mink site use factor of 1.0. Mink primarily forage on the land along waterways but may also exploit adjacent uplands in pursuit of terrestrial prey and may use other water bodies near the Study Area. In addition, mink prefer riparian cover within their foraging habitat so industrial areas where no riparian cover exists may not be used by mink (Allen 1986). Therefore, the site use factor is most likely < 1.0.

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 ideal habitat for these species. Mink prey on both terrestrial and aquatic animals; although otter share a similar diet, prey are more likely fish, crustaceans, amphibians, and reptiles. The home range of mink is significant smaller than 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 Lower Willamette River 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 include 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.

Also, mink exposure may be limited by the types of prey that they are physically able to catch. Typically, mink foraging is restricted to invertebrates and small fish in nearshore environments. Therefore, mink may have limited exposed 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 to remediation of offshore or deepwater areas.

There is uncertainty associated with the TEQ related to the derivation of mammal TEFs. The TEFs for mammals were based on *in vivo* toxicity (when data are available), and there is less variability in mammal-derived TEFs than in bird TEFs; 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.

The LOAELs used to derive mink HQs for total PCBs and TEQs have low uncertainty. 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). Field-collected carp from these studies also had reported concentrations of other chemicals (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) .

Table 8-33 presents a summary of the HQs and uncertainties for all mink COCs.

Table 8-33. Summary of Mink COCs

COC	Exposure Area(s) of HQ > 1.0	HQ(s)	TRV Uncertainty	Exposure Uncertainty
Lead	RM 9.5 to RM 10.5	4.0	Based on drinking water exposure.	Smallmouth bass prey contributed 100% of the risk based on the maximum concentration (which is over 100 times greater than the other smallmouth bass concentration available from this exposure area and 2 to 5 orders of magnitude greater than lead concentrations detected in all other bass samples).
Total PCBs	All exposure areas	19 – 33	Low uncertainty; LOAEL based on mink fed contaminated fish in the lab.	Mink is an opportunistic feeder; changing the assumed dietary portions could change the range of HQs (Section 8.1.4.4.1); however, total PCBs would still be identified as a COC.
Total TEQ[a]	All exposure areas	1.6 – 12	Low uncertainty; LOAEL based on mink fed contaminated fish in the lab.	Mink is an opportunistic feeder; changing the assumed dietary portions could change the range of HQs (Section 8.1.4.4.1); however, total TEQ would still be identified as a COC; maximum HQ (12) was based on a high sediment UCL that is greater than the maximum concentration.

[a] The dioxin/furan TEQ HQ was 2.0 in one exposure area (RM 6.5 to RM 7.5). PCB TEQ HQs ranged from 1.4 to 2.4 in all exposure areas.

HQ – hazard quotient

TEQ – toxic equivalent

PCB – polychlorinated biphenyl

8.1.4.2.6 River Otter

Nine COPCs were identified for river otter in the SLERA and refined screen. Five COPCs had HQs > 1.0 based on individual prey components (Table 8-20): lead, total PCBs, PCB TEQ, dioxin/furan TEQ, and total TEQ. HQs were calculated for these COPCs across multiple prey and incidental sediment ingestion using LOAEL-based TTCs and TSCs.

HQs based on multiple prey and incidental sediment ingestion are presented in Table 8-34.

Table 8-34. River Otter LOAEL HQs Within 3-Mile Exposure Areas

	Total HQ				
Exposure Area	**Lead[a]**	**Total PCBs[a]**	**PCB TEQ[b]**	**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. Dioxins/furans and PCB congeners were not analyzed in largescale sucker.

HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
PCB – polychlorinated biphenyl
RM – river mile
TEQ – toxic equivalent
Bold identifies HQs > 1.0.

Total PCBs and total TEQ[125] were identified as COCs for river otter because HQs based on the consumption of multiple prey were > 1.0 in at least one exposure area. HQs based on multiple prey for all exposure areas were not > 1.0 for lead and dioxin/furan TEQ; therefore, these COPCs are not expected to pose unacceptable risks to river otter and were not identified as COCs.

There are several uncertainties associated with the exposure and effects assumptions used to derive the risk estimates. The effect of varying prey portions on the HQs is presented in Section 8.1.4.4.1. There is uncertainty associated with the TEQ related to the derivation of mammal TEFs. The TEFs for mammals were based on *in vivo* toxicity (when data are available), and there is less variability in mammal-derived TEFs than in bird TEFs; 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 associated with these TEFs may result in an over- or underestimation of risk.

The LOAELs used to derive river otter HQs for total PCBs and TEQs have low uncertainty. 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). Field-collected carp from these studies also had reported concentrations of other chemicals (e.g., dioxins/furans,

[125] PCB TEQ also had an HQ > 1.0 but was not identified as a COC because total TEQ was identified as a COC.

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) .

Table 8-35 presents a summary of the HQs and uncertainties for all mink COCs.

Table 8-35. Summary of River Otter COCs

COC	Exposure Area(s) of HQ > 1.0	HQ(s)	TRV Uncertainty	Exposure Uncertainty
Total PCBs	All exposure areas	21 – 31	Low uncertainty; LOAEL based on mink fed contaminated fish in the lab.	River otter is an opportunistic feeder; changing the assumed dietary portions could change the range of HQs (Section 8.1.4.4.1); however, total PCBs would still be identified as a COC.
Total TEQa	All exposure areas	1.1 to 2.3	Low uncertainty; LOAEL based on mink fed contaminated fish in the lab.	River otter is an opportunistic feeder; changing the assumed dietary portions could change the range of HQs (Section 8.1.4.4.1); however, total TEQ would still be identified as a COC.

[a] PCB TEQ HQs were 1.3 and 1.5 in the exposure areas between RM 1.5 to RM 4.5 and above RM 10.5, respectively.

COC – chemical of concern

HQ – hazard quotient

PCB – polychlorinated biphenyl

RM – river mile

TRV – toxicity reference value

8.1.4.3 Summary of Bird and Mammal Diet COCs

Ten bird COCs were identified for at least one avian receptor: copper, lead, mercury, benzo(a)pyrene, dibutyl phthalate, total PCBs, total TEQ,[126] aldrin, DDE (as sum DDE or 4,4′-DDE), and total DDx. Three mammal COCs were identified for mink and/or river otter: lead, total PCBs, total TEQ.[16] The HQs and uncertainties associated with risk estimates are evaluated below and in Section 8.2.

8.1.4.4 Evaluation of Additional Uncertainties

Uncertainties associated with exposure assumptions, effect thresholds (TRVs), and risk characterization methods were identified in previous subsections. This subsection presents the uncertainty evaluation calculations for additional uncertainties identified as part of the wildlife dietary assessment. The uncertainty of selected prey portions for deriving risk estimates was evaluated. Per EPA's Problem Formulation (Attachment 2),

[126] Dioxin/furan TEQ and/or PCB TEQ had HQs > 1.0 but were not identified as COCs because total TEQ was identified as a COC.

risk estimates were also derived for belted kingfisher. The results of the risk characterization for belted kingfisher were compared to those for selected bird and mammal receptors to ensure that the selected receptors were protective of belted kingfisher.

8.1.4.4.1 Evaluation of Varying Prey Portions

Selected prey portions (Table 8-9) were based on dietary information presented in the literature. In the 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 a COPC would be identified as a COC) for all COPC-receptor pairs in which multiple prey species were evaluated in the diet.[127]

The prey portion uncertainty evaluation determined the range of possible HQs if all prey species could make up 0 to 100% of the diet. The range of HQs was used to determine whether any additional COPCs would have been identified as COCs (i.e., have HQs > 1.0) had different prey portions been used (other than those presented in Table 8-5) or whether any COPCs would have not been identified as COCs. Table 8-36 presents the results of this evaluation.

[127] 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 8-36. Summary of Dietary Prey Portion Uncertainty Evaluation

COPC by Receptor	LOE	HQ	Exposure Area	Selected Prey Portions Used in Risk Characterization	Prey Portion Uncertainty Evaluation	Does Uncertainty Evaluation Change COC Status?
Hooded Merganser						
Total PCBs	Diet dose	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 not be a COC if diet was primarily crayfish and peamouth; however, this is not a reasonable assumption (sculpin likely represent a portion of the merganser diet).
Lead	Diet dose	0.013 – 0.21	All exposure areas	65% sculpin, 5% peamouth, 25% clams, 5% crayfish	More than 83% if diet would have to be represented by peamouth for HQs to be > 1.0 in all exposure areas; peamouth prey portion of > 83% is not supported by the literature.	No, because probability that peamouth is > 83% of diet is low.
BEHP	Diet dose	0.0012 – 0.67	All exposure areas	70% sculpin, 25% clams, 5% crayfish	HQs are < 1.0 based on any prey portions of selected prey species.	No, would not be identified as a COC regardless of varying prey portions.
Bald Eagle						
Mercury	Diet dose	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 not be a COC if the eagle diet was comprised solely of carp; however, this is not a reasonable assumption.
Mercury	Bird egg	0.37 – 0.52	All exposure areas	45% largescale sucker, 45% carp, 5% northern pikeminnow, 5% peamouth	HQs could range from 0.29 to 3.0 based on 100% ingestion of carp and northern pikeminnow, respectively; literature on prey species does not support a high prey portion (>30%) of pikeminnow by bald eagles.	No, because probability that northern pikeminnow is > 30% of diet is low.

Table 8-36. Summary of Dietary Prey Portion Uncertainty Evaluation

COPC by Receptor	LOE	HQ	Exposure Area	Selected Prey Portions Used in Risk Characterization	Prey Portion Uncertainty Evaluation	Does Uncertainty Evaluation Change COC Status?
Total PCBs	Diet dose	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 not be a COC if carp was absent from the diet; however, this is not a reasonable assumption.
Total PCBs	Bird egg	26	All exposure areas	45% largescale sucker, 45% carp, 5% northern pikeminnow, 5% peamouth	HQs could range from 0.81 to 53 based on 100% ingestion of peamouth and carp, respectively.	No, would not be a COC if the eagle diet was composed solely of peamouth; however, this is not a reasonable assumption.
PCB TEQ	Bird egg	240	All exposure areas	100% carp	TEQ data were only available from carp and not available from other bald eagle prey species.	Unknown, because TEQ data only available from carp prey.
Dioxin/furan TEQ	Bird egg	59	All exposure areas	100% carp	TEQ data were only available from carp and not available from other bald eagle prey species.	Unknown, because TEQ data only available from carp prey.
Total TEQ	Bird egg	280	All exposure areas	100% carp	TEQ data were only available from carp and not available from other bald eagle prey species.	Unknown, because TEQ data only available from carp prey.
Sum DDE	Diet dose	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 > 1.0 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 probability that northern pikeminnow is > 45% of diet is low.
4,4′-DDE	Bird egg	9.3 – 11	All exposure areas	45% largescale sucker, 45% carp, 5% northern pikeminnow, 5% peamouth	HQs could range from 5.0 to 33 based on 100% ingestion of pikeminnow.	No, would be a COC regardless of varying prey portions.

Table 8-36. Summary of Dietary Prey Portion Uncertainty Evaluation

COPC by Receptor	LOE	HQ	Exposure Area	Selected Prey Portions Used in Risk Characterization	Prey Portion Uncertainty Evaluation	Does Uncertainty Evaluation Change COC Status?
Osprey						
Lead	Diet dose	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 > 1.0	Yes, would not be a COC if smallmouth bass represented < 1% of the diet, which is a reasonable assumption.
Mercury	Diet dose	0.25 – 0.34	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; However, literature on osprey dietary composition does not support a high prey portion (> 63%) of pikeminnow .	No, because probability that northern pikeminnow is > 63% of diet is low.
Mercury	Bird egg	0.31 – 0.44	All exposure areas	83% largescale sucker, 6% carp, 7% northern pikeminnow, 2% smallmouth bass, 2% brown bullhead	HQs could range from 0.19 to 2.0 based on 100% ingestion of bullhead and pikeminnow, respectively; literature on prey species does not support a high prey portion (> 50%) of pikeminnow by osprey.	No, because probability that northern pikeminnow is > 50% of diet is low.
BEHP	Diet dose	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 > 1.0 at RM 3.5 to RM 4.5; literature on prey species does not support a high prey portion (> 59%) of smallmouth bass by osprey.	No, because probability that smallmouth bass is > 59% of diet is low.
Total PCBs	Diet dose	1.1	RM 10.5 – RM 11.8	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.	Yes, would not be a COC if the carp or smallmouth bass were not included as part of osprey diet, which are reasonable assumptions.

Table 8-36. Summary of Dietary Prey Portion Uncertainty Evaluation

COPC by Receptor	LOE	HQ	Exposure Area	Selected Prey Portions Used in Risk Characterization	Prey Portion Uncertainty Evaluation	Does Uncertainty Evaluation Change COC Status?
Total PCBs	Bird egg	4.5 – 5.7	All exposure areas	83% largescale sucker, 6% carp, 7% northern pikeminnow, 2% smallmouth bass, 2% brown bullhead	HQs could range from 0.24 to 35 based on 100% ingestion of brown bullhead and carp, respectively.	No, would be a COC in at least one exposure area regardless of varying prey portions.
PCB TEQ	Bird egg	16 – 22	All exposure areas	89% carp, 9% smallmouth bass, 2% brown bullhead	HQs could range from 2.9 to 61 based on 100% ingestion of brown bullhead and smallmouth bass, respectively.	No, would be a COC regardless of varying prey portions.
Dioxin/ furan TEQ	Bird egg	3.9 – 10	All exposure areas	89% carp, 9% smallmouth bass, 2% brown bullhead	HQs could range from 0.73 to 73 based on 100% ingestion of brown bullhead and smallmouth bass, respectively.	No, would be a COC in at least one exposure area regardless of varying prey portions.
Total TEQ	Bird egg	19 – 25	All exposure areas	89% carp, 9% smallmouth bass, 2% brown bullhead	HQs could range from 3.7 to 82 based on 100% ingestion of brown bullhead and smallmouth bass, respectively.	No, would be a COC regardless of varying prey portions.
4,4′-DDE	Bird egg	3.3 – 4.2	All exposure areas	83% largescale sucker, 6% carp, 7% northern pikeminnow, 2% smallmouth bass, 2% brown bullhead	HQs could range from 1.2 to 12 based on 100% ingestion of brown bullhead and smallmouth bass, respectively.	No, would be a COC regardless of varying prey portions.
Mink						
Antimony	Diet dose	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 > 1.0 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 be a COC if > 63% of diet could be composed of smallmouth bass, which is a reasonable assumption.

Table 8-36. Summary of Dietary Prey Portion Uncertainty Evaluation

COPC by Receptor	LOE	HQ	Exposure Area	Selected Prey Portions Used in Risk Characterization	Prey Portion Uncertainty Evaluation	Does Uncertainty Evaluation Change COC Status?
Lead	Diet dose	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 > 1.0.	No, would not be a COC if smallmouth bass represented <5% of the diet, however, this is not a reasonable assumption given the opportunistic nature of mink and abundance of smallmouth bass in the Study Area.
Mercury	Diet dose	0.14 – 0.28	All exposure areas	20% carp, 20% sculpin, 20% largescale sucker, 20% smallmouth bass, 20% crayfish	HQs are < 1.0 based on any portions of selected prey species.	No, would not be identified as a COC regardless of varying prey portions.
Total PCBs	Diet dose	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 be a COC in at least one exposure area regardless of varying prey portions. HQs would be < 1.0 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 was composed only of crayfish or sculpin; however, ingestion of a single species is not a reasonable assumption given the opportunistic nature of mink, and HQs would still be > 1.0 in other exposure areas.

Table 8-36. Summary of Dietary Prey Portion Uncertainty Evaluation

COPC by Receptor	LOE	HQ	Exposure Area	Selected Prey Portions Used in Risk Characterization	Prey Portion Uncertainty Evaluation	Does Uncertainty Evaluation Change COC Status?
PCB TEQ	Diet dose	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 not be a COC if diet was primarily comprised of crayfish; however this is not a reasonable assumption given the opportunistic nature of mink and abundance of crayfish in the Study Area.
Dioxin/furan TEQ	Diet dose	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 not be a COC if diet was comprised only of carp; however, this is not a reasonable assumption given the opportunistic nature of mink and abundance of carp in the Study Area.
Total TEQ	Diet dose	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 not be a COC if diet was primarily comprised of crayfish; however this is not a reasonable assumption given the opportunistic nature of mink and abundance of crayfish in the Study Area.
River Otter						
Lead	Diet dose	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 > 1.0 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 be a COC if > 8% of diet could be composed of smallmouth bass, which is a reasonable assumption.

Table 8-36. Summary of Dietary Prey Portion Uncertainty Evaluation

COPC by Receptor	LOE	HQ	Exposure Area	Selected Prey Portions Used in Risk Characterization	Prey Portion Uncertainty Evaluation	Does Uncertainty Evaluation Change COC Status?
Total PCBs	Diet dose	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 be a COC regardless of varying prey portions.
PCB TEQ	Diet dose	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 not be a COC in any exposure area if diet was primarily composed of crayfish and clams; however, this is not a reasonable assumption.
Dioxin/furan TEQ	Diet dose	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 > 1.0 in one exposure area (RM 4.5 to 7.5); 100% ingestion of crayfish, sculpin, or smallmouth bass results in an HQs ranging from 1.2 to 1.5 between RM 4.5 and 7.5.	Yes; would be a COC if > 77% or 83% of diet could be composed of crayfish or sculpin, respectively, which is a reasonable assumption.
Total TEQ	Diet dose	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 not be a COC if diet was primarily composed of crayfish and clams; however, this is not a reasonable assumption.

COC – chemical of concern
COPC – chemical of potential concern
DDE – dichlorodiphenyldichloroethylene
HQ – hazard quotient
LOE – line of evidence
PCB – polychlorinated biphenyl
RM – river mile
TEQ – toxic equivalent

For most COPC-receptor pairs, varying prey portions would not change which pairs would be identified as COC-receptor pairs (Table 8-36).

For the bird receptors, assuming that any given prey species could make up 0 to 100% of the diet in a few additional COPC-receptor pairs that would have HQs > 1.0:

- Lead-hooded merganser
- Mercury-bald eagle (based on the bird egg LOE)
- Sum DDE-bald eagle (based on the dietary-dose LOE)
- Mercury-osprey (based on both the dietary-dose and bird egg LOE)
- BEHP-osprey

However, the portion of the diet that a given prey species would have to represent as part of the total diet to result in an HQ > 1.0 is not likely, based on the diet prey portions presented in the general literature and region-specific studies. Therefore, the likelihood that these bird COPC-receptor pairs may pose unacceptable risk is very low.

For the mammal receptors, assuming that any given prey species could make up 0 to 100% of the diet results in three additional COPC-receptor pairs that would have HQs > 1.0:

- Antimony-mink
- Lead-river otter
- Dioxin/furan TEQ-river otter

The portion of the diet that a given prey species would have to represent as part of the total diet to result in an HQ > 1.0 is possible for these mammal COPC-receptor pairs based on the prey portions presented in the general literature and region-specific studies. The assessment of lead and dioxin/furan TEQ exposure to an aquatic-dependent carnivore is already covered in the evaluation of mink, inasmuch as lead-mink and dioxin/furan TEQ-mink were already identified as COC-receptor pairs. Mink are protective of river otter as they have similar diets, although mink have a smaller exposure area, a higher assumed rate of incidental sediment ingestion, and a higher rate of prey ingestion relative to body weight. Unacceptable risks to mink from antimony are possible at one localized exposure area (RM 9.5 to RM 10.5) assuming that at least 63% of the diet is composed of smallmouth bass; however, the magnitude of the HQ is low (assuming 100% of the diet is composed of smallmouth bass results in an HQ of 1.6) and, therefore, the potential for risks is considered low to negligible.

A few COPC-receptor pairs would have not been identified as COCs had different dietary assumptions been used:

- Total PCBs-hooded merganser
- Mercury-bald eagle

- Total PCBs-bald eagle
- Total PCBs-osprey (based on dietary-dose LOE)
- Lead-osprey
- Lead-mink
- TEQ (PCB TEQ, dioxin/furan TEQ, and total TEQ)-mink
- TEQ (PCB TEQ and total TEQ)-river otter

For lead-osprey and total PCBs-osprey based on the dietary-dose LOE, the assumptions needed to result in HQs < 1.0 are reasonable and the identification of these COC-receptor pairs is highly uncertain. For all other COPC-receptor pairs, assumptions needed to result in HQs < 1.0 are not supported by the literature.

8.1.4.4.2 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 to the results of the risk characterization for selected bird and mammal receptors to ensure that the selected receptors were protective of belted kingfisher. Following 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. These COPCs are presented in Table 8-37.

Table 8-37. Belted Kingfisher COPCs

COPCs	
Metals	
Aluminum	Lead
Copper	Mercury
PAHs	
Benzo(a)pyrene	
Phthalates	
BEHP	Dibutyl phthalate
PCBs and Dioxins/Furans	
Total PCBs	Dioxin TEQ
PCB TEQ	Total TEQ
Pesticides	
Sum DDE	Total DDx

BEHP – bis(2-ethylhexyl) phthalate
COPC – chemical of potential concern

DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
TEF – toxicity equivalence 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

Exposure Assumptions

The exposure assumptions used to derive EPCs for belted kingfisher are presented in Table 8-38. These assumptions were based on EPA (2008j); except for the SIR, which was corrected to reflect 2% incidental sediment ingestion. Details and the rationale for the selected receptor-specific exposure parameters and uncertainties are presented in Attachment 16.

Table 8-38. 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	1.0	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
FIR – food ingestion rate
NA – not applicable (value based on best professional judgment)
SIR – sediment ingestion rate
SUF – site use factor

Belted kingfishers generally feed within 1 mile of their nesting sites but may have a foraging range up to 5 miles. The exposure scale of 1 mile based on reported foraging ranges was assumed for estimating the exposure of belted kingfishers.

Four prey species were used to represent belted kingfisher prey: juvenile Chinook salmon, peamouth, sculpin, clams. The selected diet for belted kingfisher was based on information from the literature. In the third step of the risk characterization, portions of individual prey were varied (based on diets presented in the literature) to estimate risks to belted kingfisher assuming the consumption of multiple prey species from the Study Area. 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 8.1.3.1 were used to derive receptor-specific TTCs and TSCs for the belted kingfisher. As described in Section 8.1.3.2, receptor-specific TTCs and TSCs were calculated using receptor-specific parameters (presented in Attachment 16). Table 8-39 presents the belted kingfisher TTCs and TSCs for all belted kingfisher COPCs.

Table 8-39. 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
Total DDx	µg/kg	426	4260	103,000	1,030,000

COPC – chemical of potential concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene

DDT – dichlorodiphenyltrichloroethane
LOAEL – lowest-observed-apparent-effect level
NA – not applicable (not a COPC-receptor pair)
NOAEL – no-observed-apparent-effect level
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
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)
TSC – threshold sediment concentration
TTC – threshold tissue concentration

8.1.4.4.1 Risk Characterization

This section presents the risk characterization process and results for the belted kingfisher. The same process outlined in Section 8.1.3.1 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 had HQs > 1.0 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.

HQs based on multiple prey and incidental sediment ingestion are presented in Table 8-40.

Table 8-40. Belted Kingfisher LOAEL HQs Within 1-Mile Exposure Areas

	Total HQ					
River Mile Group	**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
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 > 1.0.

HQs based on multiple prey for all exposure areas was not > 1.0 for lead, mercury, total TEQ, and sum DDE; therefore, these COPCs are not expected to pose unacceptable risks to belted kingfisher. However, BEHP and total PCBs, had HQs based on a multiple-prey diet were > 1.0 in at least one exposure area.

There is high uncertainty 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 an uncertainty factor (UF) of 10, per EPA (2008f). There is high uncertainty with this LOAEL; 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 bird than the extrapolated LOAEL. Using the literature-based BEHP LOAEL (11 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 results of the risk characterization for belted kingfisher were compared to the results of the risk characterization for selected bird and mammal receptors to ensure that the selected avian receptors were protective of belted kingfisher. The characterization of BEHP based on the selected LOAEL is too uncertain to determine that unacceptable risks to kingfisher may be occurring, and 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 were determined to be protective of belted kingfisher.

8.2 RISK CONCLUSIONS

This section presents a summary of the overall conclusions of the wildlife risk assessment. Bird and mammal COCs were identified using the dietary-dose and (for bald eagles and osprey) egg LOEs. Risk conclusions were determined by evaluating the magnitude of HQs, spatial distribution and frequency of HQs > 1.0, 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 egg LOEs were used, in part, to determine conclusions for bald eagle and osprey.

Several COPCs have been identified as COCs for one or more wildlife receptors. Not all of the COCs are equal potential contributors to ecological risk in the Study Area. Most of them are unlikely to pose unacceptable risks to wildlife populations at the levels of exposure occurring in the Study Area, even though estimated exposures are greater than TRVs. This is because most COCs only exceed TRVs to a limited extent and TRVs are largely based on organism-level (versus population-level) effects. For all receptors evaluated at the population level, LOAELs were used to derive HQs. However, LOAELs are based on organism-level effects (e.g., reduced growth, reduced clutch size, increased mortality). Because of the uncertainties and limitations of using LOAELs to extrapolate to population-level risks, the magnitude and frequency of LOAEL exceedances, as well as the endpoints used to derive the LOAEL were analyzed to determine the potential for population-level effects. It is unlikely that a COC with a limited spatial distribution of exceedances and a low HQ poses significant risks to populations. COCs with a broader distribution and higher magnitude of exceedances have greater potential for posing population-level risks. The type of effect threshold endpoint (i.e., survival, growth or reproduction) affects the interpretation of HQs simply because effects on the survival of individuals have different implications for populations than do effects on the growth of individuals. Effects on the reproductive success of individuals have different implications for populations than do effects on either survival or growth.

Even for a particular type of effect threshold endpoint (e.g., reduced survival), a particular HQ (e.g., HQ = 5) has different implications for each COC-receptor pair because the exposure-response relationship of one COC-receptor pair differs from that of other COC-receptor pairs. Short of developing structured population models and incorporating the effects of COC-receptor-specific exposure-response data, the demographic (i.e., population-level) consequences of any particular HQ simply cannot be quantified.

Populations are resilient to environmental stressors (Barnthouse et al. 2009). By corollary, it is unreasonable to assume that survival, growth, or reproduction HQs > 1.0 will necessarily be detrimental to the populations whose individuals are potentially affected.

After considering these factors, PCBs were found to be the most significant contributor to wildlife risk, and the mink population is the wildlife receptor that is most vulnerable to PCB exposure. Section 8.2.1 presents a summary of the wildlife COCs. Section 8.2.2 presents the WOE evaluation for bald eagle and osprey. Section 8.2.3 presents the risk conclusions for all wildlife COCs. In Section 11.0, the wildlife risk conclusions are combined with the risk

conclusions for other ecological receptor groups to provide a holistic view of ecological risks; COCs and receptors that indicate unacceptable ecological risks are identified, in the sense that a remediation based on those COC-receptor pairs would address potential risks posed by other COCs and to other ecological receptors.

8.2.1 Bird and Mammal COCs

Table 8-41 presents a summary the wildlife COCs identified. Total PCBs was the only COPC that was identified as a COC for all six wildlife receptors. This was not unexpected; PCBs frequently drive assessed risks at contaminated sediment sites, and PCBs have also been identified in the BHHRA to be the predominant contributor to potential human health risks in the Study Area. For two species (hooded merganser and river otter), total PCBs was the only COC identified.

Table 8-41. Summary of Wildlife COCs

	Receptor					
COC[a]	**Spotted Sandpiper**	**Hooded Merganser**	**Bald Eagle**	**Osprey**	**Mink**	**River Otter**
Copper	X					
Lead				X	X	
Mercury			X[b]			
Benzo(a)pyrene	X					
Dibutyl phthalate	X					
Total PCBs	X	X	X	X	X	X
Total TEQ	X[d]		X[c,d]	X[c,d]	X[d]	
DDx compounds	X[b,e]		X[c,f]	X[c,f]		
Aldrin	X					

[a] COCs were identified as those COPCs for which the LOAEL HQ was > 1.0 was based on the ingestion of multiple prey over relevant exposure scale for all receptors except bald eagle. For bald eagle, COCs were identified as those COPCs for which NOAEL HQ was > 1.0 were based on ingestion of multiple prey over a relevant exposure scale.

[b] Identified as a COC based on the dietary-dose LOE.

[c] Identified as a COC based on the egg LOE.

[d] Dioxin/furan TEQ and/or PCB TEQ also had HQs > 1.0 but were not identified as COCs because total TEQ was identified as a COC.

[e] DDx compounds assessed using total DDx.

[f] DDx compounds assessed using DDE (as sum DDE in the dietary-dose LOE or 4,4′-DDE in the bird egg LOE).

COC – chemical of concern

COPC – chemical of potential concern

LOAEL – lowest-observed-apparent-effect level

NOAEL – no-observed-apparent-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)

The spotted sandpiper had the greatest number of COCs with seven COCs, followed by bald eagle and osprey with four COCs each, then mink with three COCs. In total, nine COPCs were identified as COCs for at least one wildlife receptor. DDx COCs are redundant in the sense that different forms were used to assess ecological risks from exposure (i.e., 4,4′-DDE was used in the bird egg evaluation, and sum DDE and total DDx were evaluated in the dietary dose evaluation). Sum DDE, 4,4′-DDE, and total DDx are three metrics for assessing the same risk (see Section 8.2.2), so in a practical sense there are 9 COCs. They are copper, lead, mercury, benzo(a)pyrene, dibutyl phthalate, total PCBs, total TEQ, aldrin, and DDx compounds (represented by sum DDE, 4,4′-DDE, and total DDx).

8.2.2 WOE Evaluation for Piscivorous Birds

For osprey and bald eagle, both estimated dietary doses and predicted egg COPC concentrations were used to evaluate risk. These two LOEs and their associated uncertainties were weighted to arrive at risk conclusions.

Based on dietary-dose estimates, total PCBs was the only COC identified for osprey, while total PCBs and mercury were identified for bald eagle (Table 8-41). The egg LOE identified total PCBs, total TEQ and 4,4′-DDE as COCs for both osprey and bald eagle. As previously noted, different metrics were used for the dietary-dose and egg LOEs to assess risks from exposure to DDx compounds. Only 4,4′-DDE was evaluated using the bird egg LOE because it is the primary DDx compound associated with bird egg toxicity. Sum DDD, sum DDE, sum DDT, and total DDx were all evaluated using the dietary-dose LOE; however, sum DDE was the only COC identified for piscivorous birds using the dietary-dose LOE. 4,4′-DDE is a component of sum DDE and is not present at levels indicating risk by the dietary-dose LOE. Therefore, the dietary-dose and egg LOEs are conflicting for DDx compounds.

The two LOEs are in agreement in identifying total PCBs as a COC. The conclusions for the two LOEs for mercury, total TEQ, and DDx compounds conflict; therefore, a WOE evaluation is needed to determine risk conclusions for these three COCs.

8.2.2.1 Evaluation of Effects Uncertainties

Both the dietary-dose and egg LOEs were intended to be protective of the most sensitive life stage (developing embryos). Egg TRVs are based on receptor-specific field data, whereas dietary TRVs are based on the most sensitive bird species tested in the laboratory under controlled conditions.

A comparison of the fish prey TTCs from both dietary and egg TRVs for bald eagle and osprey is presented in Table 8-42. The TTCs for the egg LOE are much lower (5 to 500 times lower) than the TTCs for the dietary-dose LOE for all COPCs, with the exception of mercury. Mercury LOAEL TTCs are similar for bird eggs and diet, and NOAEL TTCs are three times greater based on bird egg TTCs.

Table 8-42. Comparison of TTCs Based on Dietary and Bird Egg TRVs

	Dietary TRV			Bird Egg TRV		
COPC	**Osprey LOAEL TTC**	**Bald Eagle NOAEL TTC**	**Reproductive Endpoint**	**Osprey LOAEL TTC**	**Bald Eagle NOAEL TTC**	**Reproductive Endpoint**
Mercury	0.304 mg/kg	0.0533[a] mg/kg	LOAEL/NOAEL – offspring production and avoidance response behavior in ducklings observed in laboratory	0.247 mg/kg	0.167 mg/kg	LOAEL – altered embryo development of mallards following exposure via laboratory diet NOAEL – associated with field bald eagle populations for which no effect on productivity was reported
Total PCBs	2,760 µg/kg	2,420 µg/kg	LOAEL/NOAEL – reproductive success (i.e., hatchability) in chickens observed in laboratory	536 µg/kg	357 µg/kg	LOAEL – associated with impaired 5-year bald eagle productivity in field NOAEL – associated with field bald eagle populations for which no effect on productivity was reported
TEQ[b]	665 ng/kg	117 ng/kg	LOAEL/NOAEL – egg production and embryo survival in ring-necked pheasants given 10 weekly injections in laboratory	3.2 ng/kg	0.23 ng/kg	LOAEL – associated with reduced productivity and eggshell thinning in Columbia River bald eagles NOAEL –associated with Willamette River osprey for which no effect on productivity was reported
DDE	1,520 µg/kg	267 µg/kg[a]	LOAEL/NOAEL – eggshell thickness, reduced eggshell strength, and nestling mortality in barn owls	44.3 µg/kg	16.5 µg/kg	LOAEL – associated with reduced bald eagle offspring productivity in field NOAEL – associated with field bald eagle populations for which no effect on productivity was reported

[a] Based on NOAEL TRV extrapolated from LOAEL TRV using an uncertainty factor of 10.
[b] Includes PCB TEQ, dioxin/furan TEQ, and total TEQ.

COPC – chemical 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
TTC – threshold tissue concentration

Inter-species extrapolation is a source of uncertainty that is unique to the dietary-dose LOE. The error introduced by this extrapolation is unknown because osprey and bald eagle embryos are not tested in the laboratory. The use of the most sensitive laboratory test species as the surrogate for untested species is a conventional approach that is typically used except in cases where there are enough tested species to derive a species sensitivity distribution or other statistical estimate based on information about a number of tested species. Use of the most sensitive laboratory test species is the norm in wildlife risk assessment unless receptor-specific experimental data are available.

Another uncertainty associated with the bird egg LOE is that the TRVs are based on field studies. Field-collected egg tissues almost certainly contain other chemicals that contribute to reproductive toxicity, which would create an underestimation bias in the egg TRVs. Field-collected data alone cannot be used to conclude that adverse effects are strictly the result of chemical contamination, as non-chemical stressors (e.g., habitat degradation, predation) can also contribute to the adverse reproductive effects observed in the field. Field data were considered unacceptable for use in the fish and invertebrate tissue-residue TRVs because of the uncertainty associated with the presence of other contaminants; however, field-based bird egg TRVs were recommended by EPA to provide thresholds that were specific to the receptors being evaluated (i.e., bald eagle and osprey). However, given the high uncertainty of bird egg TRVs, the results of the bird egg LOE are not considered reliable.

Because neither bald eagle nor osprey eggs were sampled, egg TRVs had to be extrapolated to an exposure that was modeled (i.e., maternal dose) before they could be used. The egg-to-maternal-dose extrapolation used fish data and osprey egg data collected from throughout the Willamette River. Those data were used to estimate BMFs, and the BMFs were used to extrapolate TTCs (maternal dose thresholds) using Equation 8-7.

The reliance on BMFs is a significant source of uncertainty that is unique to the egg LOE in this BERA. BMF uncertainty is discussed in Section 8.1.2.3 and Attachment 8. For 4,4′-DDE, regression analyses on the available empirical data identified no relationship between fish and egg tissue concentrations, and the DDE BMFs calculated using seven co-located fish tissue composite samples and bird egg tissues based on data reported by Henny et al. (2003) varied by nearly 2 orders of magnitude (see Section 8.1.2.3.2). The regression analysis for total TEQ did identify a significant ($p \leq 0.05$), moderately strong relationship ($r^2 = 0.58$) between fish and egg tissue concentrations, and the range of BMFs was less variable across the seven co-located pairs, so the total TEQ BMF is more reliable than the 4,4′-DDE BMF. Nonetheless, the egg-to-diet extrapolation uncertainty is unique to the egg LOE. The BMFs used in the risk characterization are based on geometric mean fish and osprey egg concentrations over a large area of the Willamette River. Using the total TEQ regression relationship reduces the total TEQ HQs by about 50% (compared the TEQ HQ based on the BMF), showing that the geometric mean BMF for total TEQ results in an overestimation of risks.

The 4,4′-DDE BMF is highly uncertain, with unknown bias. As such, the egg LOE risk estimates for DDx compounds should be treated as no better than order of magnitude

estimates. The use of field study-based effect thresholds further contributes to lack of reliability in the egg LOE for DDx compounds. The dietary-dose LOE should be used to formulate risk conclusions instead of the egg LOE.

In summary, the principal concern with the egg LOE for total TEQ is the use of field-study-based effect thresholds. Field-based NOAELs were derived from egg residues in bird populations in which no effects were reported. There is likely a large range of concentrations for which adverse effects (e.g., on productivity) would not be observed within in a population, and it is unknown where a NOAEL based on a field study falls within that range of concentrations. Field-based LOAELs 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 (e.g., habitat degradation, predation) can also contribute to adverse reproductive effects observed in the field. Field studies were deemed unacceptable for deriving fish and invertebrate tissue TRVs; and, although the field-based TRVs were used to identify COCs for the bird egg LOE, the egg TRVs should be minimally relied upon for arriving at risk conclusions.

BMF uncertainty also contributes to the unreliability of the egg LOE for total TEQ, but the effect of that uncertainty seems to be smaller than for DDx compounds (i.e., on the order of a factor of two, based on BMF to BMR comparison, rather than order of magnitude for the DDx compounds).

8.2.2.2 Evaluation of Exposure Uncertainties

Empirical fish tissue concentrations and literature-based exposure assumptions for receptors (i.e., body weight, ingestion rates, and prey portions) were used to estimate dietary concentration EPCs; therefore, uncertainties inherent in those exposure assumptions (i.e., body weight, ingestion rates and prey portions) are the same for both the dietary and bird egg LOEs.

In the bird egg LOE, additional exposure assumptions were used. Egg tissue concentrations were estimated using literature-based BMFs because empirical bird egg tissue concentrations were not available. As presented in Section 8.1.2, the use of literature-derived BMFs to predict egg tissue concentrations is highly uncertain. BMFs are based on geometric mean fish and osprey egg concentrations over a large area of the Willamette River; however, statistical analysis of co-located fish and egg concentrations did not indicate a significant relationship between bird egg tissue and fish tissue concentrations for any bird egg COPC, except for total TEQs. Using the total TEQ BMR reduces the total TEQ HQs by about 50% (compared to the TEQ HQ based on the BMF); however, this BMR was applied to Study Area fish concentrations that were much greater than the fish concentrations used to develop the BMFs. The uncertainty in this extrapolation is potentially high because the regression relationship might not be the same outside the range of data used to develop it.

Additional uncertainty is associated with the use of maximum exposure concentrations to represent prey EPCs. Given the small exposure scales evaluated, (e.g., 1-mile exposure

areas), frequently, only limited data were available to derive the UCL concentrations for each prey species within the area; in most cases, one to four samples were available per prey species in a given 1-mile exposure area. When a limited number of samples are available, the UCL cannot be reliably estimated and the maximum concentration provides a conservative estimate of the mean. The use of maximum concentrations to represent prey EPCs may result in an over or underestimate of risk.

8.2.2.3 WOE Conclusions

The reliability of the dietary-dose LOE is much higher than the reliability of the egg LOE for both of the COPCs in cases where the LOEs conflict (DDx compounds and total TEQ). Therefore, the dietary LOE was the sole LOE used in arriving at DDx compounds and total TEQ risk conclusions for osprey and bald eagle.

8.2.3 Wildlife Risk Assessment Conclusions

Dietary exposure is the main pathway by which wildlife receptors are exposed to sediment contaminants. Two LOEs were evaluated to determine the risk resulting from dietary exposure: direct exposure as measured through ingestion of prey and sediment, and, for piscivorous birds, egg tissue residues estimated from maternal dietary dose. Risk to mink from PCBs is greater than risk to any other ecological receptor population from any other chemical in the Study Area. The following subsections present a summary of the overall wildlife risk conclusions (Section 8.2.3.1) and a detailed evaluation of the risk assessment results for PCBs and mink (Section 8.2.3.2).

8.2.3.1 Overall Conclusions across All COCs

Several factors affect interpretation of the quantitative risk analysis:

- Results of multiple LOEs for osprey and bald eagle COCs
- Magnitude of HQs
- Spatial extent of HQs > 1.0
- Implications of TRV exceedances based on COC-specific toxicological endpoints
- Uncertainty of exposure and effects assumptions
- The bird egg LOE for evaluating risk to piscivorous birds (i.e., bald eagle and osprey) is associated with a high degree of uncertainty because the available BMF data and field-based TRVs are unreliable (Section 8.2.2). As an unreliable predictor of risks to piscivorous birds, the bird-egg LOE was not used to draw risk conclusions. Only the dietary-dose LOE (based on more reliable exposure and effects assumptions) was used to determine risk conclusions for bald eagle and osprey. For all other wildlife receptors, the dietary dose was the only LOE evaluated.

Of the 23 COPCs identified by the SLERA and refined screening process for the wildlife receptors, the only clear risk to wildlife is due to PCBs. Calculated risk estimates indicate that both mink and river otter in the Study Area may be experiencing reduced reproductive

success because of exposure to PCBs. Mink are estimated to have a greater risk than river otter because of their higher metabolic requirements. Spotted sandpipers and bald eagles may also face increased risk of reduced reproductive success because of PCB exposure. Overall, however, the risk to birds from PCB exposure is uncertain because of the underlying dietary assumptions used to derive risk estimates (i.e., that the spotted sandpiper diet consists only of laboratory-exposed worms and that the bald eagle diet is comprised of 100% fish from the Study Area). In addition, any potential PCB risk to bird receptors, as well as to river otters, will likely be reduced through remedies designed to mitigate the risk to mink. Because risk to mink from PCBs represents the single highest likelihood of a chemical impacting an ecological receptor population in the Study Area, a detailed evaluation of the PCB assessment to mink is presented in Section 8.2.3.2.

Risk estimates for lead indicate that osprey and mink may be at risk. The potential risk to mink, however, is driven by one extreme outlier and, therefore, may not represent the true risk to the mink population. Lead was the only COC (other than total PCBs) identified for osprey based on the dietary-dose LOE and the HQ indicating that risks to osprey may be occurring. However, field data evaluating nesting success in the LWR (including the Study Area) indicate that osprey populations have increased in recent years. Based on 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 estimates for mercury based on dietary exposure indicate that mercury may pose low risks to individual bald eagles. It should be noted, however, that mercury concentrations are elevated in fish tissue from the upriver reach and from the region at large (as reported in Hope (2003)) in addition to fish tissue in the Study Area. Similarly, risk estimates for spotted sandpiper indicate that dibutyl phthalate may pose a low risk, although sediment concentrations in the Study Area are not elevated relative to background.

The remaining COCs (i.e., copper, benzo[a]pyrene, total TEQ, DDx, and aldrin) pose either low or no unacceptable risk to the wildlife receptors evaluated, given the low magnitude of HQ values and the limited spatial extent of the exceedances, which were determined 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 dioxin/furans and dioxin-like PCBs as well as benzo(a)pyrene could not be determined based on the high uncertainty associated with the selected dietary-dose TRVs, which were based on exposure studies involving weekly injection.

Wildlife COCs, HQs, uncertainties associated with exposure and effects, and risk conclusions are summarized in Table 8-43.Wildlife COCs were combined with COCs for all other ecological receptor groups to determine the overall ecological risk conclusions (Section 11.0).

Table 8-43. Summary of Wildlife COCs and Risk Conclusions Across LOEs

Receptor by COC	LOE	HQ[a]	Exposure Considerations	Effects Considerations	Risk Conclusions
Copper					
Spotted sandpiper	Dietary dose	1.1 (RM 1.9 – RM 3.9); 1.3 (RM 4.0 – RM 6.0); 1.1 (RM 7.0 – RM 9.0)	Dietary exposure only slightly exceeds the LOAEL if spotted sandpiper are assumed to consume a diet composed solely of clams. The dietary-dose estimates associated with a diet composed of a mixture of clams and worms are below the LOAEL.	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; literature-based LOAELs and bounded NOAELs range from 29 to 66 and 16 to 47 mg/kg bw/day, respectively (Attachment 14).	Although copper was identified as a COC under the assumptions required for the BERA, it is more reasonable to conclude that copper does not pose unacceptable risk to the spotted sandpiper population based on the uncertainty in the dietary assumption (i.e., that sandpiper fed on the maximally contaminated prey type), limited spatial extent of exceedance, and uncertainty associated with the LOAEL.
Lead					
Osprey	Dietary dose	7.8 (RM 9.5 – RM 10.5)	The spatial extent of potential risk is limited to RM 9.5 to RM 10.5, and in this exposure area, the exposure estimate is driven by a single extreme outlier, indicating that most Study Area fish are not exposed to lead levels that would pose a risk to osprey. The maximum smallmouth bass concentration (1,100 mg/kg ww) detected in this exposure area is over 100 times greater than 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 in all other smallmouth bass samples, where lead was detected and concentrations ranged from 0.0048 to 1.8 mg/kg ww. HQ would be < 1.0 if smallmouth bass represented < 1% of the diet, which is a reasonable assumption.	The selected LOAEL (3.26 mg/kg bw/day) is based on an Eco SSL (EPA 2005e) and is an order of magnitude lower than the lowest acceptable literature-based LOAEL (20 mg/kg bw/day) (Attachment 14).	Nesting success of osprey was monitored along the Willamette River system between 1993 and 2001 (Henny et al. 2008). Between RM 0 and RM 26, the number of osprey nests increased from 1993 to 2001; one nest was observed in 1993 and ten nests were observed in 2001. The productivity of osprey in this section of the Willamette River in 2001 (1.75 young per all types of nest (occupied, active, and successful) was similar to the productivity of osprey in upstream sections of the Willamette River (1.77 young/active nest). Production rates of 0.7 and 0.8 young/nest have been reported to be the minimum required rate to maintain stable bald eagle and osprey populations (Henny et al. 2008; Wiemeyer et al. 1984). These data indicate that osprey populations from the LWR (including the Study Area) have increased in recent years, with productivity above that which is necessary for maintaining a stable population. Although lead was identified as a COC under the assumptions required for the BERA, it is more reasonable to conclude that lead does not pose unacceptable risk to the osprey population.
Mink	Dietary dose	4.0 (RM 9.5 – RM 10.5)	The spatial extent of potential risk is limited to RM 9.5 to RM 10.5, and in this exposure area, the exposure estimate is driven by a single extreme outlier, indicating that most Study Area fish are not exposed to lead levels that would pose risk to mink. The maximum smallmouth bass concentration (1,100 mg/kg ww) detected in this exposure area is over 100 times greater than 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 all other smallmouth bass samples (0.0048 to 1.8 mg/kg ww).	The selected LOAEL (8.9 mg/kg bw/day) is based on an Eco SSL (EPA 2005e) derived from drinking water exposure and is an order of magnitude lower than the only dietary literature-based LOAEL (90 mg/kg bw/day) (Attachment 14). The extrapolation from drinking water to dietary dose is uncertain.	The question of whether lead exposure poses unacceptable risk to mink in the Study Area hinges on the true representativeness of the extreme smallmouth bass outlier. Even if lead does pose unacceptable risk, the limited spatial extent indicates that the risk would not rise to the population level. Because the tissue chemical concentration in that outlier is so much higher than all other sample concentrations, it is possible that the source of the lead is discrete (e.g., possibly resulting from a lead sinker) rather than dispersed in sediment or water. Although lead was identified as a COC under the assumptions required for the BERA, it is more reasonable to conclude that lead does not pose unacceptable risk to the mink population.

Table 8-43. Summary of Wildlife COCs and Risk Conclusions Across LOEs

Receptor by COC	LOE	HQ[a]	Exposure Considerations	Effects Considerations	Risk Conclusions
Mercury					
Bald eagle	Dietary dose	1.2 – 1.7 (all exposure areas)	The HQ assumes that the bald eagle subsists exclusively on fish from the Study Area. The 100% site-use factor is not necessarily a conservative assumption; mercury concentrations are higher in upriver fish tissue samples than in the Study Area for species where upriver data are available (Section 7.1.4.3). Willamette-wide average mercury concentrations (as reported in Hope (2003)) are approximately two to six times higher than three of the species (i.e., carp, largescale sucker, and northern pikeminnow) from the Study Area assumed to make up 95% of the bald eagle diet.	The selected NOAEL was extrapolated from a LOAEL using uncertainty factor of 10.	The risk conclusions from the highly uncertain egg LOE are not inconsistent with the dietary-dose LOE. Given the availability of the more reliable dietary-dose LOE that assesses the same risk, this LOE should be given minimal consideration. Based on the dietary-dose LOE, is it more reasonable to conclude that mercury may pose low risks to individual bald eagles. It is should be noted that mercury exposure in the Study Area does not appear to be elevated relative to the upriver reach of the LWR and that average Willamette River-wide fish tissue mercury concentrations are greater than average Study Area fish tissue concentrations.
	Egg	< 1.0	The BMF used to estimate bird egg concentrations is highly unreliable, and given the availability of a more reliable LOE that assesses the same risk, this LOE should be given minimal consideration.	The selected NOAEL is based on field studies that were inadmissible for TRV development in the tissue LOEs for fish and benthic invertebrates.	
Benzo(a)pyrene					
Spotted sandpiper	Dietary dose	1.6 (RM 4.0 – RM 6.0)	Dietary exposure only slightly exceeds the LOAEL in RM 4.0 to RM 6.0 if spotted sandpiper are assumed to consume a diet composed solely of worms. The dietary-dose estimates associated with a diet composed of a mixture of clams and worms are below the LOAEL.	The selected LOAEL is based on the only LOAEL reported in the literature, in which pigeons were exposed to weekly intramuscular injections for five months (Hough et al. 1993). The extrapolation from an injection exposure to dietary dose is uncertain.	There was limited spatial extent of exceedances and a low magnitude of exceedance based on a conservative dietary assumption (i.e., that lab worms represent the entire spotted sandpiper diet; however, it was not possible to determine the likelihood of unacceptable risk to shorebirds from benzo(a)pyrene because of high uncertainties associated with the selected LOAEL.
Dibutyl Phthalate					
Spotted sandpiper	Dietary dose	1.4 (RM 7.0 – RM 9.0)	Dietary exposure only slightly exceeds the LOAEL in RM 7.0 to RM 9.0 if spotted sandpiper are assumed to consume a diet composed solely of clams. The dietary-dose estimates associated with a diet composed of a mixture of clams and worms are below the LOAEL. The dibutyl phthalate concentration in background sediments is higher than the average dibutyl phthalate concentration in the Study Area (Attachment 11).	The selected LOAEL was extrapolated from a NOAEL using an uncertainty factor of 10.	Although dibutyl phthalate was identified as a COC under the assumptions required for the BERA, it is more reasonable to conclude that dibutyl phthalate does not pose unacceptable risk to the spotted sandpiper population based on the conservatism in the dietary assumption, limited spatial extent of exceedance, and uncertainty associated with the extrapolated LOAEL. It also should be noted that sediment dibutyl phthalate concentrations are not elevated relative to background.
Total PCBs					
Spotted sandpiper	Dietary dose	1.7 – 12 (all exposure areas)	Site-wide exceedances assume that spotted sandpiper consume a diet composed solely of worms. The dietary-dose estimates associated with a diet composed solely of clams yield an HQ > 1.0 only in RM 7.0 to RM 9.0 (HQ = 2.2).	The selected LOAEL (0.58 mg/kg bw/day) is based on the reduced hatchability of chicken chicks (Britton and Huston 1973); chickens have high sensitivity to PCB toxicity compared with other species tested (Attachment 14).	Dietary exposure in spotted sandpiper probably exceeds the LOAEL over some of the shorebird foraging areas, and it is more reasonable to conclude that it exceeds LOAEL over all foraging areas. The effects associated with those exceedances are uncertain, but it is possible that there would be some reduction in reproductive success. Whether that would cause population-level risks to spotted sandpiper is uncertain.

Table 8-43. Summary of Wildlife COCs and Risk Conclusions Across LOEs

Receptor by COC	LOE	HQ[a]	Exposure Considerations	Effects Considerations	Risk Conclusions
Hooded merganser	Dietary dose	1.5 (RM 1.5 – RM 2.5) 1.8 (RM 6.5 – RM 7.5) 3.8 (RM 10.5 – RM 11.8)	Exceedances occurring over a limited spatial extent assume the fidelity of individual mergansers to small foraging areas.	The selected LOAEL (0.58 mg/kg bw/day) is based on the reduced hatchability of chicken chicks (Britton and Huston 1973); it is possible that the TRV for hooded merganser has been overestimated by an order of magnitude. The LOAEL for mallards (15 mg/kg dw/day) is approximately 30 times greater than the selected LOAEL (Attachment 14).	Total PCBs pose negligible risk to the hooded merganser population based on low LOAEL exceedances over a limited spatial extent and likely conservatism in the TRV.
Bald eagle	Dietary dose	3.8 – 3.9 (all exposure areas)	The exposure analysis assumed that an individual bald eagle subsists exclusively on fish from the Study Area.	The selected NOAEL is based on field studies that were inadmissible for TRV development in the tissue LOEs for fish and benthic invertebrates.	Individual bald eagles that forage exclusively on aquatic fish from the Study Area may face some increased risk of reduced reproductive success. The risk conclusions from the highly uncertain egg LOE are not inconsistent with the dietary-dose LOE. Based on the availability of the more reliable dietary-dose LOE that assesses the same risk, this LOE should be given minimal consideration.
	Egg	26 (all exposure areas)	The BMF used to estimate bird egg concentrations is highly unreliable, and based on the availability of a more reliable LOE that assesses the same risk, this LOE should be given minimal consideration.	The selected NOAEL is based on field studies that were inadmissible for TRV development in the tissue LOEs for fish and benthic invertebrates.	
Osprey	Dietary dose	1.1 (RM 10.5 – RM 11.8)	The osprey exposure assumptions used in the BERA are reliable values based on species- and region-specific literature. HQ would be < 1.0 if smallmouth bass or carp did not represent any portion of the osprey diet.	The selected LOAEL is based on field studies that were inadmissible for TRV development in the tissue LOEs for fish and benthic invertebrates.	Total PCBs pose low to negligible risk to osprey foraging from within a single localized area (RM 10.5 to RM 11.8) based on the dietary-dose LOE. The spatial extent of LOAEL exceedance is limited to RM 10.5 to RM 11.8, suggesting that total PCBs does not pose unacceptable risk to the Study Area osprey population. The risk conclusions from the highly uncertain egg LOE are not inconsistent with the dietary-dose LOE. Given the availability of the more reliable dietary-dose LOE that assesses the same risk, this LOE should be given minimal consideration.
	Egg	4.5 – 5.7 (all exposure areas)	The BMF used to estimate bird egg concentrations is highly unreliable, and based on the availability of a more reliable LOE that assesses the same risk, this LOE should be given minimal consideration.	The selected LOAEL is based on field studies that were inadmissible for TRV development in the tissue LOEs for fish and benthic invertebrates.	
Mink	Dietary dose	19 – 33 (all exposure areas)	Mink is an opportunistic feeder; changing assumed dietary portions could change the magnitude of HQs, but total PCBs would still be identified as a COC (see further evaluation of mink diet in Section 8.2.3.2).	The lowest literature-based LOAELs range from 0.037 to 0.077 mg/kg bw/day, in which adverse reproductive effects (including reduced kit body weight, delay in the onset of estrus, and reduced whelping success) were observed in mink fed field-collected carp from the Great Lakes region over a chronic period. Field-collected carp from the Great Lakes region also had reported concentrations of other organic chemicals (e.g., dioxin/furans, DDE, DDD, chlordane); however, LOAELs based on these studies are consistent with the mink reproduction effect threshold of 0.089 mg/kg bw/day, which was based on a study in which mink were fed a prepared laboratory diet for 18 months. (2001).	It is reasonable to assume, based on the information available for the BERA, that where mink occur in the Study Area, they could be at risk from exposure to PCBs and that the level of risk is sufficient to affect reproductive success in a Study Area mink population.
River otter	Dietary dose	21 – 31 (all exposure areas)	River otter is an opportunistic feeder; changing assumed dietary portions could change magnitude of HQs but total PCBs would still be identified as a COC.	The effect considerations for river otter mirror those for mink. Uncertainty is increased because of interspecies extrapolation.	It is reasonable to assume, based on the information available for the BERA, that river otter in the Study Area are at risk from exposure to PCBs and that the level of risk is sufficient to affect reproductive success in a Study Area river otter population. If it is assumed that mink and river otter consume essentially the same diet, then mink provide a conservative surrogate for river otter because the mink has higher metabolic requirements (i.e., mink eat more pound-for-pound than do river otter).

Table 8-43. Summary of Wildlife COCs and Risk Conclusions Across LOEs

Receptor by COC	LOE	HQ[a]	Exposure Considerations	Effects Considerations	Risk Conclusions
Total TEQ					
Spotted sandpiper	Dietary dose	11 (RM 1.9 – RM 3.9); 1.3 (RM 4.0 – RM 6.0); 20 (RM 7.0 – RM 9.0)[b]	Site-wide exceedances assume that spotted sandpiper consume a diet composed solely of worms. Worm prey concentrations are based on UCLs that included BSAR-predicted tissues. Adjusted steady-state lab worm tissues from bioaccumulation tests may not accurately represent prey concentrations.	The selected LOAEL is based on the lowest LOAEL reported in the literature, in which ring-necked pheasants were exposed to weekly intraperitoneal injections for 10 weeks (Nosek et al. 1992). No literature-based dietary LOAELs were available (Attachment 14). The extrapolation from an injection exposure to dietary dose is uncertain.	It was not possible to determine the likelihood of unacceptable risk to shorebirds from total TEQ because of high uncertainties associated with the selected LOAEL.
Bald eagle	Dietary dose	< 1.0	Carp is the only bald eagle prey item for which total TEQ data are available. The exposure analysis assumed that an individual bald eagle subsists exclusively on fish from the Study Area.	The dietary effect considerations for bald eagle mirror those for spotted sandpiper.	The egg LOE is highly uncertain and should not be relied upon because a more reliable dietary-dose LOE is available. TEQ risks to individual bald eagles are negligible based on the dietary-dose LOE; however, it was not possible to determine likelihood of unacceptable risk to bald eagles from total TEQ because of high uncertainties associated with the selected NOAEL.
	Egg	280[c] (all exposure areas)	The BMF used to estimate egg concentrations is unreliable, and based on the availability of a more reliable LOE that assesses the same risk, this LOE should be given minimal consideration. Carp is the only bald eagle prey item for which total TEQ data are available. The exposure analysis assumed that an individual bald eagle subsists exclusively on fish from the Study Area.	The selected NOAEL is based on field studies that were inadmissible for TRV development in the tissue LOEs for fish and benthic invertebrates.	
Osprey	Dietary dose	< 1.0	The osprey exposure assumptions used are reliable values based on species- and region-specific literature.	The dietary effect considerations for osprey mirror those for spotted sandpiper.	The egg LOE is highly uncertain and should not be relied upon because a more reliable dietary-dose LOE is available. TEQ risks to the osprey population are negligible based on the dietary-dose LOE; however, it was not possible to determine likelihood of unacceptable risk to osprey from total TEQ because of high uncertainties associated with the selected LOAEL. Furthermore, data from Henny et al. (2008) indicate that osprey populations from the LWR (including the Study Area) have increased in recent years, with productivity above that which is necessary for maintaining a stable population.
	Egg	19 – 25[d] (all exposure areas)	The BMF used to estimate egg concentrations is unreliable, and based on the availability of a more reliable LOE that assesses the same risk, this LOE should be given minimal consideration. TEQ data are not available for largescale sucker, an important osprey prey item, and it is unknown whether largescale sucker concentrations would be different from carp concentrations.	The selected LOAEL is based on field studies that were inadmissible for TRV development in the tissue LOEs for fish and benthic invertebrates.	

Table 8-43. Summary of Wildlife COCs and Risk Conclusions Across LOEs

Receptor by COC	LOE	HQ[a]	Exposure Considerations	Effects Considerations	Risk Conclusions
Mink	Dietary dose	1.6 – 12[e] (all exposure areas)	Mink is an opportunistic feeder; changing the assumed dietary portions could change range of HQs; however, total TEQ would still be identified as a COC. The use of mammal TEFs is a source of uncertainty in TEQ exposure estimates.	The selected LOAEL (2.24 ng/kg bw/day) is based on mink reproduction (Tillitt et al. 1996). Literature based LOAELs range from 2.24 to 3.6 ng/kg bw/day where adverse reproductive effects (including reduced kit body weight and kit survival) were observed in mink fed diets with varying percentages of field-collected carp from Saginaw Bay, Lake Huron for 182 days. Field-collected carp from Saginaw Bay also had reported concentrations of other organic chemicals (e.g., PCBs, DDE, DDD, chlordane); however, LOAELs based on these studies are consistent with the effect threshold (2.34 ng/kg bw/day) based on mink reproduction where mink were fed a prepared laboratory diet for 18 months. (2001).	Total TEQ HQs indicate unacceptable risks to the mink population, but total PCB HQs are higher. The dioxin/furan TEQ HQ was > 1.0 in only one exposure area (RM 6.5 to RM 7.5), where the HQ was 2.0. PCB TEQ HQs ranged from 1.4 to 2.4 in all exposure areas. The maximum total TEQ HQ includes a sediment HQ of 8.6 that is based on a sediment UCL greater than the maximum detected concentration. This statistical anomaly can be avoided by summing the PCB and dioxin TEQs for that exposure area (RM 6.5 to RM 7.5). This would result in a total TEQ of 3.5, which would still be the maximum total TEQ by exposure area (see Table 8-34).
River otter	Dietary dose	1.1 to 2.3[f] (all exposure areas)	River otter is an opportunistic feeder; changing the assumed dietary portions could change range of HQs; however, total TEQ would still be identified as a COC. The use of mammal TEFs is a source of uncertainty in TEQ exposure estimates.	The effect considerations for river otter mirror those for mink. Uncertainty is increased due to interspecies extrapolation.	Total TEQ is expected to pose a low risk to river otter that forage throughout the Study Area. Dioxin/furan TEQ HQs were all < 1.0. PCB TEQ HQs were 1.3 and 1.5 in the exposure areas between RM 1.5 and RM 4.5 and above RM 10.5, respectively.
DDx Compounds					
Spotted sandpiper	Dietary dose	1.3 – 1.4[g] (RM 7.0 – RM 9.0)	Dietary exposure only exceeds the LOAEL TRV in RM 7.0 to RM 9.0 if spotted sandpiper are assumed to consume a diet composed solely of worms. The dietary-dose estimates associated with a diet composed of clams are below the LOAEL.	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, LOAEL may overpredict risks to populations, because reproductive effects on field populations of birds are not documented for eggshell thinning of < 15-20% (Attachment 14).The selected sum DDE LOAEL is the lowest LOAEL reported in the literature (Attachment 14).	Although total DDx and sum DDE were identified as a COC under the assumptions required for the BERA, it is more reasonable to conclude that DDx compounds do not pose unacceptable risk to the spotted sandpiper population based on the conservatism in the dietary assumption, limited spatial extent of exceedance, and conservatism associated with the LOAEL.
Bald eagle	Dietary dose	< 1.0[h]	The exposure analysis assumed that an individual bald eagle subsists exclusively on fish from the Study Area.	The selected sum DDE LOAEL (0.27 mg/kg bw/day) is based on Eco SSL (EPA 2007b) and is consistent with the lowest acceptable literature based NOAEL (0.18 mg/kg bw/day), where eggshell thinning was not statistically different from the control group (1974).	The dietary-dose LOE indicates that there is no unacceptable risk to bald eagles from exposure to DDx compounds. The egg LOE is highly uncertain and should not be relied upon because a more reliable dietary-dose LOE is available.
	Egg	9.5 – 11[i] (all exposure areas)	The BMF used to estimate bird egg concentrations is highly unreliable, and based on the availability of a more reliable LOE that assesses the same risk, this LOE should be given minimal consideration.	The selected 4,4′-DDE NOAEL is based on field studies that were inadmissible for TRV development in the tissue LOEs for fish and benthic invertebrates.	

Table 8-43. Summary of Wildlife COCs and Risk Conclusions Across LOEs

Receptor by COC	LOE	HQ[a]	Exposure Considerations	Effects Considerations	Risk Conclusions
Osprey	Dietary dose	< 1.0[h]	The osprey exposure assumptions used are reliable values based on species- and region-specific literature.	The selected sum DDE 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, LOAEL may overpredict risks to populations, because reproductive effects on field populations of birds are not documented for eggshell thinning of < 15-20% (Attachment 14).	The dietary-dose LOE indicates that there is no unacceptable risk to osprey from exposure to DDx compounds. The egg LOE is highly uncertain and should not be relied upon because a more reliable dietary-dose LOE is available.
	Egg	3.3 – 4.2[i] (all exposure areas)	The BMF used to estimate bird egg concentrations is highly unreliable, and based on the availability of a more reliable LOE that assesses the same risk, this LOE should be given minimal consideration.	The selected 4,4′-DDE LOAEL is based on field studies that were inadmissible for TRV development in the tissue LOEs for fish and benthic invertebrates.	
Aldrin					
Spotted sandpiper	Dietary dose	1.7 (RM 7.0 – RM 9.0)	Dietary exposure only exceeds the LOAEL in RM 7.0 to RM 9.0 if spotted sandpiper are assumed to consume a diet composed solely of worms. Adjusted steady-state lab worm tissues from bioaccumulation tests may not accurately represent prey concentrations.	Limited toxicological data are available for assessing aldrin risks to birds.	Although aldrin was identified as a COC under the assumptions required for the BERA, it is more reasonable to conclude that aldrin does not pose unacceptable risk to the spotted sandpiper population based on the conservatism in the dietary assumption (exclusive consumption of worms), and the low magnitude and limited spatial extent of exceedance.

[a] HQ is based only on exposure areas where HQs are > 1.0. HQs are < 1.0 in all other exposure areas. HQs are based on LOAEL TRVs for all receptors except bald eagle, for which HQs are based on NOAEL TRVs.
[b] Dioxin/furan TEQ HQ is > 1.0 (HQ = 17) in only one exposure area (RM 7.0 to RM 9.0); PCB TEQ HQs are > 1.0 in two exposure areas (HQs are 10 and 11 in RM 1.9 to RM 3.9 and RM 7.0 to RM 9.0, respectively).
[c] Dioxin/furan TEQ and PCB TEQ HQs were 59 and 240, respectively, in all exposure areas.
[d] Dioxin/furan TEQ and total PCB TEQ HQs ranged from 3.9 to 4.5 and 16 to 22, respectively, in all exposure areas.
[e] Dioxin/furan TEQ HQ was 2.0 in one exposure area (RM 6.5 to RM 7.5). PCB TEQ HQs ranged from 1.4 to 2.4 in all exposure areas.
[f] Dioxin/furan TEQ HQs were all < 1.0. PCB TEQ HQs were 1.3 and 1.5 in the exposure areas between RM 1.5 and 4.5 and above RM 10.5, respectively.
[g] HQs based on sum DDE and total DDx.
[h] HQ based on sum DDE.
[i] HQ based on 4,4′-DDE.

BERA – baseline ecological risk assessment
BMF – biomagnification factor
BSAR – biota-sediment accumulation regression
COC – chemical of concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
Eco SSL – ecological screening level
EPA – US Environmental Protection Agency
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
LOE – line of evidence
LWR – Lower Willamette River
NOAEL – no-observed-adverse-effect level
PCB – polychlorinated biphenyl
TEQ – toxic equivalent
TRV – toxicity reference value
RM – river mile
TEF – toxic equivalency factor
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)
UCL – upper confidence limit on the mean

8.2.3.2 Mink and PCB Evaluation

Because risks to mink from PCBs represent the single highest probability of a chemical affecting an ecological receptor population in the Study Area, 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 mink and river otter have not been reported in the Study Area, it offers at least marginally suitable habitat, and both species have been collected from nearby locations on the Multnomah Channel and Columbia River (including RM 90, RM 108, and RM 119.5 of the Columbia River near Portland[128]) (Elliott et al. 1999; Henny et al. 1996).

8.2.3.2.1 Evaluation of Exposure Assumptions

The following subsection examines the exposure assumptions used in the mink PCB evaluation. Results of site-specific exposure modeling combined with literature-reported effects data indicate a reasonable probability, though not certainty, of risk to the mink population in the Study Area. Estimates of risk to mink from PCBs were determined by comparing their exposure as characterized by a conservative estimate of the average daily chemical dose with the daily chemical dose that has been reported in the non site-specific toxicological literature to be associated with adverse effects on mink.[129] 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. Mink are opportunistic feeders, consuming a range of prey, such as muskrats, fish, frogs, crayfish, small mammals, and birds found near water (Csuti et al. 2001). A study of mink diets from the Columbia River upstream from the Willamette River[130] reported that birds, mammals, and fish were consumed at similar frequencies; crayfish were consumed at a higher frequency (WDG 1980).[131] For this BERA, it was conservatively assumed that the mink diet consists solely of fish and crayfish from the Study Area, although (WDG 1980) reported that remains of fish or crayfish were found in only 32 and 47% of mink feces, respectively. The literature also notes that mink diets vary seasonally; their diets may shift towards fish in the winter and birds in the summer. The default dietary

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[128] The mouth of the LWR is located at RM 101 of the Columbia River.

[129] In Section 8.1, the dietary risk equation was mathematically rearranged so that LOAELs were expressed as threshold tissue and sediment chemical concentrations (TTC and TSC, respectively) 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.

[130] Data were collected between RM 107 and RM 546 of the Columbia River including those from 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.

[131] Data were based on the percent frequency of occurance 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.

assumption used in the wildlife risk evaluation of mink, based on the relative abundance of fish and crayfish in the Study Area, is that mink eat equal portions of a diet composed entirely of crayfish, largescale sucker, carp, sculpin, and smallmouth bass (i.e., that each component is 20% of the diet).[132]

Table 8-44 presents mink exposure to PCBs in the diet (expressed as a dose), the selected dose-based LOAEL, and resulting HQs within 1-mile exposure areas of the Study Area. Calculated doses for all exposure areas were greater than the selected LOAEL.

Table 8-44. 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

To evaluate the potential for risks 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. Assuming 33% ingestion each of juvenile Chinook salmon, smallmouth bass, and brown bullhead,[133] an exposure dose of

[132] 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.

[133] Smallmouth bass, brown bullhead, and juvenile Chinook salmon EPCs were estimated as the maximum concentration for each species because limited data were available (n < 6) for each species.

0.021 mg/kg bw/day was calculated. An HQ of 0.57 was calculated based on this upriver reach dose (0.021 mg/kg bw/day) and a LOAEL of 0.037 mg/kg bw/day. Thus, negligible risks to mink are expected on the basis of the limited data from the upriver reach; however, it should be noted that tissue data are not available for most of the predominant mink prey species (i.e., carp, largescale sucker, crayfish, and sculpin). Thus 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, all of which may potentially represent mink prey) served as the sole food item in this analysis. Results are shown for each 1-mile exposure area in Table 8-45 (mink dietary PCB doses) and Table 8-46 (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. It should be noted that the dietary doses and HQs presented in Tables 8-45 and 8-46, respectively, do not account for exposure to PCBs via incidental ingestion of sediment. Incidental ingestion of sediment in mink was assumed to be about 9% of dietary intake on a mass basis, based on the foraging habits of raccoons (Beyer et al. 1994). However, sediment represents a trivial fraction of overall exposure (i.e., sediment contributes < 1% of the exposure dose and HQ), and therefore, the exposure doses and HQs are expected to be approximately the same whether ingestion of incidental sediment is included or excluded. Reducing the sediment ingestion rate would not affect risk estimates.

Table 8-45. 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
DL – detection limit
NA – not applicable
ND – no data available
PCB – polychlorinated biphenyl
RM – river mile

Italics identify dietary doses derived from maximum concentration; UCL concentration could not be derived (n detects < 6).

Table 8-46. 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.

DL – detection limit
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 > 1.0.

Dietary doses and HQs associated with any prey portion assumption can be determined from the information presented in Tables 8-45 and 8-46 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 8-44, again showing that HQs >1.0 are predicted for all 1-mile exposure areas.

Because mink are opportunistic hunters, which eat more abundant prey more frequently than less abundant prey (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 used in this sensitivity analysis are presented in Table 8-46. These were used to graph the predicted ranges of HQs for mink (Figure 8-4). For a given 1-mile exposure area, the minimum and maximum HQs were calculated by selecting the prey portion for each prey species in the range shown in Table 8-46 that resulted in the highest and lowest HQ. Figure 8-4 shows that the HQs for all exposure areas are always > 1.0. The default diet HQs consistently are near the high end of the ranges found through this sensitivity analysis (Figure 8-4). This is because the highest PCB tissue concentrations occurred in the prey species that have the highest maximum relative abundance (i.e., carp, crayfish, largescale sucker, northern pikeminnow, peamouth, sculpin, and smallmouth bass).

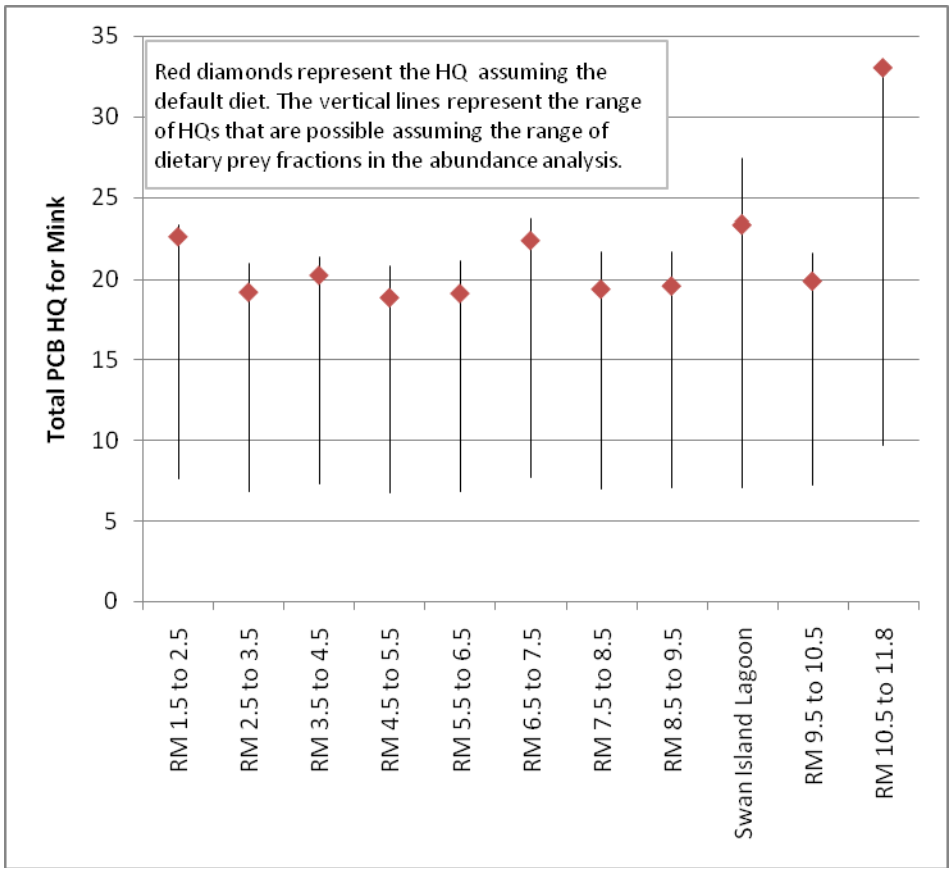

Figure 8-4. Range of Total PCB Mink HQs Based on Abundance of Mink Prey Species

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 mink diets (5% of diet for carp and 35% of diet for largescale sucker) would result in HQs > 1.0 throughout the Study Area (i.e., HQ in all exposure areas would be 6.5). 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 site use factor of 1.0 was assumed for mink to calculate 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 birds and mammals mink might consume are not entirely dependent on the Study Area, consumption of such birds and mammals would reduce Study Area-related risks 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 is composed of 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 food ingestion rates and body weights. HQs increase with food ingestion rate and decrease with body weight; however, because food ingestion rate is a function of body weight, assuming different body weights does not affect risk estimates. No additional food ingestion rates other than 16% body weight per day were identified for female mink; food ingestion rates for male mink ranged from 12 to 22% body weight per day (EPA 1993). These food ingestion rates bracket those assumed for female mink, so different food ingestion rates could result in higher or lower risks.

Based on 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 site use factor still results in estimated risks to mink from PCB exposure.

8.2.3.2.2 Evaluation of Selected LOAEL TRV

The following subsection evaluates the selected TRV study that was used to derive the LOAEL by which risks to mink from PCBs were 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 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 rates 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/d); adverse effects on adult mink were generally not observed until much higher dosages (0.32 to 2000 mg/kg bw/day).

Contrary to other receptors and COCs for which toxicological data specific to the receptor are generally lacking, literature-reported adverse effects data specific to mink and PCBs are fairly numerous. 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 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; Brunström et al. 2001; Ringer 1983; Wren et al. 1987). 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 8-5. 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 0.057 mg/kg bw/day. Calculated PCB doses for mink from the Study Area (Table 8-44; Figure 8-5) 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.

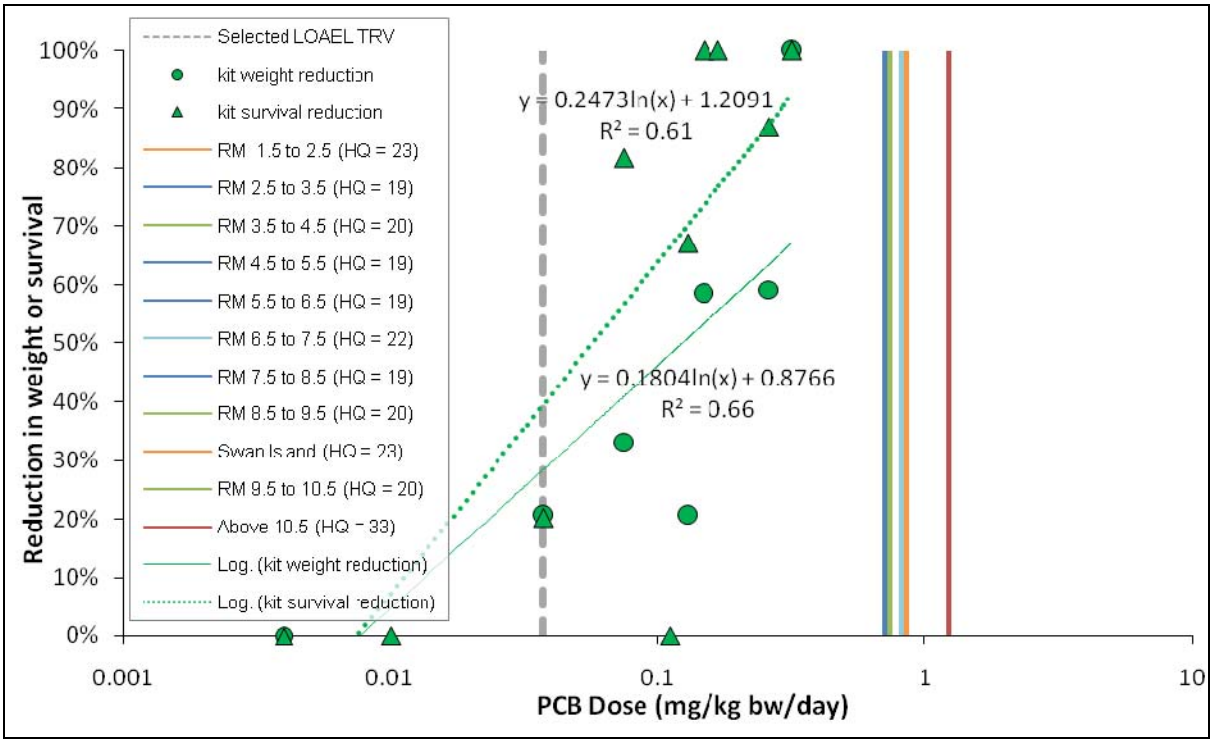

Figure 8-5. Toxicity of PCBs on Mink Kit Body Weight and Survival 5 or 6 Weeks after Whelping

8.2.3.2.3 Conclusions

Based on this analysis, it is apparent that Study Area PCB concentrations result in risk to mink, even when alternative exposure parameters are evaluated. The exceedance of the LOAEL TRV is expected to result in population-level risks to mink because the LOAEL is associated with adverse effects on mink reproduction, which is likely to affect the overall health of the mink population. However, mink prefer riparian cover for foraging habitat, so industrialized areas within the Study Area likely represent poor habitat quality, resulting in

overestimates of exposure. Field studies in the LCR have indicated the presence of river otter, but declines in mink populations despite the presence of habitat areas present in the LCR. The impact of mink risks from PCBs on overall ecological risk conclusions is discussed 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 screening-level thresholds that were 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[134] during the 2002 Portland Harbor amphibian and aquatic plant reconnaissance survey (Integral et al. 2004a). The life stage likely to be affected is the larval stage when eggs hatch in March through April.

The ecological CSM (Figure 3-2) classified ingestion of prey, direct contact with surface water, and direct contact with TZW as complete and significant exposure pathways for amphibians.[2] 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 is limited. No attempt was made to further characterize TZW exposure; accordingly, results of the TZW assessment were not incorporated into the risk conclusions section.

Although the CSM identifies dietary exposure as a complete and significant exposure pathway for amphibians (Figure 3-2), this pathway was not evaluated per EPA's Problem Formulation (Attachment 2) and because dietary-based TRVs for quantitative assessment are not available. In addition, exposure to Study Area-contaminated prey may be reduced because of the red-legged frog's reliance on riparian habitat and terrestrial prey for portions of its life. Although highly associated with aquatic habitats, adults of some species or subspecies of red-legged frogs often live for extended periods (i.e., months) in dense riparian habitats, feeding on mice, other amphibians (most often tree frogs), and terrestrial insects (Hayes and Tennant 1985). The evaluation of other small-home-range receptors (e.g., sculpin) is likely to be protective of the amphibian dietary exposure pathway (the home range for amphibians is wide, with frog movement up to 2 miles (USFWS 2002), whereas the sculpin has a much smaller home range, as presented in Section 7.0).

9.1 SURFACE WATER ASSESSMENT

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 amphibians. All surface water samples taken from amphibian habitat areas were evaluated (Map 9-1). The comparison of surface water concentrations to water TRVs was conducted on an individual sample basis per EPA direction (Attachment 2). COPCs with HQs > 1.0 for any individual surface water sample within amphibian habitat areas were identified as COCs. Surface

[134] 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.

water concentrations were also evaluated on a site-wide basis including all samples location within amphibian habitat areas. For all COCs, the spatial distribution and magnitude of HQs, the seasonal and sampling method patterns of HQs, and the associated exposure and effects assumptions were evaluated to provide a more detailed assessment of impacts on amphibian populations. The evaluation of COCs and associated uncertainties were further evaluated to arrive at risk conclusions for amphibians (Section 9.3).

The details of this assessment are presented as follows:

- Section 9.1.1 presents a summary of the COPCs evaluated.
- Section 9.1.2 presents an overview of how exposure concentrations were derived. All surface water chemical concentrations are presented in Attachment 4.
- Section 9.1.3 presents a summary of the effects data. Details on the development of the water TRVs are presented in Attachment 10.
- Section 9.1.4 presents the risk characterization results, COC-receptor pairs, and associated uncertainties. These COCs are further assessed in the amphibian risk conclusions section (Section 9.3).

Figure 9-1 presents a flowchart of the surface water assessment section organization.

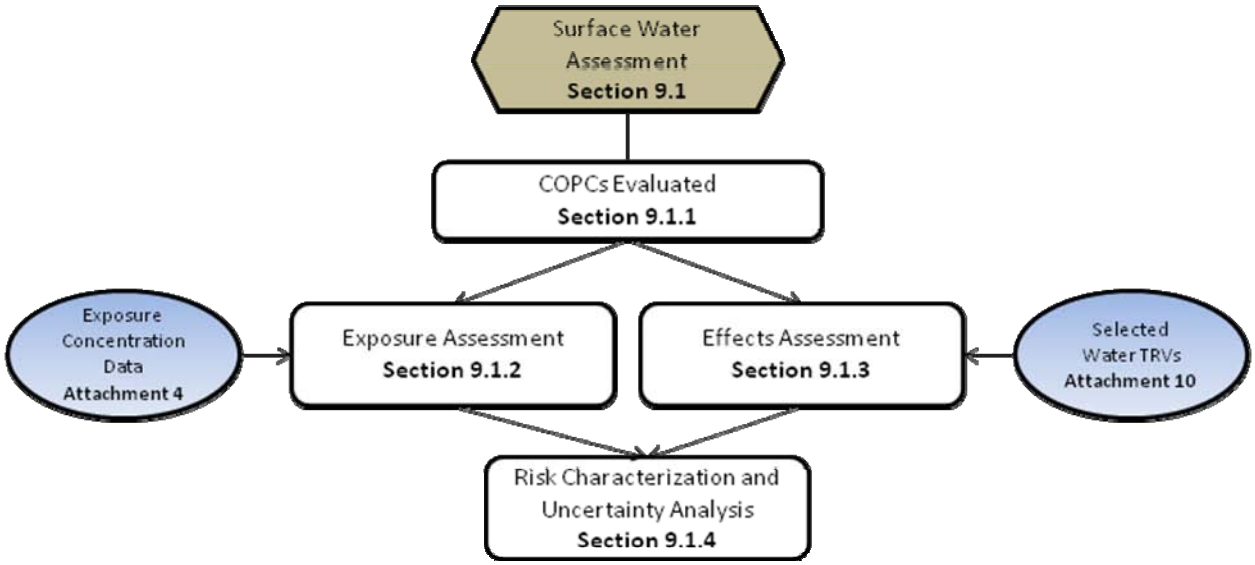

Figure 9-1. Overview of Amphibian Surface Water Section Organization

9.1.1 COPCs Evaluated

Twelve[135] surface water COPCs were identified based on the results of the SLERA and refined screen (Attachment 5). These surface water COPCs are presented in Table 6-26. Four surface water COIs could not be evaluated because no toxicological data were available. These COIs (4-chloroaniline, aniline, 2,4-DB, and MCPP) were infrequently detected and were detected in isolated areas and during different sampling periods.

9.1.2 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 from the study are presented in the benthic risk assessment (Section 6.5.2).

9.1.2.1 Exposure Areas

For amphibians, only surface water samples collected within quiescent areas or amphibian/aquatic plant habitat areas were used to represent EPCs for amphibians, as identified on Map 9-1. Quiescent areas were defined as those areas in the Study Area potentially capable of sustaining aquatic plant growth because of slow moving, shallow water. Amphibian habitat areas were determined during a reconnaissance survey conducted in the summer of 2002 to confirm the presence of amphibians and aquatic plants and to delineate potential amphibian/aquatic plant habitats within the Initial Study Area (RM 3.5 to RM 9.2). Amphibian habitat areas were identified as areas with low-sloping beaches and riprapped or rocky banks. During the survey, the northern red-legged frog was observed and the Pacific tree frog call was heard (Integral et al. 2004a). Details of the 2002 Portland Harbor amphibian and aquatic plant reconnaissance survey are presented in Integral et al. (2004a).

9.1.2.2 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). Ponds, pools, or other quiescent water are typical breeding sites. Eggs are often laid on submerged or emergent vegetation. Amphibians are exposed to surface water chemical concentrations at an early life stage (eggs) because an aquatic habitat is required for laying eggs. Tadpoles also represent a sensitive life stage for amphibians. Red-legged frog tadpoles take 11 to 20 weeks to develop into terrestrial adults (USFWS 2002), so the estimated exposure period for tadpoles is from spring to summer. During wet weather,

[135] Three individual DDT metabolites identified in the SLERA (2,4′-DDD, 2,4′-DDT, and 4,4′-DDD) were evaluated as total DDx and were not evaluated individually. The compound 4,4′-DDT was evaluated both individually and as total DDx because the TRV for DDx is based on 4,4′-DDT.

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 still 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 exposure concentrations for amphibians 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.

There is additional uncertainty 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.

9.1.2.3 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 – Aluminum, zinc, and butyltin were only analyzed in peristaltic pump samples. Aluminum (total fraction) was detected in all samples within amphibian exposure areas at concentrations of 41 to 1,820 µg/L, and zinc (dissolved) in about half at concentrations of 0.9 to 41.9 µg/L. Butyltin 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, and the maximum aluminum concentration was collected during a January high-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 detection limits. Detected concentrations of PAHs were substantially higher in the peristaltic pump samples (mean concentrations ranged from 32 to 76,000 ng/L) than the XAD samples (mean concentrations ranged from 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 were in peristaltic samples collected for naphthalene at RM 6.4 property 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 detection limits. 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, where ethylbenzene was detected (concentration was 3.46 µg/L) and trichloroethene was not detected. There is uncertainty in the exposure data for VOCs because they were analyzed in only one sample within an amphibian exposure area (located at RM 6.4 on the west bank of the Study Area) during one sampling event in May 2005. Data from one sample do 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 detection limits. 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 1999). The qualification indicates that the analyst believed that the result was the result of 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).

9.1.2.4 Surface Water EPCs

Surface water EPCs in this assessment were 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 habitat areas. Site-wide amphibian habitat UCLs were calculated for all surface water COPCs, with the exception of ethylbenzene and trichloroethene. These COPCs were not evaluated on a

site-wide basis because VOCs were only analyzed in one sample located within an amphibian habitat area collected between RM 5.5 and 6.5. 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 detection limits and higher accuracy of results. 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. 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. 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.

Surface water concentrations 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

There are uncertainties associated with the use of multiple sampling types and methods (e.g., duration of sampling time for a single-point grab sample versus an integrated transect sample or volume of water sampled in an XAD versus peristaltic sample) in the evaluation of ecological exposure to surface water. Surface water samples were collected both as single-point samples and as transect (vertical and/or horizontal) 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 only sampled at 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 both transect, single-point, XAD, and peristaltic samples allows for the evaluation of surface water data over a larger spatial scale; however, the relevance of ecological exposure to surface water data collected from the various sampling types is highly uncertain.

9.1.3 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 6.5.3 and Table 6-27 in the benthic risk assessment present the water TRVs developed for all surface water COPCs. Water TRVs were primarily developed based on fish and invertebrate species effects. Therefore, there are uncertainties associated with the use of these as TRVs for evaluating risk to amphibians.

Because the selected AWQC for total PCBs and 4,4′-DDT were based on the protection of mammals and birds, respectively, risk estimates for aquatic receptors based on these TRVs are uncertain. Therefore, alternative TRVs specifically protective of fish and invertebrates were developed in this BERA using methods consistent with those used for AWQC derivation.

Based on the AWQC document for PCBs (EPA 1980c), toxicity data were insufficient to allow derivation of an FAV, or an FCV. Therefore, the PCB FAV was calculated in this BERA using additional toxicity data for aquatic organisms reported in EPA's ECOTOX online database published since 1979 (ECOTOX 2009). When these additional data were added to those for aquatic organisms included in the AWQC document for PCBs (EPA 1980c), the data were sufficient to derive an FAV in accordance with the methods used to establish AWQC values. An FAV of 1.6 µg/L was derived and then divided by the geometric mean of the ACRs presented in the AWQC document for PCBs (8.39), to determine an FCV of 0.19 µg/L, which is specific to aquatic organisms. This concentration (0.19 µg/L) was evaluated as an alternative water TRV for total PCBs; however, slightly lower chronic values are reported in the AWQC document for fish (lowest chronic value of 0.098 µg/L was reported for fathead minnow) and plants (lowest diatom value is 0.1 µg/L). 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. Although both the selected PCB water TRV (0.014 µg/L) and the alternative water TRV (0.19 µg/L) were used to derive water HQs, only the alternative TRV was used to determine risk conclusions.

Based on the AWQC document for DDTs (EPA 1980a), sufficient toxicity data for aquatic organisms were available to derive an FAV directly for 4,4′-DDT (1.1 µg/L) but not an FCV or an ACR. Only a single ACR (65) was identified in the AWQC document for DDTs, and Suter and Tsao (1996) recommend using 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) from four studies of chemicals with a DDT-like mode of action. Dividing the FAV (1.1 µg/L) by the median reported ACR of 3.6 results in a chronic value of 0.31 µg/L.. In accordance with the methods used to establish AWQC, it is appropriate to use a final tissue-residue value 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 10th percentile fish tissue-residue LOAEL (1.6 mg/kg ww) by a BAF of 142,960[136] (derived from the DDT AWQC document). This water concentration of 0.011 µg/L is lower than the FAV divided by the ACR, and was evaluated as an alternative water TRV for DDx compounds in water. For evaluating the direct exposure of aquatic organisms to water, this alternative TRV is considered more appropriate than the AWQC (0.0010 µg/L), which are based on the protection of brown pelican via ingestion of contaminated prey. Although both the selected DDT water TRV

[136] A BAF of 142,960 was based on the lipid normalized BAF (17,870) and anchovy lipid percent (8%) as presented in the DDT AWQC document.

(0.001µg/L) and the alternative water TRV (0.011 µg/L) were used to derive water HQs, only the alternative TRV was used 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 another receptor group (e.g., benthic invertebrates or fish). 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. 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 9.1.4.

The water TRV for DDx compounds based on the 4,4′-DDT AWQC is derived from the effects data for only one bird species (brown pelican). Birds are known to be more sensitive to DDx compounds, and the water TRV that is protective of birds is of questionable relevance and likely overestimates risks to amphibians. Similarly, the total PCB 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 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 (and determine COCs), only the alternative TRV was used to determine risk conclusions.

9.1.4 Risk Characterization and Uncertainty Analysis

This section presents the risk estimates for amphibians based on the surface water LOE. An HQ was derived for all COPCs using Equation 6-1 and used to quantify surface water risk estimates. The EPC and TRV are represented by surface water concentrations.

Section 9.1.4.1 presents the overall approach used to characterize risks via surface water to amphibians. Section 9.1.4.2 presents the risk characterization results and uncertainty evaluation. Section 9.1.4.3 presents an evaluation of background concentrations. Section 9.1.4.4 summarizes surface water COCs.

9.1.4.1 Risk Characterization Process

The risk characterization process for assessing risks to amphibians via surface water was conducted by evaluating individual surface water samples. HQs were determined on an individual surface water sample-by-sample basis within the Study Area in accordance with EPA's Problem Formulation (Attachment 2). COPCs with HQs > 1.0 for any individual surface water sample (within amphibian exposure areas) were identified as COCs. In addition, surface water concentrations were evaluated on a site-wide basis including all samples location within amphibian habitat areas. For all COCs, the spatial distribution and magnitude of the HQs, the seasonal and sampling method patterns of the HQs, and the associated exposure and effects assumptions were evaluated to provide a more detailed assessment of impacts on the amphibians. The evaluation of COCs and associated uncertainties were further evaluated to arrive at risk conclusions for amphibians (Section 9.3).

9.1.4.2 Risk Characterization Results and Uncertainty Evaluation

Table 9-1 presents a summary of the individual HQs calculated across all surface water samples collected within amphibian exposure areas for all surface water COPCs. Table 9-2

presents the HQs calculated site-wide UCLs based on the surface water samples located within the amphibian habitat areas.

Table 9-1. Number of Surface Water Samples in Amphibian Exposure Areas with HQs > 1.0

COPC	Number of Samples with HQs > 1.0 of Total Number	Percent of Samples with HQs > 1.0
Metals		
Aluminum (total)	**88/95 (21)**	93%
Zinc (dissolved)	**1/117 (1.1)**	< 1%
Butyltins		
Butyltin ion	0/117	0%
PAHs		
Benzo(a)anthracene	**2/158 (10)**	1.3%
Benzo(a)pyrene	**3/158 (14)**	1.9%
Naphthalene	**1/159 (50)**	< 1%
Phthalates		
BEHP	**1[a]/129 (1.2)**	< 1%
PCBs		
Total PCBs	**2/111 (1.2)[b]**	1.8%
Pesticides		
4,4′-DDT	**13[c]/121 (3.9)[d]**	11%
Total DDx	**26[c]/121 (20)[d]**	21%
VOCs		
Ethylbenzene	0/1	0%
Trichloroethene	0/1	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] None of the 111 samples had total PCB concentrations greater than the alternate total PCB TRV of 0.19 µg/L.

[c] An additional three samples had DLs that were greater than the TRV. The maximum HQ based on a DL is 1.6 for both 4,4′-DDT and total DDx.

[d] One of 121 samples had 4,4′-DDT and total DDx concentrations greater than the alternate 4,4′-DDT TRV of 0.011 µg/L (HQs were 1.7 and 1.8, respectively).

BEHP – bis(2-ethylhexyl) phthalate
COPC – chemical of potential concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
DL – detection limit
HQ – hazard quotient
Bold identifies HQs > 1.0.
NA – not analyzed
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)
VOC – volatile organic compound

Table 9-2. Summary of Site-Wide Amphibian Habitat Surface Water UCL HQs

COPC	Unit	UCL	Water TRV	HQ
Metals				
Aluminum	µg/L	445	87	5.1
Zinc	µg/L	2.9	36.5	0.079
Butyltins				
Butyltin 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	14	0.18[a]
Pesticides				
4,4′-DDT	ng/L	0.53	1	0.53[b]
Total DDx	ng/L	1.9	1[c]	1.9[b]

[a] UCL did not exceed the alternative total PCB TRV of 190 ng/L either.
[b] UCLs for 4,4′-DDT and total DDx did not exceed the alternative 4,4′-DDT of 11 ng/L (HQs were 0.063 and 0.15, respectively).
[c] Criteria for 4,4′-DDT used to evaluate total DDx.

BEHP – bis(2-ethylhexyl) phthalate
COPC – chemical of potential concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
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
UCL – upper confidence limit on the mean
Bold identifies HQs > 1.0.

The following COPCs were identified as COCs because their HQs were > 1.0 in at least one surface water sample located within amphibian exposure areas: aluminum, zinc, benzo(a)anthracene, benzo(a)pyrene, naphthalene, BEHP, total PCBs, and DDx compounds (as 4,4′-DDT and total DDx). Maps 9-2 through 9-6 present the stations with HQ > 1.0 for all COCs, with the exception of aluminum. Since aluminum HQs > 1.0 were widespread, these locations were not mapped. Site-wide amphibian habitat HQs were > 1.0 for three COPCs (i.e., aluminum, naphthalene, and total DDx). There is high uncertainty with the identification of total PCBs and total DDx as COCs for amphibians because HQs were < 1.0 using the alternative TRV that is protective of direct exposure to surface water.

A discussion of these COCs, including an evaluation of the key uncertainties and the frequency and location and HQs > 1.0 is presented below. Key uncertainties associated with

the appropriateness of using water TRVs for assessing risks to amphibians are also discussed.

Metals – Aluminum HQs were > 1.0 in 93% of all individual samples; these samples were located between RM 2.0 and RM 9.7 and included samples collected during the March 2005 event, the sensitive amphibian exposure period. The site-wide HQs for aluminum was also > 1.0. Zinc HQs were > 1.0 in only one sample (HQ = 1.1) and the site-wide HQ was < 1.0. This sample was collected in November 2004. There is uncertainty associated with the TRVs for aluminum (87 µg/L) and zinc (36.5 µg/L) because both are based on risk to fish and aquatic invertebrates. Amphibian-specific toxicity data for both aluminum and 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 aluminum LC50 and LC10 (concentration that is lethal to 10% of an exposed population) values of 50 and 10 µg/L, respectively, and zinc LC50 and LC10 values of 10 and 3 µg/L, respectively (Birge et al. 2000). Amphibians appear to be as sensitive or more sensitive to aluminum and zinc than assumed when using the fish- and invertebrate-based AWQC TRVs.

PAHs and BEHP – PAH (i.e., benzo[a]anthracene, benzo[a]pyrene, and naphthalene) and BEHP HQs were all > 1.0 in less than 2% of samples; HQs were < 1.0 for these COCs 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 were < 1.0 and the site-wide HQ for naphthalene was 3.2. There is uncertainty associated with the TRVs for PAHs (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 were 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,[137] and 500 µg/L of naphthalene (Sparling 2000). Ultraviolet (UV) radiation (via sunlight) can affect the toxicity of PAHs to amphibians; lower thresholds of benzo(a)pyrene associated with effects have been reported in amphibians when they are also exposed to UV radiation; 100% mortality was reported in newt tadpoles following exposure to UV-A radiation and 6.25 to 12.5 µg/L of benzo(a)pyrene (Sparling 2000). Amphibians appear to be as sensitive or more sensitive to PAHs than assumed when using the fish- and invertebrate-based Tier II TRVs.

Total PCBs – Two samples had total PCB HQs >1.0, with a maximum HQ of 1.2. One of these samples was collected during the March 2005 event. There is high uncertainty associated with the PCB TRV of 0.014 µg/L because it is based on risk to mink. Using the alternative total PCB TRV of 0.19 µg/L, no samples exceeded the TRV. The site-wide HQs using both the selected TRV and alternative TRV were < 1.0. Limited amphibian-specific

[137] Induction of micronucleated erythrocytes were observed at 3.12 and 6.25 µg/L of benzo(a)anrthacene with UV-A exposure and at 750 µg/L of benzo(a)anrthacene with no UV-A exposure.

PCB toxicological data are available. Acute and chronic PCB Aroclor 1254 amphibian thresholds of 63 and 6.3 µg/L were derived from a 96-hour study (Zhou et al. 2004), in which 70% mortality was observed. Uncertainty factors of 4 and 40 were applied to the reported water concentration in this study associated with mortality to derive an acute and chronic threshold, respectively. PCB Aroclor 1254 LC50s for 4-day post-hatch and hatchling amphibians 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 have higher sensitivities to PCBs than do amphibians. Due to the fact that amphibian thresholds are higher than thresholds for fish, risks to amphibians from PCBs in water may be overestimated using the water TRV.

DDx – Total DDx HQs were > 1.0 in all samples where individual DDT metabolite HQs were also > 1.0, therefore this discussion focuses on total DDx only. The site-wide total DDx HQ using the selected TRV was 1.9. Samples with detected concentrations of total DDx had HQs > 1.0 at locations throughout the Study Area (from RM 2.0 to RM 9.9) during all types of flow conditions. Twenty-three percent of the samples (n = 6) with total DDx HQs > 1.0 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 of 20 µg/L, are considered highly uncertain. Total DDx HQs based on non-N qualified data (n = 20) range from 1.2 to 9.8, and 15% of these samples (n = 3) were collected during the targeted amphibian egg exposure period in March 2005.

There is uncertainty associated with the total DDx evaluation because the AWQC-based TRV (0.001 µg/L) was derived from bird effects data. Using the alternative DDT TRV of 0.011 µg/L, only one sample (W001 located at RM 2.0) had a total DDx concentration that exceeded the TRV; however, this result was N-qualified. The total DDx HQ for this sample based on the alternative TRV was 1.8. The site-wide total DDx HQ using the alternative TRV was 0.17. 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 DDx compounds varies with stage of development (Sparling 2000). High mortality (80 to 100%) was reported in frog and toad tadpoles at varying 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), the thresholds reported for amphibians.

9.1.4.3 Evaluation of Background Concentrations

Background aluminum concentrations were established as part of the RI (see Section 7.0 of the draft RI). Background and Study Area concentrations in sediment and surface water were compared and are presented in Attachment 11 and Section 6.5.4.3. Aluminum concentrations in sediment and surface water for the Study Area were 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 risks 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 concentration of zinc (2.5 µg/L) was greater than the range of zinc concentrations detected in background[138] (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 solely be attributed to background.

9.1.4.4 Summary of Surface Water COCs

The following surface water COCs were identified for amphibians: aluminum, zinc, benzo(a)anthracene, benzo(a)pyrene, naphthalene, BEHP, total PCBs, and DDx compounds.

9.2 TZW ASSESSMENT

The TZW assessment is presented in detail of Section 6.6. A summary of the TZW assessment is provided in this section.

A TZW[139] exposure pathway was evaluated for amphibians per EPA's Problem Formulation (Attachment 2). TZW was screened against surface water screening values, without regard to the presence or absence of amphibian habitat within the TZW assessment areas. The TZW data are limited, so the results of the TZW assessment were not incorporated into the risk conclusions section (Section 9.3).

TZW was sampled along the shoreline of nine facilities within the Study Area. The TZW exposure pathway for specific aquatic receptors (i.e., benthic invertebrates, benthic fish, amphibians, and aquatic plants) was evaluated by comparing TZW concentrations to water TRVs. Sixty TZW COPCs were identified in the SLERA (Attachment 5) based on a comparison of the maximum detected concentrations in any one TZW sample to water TRVs. A sample-by-sample summary of TRV exceedances of these COPCs along the shoreline of the nine facilities is presented in Table 6-33. All facilities had one or more TZW chemical concentrations greater than TRVs; however, many HQs were low (< 3.0). The complete results of this evaluation are presented in Section 6.6.

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[138] Zinc concentrations were detected in only three of twenty-two surface water samples included in the background data set (see Section 7.0 of the draft RI).

[139] 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.

There is high uncertainty associated with the TZW evaluation. Because of the spatial focus of the TZW data collection, the TZW chemical concentrations are not representative of conditions throughout the Study Area. The TZW collected from the Study Area is a snapshot of only the specific areas where the data were collected (i.e., no Study Area- wide spatial inferences can be made). Furthermore, there are uncertainties regarding the relevance of certain TRVs for effects on receptors because of the number of studies and species used to derive those values. Uncertainties regarding water TRVs used to evaluate TZW are discussed in Section 6.6.6. Consistent with the agreement between the LWG and EPA managers, COCs were not identified on the basis of the TZW evaluation.

9.3 RISK CONCLUSIONS

This section presents a summary of the overall risk conclusions for the amphibian risk assessment. Amphibian COCs were identified based only on the surface water LOE. Risk conclusions incorporated the magnitude of HQs, spatial distribution and frequency of HQ > 1.0, and the uncertainty of exposure and effects assumptions for estimating population-level risks to amphibians. Amphibians were selected to be a receptor that is protective of reptiles; therefore, risk conclusions for amphibians are applicable to reptiles.

Table 9-3 summarizes the HQs and exposure and effects uncertainties for each of the surface water COCs. In general, surface water concentrations of several COCs are below amphibian-specific thresholds or have HQs > 1.0 during non-reproductive periods (and may not be present in the Study Area), indicating negligible risks to amphibian populations. There is high uncertainty concerning the relevance of the selected TRVs for individual PAHs to amphibian populations because no amphibian-specific thresholds are available for comparison. In addition, aluminum concentrations in background areas indicate that aluminum is not attributed to anthropogenic sources in the Study Area and is not expected to be toxic to amphibians at naturally occurring concentrations.

Table 9-3. Summary of Amphibian Surface Water COCs

COC	Risk Exposure Area	HQs[a]	Frequency of All Samples with HQs > 1.0	Sampling Event with HQs > 1.0	Key Uncertainties	Risk Conclusions
Metals						
Aluminum	Site-wide	1.0– 21	93%	All except the January 2006 high-flow event	Aluminum concentrations in background areas indicate that aluminum is not attributed to anthropogenic sources in the Study Area and is not expected to be toxic to amphibians at naturally occurring concentrations; TRV is based on toxicity to fish and invertebrates.	Aluminum concentrations in background areas indicate that aluminum is not attributed to anthropogenic sources in the Study Area and is not expected to be toxic to amphibians at naturally occurring concentrations; therefore, negligible risks to amphibians from aluminum are expected.
Zinc	Individual samples with HQs > 1.0 at RM 9.7,west	1.1	< 1%	November 2004 low-flow event	HQ was > 1.0 in one sample only; TRV is based on toxicity to fish and invertebrates; amphibians may be more sensitive to zinc than fish/invertebrates. Site-wide HQ is < 1.0.	A low magnitude and frequency of HQ > 1.0 was calculated during a non-reproductive period for amphibians. There is uncertainty associated with whether the selected TRV is protective of amphibians. Although zinc is identified as a COC in surface water under the assumptions required for the BERA, it is more reasonable to conclude that zinc does not pose unacceptable risk to amphibian populations based on the low magnitude of exceedance and limited spatial extent of exceedances.

Table 9-3. Summary of Amphibian Surface Water COCs

COC	Risk Exposure Area	HQs[a]	Frequency of All Samples with HQs > 1.0	Sampling Event with HQs > 1.0	Key Uncertainties	Risk Conclusions
PAHs						
Benzo(a)anthracene	Individual samples with HQs > 1.0 at RM 6.1, west; and RM 6.3,west	4.1 – 10	1.3%	July 2005 low-flow event and winter 2007 high-flow event	HQ was > 1.0 only in peristaltic samples at W012 and W031; TRV is based on extrapolated *Daphnia* acute LC50; limited toxicological data indicate that amphibians may be less sensitive to PAHs than are fish/invertebrates. Site-wide HQ is < 1.0.	A low frequency of HQ > 1.0 was calculated during a non-reproductive period for amphibians. There is high uncertainty concerning the relevance of the TRV to amphibian populations. Although benzo(a)anthracene was identified as a COC in surface water under the assumptions required for the BERA, it is more plausible that benzo(a)anthracene does not pose unacceptable risk toamphibian populations based on the limited spatial extent of exceedances.
Benzo(a)pyrene	Individual samples with HQs > 1.0 at RM 6.1,west; and RM 6.3,west	1.4 – 14	1.9%	November 2004 and July 2005 low-flow events, and winter 2007 high-flow event	HQ was > 1.0 in peristaltic samples only at W012 and W031; TRV is based on extrapolated *Daphnia* acute LC50; limited toxicological data indicate that amphibians may be less sensitive to PAHs than are fish/invertebrates. Site-wide HQ is < 1.0.	A low frequency of HQ > 1.0 was calculated during a non-reproductive period for amphibians. There is high uncertainty concerning the relevance of the TRV to amphibian populations. Although benzo(a)pyrene was identified as a COC in surface water under the assumptions required for the BERA, it is more reasonable to conclude that benzo(a)pyrene does not pose unacceptable risk to amphibian populations based on the limited spatial extent of exceedances.

Table 9-3. Summary of Amphibian Surface Water COCs

COC	Risk Exposure Area	HQs[a]	Frequency of All Samples with HQs > 1.0	Sampling Event with HQs > 1.0	Key Uncertainties	Risk Conclusions
Naphthalene	Individual samples with HQs > 1.0 at RM 6.4,west	50	< 1%	May 2005	HQs were > 1.0 in one sample only; TRV is based on risk to fish and invertebrates; limited toxicological data indicate that amphibians may be less sensitive to PAHs than are fish/invertebrates (i.e., all surface water concentrations are less than amphibian toxicity thresholds).	A low frequency of HQ > 1.0 was calculated during amphibian egg exposure (May 2005). There is high uncertainty concerning the relevance of the TRV to amphibian populations. Based on an amphibian-specific TRV, negligible risks to amphibians from naphthalene exposure are expected.
Phthalates						
BEHP	Individual sample with HQs > 1.0 at RM 6.7,east (Willamette Cove)	1.2	< 1%	Winter 2007 high-flow event	HQs were > 1.0 in one sample only; TRV is based on risk to fish, invertebrates, and amphibians. Site-wide HQ is < 1.0	The low magnitude and frequency of surface water HQ > 1.0 (during a non-reproductive period for amphibians) indicates that negligible risks to amphibians are expected.

Table 9-3. Summary of Amphibian Surface Water COCs

COC	Risk Exposure Area	HQs[a]	Frequency of All Samples with HQs > 1.0	Sampling Event with HQs > 1.0	Key Uncertainties	Risk Conclusions
PCBs						
Total PCBs	Individual samples with HQs > 1.0 at RM 3.7 (International Slip); and RM 6.9,east (Willamette Cove)	1.1 – 1.2	1.8%	November 2004 low-flow event and March 2005 low-flow event	HQs were > 1.0 in peristaltic samples only at W014 and W004; site-wide HQ is < 1.0; TRV is based on risks to mink; no samples exceed the alternative water TRV that is protective of direct exposure to water; amphibian-specific thresholds are higher than selected and alternative TRVs.	A low magnitude and frequency of HQ > 1.0 was calculated, and no samples exceeded the alternative water TRV that is based on direct exposure to water. Given that amphibian thresholds are higher than the selected and alternative TRV and that no samples exceeded the alternative TRV that is based on direct exposure to surface water, no unacceptable risks to amphibian populations, are expected.

Table 9-3. Summary of Amphibian Surface Water COCs

COC	Risk Exposure Area	HQs[a]	Frequency of All Samples with HQs > 1.0	Sampling Event with HQs > 1.0	Key Uncertainties	Risk Conclusions
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 January 2007 high-flow event	23% of samples with total DDx HQs > 1.0 (n = 6 samples) were based on N-qualified data, where HQs ranged from 1.4 to 20; total DDx HQs based on non-N-qualified data (n = 20) ranged from 1.2 to 9.8, where 15% of these samples were collected during the May 2005 event; TRV (0.001 µg /L) is based on risk to birds. One sample exceeds alternative water TRV that is protective of direct exposure to surface water (HQ = 1.8); however, this sample (W001 at RM 2.0) is N-qualified.; limited toxicological data indicate that amphibian thresholds are much higher than selected and alternative TRV.	Total DDx HQs > 1.0 in 21% of samples (n = 26); however, about one-fourth of HQs > 1.0 are based on N-qualified data. Given that amphibian thresholds are higher than the selected and alternative TRV and the only sample that exceeded the alternative TRV based on direct exposure to surface water indicates sample interference, no unacceptable risks to the amphibians from surface water are expected.

[a] HQs are presented only for water samples with HQs > 1.0. HQs in all other water samples are < 1.0.

BEHP – bis(2-ethylhexyl) phthalate
COC – chemical of concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
HQ – hazard quotient
J – estimated concentration
N – tentative identification
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
RM – river mile
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)
TRV – toxicity reference value

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.[140] The surface water assessment is presented in Section 10.1. The direct sediment contact pathway was 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.[141] The TZW evaluation is presented in Section 10.2 As required by EPA, TZW was screened against surface water screening values. TZW data is limited. No attempt was made to further characterize TZW exposure; accordingly, results of the TZW assessment were not incorporated into the risk conclusions section. Risk conclusions to aquatic plants are presented in Section 10.3.

10.1 SURFACE WATER ASSESSMENT

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 aquatic plants. All surface water samples taken from aquatic plant habitat areas were evaluated (Map 9-1).The comparison of surface water concentrations to water TRVs was conducted on an individual sample basis per EPA direction (Attachment 2). COPCs with HQs > 1.0 for any individual surface water sample within aquatic plant exposure areas were identified as COCs. Surface water concentrations were also evaluated on a site-wide basis including all samples location within aquatic plant habitat areas. For all COCs, the spatial distribution and magnitude of HQs, the seasonal and sampling method patterns of HQs, and the associated exposure and effects assumptions were evaluated to provide a more detailed characterization of any potential risks to the aquatic plant community. The evaluation of COCs and associated uncertainties were further evaluated to arrive at risk conclusions for the aquatic plant community (Section 10.3).

The details of this assessment are presented as follows:

- Section 10.1.1 presents a summary of the COPCs evaluated.
- Section 10.1.2 presents an overview of how exposure concentrations were derived. All surface water chemical concentrations are presented in Attachment 4.

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[140] 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.

[141] 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.

- Section 10.1.3 presents a summary of the effects data. Details on the development of the water TRVs are presented in Attachment 10.
- Section 10.1.4 presents the risk characterization results, COCs, and associated uncertainties. These COCs are further assessed in the aquatic plant risk conclusions (Section 10.3).

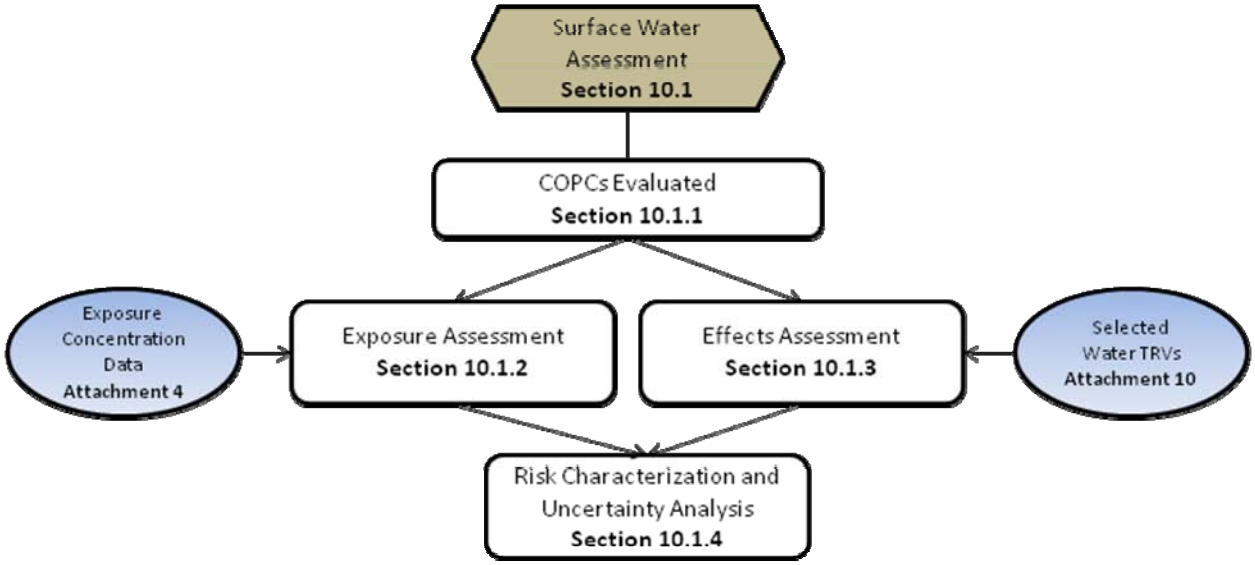

Figure 10-1. Overview of Aquatic Plant Surface Water Section Organization

10.1.1 COPCs Evaluated

Twelve[142] surface water COPCs were identified based on the results of the SLERA and refined screen (Attachment 5). These surface water COPCs are presented in Table 6-26. Four surface water COIs could not be evaluated because no toxicological data were available. These COIs (4-chloroaniline, aniline, 2,4-DB, and MCPP) were infrequently detected and were detected in isolated areas and during different sampling periods.

10.1.2 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 and rationale and sample types) and general trends in COPC concentrations from the Study Area are presented in the benthic risk assessment (Section 6.5.2). Quiescent areas or amphibian/aquatic plant habitats were identified as aquatic plant exposure areas (Map 9-1). Quiescent areas were defined as slow-moving, shallow waters in the Study Area potentially

[142] Three individual DDT metabolites identified in the SLERA (2,4′-DDD, 2,4′-DDT, and 4,4′-DDD) 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 DDx is based on 4,4′-DDT.

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 were 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, Toni Pennington (2002). Aquatic plant habitats were identified 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 river banks. 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 plant communities along the Study Area shoreline are generally dominated by exotic species such as reed canary grass and Himalayan blackberry or 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).

There is uncertainty 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 were 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 were represented by detected concentrations in all individual surface water samples collected within quiescent areas or amphibian/aquatic plant habitat areas. Surface water EPCs were also represented by site-wide UCL concentrations calculated using all surface water sampling locations collected within amphibian habitat areas.[143] Site-wide amphibian habitat UCLs were calculated for all surface water COPCs, with the exception of ethylbenzene and trichloroethene. These

[143] The site-wide UCL concentration was limited by only those surface water samples located within the amphibian/aquatic plant habitat areas identified between RM 3.5 and RM 9.2.

COPCs were not evaluated on a site-wide basis because VOCs were only analyzed in one sample located within an amphibian habitat area collected between RM 5.5 and 6.5. 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 detection limits and higher accuracy of results.

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. 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. 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.

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

There are uncertainties associated with the use of multiple sampling types and methods (e.g., duration of sampling time for a single point grab sample versus an integrated transect sample or volume of water sampled in an XAD versus peristaltic sample) in the evaluation of ecological exposure to surface water. Surface water samples were collected both as single-point samples and as transect (vertical and/or horizontal) 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 only sampled at 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 both transect, single-point, XAD, and peristaltic samples allows for the evaluation of surface water data over a larger spatial scale; however, the relevance of ecological exposure to surface water data collected from the various sampling types is highly uncertain.

10.1.3 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 6.5.3 and Table 6-27 in the benthic risk assessment present the water TRVs developed for all surface water COPCs. Water TRVs were primarily developed based on fish and invertebrate species effects.

Therefore, there are significant uncertainties associated with the use of these as TRVs for evaluating risks to aquatic plants.

Because the selected AWQCs for total PCBs and 4,4′-DDT were based on protection of mammals and birds, respectively, risk estimates for aquatic receptors based on these TRVs are uncertain. Therefore, alternative TRVs specifically protective of fish and invertebrates were developed in this BERA using methods consistent with those used for AWQC derivation.

Based on the AWQC document for PCBs (EPA 1980c), toxicity data were insufficient to allow derivation of an FAV or an FCV. Therefore, the PCB FAV was calculated in this BERA using additional toxicity data for aquatic organisms reported in EPA's ECOTOX online database published since 1979 (ECOTOX 2009). When these additional data were added to those for aquatic organisms included in the AWQC document for PCBs (EPA 1980c), the data were sufficient to derive an FAV in accordance with the methods used to establish AWQC values. An FAV of 1.6 µg/L was derived and then divided by the geometric mean of the ACRs presented in the AWQC document for PCBs (8.39), to determine an FCV of 0.19 µg/L, which is specific to aquatic organisms. This concentration (0.19 µg/L) was evaluated as an alternative water TRV for total PCBs; however, slightly lower chronic values are reported in the AWQC document for fish (lowest chronic value of 0.098 µg/L was reported for fathead minnow) and plants (lowest diatom value is 0.1 µg/L). 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. Although both the selected PCB water TRV (0.014 µg/L) and the alternative water TRV (0.19 µg/L) were used to derive water HQs, only the alternative TRV was used to determine risk conclusions.

Based on the AWQC document for DDTs (EPA 1980a), sufficient toxicity data for aquatic organisms were available to derive an FAV directly for 4,4′-DDT (1.1 µg/L), but not an FCV or an ACR. Only a single ACR (65) was identified in the AWQC document for DDTs, and Suter and Tsao (1996) recommend using 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) from four studies of chemicals with a DDT-like mode of action. Dividing the FAV (1.1 µg/L) by the median reported ACR of 3.6 results in a chronic value of 0.31µg/L.. In accordance with the methods used to establish AWQCs, it is appropriate to use a final tissue-residue value 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 10th percentile fish tissue residue LOAEL (1.6 mg/kg ww) by a BAF of 142,960[144] (derived from the DDT AWQC document). This concentration of 0.011 µg/L is lower than the FAV divided by the ACR and was evaluated as an alternative water TRV for DDx compounds in water. For evaluating direct exposure

[144] A BAF of 142,960 was based on the lipid-normalized BAF (17,870) and anchovy lipid percent (8%) as presented in the DDT AWQC document.

of aquatic organisms to water, this alternative TRV is considered more appropriate than the AWQC (0.0010 µg/L), which are based on the protection of brown pelican via ingestion of contaminated prey. Although both the selected DDT water TRV (0.001µg/L) and the alternative water TRV (0.011 µg/L) were used to derive water HQs, only the alternative TRV was used 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 over- or underestimate effects to aquatic plants. The water TRVs for all COPCs were established by the AWQC and Tier II criteria, which are based on or included toxicological data for invertebrate (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) so use of algae toxicity data to assess plant community risks is uncertain. The TRV uncertainties are discussed on a chemical-specific basis in Section 10.1.4.

The water TRV for DDx compounds based on the 4,4′-DDT AWQC is derived from the effects data for only one bird species (brown pelican). Birds are known to be more sensitive to DDx compounds, and the water TRV that is protective of birds is of questionable relevance and might overestimate risks to aquatic plants. Similarly, the total PCB 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 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 (and determine COCs), only the alternative TRV was used to determine risk conclusions.

10.1.4 Risk Characterization and Uncertainty Analysis

This section presents the risk estimates for aquatic plants based on the surface water LOE. An HQ was 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 10.1.4.1 presents the overall approach used to characterize risks via surface water to aquatic plants. Section 10.1.4.2 presents the risk characterization results and uncertainty evaluation. Section 10.1.4.3 presents an evaluation of background concentrations. Section 10.1.4.4 summarizes surface water COCs.

10.1.4.1 Risk Characterization Process

The risk characterization process for assessing risks to aquatic plants via surface water was conducted by evaluating individual surface water samples. HQs were determined on a sample-by-sample basis for all surface water samples within the Study Area in accordance with EPA's Problem Formulation (Attachment 2). COPCs with HQs > 1.0 for any individual surface water sample (within aquatic plant exposure areas) were identified as COCs. In addition, surface water concentrations were evaluated on a site-wide basis including all samples location within aquatic plant habitat areas. For all COCs, 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 aquatic plants. The evaluation of COCs and associated uncertainties were further evaluated to arrive at risk conclusions for aquatic plants (Section 10.3).

10.1.4.2 Risk Characterization Results and Uncertainty Evaluation

HQs calculated for all surface water COPCs within aquatic plant exposure areas are the same as those presented for amphibians (Section 9.1.4 and Table 9-1). Samples with HQs > 1.0 are shown on Maps 9-2 through 9-6. Table 9-2 presents the HQs calculated site-wide UCLs based on the surface water samples located within the aquatic plant habitat areas (the amphibian habitat areas are the same as the aquatic plant habitat areas).

The following COPCs were identified as COCs for aquatic plants: aluminum, zinc, benzo(a)anthracene, benzo(a)pyrene, naphthalene, BEHP, total PCBs, and DDx compounds (as 4,4′-DDT, and total DDx). Site-wide amphibian habitat HQs were > 1.0 for three COPCs (i.e., aluminum, naphthalene, and total DDx). There is high uncertainty with the identification of total PCBs and total DDx as COCs for amphibians because HQs were < 1.0 using the alternative TRV that is protective of direct exposure to surface water.

A discussion of these COCs, including an evaluation of the key uncertainties and the frequency and location and HQs > 1.0, is presented below. Key uncertainties associated the appropriateness of using water TRVs for assessing risks to aquatic plants are also discussed.

Metals – Aluminum HQs were > 1.0 in 93% of all individual samples; these samples were located between RM 2.0 and RM 9.7 and included samples collected during all sampling events. The site-wide HQs for aluminum were also > 1.0. There is uncertainty associated with the TRV for aluminum (87 µg/L) because it is based on risks to fish and aquatic invertebrates. Water toxicity data for two algae and two aquatic plants were included in the aluminum water quality criteria document, with algae LOAELs ranging from 460 µg/L for reduced biomass of green algae to 6,480 µg/L for complete mortality of a diatom. Vascular plant LOAELs ranged from 2,500 µg/L for reduced root weight of water milfoil to > 45,700 µg/L for reduced frond production of duckweed. The 4-day aluminum lowest LCV identified for algae by Suter and Tsao (1996) was 460 µg/L, based on *Selenastrum capricornutum,* as reported by EPA (1988). Based on these limited data, aquatic plants appear to be less sensitive than fish and invertebrates to aluminum.

Zinc HQs were > 1.0 in only one sample (HQ = 1.1) and the site-wide HQ was < 1.0. There is uncertainty 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, with LOAELs ranging from 30 to 200,000 µg/L (EPA 1987). Algae were both the most and least sensitive species tested. Wang et al. (1997) also reported highly variable toxicity values (ranging from 10 to > 100,000 µg/L) for various plant species (including algae) measured in laboratory toxicity tests. The 7-day zinc LCV identified for algae by Suter and Tsao (1996) was 30 µg/L,[145]

[145] The algae-specific toxicity threshold for zinc (30 µg/L) was based on a hardness of 14.9 mg/L $CaCO_3$, and the water TRV for zinc (36.5 µg/L) was based on a hardness of 25 mg/L $CaCO_3$).

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 – PAH (i.e., benzo[a]anthracene, benzo[a]pyrene, and naphthalene) and BEHP HQs were each > 1.0 in less than 2% of samples. The site-wide HQs for benzo(a)pyrene, benozo(a)anthracene, and BEHP were < 1.0 and the site-wide HQ for naphthalene was 3.2. There is uncertainty associated with the PAH and BEHP TRVs because they were based on invertebrate, fish, and/or amphibians toxicity data. 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 were available for naphthalene. The naphthalene TRV (12 µg/L) is 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 ultraviolet radiation. The naphthalene algae LCV presented in Suter and Tsao (1996) as reported by EPA (1980b) was 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 were not available for benzo(a)anthracene, benzo(a)pyrene, and BEHP.

Total PCBs – Total PCBs had HQs >1.0 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 of these samples were collected during low-flow events. There is high uncertainty associated with the PCB TRV (0.014 µg /L) because it is based on risks to mink. Using the alternative total PCB TRV of 0.19 µg/L, no samples exceeded the TRV. The site-wide HQs using both the selected TRV and alternative TRV were < 1.0. Limited plant-specific PCB toxicological data are available. The AWQC document did not included any appropriate data for freshwater aquatic plants but did identify adverse effects in saltwater diatoms ranging 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) was 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 risks 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 on 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 HQs were > 1.0 in all samples in which individual DDx metabolite HQs were > 1.0; therefore, this discussion focuses only on total DDx. The site-wide total DDx HQ using the selected TRV was 1.9. However, the AWQC-based TRV (0.001 µg/L) was derived from bird effects (eggshell thinning) data. Using the alternative DDT TRV of 0.011 µg/L, only one sample (W001 located at RM 2.0) had a total DDx concentration that

exceeded the TRV; however, this result was N-qualified. The total DDx HQ in this sample based on the alternative TRV was 1.8. The site-wide total DDx HQ using the alternative TRV was 0.17. Toxicity data for four species of algae were included in the DDT quality criteria document, with LOAELs ranging from 0.3 to 800 µg/L (EPA 1980a). The lowest LOAEL (0.3 µg/L) was reported for growth and morphology affects for *Chlorella* sp. Wang et al. (1997) reports phytotoxicity at much higher DDT concentrations (greater than 100,000 µg/L). The selected and alternative total DDx water TRVs are substantially lower than the reported algae-specific and phytotoxicity thresholds, indicating that risks to aquatic plants from DDx compounds in water are likely substantially overestimated.

10.1.4.3 Evaluation of Background Concentrations

Background aluminum concentrations were established as part of the RI (see Section 7.0 of the draft RI). Background concentrations in sediment and surface water are compared in Attachment 11 and discussed in Section 6.5.4.3. Aluminum concentrations in sediment and surface water for the Study Area were 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 risks 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) was greater than the range of zinc concentrations detected in background[146] (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 solely be attributed to background.

10.1.4.4 Summary of Surface Water COCs

The following surface water COCs were identified for aquatic plants: aluminum, zinc, benzo(a)anthracene, benzo(a)pyrene, naphthalene, BEHP, total PCBs, and DDx compounds.

[146] Zinc concentrations were detected in only three of twenty-two surface water samples included in the background data set (see Section 7.0 of the draft RI).

10.2 TZW ASSESSMENT

The TZW assessment is presented in detail in Section 6.6. A summary of the TZW assessment is provided in this section.

A TZW[147] exposure pathway was evaluated for aquatic plants per EPA's Problem Formulation (Attachment 2). TZW was screened against surface water screening values, without regard to the presence or absence of aquatic plant habitat within the TZW assessment areas. TZW data are limited, so the results of the TZW assessment were not incorporated into the risk conclusions section (Section 10.3).

TZW was sampled along the shoreline of nine facilities within the Study Area. The TZW exposure pathway for specific aquatic receptors (i.e., benthic invertebrates, benthic fish, amphibians, and aquatic plants) was evaluated by comparing TZW concentrations to water TRVs. Sixty TZW COPCs were identified in the SLERA (Attachment 5) based on a comparison of the maximum detected concentrations in any one TZW sample to water TRVs. A sample-by-sample summary of TRV exceedances of these COPCs along each of the nine facilities is presented in Table 6-33. All facilities had one or more TZW chemical concentrations greater than TRVs; however, many HQs were low (< 3.0). The complete results of this evaluation are presented in Section 6.6.

There is high uncertainty associated with the TZW evaluation. Because of the spatial focus of the TZW data collection, the TZW chemical concentrations are not representative of conditions throughout the Study Area. The TZW collected from the Study Area is a snapshot of only the specific areas where the data were collected (i.e., no Study Area- wide spatial inferences can be made). Furthermore, there are uncertainties regarding the relevance of certain TRVs for effects on receptors because of the number of studies and species used to derive those values. Aquatic vascular plants may be more directly exposed to TZW through their roots; however, aquatic vegetation habitat tends be present outside areas of known or likely contaminated groundwater discharge zones. Soil or sediment exposure in aquatic and terrestrial vascular plants is mediated via interstitial water in both habitats. However, the relationship between the solid and aqueous phases is highly complex and exposure via the aqueous (i.e., interstitial water or porewater) phase is never expressed independent of the solid phase. Consequently, while direct contact with TZW is conceptually correct, it is not a pathway that can be meaningfully characterized separately from bulk analyses of soil or sediment which incorporates both the solid and aqueous phases. In addition, the relevance of assessing risks to aquatic plants using toxicity data based on fish and invertebrate responses, as represented by the water TRVs, is uncertain. Uncertainties regarding water TRVs used to evaluate TZW are discussed in Section 6.6.6. Consistent with the agreement between the LWG and EPA managers, COCs were not identified on the basis of the TZW evaluation.

[147] 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.

10.3 RISK CONCLUSIONS

This section presents a summary of the overall risk conclusions of the aquatic plant risk assessment. Aquatic plant COCs were identified based only on the surface water LOE. Risk conclusions incorporated the magnitude of HQs, spatial distribution and frequency of HQs > 1.0, and the uncertainty of exposure and effects assumptions for estimating risks to the aquatic plant community.

Table 10-1 summarizes the HQs and exposure and effects uncertainties for each of the surface water COCs. In general, surface water concentrations of COCs are below algae-specific thresholds or have a low magnitude and frequency of samples with HQs > 1.0, indicating low to negligible risks to the aquatic plant community. There is high 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 addition, aluminum concentrations in background areas indicate that aluminum is not attributed to anthropogenic sources in the Study Area and is not expected to be toxic to aquatic plants at naturally occurring concentrations.

Table 10-1. Summary of Aquatic Plants Surface Water COCs

COC	Risk Exposure Area	HQs[a]	Frequency of All Samples with HQs > 1.0	Sampling Event with HQs > 1.0	Key Uncertainties	Risk Conclusions
Metals						
Aluminum	Site-wide	1.0–21	93%	All except the January 2006 high-flow event	Aluminum concentrations in background areas indicate aluminum is not attributed to anthropogenic sources in the Study Area and not expected to be toxic to plants at naturally occurring concentrations. TRV is based on toxicity to fish and invertebrates; algae data indicate that aquatic plants may be less sensitive to aluminum than are fish or invertebrates.	Aluminum concentrations in background areas indicate aluminum is not attributed to anthropogenic sources in the Study Area and not expected to be toxic to aquatic plants at naturally occurring concentrations; therefore, negligible risks to aquatic plants from aluminum are expected.
Zinc	Individual samples with HQs > 1.0 at RM 9.7, west	1.1	< 1%	November 2004 low-flow event	HQ was > 1.0 in one sample only; TRV is based on toxicity to fish and invertebrates; algae data indicate that aquatic plants toxicity to zinc is highly variable. Site-wide HQ is < 1.0	A low magnitude and frequency of HQ > 1.0 was calculated. There is high uncertainty concerning the relevance of the TRV to the aquatic plant community and high variability of algae-specific toxicity data. Although zinc is identified as a COC in surface water under the assumptions required for the BERA, it is more reasonable to conclude that zinc does not pose unacceptable risk to plant populations based on the low magnitude of exceedance and limited spatial extent of exceedances.

Table 10-1. Summary of Aquatic Plants Surface Water COCs

COC	Risk Exposure Area	HQs[a]	Frequency of All Samples with HQs > 1.0	Sampling Event with HQs > 1.0	Key Uncertainties	Risk Conclusions
PAHs						
Benzo(a)anthracene	Individual samples with HQs > 1.0 at RM 6.1, west; and RM 6.3, west	4.1 – 10	1.3%	July 2005 low-flow event and winter 2007 high-flow event	HQs were > 1.0 in peristaltic samples only at W012 and W031; TRV is based on extrapolated *Daphnia* acute LC50. Site-wide HQ is < 1.0	A low frequency of HQ > 1.0 was calculated. There is high uncertainty concerning the relevance of the TRV to the aquatic plant community and lack of plant-specific toxicity data. Although benzo(a)anthracene was identified as a COC in surface water under the assumptions required for the BERA, it is more plausible that benzo(a)anthracene does not pose unacceptable risk to plant populations based on the limited spatial extent of exceedances.
Benzo(a)pyrene	Individual samples with HQs > 1.0 at RM 6.1, west; and RM 6.3, west	1.4 – 14	1.9%	November 2004 and July 2005 low-flow events, and winter 2007 high-flow event	HQs were > 1.0 in peristaltic samples only at W012 and W031; TRV is based on extrapolated *Daphnia* acute LC50. Site-wide HQ is < 1.0	A low frequency of HQ > 1.0 was calculated. There is high uncertainty concerning the relevance of the TRV to the aquatic plant community and lack of plant-specific toxicity data.
Naphthalene	Individual samples with HQs > 1.0 RM 6.4, west	50	< 1%	May 2005	HQ was > 1.0 in only one sample; TRV is based on risk to fish and invertebrates; algae data indicate that aquatic plants may be less sensitive to naphthalene than are fish or invertebrates (i.e., all surface water concentrations are less than algae toxicity thresholds).	A low frequency of HQ > 1.0 was calculated. Given that no samples exceed the algae-specific TRV, negligible risks to aquatic plants from naphthalene are expected.

Table 10-1. Summary of Aquatic Plants Surface Water COCs

COC	Risk Exposure Area	HQs[a]	Frequency of All Samples with HQs > 1.0	Sampling Event with HQs > 1.0	Key Uncertainties	Risk Conclusions
Phthalates						
BEHP	Individual sample with HQ > 1.0 at RM 6.7, east (Willamette Cove)	1.2	< 1%	Winter 2007 high-flow event	HQ was > 1.0 in only one sample; TRV is based on risk to fish and invertebrates. Site-wide HQ is < 1.0	A low magnitude and frequency of HQ > 1.0 was calculated. There is high uncertainty concerning the relevance of the TRV to the aquatic plant community and lack of plant-specific toxicity data. Although BEHP was identified as a COC in surface water under the assumptions required for the BERA, it is more plausible that BEHP does not pose unacceptable risk to plant populations based on the limited spatial extent of exceedances.
PCBs						
Total PCBs	Individual samples with HQs > 1.0 at RM 3.7 (International Slip); and RM 6.9, east (Willamette Cove)	1.1 – 1.2	1.8%	November 2004 low-flow event and March 2005 low-flow event	HQs were > 1.0 in peristaltic samples only at W014 and W004; site-wide HQ is < 1.0; TRV is based on one study with fathead minnow; algae data indicate that aquatic plants may be less sensitive than fish (i.e., all surface water concentrations are less than algae toxicity thresholds).	A low magnitude and frequency of HQ > 1.0 was calculated. Given that no samples exceed the alternative water TRV that is based on direct exposure to surface water or an algae-specific TRV, negligible risks to aquatic plants from PCB exposure are expected.

Table 10-1. Summary of Aquatic Plants Surface Water COCs

COC	Risk Exposure Area	HQs[a]	Frequency of All Samples with HQs > 1.0	Sampling Event with HQs > 1.0	Key Uncertainties	Risk Conclusions
Pesticides						
Total DDx	Site-wide	1.2 – 20	21%	November 2004, March 2005, and July 2005 low-flow events, November 2006 storm runoff event and January 2007 high-flow event	23% of samples with total DDx HQs > 1.0 (n = 6 samples) are based on N-qualified data where HQs ranged from 1.4 to 20; HQs based on non-N-qualified data ranged from 1.2 to 9.8; TRV (0.001 µg /L) is based on risk to birds. One sample exceeds alternative water TRV that is protective of directed exposure to surface water (HQ = 1.8); however, this sample (W001 at RM 2.0) is N-qualified. Algae data indicate that aquatic plants thresholds are much higher than the selected and alternative TRV (and all surface water concentrations are less than algae toxicity thresholds).	Total DDx HQs were > 1.0 in 21% of samples (n = 26); however, about 25% of HQs > 1.0 are based on N-qualified data. Given that that algae-specific thresholds are higher than the selected and alternative TRV and that the only sample exceeded the alternative TRV based on direct exposure to surface water indicate sample interference, no unacceptable risks to aquatic plants from surface water are expected.

[a] HQs are presented only for water samples with exceedances of chronic TRVs. HQs in all other water samples were < 1.0.

BEHP – bis(2-ethylhexyl) phthalate
COC – chemical of concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
HQ – hazard quotient
J – estimated concentration
N – tentative identification
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
RM – river mile
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)

11.0 ECOLOGICAL RISK CONCLUSIONS

This section presents the overall risk conclusions for the BERA. Risk estimates in this BERA were calculated following CERCLA guidance (EPA 1997a, 1998) and EPA's Problem Formulation (Attachment 2). The conclusions of the BERA, along with those of the BHHRA (Appendix F of the draft RI) will be used to provide information to risk managers on potential 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 remainder of this section is organized as follows. Section 11.1 presents a summary of the identified COCs for all ecological receptors. Section 11.2 presents a discussion of background and upriver concentrations. The risk conclusions across all ecological receptor groups were combined in Section 11.3 to provide a holistic view of ecological risks and to identify the COC-receptor pairs that indicate potential unacceptable ecological risks, in the sense that remediation based on those COC-receptor pairs would address potential unacceptable risks posed by other COCs and to other ecological receptors. General recommendations from the risk assessors to risk managers are presented in Section 11.4.

11.1 SUMMARY OF ECOLOGICAL COCS

COC identification was an iterative process with decreasing conservatism with each iteration. If a particular COPC-LOE combination was not found to pose unacceptable risk to an ecological receptor, it was not carried forward in the risk assessment process.

For sediments, a COPC was identified as a COC if its concentration was higher than its site-specific SQG. SQGs were developed only for those chemicals demonstrating a predictive relationship between sediment chemistry and toxicity.

For the TZW LOE, only a screening level assessment was conducted; sixty COPCs were identified (Table 6-33) but COCs were not. The TZW collected from the Study Area is a snapshot of the limited potential ground water discharge areas where the data were collected. Furthermore, the ability of benthic organisms to limit their exposure to anoxic porewater calls into question the ecological relevance of a TZW exceedance. The screening level assessment was sufficient to conclude that potential effects from TZW exposure would generally occur in locations where effects are also predicted from sediment exposure.

All other receptor-COPC-LOE combinations were carried forward through refined stages of risk assessment. The process proceeded from more conservative to more realistic exposure and effects assumptions until such time that a conclusion about unacceptable risk to the

population or community assessment endpoint[148] was warranted, or the limits of the available exposure and effects data were reached.

For all LOEs other than TZW and benthic sediment toxicity, COCs were identified as those COPCs with HQs > 1.0 (i.e., concentration greater than an effect threshold) based on ecologically relevant exposure scales and dietary assumptions. The BERA conclusions are indeterminate for COCs that made the list based on data limitations. In some cases, such as where the spatial extent of TRV exceedances is small, or where TRVs are based on organism-level effects that do not translate into population-level responses, a conclusion that a chemical poses negligible risk to a receptor is warranted even for COCs.

Population versus Organism Risks

In the Portland Harbor BERA, effects (i.e., survival growth, or reproduction) on individual organisms were conservatively used to represent population-level effects (e.g., species diversity, population growth rate, or age structure of the population) in determining risks to many of the target receptors. This approach follows the conventional practice in ecological risk assessment of using organism-level measurement and effects endpoints; however, organism-to-population extrapolation is a source of uncertainty and, in many but not all cases, a source of conservatism in risk estimates. Populations are resilient to environmental stressors. Evidence of effects on individual organisms for evaluating risks to populations rests on inferences that have little or no scientific basis because populations compensate for adverse effects on organisms through mechanisms such as physiological and genetic adaptation, biological compensation, and other ecological processes (Caswell 2001; Pastorok et al. 2001; Vandermeer and Goldberg 2003; Barnthouse et al. 2009).

Several methods have been used in an attempt to address potential population-level effects, but no consensus on approach currently exists. In other ERAs, risks to populations have been assessed using the 95th UCL concentrations in exposure media representing the site or some portion of the site (Windward 2007; MacDonald Environmental 2002). The Oregon Department of Environmental Quality (ODEQ 1998) has defined acceptable risk to a population as ≤ 10% chance that > 20% of the total population would receive an adverse exposure (adverse exposure is defined as greater than the LD50 or LC50 for a hazardous substance based on studies with route and exposure duration simulating field exposure conditions of the ecological receptor). No such analysis was conducted for the Portland Harbor BERA, because no agreement on approach was reached for this site, and they were not identified in Problem Formulation documents. However, more explicit analysis could be helpful in the FS and post-FS stages of the project to help identify the most appropriate risk management actions needed to protect the Assessment Endpoints.

The uncertainties associated with risk estimates for individual COCs, the spatial distribution of COC exceedances, the magnitude of exceedance, and the type of effect threshold all play a role in identifying whether chemicals pose a population-level risk. The likelihood that a COC with a limited spatial distribution of exceedances and a low HQ poses significant risks to populations is low. COCs with a broader distribution and higher magnitude of exceedances have a greater potential for posing population-level risks. The type of effect threshold endpoint (i.e., survival, growth, or reproduction) affects the interpretation of HQs simply because effects on the survival of individuals have different implications for populations than do effects on the growth of individuals. Effects on the reproductive success of individuals have different implications for populations than do effects on either survival or growth.

Even for a particular type of endpoint (e.g., reduced survival), a particular HQ (e.g., HQ = 5) has different implications for each COC-receptor pair, because the exposure-response relationship of one COC-receptor pair differs from that of other COC-receptor pairs. Short of developing structured population models and incorporating the effects of COC-receptor-specific exposure-response data, the demographic (i.e., population-level) consequences of any particular HQ simply cannot be quantified.

[148] The assessment endpoint for the evaluation of special-status-species receptors (i.e., juvenile Chinook salmon, Pacific lamprey ammocoetes, and bald eagles) was at the organism level.

Table 11-1 presents a summary of the COCs (individual chemicals, sums, or totals) identified for each ecological receptor group. In total, 31 ecological COCs were identified in this BERA:

- **Benthic invertebrates** – Twenty-seven COCs (individual chemicals, sums, or totals) were identified based on exceedance of the SQG, tissue TRV, or surface water TRV.
- **Fish** – Seventeen COCs were identified using the tissue-residue, dietary-dose, and surface water LOEs.
- **Wildlife** – Ten COCs were identified for birds using the dietary-dose and egg LOEs, and three COCs were identified for mammals using the dietary-dose LOE.
- **Amphibians** – Eight COCs were identified using the surface water LOE.
- **Aquatic plants** – Eight COCs were identified using the surface water LOE.

Table 11-1. Summary of COCs Across all Ecological Receptor Groups

COC[a]	Benthic Invertebrates	Fish	Birds	Mammals	Amphibians	Aquatic Plants
Metals						
Aluminum	X	X			X	X
Arsenic	X					
Cadmium	X	X				
Copper	X	X	X			
Lead		X	X	X		
Mercury	X	X	X			
Silver	X					
Zinc	X	X			X	X
Butyltins						
Tributyltin	X	X				
Butyltin	X	X				
PAHs						
Benzo(a)anthracene	X	X			X	X
Benzo(a)pyrene	X	X	X		X	X
Naphthalene	X	X			X	X

Table 11-1. Summary of COCs Across all Ecological Receptor Groups

COC[a]	Benthic Invertebrates	Fish	Birds	Mammals	Amphibians	Aquatic Plants
Total benzofluoranthenes	X					
Total HPAHs	X					
Total LPAHs	X					
Phthalates						
BEHP	X	X			X	X
Dibutyl phthalate			X			
Phenols						
Phenol	X					
SVOCS						
Benzyl alcohol	X					
Carbazole	X					
PCBs						
Total PCBs	X	X	X	X	X	X
Dioxins/Furans						
Total TEQ			X[b]	X[b]		
VOCs						
Trichloroethene	X	X				
Ethylbenzene	X	X				
Pesticides						
Aldrin			X			
DDx compounds	X[c]	X	X		X	X
Dieldrin	X					
Endrin	X					
Endrin ketone	X					
delta-HCH	X					

[a] COCs were identified as those COPCs for which HQ were > 1.0 based on a relevant exposure scale.
[b] Dioxins, furans, and dioxin-like PCB congeners expressed as TEQs.

[c] 4,4'-DDD and total DDx.

BEHP – bis(2-ethylhexyl) phthalate
COC – chemical of concern
COPC – chemical of potential concern
DDD – dichlorodiphenyldichloroethane
DDE – dichlorodiphenyldichloroethylene
DDT – dichlorodiphenyltrichloroethane
HCH – hexachlorocyclohexane
HQ – hazard quotient
HPAH – high-molecular-weight polycyclic aromatic hydrocarbon
LPAH – low-molecular-weight 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)
TPH – total petroleum hydrocarbons

11.2 BACKGROUND AND UPRIVER CONCENTRATIONS

Some chemicals identified as COCs for ecological receptors in the Study Area occur at concentrations similar to the sediment and surface water background levels defined in Section 7.0 of the draft RI 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). Although risk estimates indicate the potential for unacceptable risks in the Study Area, some risks are associated with regional rather than Study-Area-specific contamination.

Attachment 11 presents a comparison for all ecological COCs of background and Study Area UCLs in sediment and surface water. For aluminum, di(n)butyl phthalate, benzyl alcohol, and alpha-endosulfan, background sediment UCLs were the same as or higher than Study Area UCLs. The background surface water UCL concentration was higher than the Study Area UCL only for aluminum. Attachment 11 also includes a comparison of fish tissue concentrations from the upriver reach and the Study Area for all wildlife COCs and fish tissue-residue COCs. Although fish tissue data from the upriver reach were insufficient to allow calculation of UCLs, their concentrations were similar to those in the Study Area for aluminum, mercury, and copper, as presented in Section 7.1.4.3.

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. These COCs are identified and discussed in Section 11.3.

11.3 ECOLOGICAL RISK CONCLUSIONS

PCBs are the most significant contributor to unacceptable ecological risk in the Study Area. Total PCBs was the only COPC identified as a COC for all six ecological receptor groups (Table 11-1). This was not unexpected; PCBs frequently dominate assessed risks at contaminated sediment sites. PCBs have also been identified in the BHHRA (Appendix F of the draft RI) as the predominant contributor to potential human health risks in the Study Area. Unacceptable risk to mink from PCBs represents the single highest likelihood that a

chemical will impact an ecological receptor population in the Study Area. Unacceptable risks to other fish, wildlife, amphibians and plants associated with PCBs and other COCs (Table 11-2) would be reduced or eliminated by sediment remedies that address mink PCB risks. The extent of risk reduction will become apparent as AOPCs are developed for the FS. The detailed evaluation of the PCB risks to mink summarized in Section 11.3.1 illustrates the relevance of the risk results to site-wide decisions.

COCs occur at levels that are projected to pose unacceptable benthic risks for about 5% 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. Toxicity testing and modeling and benthic tissue-residue data identified metals, TBT, PAHs, SVOCs, phenol, BEHP, total PCBs, and pesticides as COCs, based on a point-by-point assessment of potential effects on benthic organisms. A WOE was assessed to extrapolate from the organism-level measurement endpoints to the population and community-level assessment endpoints. Based on that evaluation, the COCs that pose potential unacceptable risk to the benthic community or populations are PAHs, PCBs, and DDx compounds.

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 (steeply sloped areas, areas of sediment transport 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, which is also the reach with the greatest area associated with potential unacceptable risk to the benthic community, suggesting possible chemical toxicity, among other potential factors, as the reason for the presence of lower successional stages in this reach. 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 some depositional areas that have received historical releases of contamination.

Table 11-2 presents a summary of risk conclusions for each COC across all ecological receptor groups[149]. Only receptor groups for which a COC was identified are presented in Table 11-2. No unacceptable risk is predicted for receptor groups that are not listed under a given COC. Risk conclusions were determined by evaluating the magnitude of HQs, spatial distribution and frequency of HQs > 1.0, the results of multiple LOEs, and the uncertainty of exposure and effects assumptions for estimating appropriate community or population-level risks. Background sediment and surface water concentrations and upriver fish tissue concentrations were considered, as appropriate, for context. Background concentrations were not, however, "subtracted out" or otherwise used to discount unacceptable ecological risks. Although HQs for at least one ecological LOE were > 1.0 for aluminum and dibutyl phthalate, the Study Area concentrations were not elevated compared to background, indicating that unacceptable ecological risk in the Study Area associated with these COCs may be due in whole or in part to sources outside the Study Area. Aluminum, mercury, and copper concentrations in Study Area fish tissue were generally similar to or less than concentrations in upriver reach fish tissue samples.

[149] A COC is not necessarily equated with unacceptable risk. As previously stated, for all LOEs other than TZW and benthic sediment toxicity, COCs were identified as those COPCs with HQs > 1.0 (i.e., concentration greater than an effect threshold) based on ecologically relevant exposure scales and dietary assumptions. The BERA conclusions are indeterminate for COCs that made the list based on data limitations. In some cases, such as where the spatial extent of TRV exceedances is small or where TRVs are based on organism-level effects that do not translate into population-level responses, a conclusion that a chemical poses negligible risk to a receptor is warranted even for COCs.

Table 11-2. Summary of COCs and Risk Conclusions Across All Ecological Receptors

COC	Receptor Group	Unacceptable Risk?	Justification for Risk Conclusions
Metals			
Aluminum	Benthic invertebrates	No	Aluminum and other trace elements are major constituents of the mineral fraction of sediment but contribute to the analytical chemical results as a result of the acid extraction step during analysis. Because aluminum is not biologically available and not toxic at naturally occurring concentrations generally found in surface water, aluminum is not expected to pose unacceptable risks to benthic invertebrates, fish, amphibians, or aquatic plants. Furthermore, aluminum concentrations in background areas indicate that aluminum is not attributable to anthropogenic sources in the Study Area.
	Fish	No	
	Amphibians/ aquatic plants	No	
Arsenic	Benthic invertebrates	No	Arsenic poses negligible risks to the benthic community or populations because of two exceedances of the TRV, the low magnitude of HQ, high uncertainty with respect to metals toxicity based on the tissue LOE, and two LOEs (site-specific SQGs and surface water) that identify no unacceptable risk associated with arsenic..
Cadmium	Benthic invertebrates	No	Cadmium poses negligible risks to the benthic community or populations because of the low frequency and limited spatial extent of EFs > 1.0 and two LOEs (tissue residue and surface water) that identify no unacceptable risk associated with cadmium. Only one bioassay station was located within the areas with SQG exceedances; this stations was toxic based on all four endpoints and exceeded the SQGs for numerous chemicals.
	Fish	No	Although cadmium was identified as a COC for sculpin and juvenile Chinook salmon based on the dietary-dose LOE under the assumptions required for the BERA, it is more reasonable to conclude that cadmium does not pose unacceptable risk to sculpin populations or individual juvenile Chinook salmon based on the low magnitude of exceedances of a conservative TRVs (HQs ranged from 1.1 to 3.5 and the selected TRVs are 2 to 3 orders of magnitude lower than other thresholds reported in the literature), and because the surface water LOE does not identify cadmium as a COC.
Copper	Benthic invertebrates	No	Copper poses negligible risks to the benthic community or populations because of the low magnitude of EFs/HQs > 1.0, high uncertainty with respect to metals toxicity based on the tissue LOE, and one LOE (surface water) that identify no unacceptable risk associated with copper. Using the tissue-residue LOE for metals is highly uncertain because benthic invertebrates regulate metals, confounding inter-species extrapolation, and the tissue-residue TRV for copper cannot reliably predict toxicity. Only one bioassay station was located within the areas with SQG exceedances; the station was toxic based on the *Hyalella* mortality and growth endpoints.
	Birds	No	Although copper was identified as a COC for spotted sandpiper under the assumptions required for the BERA, it is more reasonable to conclude that copper does not pose an unacceptable risk to the spotted sandpiper population based on the uncertainty in the dietary assumption (i.e., that sandpiper fed exclusively on the maximally contaminated prey type), low magnitude of exceedances of a conservative and uncertain TRV (HQs ranged from 1.1 to 1.3 and selected LOAEL was based on an Eco SSL which was lower than all bounded NOAELs and LOAELs reported in the literature).
	Fish	No	Although copper was identified as a COC for juvenile Chinook salmon, northern pikeminnow, and sculpin based on the dietary-dose LOE (HQs ranged from 1.1 to 3.6) and for sculpin and Pacific lamprey based on the tissue LOE (HQs ranged from 1.1 to 2.3) based on the assumptions required for the BERA, it is more reasonable to conclude that copper does not pose an unacceptable risk to fish populations because of the low magnitude of HQ exceedances, uncertainties associated with tissue residue and dietary effects data, and because the surface water LOE does not identify copper as a COC. The HQs from the dietary LOE are uncertain because HQs are based on conservative and uncertain TRVs that could not be replicated and that are at or near the nutritional range requirements for fish. When the next lowest dietary TRVs are used, all HQs are < 1.0; therefore dietary risks are assumed to be over-predicted. In addition, the risk conclusions from the tissue-residue LOE are uncertain because fish regulate copper concentrations in tissue, and the measured tissue-residues are relatively close to concentrations associated with nutritional deficiencies. Given the uncertainties of the tissue and dietary LOEs, the surface water LOE and alternative dietary TRV HQs provide the strongest LOEs and indicate no unacceptable risk to fish in the Study Area. It should also be noted that upriver fish tissue copper concentrations of various fish species (including lamprey) are similar to those of Study Area fish.
Lead	Birds	No	Lead was identified as a COC for osprey based on the dietary-dose LOE in one exposure area (RM 9.5 to RM 10.5 with an HQ of 7.8) based on an exceedance of the LOAEL by the smallmouth bass sample from this exposure area which had an unusually high lead concentration. Nesting success of osprey was monitored along the Willamette River system between 1993 and 2001 (Henny et al. 2008). These data indicate that osprey populations from the LWR (including the Study Area) have increased in recent years, with productivity above that which is necessary for maintaining a stable population. Although lead was identified as a COC under the assumptions required for the BERA, it is more reasonable to conclude that lead does not pose an unacceptable risk to the osprey population based on the limited spatial extent of LOAEL exceedance (in only one exposure area) and field data available for osprey populations.

Table 11-2. Summary of COCs and Risk Conclusions Across All Ecological Receptors

COC	Receptor Group	Unacceptable Risk?	Justification for Risk Conclusions
	Mammals	No	Lead was identified as a COC for mink based on the dietary-dose LOE in one exposure area (RM 9.5 to RM 10.5 with an HQ of 4.0). The question of whether lead exposure poses unacceptable risk to mink in the Study Area hinges on the true representativeness of the extremely high lead concentration in the single smallmouth bass sample from RM 9.5 to RM 10.5. Even if lead does pose an unacceptable risk within this exposure area, the limited spatial extent of TRV exceedance, and the uncertain and conservative TRV used (the selected LOAEL was based on an Eco SSL derived from a drinking water exposure that was one order of magnitude lower than the literature-based dietary LOAEL) indicates that the risk would not rise to the population level.
	Fish	No	Lead was identified as a COC for peamouth and smallmouth bass based on a tissue TRV exceedance in one sample of each species (HQs of 2.7 and 280, respectively). Lead was not identified as a COPC based on the surface water and dietary LOEs indicating no unacceptable risk is expected from water and dietary exposure pathways. The risk conclusions from the tissue-residue LOE are uncertain because a limited number of toxicity studies were available. Even if risks from lead are elevated for some few smallmouth bass or peamouth, the limited frequency of TRV exceedances indicates that the risks would not rise to the population-level. Because the tissue chemical concentration in the single smallmouth bass outlier (located between RM 9.5 and RM 10.5) is so much higher than all other sample concentrations, it is reasonable to speculate that the source of the lead is discrete (e.g., possibly resulting from a lead sinker) rather than dispersed in sediment or water.
Mercury	Benthic invertebrates	No	Mercury poses negligible risks to site wide benthic community or populations. Because of the low frequency and limited spatial extent of EFs > 1.0 and two LOEs (tissue residue and surface water) that identify no risk associated with mercury. Only one bioassay station was located within the areas with SQG exceedances; this station was toxic based on all four endpoints and exceeded the SQGs for numerous chemicals. Mercury concentration in sediment is elevated and may affect individual organisms in Willamette Cove, in Cascade General Shipyard, and at the southeast corner of Swan Island Lagoon. Potential effects on benthic community in Willamette Cove based on a high EF and elevated concentrations in the PBRA as compared to the maximum concentrations in the non-toxic bioassay station and at Cascade General Shipyard based on an elevated EF.
	Birds	Yes (background)[b]	Mercury was identified as a COC for bald eagle based on the dietary-dose LOE in all exposure areas. Mercury was not identified as a COC based on the egg LOE; However, given the availability of the more reliable dietary-dose LOE that assesses the same risk, the egg LOE should be given minimal consideration. Based on the dietary-dose LOE, mercury might pose low risks to individual bald eagles if they subsist exclusively on fish (HQs ranged from 1.2 to 1.7 in all exposure areas). There is some uncertainty with the selected TRV which is a NOAEL that was extrapolated from a LOAEL. It should be noted that mercury exposure of eagles feeding in the Study Area is no greater than exposure when feeding in the upriver reach of the LWR and that average Willamette River-wide fish tissue mercury concentrations are greater than average Study Area fish tissue concentrations.
	Fish	No	Although mercury was identified as a COC for northern pikeminnow based on the tissue LOE under the assumptions required for the BERA, it is more reasonable to conclude that mercury does not pose unacceptable risk to northern pikeminnow populations based on the low magnitude and limited spatial extent of TRV exceedances (HQ was in 1.1 in one exposure area from RM 6.5 to RM 7.5), and because the surface water and dietary-dose LOEs do not identify mercury as a COC. It should be noted that upriver fish tissue mercury concentrations are similar to (or greater than) those of Study Area fish and that average Willamette River-wide northern pikeminnow tissue mercury concentrations are greater than average Study Area northern pikeminnow tissue concentrations.
Silver	Benthic invertebrates	No	Silver poses negligible risks to the benthic community or populations because of the low magnitude, low frequency, and limited spatial extent of EFs > 1.0 and two LOEs (tissue residue and surface water) that identify no unacceptable risk associated with silver. Organism level effects may be observed at the southern end of Swan Island Lagoon. Only one bioassay station was located within the areas with SQG exceedances; this station was toxic based on all four endpoints and exceeded the SQGs for numerous chemicals.
Zinc	Benthic invertebrates	No	Zinc poses negligible risks to the benthic community or populations because of the low magnitude of HQs > 1.0, high uncertainty with respect to metals toxicity based on the tissue LOE, and one LOE (site-specific SQG) that identify no unacceptable risk associated with zinc. Using the tissue-residue LOE for metals is highly uncertain because benthic invertebrates regulate metals, confounding inter-species extrapolation, and the tissue-residue TRV for zinc cannot reliably predict toxicity.
	Fish	No	Although zinc is identified as a COC based on the surface water LOE for small-home-range fish (i.e., sculpin, smallmouth bass, and northern pikeminnow) under the assumptions required for the BERA, it is more likely that zinc does not pose unacceptable risk to fish populations based on the low magnitude, limited spatial extent of TRV exceedances (only one sample exceeded at RM 9.7 on the west bank with an HQ of 1.1), uncertainty associated with the tissue LOE, and because the dietary LOE does not identify zinc as a COC. The tissue LOE is to uncertain because fish regulate metals and the tissue-residue TRV for zinc cannot reliably predict toxicity.
	Amphibians/ aquatic plants	Unknown	Zinc was identified as a COC based on the exceedance of the water TRV in one sample at RM 9.7 on the west bank where the HQ was 1.1 during a non-reproductive period for amphibians. Given the high uncertainty concerning the relevance of the water TRV derived from invertebrate and fish data to amphibian populations and the aquatic plant community (highly variable algae-specific toxicity data) , risk conclusions cannot be made for these receptors.

Table 11-2. Summary of COCs and Risk Conclusions Across All Ecological Receptors

COC	Receptor Group	Unacceptable Risk?	Justification for Risk Conclusions
Butyltins			
Butyltin	Benthic invertebrates	No	Butyltin was identified as a surface water COC using the TBT TRV as a surrogate. Because of the low magnitude and frequency of surface water samples with HQs > 1.0 (only one sample exceeded at RM 11 on the west bank with and HQ of 1.2), uncertainty associated with use of TBT effects data as a surrogate for butyltin, and two LOEs (site-specific SQG and tissue-residue) that identify no unacceptable risk associated with butyltin, no unacceptable risks to the benthic invertebrate community are expected from surface water.
	Fish	No	Butyltin was identified as a surface water COC for small-home-range fish (i.e., sculpin, smallmouth bass, and northern pikeminnow) using the TBT TRV as a surrogate. Because of the low magnitude and frequency of surface water samples with HQs > 1.0 (only one sample exceeded at RM 11 on the west bank with and HQ of 1.2) and uncertainty associated with use of TBT effects data as a surrogate for butyltin, no unacceptable risks to small-home-range fish are expected from surface water.
TBT	Benthic invertebrates	No	TBT poses negligible risks to the harbor-wide benthic community or populations because of limited spatial extent and high uncertainty associated with predicted tissue TBT concentration and two LOEs (site-specific SQG and surface water) that identify no risk associated with TBT. Sediment concentration range used to develop BSAR was an order of magnitude lower than concentration on which the predicted tissue exceedance was based. TRV probably over-predicts effects to invertebrates. Organism-level effects might occur in the Cascade General Shipyard, and in some locations in Swan Island Lagoon.
	Fish	No	No unacceptable risks to any fish receptors from TBT are expected based on the surface water or tissue LOE. TBT was identified as a COC for all fish receptors based on the dietary-dose LOE (HQs ranged from 1.3 to 69); however, there are uncertainties associated with the dietary TRV (only a single uncertain LOAEL was identified). Elevated TBT fish prey concentrations were restricted to the mouth of Swan Island Lagoon and at the International Slip. Thus, while TBT was identified as a COC based on the dietary LOE using a TRV associated with high uncertainty, it is more likely that TBT does not pose unacceptable risk to fish populations based on the tissue and surface water LOEs and based on the fact that dietary risks from exposure to TBT are driven by localized areas. Therefore, unacceptable risks to fish populations are not likely.
PAHs			
Benzo(a) anthracene	Fish	No	Although benzo(a)anthracene was identified as a COC based on the exceedance of the water TRV for small-home-range fish (i.e., sculpin, smallmouth bass, and northern pikeminnow) under the assumptions required for the BERA, it is more plausible that benzo(a)anthracene does not pose an unacceptable risk to fish populations based on the limited frequency and spatial extent of exceedances (only two samples exceeded at RM 6.1 and RM 6.3 on the west bank with HQs of 4.1 and 10, respectively), uncertainty associated with the water TRV, and because benzo(a)anthracene and PAH mixtures were not identified as COCs for the dietary LOE. The surface water TRV is uncertain because it was extrapolated based on acute *Daphnia* toxicity data. Because of the uncertainties associated with the surface water LOE (specifically the water effect threshold), risk conclusions for PAHs should be based on the results of the dietary LOE.
	Amphibians/ aquatic plants	Unknown	Benzo(a)anthracene was identified as a COC for plants based on exceedance of the TRV by two water samples at RM 6.1 and RM 6.3 on the west bank with HQs of 4.1 and 10, respectively. Given the high uncertainty concerning the relevance of the water TRVs for individual PAHs to the aquatic plant community and amphibian populations because limited amphibian-specific thresholds are available and no aquatic plant or algae-specific thresholds are available, risk conclusions cannot be made for these receptors.
Benzo(a)pyrene	Birds	Unknown	Benzo(a)pyrene was identified as a COC for spotted sandpiper based on the exceedance of the dietary-dose LOAEL in one exposure area (RM 4.0 and RM 6.0) with an HQ of 1.6 based on ingestion of worms only. The uncertainty in the dietary assumption that lab worms represent the entire spotted sandpiper diet and the limited spatial extent and magnitude of exceedance indicate that there is no unacceptable risk to spotted sandpiper populations; however, it was not possible to conclusively determine the likelihood of an unacceptable risk to shorebirds from benzo(a)pyrene because of high uncertainty associated with the LOAEL (LOAEL is based on a weekly injection of benzo(a)pyrene into pigeon rather than an ecologically relevant exposure; injection studies were considered an unacceptable form of exposure in the selection of TRVs).
	Fish	No	Benzo[a]pyrene or PAH mixtures were not identified as COCs based on the dietary LOE so no unacceptable risks are expected. Although, based on the exceedance of the water TRV, benzo(a)pyrene was identified as a COC for small-home-range fish (i.e., sculpin, smallmouth bass, and northern pikeminnow), there was a limited frequency and spatial extent (only two samples exceeded at RM 6.1 and RM 6.3 on the west bank with HQs of 1.4 and 14, respectively) of exceedances and some uncertainty associated with the water TRV (TRV was extrapolated based on acute *Daphnia* toxicity data). Because of the uncertainties associated with the surface water LOE (specifically the water effect threshold), the dietary LOE is considered a stronger LOE; however, results of both LOEs suggest no unacceptable risks to fish populations.
	Amphibians/ aquatic plants	Unknown	Benzo(a)pyrene was identified as a COC for plants based on exceedance of the TRV by two water samples at RM 6.1 and RM 6.3 on the west bank with HQs of 1.4 and 14, respectively. Given the high uncertainty concerning the relevance of the water TRVs for individual PAHs to the aquatic plant community and amphibian populations because limited amphibian-specific thresholds are available and no aquatic plant or algae-specific thresholds are available, risk conclusions cannot be made for these receptors.

Table 11-2. Summary of COCs and Risk Conclusions Across All Ecological Receptors

COC	Receptor Group	Unacceptable Risk?	Justification for Risk Conclusions
Naphthalene	Fish	No	PAH mixtures were not identified as COCs based on the dietary LOE. Although naphthalene was identified as a COC for small-home-range fish based on the exceedance of the water TRV, (HQs for sculpin and smallmouth bass/northern pikeminnow were 14 and 2.8, respectively), there was a low frequency and limited spatial extent of exceedances. Due to the low frequency (4% of all samples) and limited spatial extent of exceedances (samples with HQs > 1.0 were limited to a single localized area from RM 6.4 to 6.5), no unacceptable risks to fish populations are expected.
	Amphibians/ aquatic plants	No	Although naphthalene was identified as a COC for amphibians and aquatic plants based on the exceedance of the water TRV by one samples at RM 6.4 on the west bank with an HQ of 50, it is reasonable to conclude that naphthalene does not pose an unacceptable risks to amphibians populations or aquatic plant communities because of the limited spatial extent and because no samples exceeded amphibian- or algae-specific water thresholds.
PAHs[a]	Benthic invertebrates	Yes	PAHs were identified as COCs based on SQG and water TRV exceedances. Unacceptable risks to the benthic community or populations were estimated to occur on the west side of the river between approximately RM 6.1 and RM 6.6 and in three areas in the navigational channel between approximately RM 5.1 and RM 5.8. PAHs were not assessed using the tissue-residue LOE. Fifteen bioassay stations were located within the areas with SQG exceedances; four of these stations were non-toxic, four were toxic based on all four endpoints, and seven were toxic based on 1 to 3 endpoints.
SVOCs			
Benzyl alcohol	Benthic invertebrates	No	Benzyl alcohol poses negligible risks to the benthic community or populations because of the low frequency and limited spatial extent of SQG exceedances (detected concentration exceeded SQG at only a few locations) and two LOEs (tissue residue and surface water) that identify no unacceptable risk associated with benzyl alcohol. One bioassay station was located in the area with SQG exceedance; the bioassay station was toxic based on all four endpoints. Two other chemicals, phenol and total PCBs, exceeded the SQGs at this station.
Carbazole	Benthic invertebrates	No	Carbazole poses negligible risks to the benthic community or populations because of the low frequency and limited spatial extent of EFs > 1.0 and two LOEs (tissue residue and surface water) that identify no unacceptable risk associated with carbazole. Six bioassays were located within the areas with SQG exceedances; three of these stations were toxic based on all four endpoints, and three were toxic based on 1 to 2 endpoints. The seven locations with relatively high EFs (between 7.5 and 29) are co-located with relatively high LPAH concentrations. Quantification of carbazole is uncertain.
Phenols			
Phenol	Benthic invertebrates	No	Isolated SQG exceedances do not represent a population- or community-level effect. Two LOEs (tissue residue and surface water) identify no unacceptable risk associated with phenol. One bioassay station was located in the area with SQG exceedance; the bioassay station was toxic based on all four endpoints. Two other chemicals, benzyl alcohol and total PCBs, exceeded the SQGs at this station. Quantification of phenols is uncertain.
Phthalates			
BEHP	Benthic invertebrates	No	BEHP poses negligible risks to the benthic community or populations because of the low magnitude and frequency of HQs > 1.0 and one LOE (site-specific SQG) that identify no unacceptable risk associated with BEHP. The single tissue TRV exceedance was for a laboratory-exposed clam. The water TRV was exceeded during two sampling events (two individual samples exceeded at a horizontal transect sampling location at RM 3.9 and at RM 6.7 on the east bank [Willamette Cove] with HQs of 1.2 and 2.3).
	Fish	No	BEHP concentrations in one sculpin tissue sample and two smallmouth bass samples were greater than the only acceptable NOAEL identified (no LOAEL was available); however, an HQ > 1.0 based on an unbounded NOAEL does not indicate risks. Although BEHP was identified as a surface water COC under the assumptions required for the BERA, it is more reasonable to conclude that BEHP does not pose an unacceptable risk to small-home-range fish (i.e., sculpin, smallmouth bass, and northern pikeminnow) populations based on the low frequency and magnitude of exceedance and limited spatial extent of exceedances (only two individual samples exceeded at a horizontal transect sampling location at RM 3.9 and at RM 6.7 on the east bank [Willamette Cove] with HQs of 1.2 and 2.3).
	Amphibians	No	Although BEHP was identified as a COC for amphibians based on surface water LOE, it is reasonable to conclude that BEHP does not pose an unacceptable risks to amphibians populations because the only sample with an HQ > 1.0 (collected at RM 6.7 on the east bank [Willamette Cove]) was collected during a non-reproductive period for amphibians and the HQ magnitude was low (HQ = 1.2).
	Aquatic plants	Unknown	BEHP was identified as a COC for aquatic plants based on exceedance of the TRV by one water samples at RM 6.7 on the east bank with an HQ of 1.2.Given the high uncertainty concerning the relevance of the water TRV to the aquatic plant community, because no aquatic plant or algae-specific thresholds are available, risk conclusions cannot be made for the aquatic plant community.

Table 11-2. Summary of COCs and Risk Conclusions Across All Ecological Receptors

COC	Receptor Group	Unacceptable Risk?	Justification for Risk Conclusions
Dibutyl phthalate	Birds	No	Although dibutyl phthalate was identified as a COC for spotted sandpiper under the assumptions required for the BERA, it is more reasonable to conclude that dibutyl phthalate does not pose an unacceptable risk to the spotted sandpiper population based on the conservatism in the dietary assumption that sandpipers consume solely the most contaminated prey item, the limited spatial extent of exceedance (the only exposure area with an HQ was 1.4 in the exposure area between RM 7.0 and RM 9.0), and the uncertainty associated with extrapolating the LOAEL from a NOAEL. It also should be noted that sediment dibutyl phthalate concentrations are not elevated relative to background.
PCBs			
Total PCBs	Benthic invertebrates	Yes	Probable benthic risks exist based on the magnitude, frequency, and spatial extent of EFs/HQs > 1.0 in sediment and tissue. There are probable risks to the benthic community along the east/north shoreline between RM 2.1 and RM 2.4 based on bioaccumulation risk; on the south side of International Slip based on bioaccumulation risk, sediment concentrations greater than the maximum concentration at the non-toxic bioassay stations, and toxic bioassay stations; in localized areas in Swan Island Lagoon and at RM 8.8 based on predicted bioaccumulation risks; and along the east/north shoreline near RM 11.3 based on bioaccumulation risk. The slips of Terminal 4 were identified as possible risk areas based on SQG exceedances. The risk is uncertain because no bioassays were performed in these slip areas, and there is no predicted bioaccumulation risk.
	Birds	Yes	Total PCBs were identified as a COC for all bird receptors based on dietary and egg LOEs in exposure areas throughout the Study Area. The risk conclusions for bald eagle and osprey from the highly uncertain egg LOE are consistent with the dietary-dose LOE; however, HQs were of much higher magnitude based on the bird egg LOE. Given the availability of the more reliable dietary-dose LOE that assesses the same risk, the egg LOE should not be used to determine risk conclusions. Total PCBs are expected to pose low risk to the hooded merganser and osprey populations given the low LOAEL exceedances using the dietary-dose LOE (HQ ranged from 1.1 to 3.8) and likely conservatism in the TRV (LOAEL is based on the reduced hatchability of chicken chicks and may overestimate risks to other avian species). Dietary exposure in spotted sandpiper exceeds the LOAEL over some of the shorebird foraging areas (HQs ranged from 1.7 to 12 in all exposure areas). Individual bald eagles that forage exclusively on aquatic fish from the Study Area also may face some increased risk of reduced reproductive success. The effects associated with spotted sandpiper and bald eagle exceedances are uncertain, but it is more reasonable to conclude that there would be some reduction in reproductive success (dietary-dose HQ ranged from 3.8 to 3.9 in all exposure areas). It should be noted that any potential PCB risk to spotted sandpiper, hooded merganser, osprey populations or individual bald eagles associated with Study Area sediment contamination are coincident with unacceptable risk to mink because the magnitude and extent of risks to mink are greater and have lower associated uncertainty than for any other receptor.
	Mammals	Yes	Total PCBs were identified as a COC for both mink and river otter in all exposure areas of the Study Area with HQs ranging from 19 to 33. It is reasonable to conclude, based on the information available for the BERA, that mink and river otter in the Study Area might be at risk from exposure to PCBs and that the level of risk is sufficient to affect reproductive success in Study Area mink and river otter populations. The selected LOAEL is the lowest of several studies investigating PCB effects on mink and corresponds with a magnitude of effect on reproduction that would plausibly result in reduced population viability. The dietary exposure of mink to PCBs is the single greatest ecological risk in the Study Area. If it is assumed that mink and river otter consume essentially the same diet, then mink provide a conservative surrogate for river otter because mink have higher metabolic requirements and a smaller foraging range.
	Fish	No	Although total PCBs were identified as a COC for sculpin (based exceedance of on the water TRV by two samples at RM 3.7 [International Slip] and RM 6.9 [Willamette Cove] with HQs of 1.1 and 1.2, respectively). and largescale sucker, sculpin, smallmouth bass, and northern pikeminnow based on the tissue LOE (HQ exceedances ranged from 1.1 to 7.1) and for empirical sculpin tissue (HQ exceedances range from 2.5 to 9.4) under the assumptions required for the BERA, it is more reasonable to conclude that total PCBs do not pose an unacceptable risk to fish populations based on the high uncertainty associated with the 10th percentile LOAEL (which is expected to overestimate risks to fish because of the toxicity data included in the SSD used to derive LOAEL) and based on the fact that no surface water samples exceeded the alternative water TRV . The water TRV is highly uncertain because it is based on protection of mink. HQs calculated based on an alternative water TRV that is protective of direct exposure to surface water are less than 1.0. Several areas were identified where predicted sculpin HQs were > 1.0; however, these areas were near or at the same locations where empirical sculpin tissue samples had HQs < 1.0 and empirical data are assumed to be more representative of tissue concentrations in Study Area sculpin than are predicted tissue chemical concentrations. Any potential PCB risk to fish populations associated with Study Area sediment contamination are coincident with an unacceptable risk to mink because the magnitude and extent of risks to mink are greater and have lower associated uncertainty than for any other receptor.
	Amphibians/ aquatic plants	No	Although total PCBs was identified as a COC for amphibians and aquatic plants based on the surface water LOE due to exceedance of the TRV by two samples at RM 3.7 [International Slip] and RM 6.9 [Willamette Cove] with HQs of 1.1 and 1.2, respectively; it is reasonable to conclude that total PCBs do not pose an unacceptable risks to amphibians populations or aquatic plant communities because no samples exceeded the alternative water TRV that is based on direct exposure to surface water or the algae-specific water threshold.

Table 11-2. Summary of COCs and Risk Conclusions Across All Ecological Receptors

COC	Receptor Group	Unacceptable Risk?	Justification for Risk Conclusions
Dioxins/Furans			
Total TEQ	Birds	Unknown	Total TEQ (representing dioxins and dioxin-like PCBs) was identified as a COC for spotted sandpiper based on the dietary LOE (in three of four exposure areas: RM 1.9 to RM 3.9, RM 4.0 to 6.0, and RM 7.0 to RM 9.0 with HQs ranging from 1.3 to 20) and for bald eagle and osprey based on the egg LOE (in all exposure areas throughout the Study Area). The egg LOE is highly uncertain due to the fact that TRVs were based on field studies and because of the uncertainties associated with the BMRs used to predict bird egg tissue concentrations and therefore, the bird egg LOE should not be used to determine risk conclusions. It is not possible to determine the likelihood of dietary risk to birds from dioxins using the dietary-dose LOE because of the high uncertainty associated with the TRV, which was based on a weekly injection of 2,3,7,8-TCDD into ring-necked pheasant (injection studies were considered an unacceptable form of exposure in the selection of TRVs) rather than an ecologically relevant exposure. Furthermore, data from Henny et al. (2008) indicate that osprey populations from the LWR (including the Study Area) have increased in recent years, with productivity above that which is necessary for maintaining a stable population.
	Mammals	Yes	Total TEQ (representing dioxins and dioxin-like PCBs) was identified as a COC for both mink and river otter in all exposure areas of the Study Area with HQs ranging from 1.6 to 12 for mink and 1.1 to 2.3 for river otter. The selected LOAEL is the lowest of several studies investigating dioxin effects on mink and corresponds with a magnitude of effect on reproduction that would plausibly result in reduced population viability. Total TEQ HQs indicate low risks to the mink and river otter populations, but total PCB HQs are higher.
Pesticides			
Aldrin	Birds	No	Although aldrin was identified as a COC for spotted sandpiper under the assumptions required for the BERA, it is reasonable to conclude that aldrin does not pose an unacceptable risk to the spotted sandpiper population based on the conservatism in the dietary assumption (exclusive consumption of worms) and the low magnitude and limited spatial extent of exceedance (the only exposure area with an HQ > 1.0 [HQ = 1.7] was between RM 7.0 and RM 9.0).
delta-HCH	Benthic invertebrates	No	Delta-HCH poses negligible risks to the benthic community or populations based on the low magnitude, frequency, number of locations with EFs > 1.0 and two LOEs (tissue residue and surface water) that identify no unacceptable risk associated with delta-HCH. Two bioassays were located within the areas with SQG exceedances; one of these stations was non-toxic, and the other station was toxic based on all four endpoints. PAHs, total DDX, and carbazole exceeded their SQGs at this station. Most reported values based on detection limits.
Dieldrin	Benthic invertebrates	No	Dieldrin poses negligible risks to the benthic community or populations based on the low magnitude, frequency, number of locations with EFs > 1.0 and two LOEs (tissue residue and surface water) that identify no unacceptable risk associated with dieldrin. Only one bioassay station was located within the areas with SQG exceedances; this stations was toxic based on all four endpoints and exceeded the SQGs for numerous chemicals. Most reported values based on detection limits.
DDx	Benthic invertebrates	Yes	Probable benthic risks exist based on the magnitude, frequency, and spatial extent of SQG and tissue TRV exceedances from approximately RM 6.8?? to RM 7.5.
	Birds	No	Although total DDx and sum DDE were identified as COCs for spotted sandpiper under the assumptions required for the BERA, it is more reasonable to conclude that DDx compounds do not pose unacceptable risk to the spotted sandpiper population based on the conservatism in the dietary assumption (that sandpipers exclusively consume worms), limited spatial extent of exceedance (the only LOAEL exceedance was between RM 7.0 and RM 9.0 with a total DDx and sum DDE HQ of 1.4 and 1.3, respectively), and conservatism associated with the LOAEL (the selected total DDx LOAEL based on 6% eggshell thinning may overpredict risks to populations, because reproductive effects on field populations of birds are not documented for eggshell thinning of < 15-20%). 4,4′-DDE was identified as a COC for bald eagle and osprey based on the egg LOE in all exposure areas throughout the Study Area; however, the risk conclusions from the egg LOE are highly uncertain and should not be used to determine risk conclusions because a more reliable dietary-dose LOE is available. The dietary-dose LOE indicates that there is no unacceptable risk to osprey and bald eagles from exposure to DDx compounds.
	Fish	No	Although total DDx was identified as a COC for sculpin based on the tissue LOE and for all fish based on the surface water LOE under the assumptions required for the BERA, it is more reasonable to conclude that it does not pose an unacceptable risk to sculpin or other fish populations because of the low magnitude and spatial extent of sculpin tissue HQs (one sample exceeded the 10th percentile LOAEL at RM 7.3 on the west bank with an HQ of 1.9 and predicted HQs are > 1.0 in areas along RM 7.1 to RM 7.4 on the west bank), uncertainty associated with the water TRV, and because DDx compounds were not identified as a COC for all other fish receptors based on the tissue LOE. The water TRV is highly uncertain because it is based on protection of brown pelican reproduction. HQs calculated based on an alternative water TRV that is protective of direct exposure to surface water are less than 1.0 for all but one sample; however, this sample (at RM 2.0 on the east bank with an HQ of 1.8) is N-qualified. Even if risks from total DDx are elevated for some few sculpin, the limited frequency of TRV exceedances indicates that the risks would not rise to the population-level.
	Amphibians/ aquatic plants	No	Although total DDx was identified as a COC for amphibians and aquatic plants based on exceedance of the surface water LOE, it is reasonable to conclude that total DDx do not pose an unacceptable risks to amphibians populations or aquatic plant communities because the only sample that exceeded the alternative TRV based on direct exposure to surface water (at RM 2.0 on the east bank with an HQ of 1.8) indicated sample interference (i.e., N-qualified) and no samples exceeded the algae-specific water threshold.

Table 11-2. Summary of COCs and Risk Conclusions Across All Ecological Receptors

COC	Receptor Group	Unacceptable Risk?	Justification for Risk Conclusions
Endrin	Benthic invertebrates	No	Endrin poses negligible risks to the benthic community or populations based on low magnitude and frequency of EFs > 1.0 and two LOEs (tissue residue and surface water) that identify no unacceptable risk associated with endrin. Two bioassays were located within the areas with SQG exceedances, and both stations were toxic based on the *Chironomus* growth and mortality endpoints. PAHs exceeded their SQGs at both stations, and carbazole exceeded its SQG at one station.
Endrin ketone	Benthic invertebrates	No	Endrin ketone poses negligible risks to the benthic community or populations based on the low magnitude and frequency of EFs > 1.0 and two LOEs (tissue residue and surface water) that identify no unacceptable risk associated with endrin ketone. Only one bioassay station was located within the areas with SQG exceedances; this station was toxic based on all four endpoints and exceeded the SQGs for numerous chemicals.
VOCs			
Ethylbenzene	Benthic invertebrates	No	Ethylbenzene was identified as a COC based on surface water and a single sample exceeding the TRV with an HQ of 1.6 on the west bank at RM 6.4. Negligible risks to the benthic community or populations.
Trichoroethene	Benthic invertebrates	No	Trichoroethene was identified as a COC based on surface water and a single sample exceeding the TRV with an HQ of 4.1 on the west bank at RM 6.4. Negligible risks to the benthic community or populations.
	Fish	No	Although trichloroethene was identified as a COC for sculpin based on the surface water LOE under the assumptions required for the BERA, it is more reasonable to conclude that trichloroethene does not pose an unacceptable risk to sculpin populations based on the limited spatial extent of TRV exceedances (only one sample exceeded at RM 6.4 on the west bank with an HQ of 4.1).

[a] The individual PAHs and PAH groups that were identified as benthic COCs were benzo(a)anthracene, benzo(a)pyrene, naphthalene, total benzofluoranthenes, total LPAHs and total HPAHs.

[b] Risks are elevated within the Study Area; however, exposure in the Study Area is not greater than exposure upriver of the Study Area so risk is not attributable to an anthropogenic source in the Study Area.

BEHP – bis(2-ethylhexyl) phthalate
BERA – baseline ecological risk assessment
BMR – biomagnification regression
COC – chemical of concern
HQ – hazard quotient
LOAEL – lowest-observed-adverse-effect level
LOE – line of evidence
LWR – Lower Willamette River
NOAEL – no-observed-adverse-effect level
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
RM – river mile
TBT – tributyltin
TEQ – toxic equivalent
TRV – toxicity reference value
VOC – volatile organic compound

11.3.1 Risk to Mink from PCBs

Unacceptable risks to mink from PCBs represent the single highest probability of a chemical affecting an ecological receptor population in the Study Area. As such, a detailed evaluation of the PCB assessment to mink was presented in Section 8.2.3.2, including an uncertainty evaluation and discussion of the exposure and effects assumptions. The following section presents a summary of the mink PCB risk analysis.

Survival, growth, and reproduction in mink and river otter populations were identified as the assessment endpoints to be evaluated in determining risks to mammals that live in or get their food from the Study Area. Although mink and river otter have not been reported in the Study Area, it offers at least marginally suitable habitat and both species have both been collected from nearby locations on the Multnomah Channel and Columbia River (including RM 90, RM 108, and RM 119.5 of the Columbia River near Portland[150]) (Elliott et al. 1999; Henny et al. 1996).

Mink and river otter are semi-aquatic mammals closely associated with water and riparian habitats. Other aquatic or semi-aquatic mammals in the Study Area include beaver, muskrat, raccoon, nutria, and California sea lion. Risks to mink and river otter were assessed because these two species have relatively small home ranges and eat large amounts of fish and crustaceans, causing them to have the highest potential exposure to chemicals from the Study Area. Exposure to site chemicals by the other aquatic-dependent mammals is expected to be lower because, compared to mink and river otter, they are less associated with aquatic habitats (raccoon), use the Study Area as a small fraction of their foraging range (California sea lion), or eat lower on the food chain (beaver, muskrat, raccoon, and nutria) (see EPA's Problem Formulation in Attachment 2).

Furthermore, mink have been shown to be affected by PCBs and dioxins at lower daily doses than most other animals (Attachment 14). The sensitivity of river otter to these chemicals is not well studied. However, because mink generally eat more than river otters relative to their body size, mink get higher dosages of chemical contaminants from the same food, assuming the same diet for both species. Lowering the risk to mink, with its small home range, high-trophic-level diet, high food consumption rate, and sensitivity to PCBs, is also protective of risks to the other aquatic-dependent mammals in the Study Area, including river otter.

Estimates of risk to mink from PCBs were determined in Section 8.1. As a conservative assumption for this BERA, it was assumed that the mink diet consists solely of fish and crayfish from the Study Area. A study of riparian habitats along the Columbia River by the Washington Department of Game (WDG 1980) reported that fish and crayfish comprise only about one-third of the mink diet. The default dietary assumption used in the wildlife risk evaluation, based on data on the relative abundance of fish and crayfish in the Study Area, is that mink eat equal portions of crayfish, largescale sucker, carp, sculpin, and smallmouth bass (i.e., that each component is 20% of the diet). Even under this default dietary exposure

[150] The mouth of the LWR is located at RM 101 of the Columbia River.

assumption, calculated doses for all exposure areas were greater than the selected LOAEL, with HQs ranging from 19 to 33.

Because mink are opportunistic hunters that eat more abundant prey more frequently than less abundant prey (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 fish in the Study Area (this analysis is presented in Attachment 16). For each 1-mile exposure area, minimum and maximum HQs were calculated by selecting the prey portion for each species (in its range of relative occurrences) that resulted in the highest and lowest HQ. For all exposure areas, HQs were always > 1.0, indicating PCB risks to mink regardless of the portions of prey ingested based on the relative occurrence of the potential prey species. The analysis presented in Section 8.2.3.2 indicates that all reasonable mink prey portion exposure scenarios, including that based on relative percent occurrence for the Study Area, result in dietary doses that exceed the selected LOAEL. In addition, it was determined that decreasing the selected incidental sediment ingestion rate (to < 9%) or SUF (assuming that up to two-thirds of the mink diet is comprised of birds or mammals with no Study Area-related PCBs) still results in estimated risks to mink from PCB exposure.

A detailed examination of the effects data in Section 8.2.3.2 also supports the conclusion that unacceptable risks to mink populations in the Study Area might be occurring. Contrary to other receptors and COCs for which toxicological data specific to the receptor are generally lacking, literature-reported adverse effects data specific to mink and PCBs are fairly numerous. Five toxicological papers 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 (Wren et al. 1987; Restum et al. 1998; Tillitt et al. 1996; Heaton et al. 1995; Hornshaw et al. 1983). Calculated average daily PCB doses for mink from the Study Area are higher than all of the reported doses associated with reduction 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.

11.3.2 Other Ecological COC-Receptor Pairs

As previously stated, unacceptable risk to mink from PCBs represents the single highest likelihood that a chemical will impact an ecological receptor population in the Study Area. With the exception of some areas of unacceptable risk to the benthic community (predominantly on the west side of the Study Area between RM 5.0 and RM 7.0), the areas with elevated total PCB concentrations in sediment that indicate unacceptable risks to mink include the areas where elevated sediment concentrations of other COCs cause TRV exceedances for other receptors.

The information summarized in Table 11-2 shows that most COCs identified for fish and wildlife are unlikely to pose unacceptable risks to populations at the levels of exposure that occur in the Study Area, even though the BERA has estimated exposures greater than TRVs. This conclusion was reached because most COCs exceed TRVs by only a small magnitude within a limited portion of the Study Area, and TRVs are largely based on effects to individual organisms rather than to populations. Mink risks from dioxins, furans, and dioxin-

like PCBs combined (expressed as total TEQ) also indicate low risks to the mink, but HQs for total PCBs are higher and cover a larger spatial area. Consequently, management decisions for total PCB risks would be protective of dioxin TEQ risks in areas where elevated sediment concentrations of dioxins and PCBs are co-located.

Based on SPI survey data, the benthic community in the Study Area is typical of a large river system that is strongly influenced by physical processes. Benthic community impacts from sediment contamination appear to be limited to depositional areas that have been impacted by historical releases of contamination. With the exception of a small PAH SQG exceedance area along the eastern shoreline at approximately RM 2.8, the only unique ecorisk area contributed by benthic risk was in the middle segment of the Study Area (RM 5.0 to RM 9.0). Specifically, it was the area along the western side of the river between approximately RM 6.1 and RM 6.6; the three areas in the channel between approximately RM 5.6 and RM 5.8, between RM 5.3 and RM 5.5, and between RM 5.1 and RM 5.2; and scattered locations on the western side of the river from approximately RM 6.9 to RM 7.6.[151] Benthic risk from approximately RM 6.9 to RM 7.6 was associated with DDx compounds. Benthic risk in the aforementioned locations between approximately RM 5.1 and RM 6.6 was associated with PAHs and, to a more limited extent, DDx compounds.

The conclusions from the amphibian and aquatic plant risk assessments, although uncertain, indicate that no unacceptable risks are expected to these receptor populations/communities.

- With the possible exception of the aforementioned locations, the BERA found that DDx compounds pose negligible ecological risks: Total DDx was identified as a COC for fish, plants, and amphibians based on the surface water LOE using the 4,4'-DDT AWQC. However, the AWQC-derived water TRV is not appropriate for evaluating risks to aquatic organisms directly exposed to surface water because it is based on dietary exposure of birds to 4,4'-DDT. An alternative total DDx water TRV protective of direct exposure to surface water was derived. Only one surface water sample exceeded this alternative water TRV (HQ = 1.8); however, this sample (W001 at RM 2.0) was N-qualified and DDx concentrations are low in nearby surface sediments. The indication of sample 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 aquatic organisms from surface water are expected.
- DDx was also identified as a COC for birds. For spotted sandpiper, the total DDx and sum DDE dietary TRVs were exceeded (by a factor of 1.4 and 1.3, respectively) only between RM 7.0 and RM 9.0 of the Study Area and only when it was assumed that spotted sandpiper eat only worms (as represented by laboratory bioaccumulation data). However, based on the conservatism in the dietary assumption that sandpipers exclusively consume worms, the limited spatial extent of exceedance, and the

[151] The area from approximately RM 6.9 to RM 7.6 continuous. Portions of it pose PCB risks to higher tropic level receptors, which is why the areas outside the PCB footprint are characterized as "scattered."

conservatism associated with the total DDx LOAEL (the selected LOAEL based on 6% eggshell thinning likely overpredicts risks to populations, because reproductive effects on field populations of birds are not documented for eggshell thinning of < 15 to 20%), no unacceptable risks to spotted sandpiper populations from DDx compounds are expected.

- 4,4′-DDE was identified as a COC for bald eagle and osprey based on the egg LOE; however, the risk conclusions from the egg LOE are highly uncertain. The egg LOAEL is based on adverse effects observed in field-collected eggs from sites contaminated by multiple chemicals, and the egg NOAEL is based on field-collected egg tissue concentrations associated with no effects. Thus, the effect of any single chemical cannot be reliably determined from these data. In addition, there is only a poor relationship between DDx compound concentrations in eggs and prey data used to estimate exposure, so the relationship between exposure and effects can only be estimated within about an order of magnitude. Because of these uncertainties, the egg LOE is not a reliable LOE and was not used to determine risk conclusions for piscivorous birds. The dietary-dose LOE indicates that there is no unacceptable risk to osprey and bald eagles from exposure to DDx compounds and therefore, no unacceptable risks to these piscivorous birds are expected.
- Total DDx was identified as a COC for sculpin based on the tissue LOE. Tissue concentrations of DDTs in a single sculpin tissue sample from RM 7.3 on the west side of the river exceeded the LOAEL by a factor of 1.9. Total DDx concentrations in the other 37 sculpin samples from other locations in the Study Area and samples of all other fish species were below the LOAEL. The low magnitude and frequency of tissue HQs >1.0 indicate that DDx compounds do not pose risks to the sculpin population.

11.3.3 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 was 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 risks in the erosional areas. Attachment 18 presents a comparison of current and future risk predictions in the erosional areas of the Study Area. The majority of the erosional sediments (approximately 83%) did not change status in predicted risks to the benthic community (i.e., the sediment quality was similar at the erosional depth and the surface). This is not surprising because the erosional sediments are predicted to be primarily sands. Of the remaining erosional sediments, approximately 12% are predicted to be contaminated in the erosional scenario. The last 5% of the erosional area is predicted to be cleaner after the erosional event.

Next Steps

This BERA has characterized baseline ecological risks in the Study Area, but it has not shown "where risk occurs" or fully explained how ecological risks might best be managed. Both of these elements must be understood in order to complete the Remedial Investigation/Feasibility Study (RI/FS), and both will be addressed as part of the FS.

Answering the question of where risk occurs requires an understanding of how COCs get from sediment and other source media to the tissues of the ecological receptors that were evaluated in the BERA, and their prey. Bioaccumulation modeling has been conducted to help answer that question, and the LWG and EPA are already using that modeling to estimate risk-based preliminary remediation goals (PRGs) and identify areas of potential concern (AOPCs). The PRGs and AOPCs identified through this process, and refined based on risk and other considerations, will be used in the FS for evaluating remedial action alternatives.

For the benthic community, bioaccumulation modeling is less significant because benthic organisms are more directly exposed to sediments than are organisms higher on the food chain. The potential benthic risk areas (PBRAs) identified in the BERA are based directly on sediment COC concentrations. The question of where benthic community risks occur hinges more on decisions about what level, spatial extent and certitude of toxicity to benthic organisms constitutes a risk to the benthic community. The draft BERA answers these questions, but the LWG and EPA understand that these answers might need to be collaboratively refined.

The BERA has identified and described a number of factors that create uncertainty about the true magnitude and extent of baseline ecological risks in the Study Area. These include, for example:

uncertainties about whether exceedances of effects thresholds truly pose risks to individual organisms;

uncertainties about how much individual organisms are truly exposed to COCs (e.g., whether they get all their food from the Study Area, and whether and how much they use areas of what would be less suitable habitat within the Study Area even in the absence of COCs);

uncertainties about whether estimated risks to organisms translate into population-level risks.

These questions are not completely answerable within the framework of the RI/FS, and so managers will have to make practical decisions about what and how much should be done to reduce uncertain risks, or to better understand their true magnitude and extent (e.g., through long-term monitoring, and/or through additional analyses of uncertainty about the effectiveness of alternative risk management decisions).

11.4 CONCLUSIONS AND RISK MANAGEMENT RECOMMENDATIONS

This section identifies COCs that might cause unacceptable ecological risk (a plausible chance of adverse effects on an ecological assessment endpoint) in the Study Area, and provides management recommendations based on the relative magnitude of effects and uncertainties associated with the potential unacceptable risks. Unacceptable ecological risks were estimated in the Study Area for the following COCs and assessment endpoints:

- **PCBs** – Total PCBs were found to potentially pose unacceptable risk to the protection and maintenance of populations of mink, river otter, osprey, hooded merganser, and spotted sandpiper; the protection of individual bald eagles; and the protection of the benthic community.
- **Dioxins/furans** –Dioxins, furans, and dioxin-like PCBs were found to potentially pose unacceptable risk to protection and maintenance of mink and river otter populations. Risks to birds from dioxins, furans, and dioxin-like PCBs could not be determined because of the uncertainties associated with the effect thresholds that were used to derive quantitative risk estimates.

- **Mercury** – Mercury was found to potentially pose unacceptable risk to the protection of individual bald eagles.
- **PAHs** – PAHs were found to potentially pose unacceptable risk to the protection and maintenance of the benthic community.
- **DDx compounds** – DDx compounds were found to potentially pose unacceptable risk to the protection and maintenance of the benthic community.

Other COCs were found not to pose unacceptable risk to the ecological assessment endpoints assessed in the BERA.

The results of the BERA will be used to produce risk-based PRGs and AOPCs for the FS. In addition, the BERA may be consulted by risk managers as they deliberate practical risk management objectives during the course of the FS.

The BERA concludes with the following key findings and brief recommendations for risk managers:

Bioaccumulation of PCBs by receptors and their prey poses the most significant potential unacceptable ecological risks of all COCs evaluated. Mink were identified as the ecological receptor at greatest risk and as having the largest spatial extent of potential unacceptable risk. Most unacceptable ecological risks from other COC-receptor pairs are spatially co-located with mink PCB risks. The unacceptable ecological risks for other COC-receptor pairs likely would be reduced or eliminated as a collateral benefit of mink PCB remedies. The extent of the collateral benefit will become apparent as AOPCs are developed for the FS.

Other areas of unacceptable risk were identified for the benthic invertebrate community based on exceedances of PAH and DDx SQGs, primarily in portions of the middle reach of the Study Area (RM 5.0 to RM 8.0).

The spatial evaluation identified approximately 5% of the Study Area as posing unacceptable risk to the benthic community (i.e., Level 3 SQGs or exceeding tissue TRVs).

Surface water and TZW results do not identify any unique ecological risk areas, but they do provide useful information for evaluating potential remedies in the FS.

Exposure to mercury concentrations in fish was found to potentially pose unacceptable risk to individual bald eagles; however, mercury contamination is a greater Willamette River issue requiring watershed-scale risk management.

Anthropogenic background is an important factor, in addition to site-specific releases, contributing to unacceptable ecological risk in the Study Area.

The possible re-exposure of buried contaminated sediment within the Study Area as the result of a high flow event is unlikely to increase unacceptable ecological risk.

These key findings provide the framework for analyzing and comparing alternatives to remediate areas of elevated risk to ecological receptors during the FS.

The draft BERA identified uncertainties associated with the risk conclusions. Following the methods of EPA's Problem Formulation, the identification of COCs was conducted using conservative methods and assumptions. Throughout the risk evaluation process, assumptions were made to provide a margin of conservatism in ecological risk estimates. This was applied even during COC identification, with the consequence that not all COCs pose unacceptable ecological risk. The most important conservative assumption in the draft BERA's COC selection process was the assumption that effects on organisms translate into effects on ecological populations and communities. Populations compensate for individual losses through a variety of ecological processes, so individual-level risks do not necessarily imply population-level risks.

Other examples of conservatism include assumptions about chemical bioavailability and assumptions that reduced effect thresholds (toxicity reference values [TRVs]) to levels that, for example in the case of essential metals, had to be readjusted upward because they were below nutritional thresholds. To account for the conservative assumptions used to identify COCs, additional considerations were used to identify only those COCs that likely pose unacceptable risks to ecological populations and communities. These additional considerations included spatial analysis and further evaluation of the ecological relevance of exposure assumptions, as well as the relevance of selected toxicity thresholds to assessing unacceptable risks to populations or communities.

An additional uncertainty is the extent of co-occurrence of chemicals (present in a form that can exert a toxic effect) and ecological receptors. For most ecological receptors, all exposure areas throughout the Study Area were assumed to provide some type of habitat; however, a lack of habitat in some areas could contribute to an overestimation of exposure. Risk managers should consider these and other uncertainties when applying the results of the predicted risks in this draft BERA to risk management decisions.

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