PORTLAND HARBOR RI/FS

DRAFT FINAL REMEDIAL INVESTIGATION REPORT

APPENDIX F

BASELINE HUMAN HEALTH RISK ASSESSMENT

DRAFT FINAL

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May 2, 2011

Prepared for The Lower Willamette Group

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RECOMMENDED FOR INCLUSION IN ADMINISTRATIVE RECORD

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ACG	analytical concentration goal
ADAF	age-dependent adjustment factor
ALM	Adult Lead Methodology
AOPC	Area of Potential Concern
ATSDR	Agency for Toxic Substances and Disease Registry
AWQC	Ambient Water Quality Criteria
BEHP	Bis 2-ethylhexyl phthalate
BERA	baseline ecological risk assessment
BHHRA	baseline human health risk assessment
Cal EPA	California Environmental Protection Agency
CDC	Centers for Disease Control
CDI	chronic daily intake
CERCLA	Comprehensive Environmental Response, Compensation, and Liability Act
cm	centimeter
cm/hr	centimeters per hour
CNS	central nervous system
COI	contaminant[1] of interest
COPC	contaminant[1] of potential concern
CRITFC	Columbia River Inter-tribal Fish Commission
CSM	conceptual site model
CT	central tendency
DA_{event}	absorbed dose per event
DDD	dichlorodiphenyldichloroethane
DDE	dichlorodiphenyldichloroethylene
DDT	dichlorodiphenyltrichloroethane
delta-HCH	delta-hexachlorocyclohexane
DEQ	Oregon Department of Environmental Quality
DL	detection limit
DQO	data quality objective
E	east
EPA	United States Environmental Protection Agency
EPC	exposure point concentration
EPD	effective predictive domain
FS	feasibility study
g/day	grams per day
GI	gastrointestinal
GSI	Groundwater Solutions, Inc.
HEAST	Health Effects Assessment Summary Table
HHRA	human health risk assessment

[1] Prior deliverables and some of the tables and figures attached to this document may use the term “Chemical of Interest” or “Chemical of Potential Concern”, which as the same meaning as “Contaminant of Interest” or “Contaminant of Potential Concern”, respectively, and refers to “contaminants” as defined in 42 USC 9601(33).

HI	hazard index
HQ	hazard quotient
IEUBK	Integrated Exposure Uptake Biokinetic
IRAF	Infant Risk Adjustment Factor
IRIS	Integrated Risk Information System
ISA	initial study area
K_p	dermal permeability coefficient
l/day	liters per day
LADI	lifetime average daily intake
LOAEL	lowest observed adverse effects level
LWG	Lower Willamette Group
LWR	Lower Willamette River
μg/dl	microgram per deciliter
μg/kg	microgram per kilogram
μg/l	microgram per liter
MCL	Maximum Contaminant Level
MCPP	2-(4-Chloro-2-methylphenoxy)propanoic acid
mg/kg	milligram per kilogram
ml/day	milliliters per day
ml/hr	milliliters per hour
MRL	method reporting limit
NHANES	National Health and Nutrition Evaluation Survey
NLM	National Library of Medicine
OAR	Oregon Administrative Rules
ODFW	Oregon Department of Fish and Wildlife
ODHS	Oregon Department of Human Services
pg/g	picograms per gram
PAH	polycyclic aromatic hydrocarbon
PBDE	polybrominated diphenyl ether
PCB	polychlorinated biphenyl
PEF	potency equivalency factor
PPRTV	Provisional Peer Reviewed Toxicity Value
PRG	preliminary remediation goal
RBC	risk-based concentration
RfD	reference dose
RG	remediation goal
RI/FS	remedial investigation/feasibility study
RM	river mile
RME	reasonable maximum exposure
RSL	Regional Screening Level
SCRA	site characterization and risk assessment
SF	slope factor
STSC	Superfund Health Risk Technical Support Center
SVOC	semivolatile organic compound
TCDD	tetrachlorodibenzo-p-dioxin

TEF	toxic equivalency factor
TEQ	toxic equivalent
TZW	transition zone water
UCL	upper confidence limit
95% UCL/max	95% UCL or maximum
USDA	United States Department of Agriculture
VOC	volatile organic compound
W	west
WHO	World Health Organization
XAD	XAD-2 Infiltrex™ 300 system

GLOSSARY

Term	Definition
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
central tendency	a measure of the middle or expected value of a dataset
contaminant of concern	the subset of contaminants[2] of potential concern with exposure concentrations that exceed EPA target risk levels
contaminant of interest	contaminant[2] detected in the Study Area for all exposure media (i.e., surface water, transition zone water, sediment, and tissue)
contaminant of potential concern	the subset of contaminants[2] of interest with maximum detected concentrations that are greater than screening levels
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 human receptors at a site
congener	a specific chemical within a group of structurally related chemicals (e.g., PCB congeners)
human health risk assessment	a process to evaluate the likelihood that adverse effects to human health might occur or are occurring as a result of exposure to one or more contaminants
dose	the quantity of a 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
empirical data	data quantified in a laboratory
exposure assessment	the part of a risk assessment that characterizes the chemical exposure of a receptor

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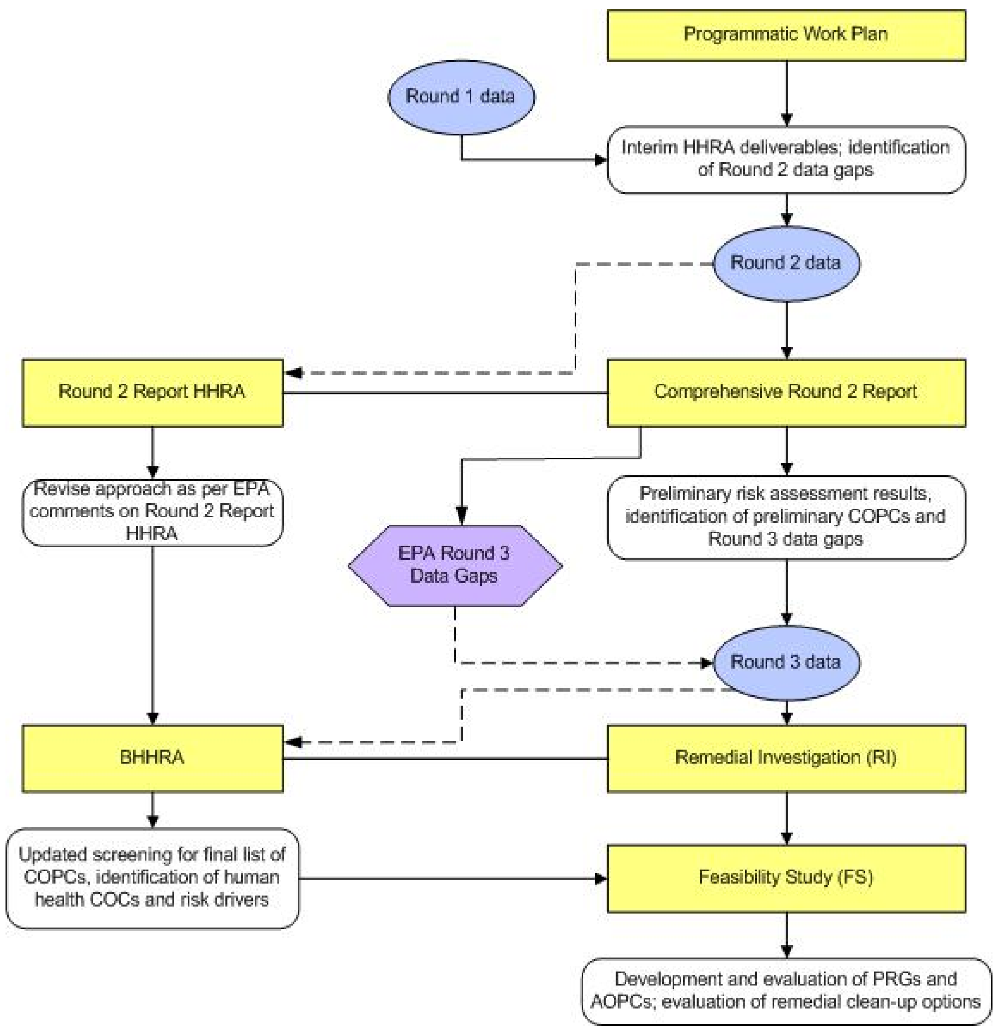
Term	Definition
exposure pathway	physical route by which a contaminant moves from a source to a human receptor
exposure point	the location or circumstances in which a human receptor is assumed to contact a contaminant
exposure point concentration	the value that represents the estimated concentration of a contaminant at the exposure point
exposure area	size of the area through which a receptor might come in contact with a contaminant as determined by human uses
hazard quotient	the quotient of the exposure level of a chemical divided by the toxicity value based on noncarcinogenic effects (i.e., reference dose)
predicted data	data not quantified in a laboratory but estimated using a model
reasonable maximum exposure	the maximum exposure reasonably expected to occur in a population
receptor	The exposed individual relative to the exposure pathway considered
risk	the likelihood that a specific human receptor 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. Specifically for carcinogenic effects, risk is estimated as the incremental probability of an individual developing cancer over a lifetime as a result of exposure to a potential carcinogen. Specifically for noncarcinogenic (systemic) effects, risk is not expressed as a probability but rather is evaluated by comparing an exposure level over a period of time to a reference dose derived for a similar exposure period.
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
slope factor	toxicity value for evaluating the probability of an individual developing cancer from exposure to contaminant levels over a lifetime
Study Area	the portion of the Lower Willamette River that extends from River Mile 1.9 to River Mile 11.8

Term	Definition
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
transition zone water	Pore water associated with the upper layer of the sediment column; may contain both groundwater and surface water
uncertainty	a component of risk resulting from imperfect knowledge of the degree of hazard or of its spatial and temporal distribution
upper confidence limit on the mean	a high-end statistical measure of central tendency
variability	a component of risk resulting from true heterogeneity in exposure variables or responses, such as dose-response differences within a population or differences in contaminant levels in the environment

EXECUTIVE SUMMARY

The baseline human health risk assessment (BHHRA) was conducted as part of the Remedial Investigation Report (RI Report) for the Portland Harbor Superfund Site (Site). The BHHRA is an analysis of potential adverse health effects (current or future) caused by hazardous substance releases from a site in the absence of any actions to control or mitigate these releases. The results of the BHHRA are used to develop remedial action objectives and to assist in risk management decisions for the Site. Figure ES-1 presents an overview of how the development and production of the BHHRA fits in with the overall Remedial Investigation/Feasibility Study (RI/FS) process for the Portland Harbor Superfund Site.

Figure ES-1 Portland Harbor RI/FS Process and BHHRA



The general objective of the BHHRA is to assess the potential risks to human health from exposure to site-related chemicals present in or entering into environmental media (i.e., water or sediment) or bioaccumulating in the food chain, to assist in determining the need for remedial action, to assist in providing a basis for determining concentrations of chemicals that can remain in place and still be protective of public health, and to assist in providing a basis for comparing the effectiveness of various remedial alternatives. Specifically, this included evaluating whether exposure to chemicals in sediment, surface water, groundwater seeps, or biota may result in unacceptable risks to human health.

The BHHRA followed the approach that was documented in the Programmatic Work Plan (Integral et al. 2004) and subsequent interim deliverables. It also reflects numerous discussions, directives, and agreements on risk assessment techniques for the Site with or from the United States Environmental Protection Agency (EPA), Oregon Department of Environmental Quality (DEQ), Oregon Department of Human Services (ODHS), and Native American Tribes. To minimize the chances of underestimating risks, the BHHRA incorporated conservative (i.e. health-protective) assumptions into the identification of exposure scenarios, the estimates of exposure, and the use of toxicity values.

Industrial use of Portland Harbor and adjacent areas of the Lower Willamette River (LWR) has been extensive. Portland Harbor generally refers to a heavily industrialized reach of the LWR between river mile (RM) 0 and RM 11.8, the extent of the navigation channel. The approximate 10-mile portion of Portland Harbor from RM 1.9 to 11.8 is referred to as the Study Area, which is the focus of the BHHRA. Potential human uses of Portland Harbor were considered in identifying the exposure scenarios and exposure media for evaluation in the BHHRA.

ES.1 BHHRA DATASET

The BHHRA dataset includes those data used for direct human health exposure pathways that were quantitatively evaluated in the risk characterization sections of the document: surface sediment (0 to 30.5 centimeter (cm) in depth), surface water, groundwater seep water, clam and crayfish tissue, and fish tissue. Other matrices included in the site characterization and risk assessment (SCRA) dataset (e.g., subsurface sediment) were not evaluated in the BHHRA because they were not relevant to the exposure scenarios evaluated. Although the BHHRA focused on the Study Area, data from outside the Study Area, from downstream to RM 1.0, including Multnomah Channel, and upstream to RM 12.2, were also used to assess risk, per an agreement with EPA. The following summarizes the data used in the BHHRA by medium:

- Beach sediment: Composite beach sediment samples that were collected from designated human use areas within the Study Area were included in the BHHRA dataset.

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- In-water sediment: In-water sediment (i.e., not beach sediment) samples that were collected from the top 30.5 cm in depth between the bank and the navigation channel were included in the BHHRA dataset.
- Surface water: All Round 2 and Round 3 surface water data collected within the Study Area and in Multnomah Channel were included in the BHHRA dataset.
- Groundwater seep: Data from Outfall 22B, which discharges in a potential human use area, were included in the BHHRA dataset. Samples collected from this outfall as part of a stormwater sampling event were excluded from the BHHRA groundwater seep dataset.
- Fish tissue: Composite samples, both whole body and fillet with skin (fillet without skin samples were analyzed for mercury only), of target resident fish species (smallmouth bass, brown bullhead, black crappie, and common carp) were included in the BHHRA dataset. Composite samples of adult Chinook salmon (whole body, fillet with skin, and fillet without skin), adult lamprey (whole body only), and sturgeon (fillet without skin only) were also included in the BHHRA dataset.
- Shellfish tissue: Field-collected composite samples of crayfish and clam tissue (depurated and undepurated) were included in the BHHRA dataset.

ES.2 BHHRA EXPOSURE SCENARIOS

The risk characterization in the BHHRA evaluated the following exposure scenarios, as provided in the approved Programmatic Work Plan and subsequent agreements with or directives from the EPA related to the BHHRA approach:

	Beach Sediment: Ingestion and dermal absorption	**In-water Sediment:** Ingestion and dermal absorption	**Surface Water:** Ingestion and dermal Absorption	**Groundwater Seeps:** Ingestion and dermal absorption	**Fish/ Shellfish:** Ingestion	**Infant Consumption of Human Milk**
Workers	●	●				●
Transients	●		●	●		
Beach Users	●		●			
Fishers	●	●			●	●
Divers		●	●			●
Domestic Users			●			

- Dockside worker — direct exposure to (i.e., ingestion of and dermal contact with) beach sediment, infant ingestion of human breast milk.
- In-water worker — direct exposure to in-water sediment, infant ingestion of human breast milk.
- Transient — direct exposure to beach sediment, surface water (for bathing and drinking water scenarios), and groundwater seeps.
- Adult and child recreational beach user — direct exposure to beach sediment and surface water (for swimming scenarios).
- Tribal fisher — direct exposure to beach sediment or in-water sediment, fish consumption, and infant ingestion of human breast milk.
- Fisher — direct exposure to beach sediment or in-water sediment, fish consumption, shellfish consumption, and infant ingestion of human breast milk.
- Diver — direct exposure to in-water sediment and surface water, infant ingestion of human breast milk.
- Domestic water user – direct exposure to untreated surface water hypothetically used as a drinking water source in the future.

Exposures to beach sediment were assessed per beach, and exposures to groundwater seeps were assessed per seep. Exposures to in-water sediment, surface water, and fish and shellfish tissue were assessed on both localized and Study Area-wide scales. Details of each exposure scenario and associated exposure parameters are provided in Section 3 of this BHHRA.

Of these scenarios, the following were evaluated at the direction of EPA: clam tissue ingestion, fish ingestion for single-species diets, exposure to in-water sediment and surface water by commercial divers, and hypothetical exposure to untreated surface water by a domestic user. Even though surface water in the LWR within Portland Harbor is not currently used as a domestic water source, under OAR 340-041-0340 Table 340A, domestic water supply is a designated beneficial use of the Willamette River, with adequate pretreatment. Divers and clam consumption by fishers were not included in the original Programmatic Work Plan but were included in the BHHRA as directed by EPA. Asian clams (*Corbicula* sp.) are the only clam species that were found in the Study Area during sampling events and, in addition to crayfish, were evaluated for shellfish consumption in the BHHRA. Although harvest and possession of Asian clams is illegal in the State of Oregon, conversations with transients indicated shellfish (both crayfish and clams) are eaten by them (Wagner 2004). In addition, crayfish are commercially harvested in the Willamette River, although the extent of this harvest within the Portland Harbor Superfund Site is not known.

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ES.3 BHHRA EXPOSURE ASSESSMENT

The exposure assessment incorporated the reasonable maximum exposure (RME) approach described by EPA (1989). The RME is intended to be a conservative exposure level that is still within the range of possible exposures. Consistent with EPA (1989), the exposure assessment also used central tendency (CT) values, which represent average exposures, for certain exposure assumptions. For some exposure scenarios, such as fish consumption, exposure assumptions were directed by EPA. Exposure point concentrations (EPCs) were calculated for the 95% upper confidence limit on the arithmetic mean (95% UCL) and the arithmetic mean for each exposure area. In some exposure areas, the maximum concentration was used instead of the 95% UCL. Therefore, the EPCs are referred to as the 95% UCL/max and mean throughout the BHHRA.

EPCs for sediment, surface water, and tissue were calculated for individual exposure areas and on a Study Area-wide basis. The spatial scale of the individual exposure areas and the resulting data included in the calculation of those EPCs were predetermined through discussions with EPA based on assumptions about potential human uses as well as the species' home ranges in the case of tissue EPCs. Exposure areas were designated throughout the Study Area based on the predetermined spatial scales.

Assumptions about each population evaluated in the BHHRA were used to select exposure parameters to calculate the pathway-specific chemical intakes. Site-specific values are not available for all populations and pathways. Therefore, default values were used where site-specific values are not available. Where default values are not available, best professional judgment based on knowledge of human uses of the Study Area or requirements from EPA were used. Uncertainties that are inherent in exposure assessment are attributed to both variability in the population assessed and also the degree of knowledge associated with exposure assumptions. These uncertainties associated with the exposure assessment impact the risk estimates (EPA 1989).

ES.4 BHHRA TOXICITY ASSESSMENT

Toxicity values provide a quantitative estimate of the potential for adverse effects resulting from exposure to a chemical. Cancer and noncancer toxicity values are used in human health risk assessments to quantify the likelihood of adverse effects occurring at different levels of exposure to a chemical. Toxicity values are often based on the results of animal studies, and the extrapolation of toxicological data from animal studies to humans can be one of the largest sources of uncertainty in a risk assessment. Modifying factors, which typically range from two to three orders of magnitude (100 to 1,000 times), are often used by EPA in deriving toxicity values for human health given the level of confidence in the toxicological data, the intra-species differences (i.e., animal to human), and the inter-species differences to account for sensitive human subpopulations.

Some toxicity values are based on exposure to chemical mixtures and not to individual chemicals. This is because these chemicals are commonly present as mixtures in the environment, and the individual components of the mixtures have similar modes of toxicity (such as dioxins). The chemicals that were evaluated in the BHHRA for toxicity as mixtures include: chlordanes, dichlorodiphenyldichloroethane (DDD), dichlorodiphenyldichloroethylene (DDE), and dichlorodiphenyltrichloroethane (DDT); endosulfan; polychlorinated biphenyl (PCBs); and dioxins and furans.

ES.5 BHHRA RISK CHARACTERIZATION

Consistent with DEQ (DEQ 2000a) and EPA guidance (EPA 1989), noncarcinogenic and carcinogenic effects were evaluated separately in the BHHRA. To characterize potential noncarcinogenic effects, comparisons were made between projected intakes of substances and toxicity values. To characterize potential carcinogenic effects, projected intakes and chemical-specific, dose-response data were used to estimate the probability that an individual will develop cancer over a lifetime.

Hazard quotients (HQs) were calculated for noncarcinogenic contaminants of potential concern (COPCs) to estimate the potential for noncarcinogenic effects. The HQs were then summed to yield cumulative hazard indices (HIs) for each exposure area and for the entire Study Area. Estimated HIs were compared to a target HI of 1. For exposure areas exceeding a cumulative HI of 1, endpoint-specific HIs were then calculated and compared to a target HI of 1, below which remedial action at a Superfund site is generally not warranted (EPA 1991a).

Table ES-1 shows the ranges of cancer risks and HIs for each receptor and medium. The exposure pathway with the highest range of HI estimates is consumption of fish tissue. For the most part, exposure scenarios other than fish and shellfish consumption did not exceed a target HI of 1. The ranges of HI estimates are due to the evaluation of different exposure areas, RME and CT scenarios for sediment and water, and multiple ingestion rates and diets for tissue consumption. For example, the range of HI estimates for tissue encompass results for both adult and child consumers, results from three different ingestion rates for each receptor, and results from five different diet compositions.

Potential cancer risks were calculated for carcinogenic COPCs. This calculated risk is expressed as the probability of an individual developing cancer over a lifetime as a result of exposure to the potential carcinogen, and is a health protective estimate of the incremental probability of excess individual lifetime cancer risk. Estimated total cancer risks (summed across all chemicals) were compared to a 1×10^{-4} to 1×10^{-6} risk range, which is the "target range" within which the EPA strives to manage risk as a part of the Superfund program (EPA 1991a). The DEQ target risk levels are 1 x 10^{-6} for individual carcinogens and 1 x 10^{-5} for total cancer risks.

As shown below in Table ES-1, the exposure pathway with the highest range of cancer risk estimates is consumption of fish tissue. For the most part, exposure scenarios other than fish and shellfish consumption were within or below the target risk range of 1 x 10^{-4} to 1 x 10^{-6}. The ranges of cancer risk estimates are due to the evaluation of different exposure areas, RME and CT scenarios for sediment and water, and multiple ingestion rates and diets for tissue consumption. Round 1 fillet tissue samples were not analyzed for PCB, dioxin, or furan congeners. Therefore, the risks from consumption of black crappie and brown bullhead fillet tissue, which were only analyzed in Round 1, likely underestimate the actual risks. However, a range of risks was calculated for fish consumption scenarios, which included samples that were analyzed for congeners, so the lack of analysis of chemicals in certain samples should not impact the overall conclusions of this BHHRA.

Table ES-1. Ranges of Estimated Cumulative Excess Lifetime Cancer Risks and Hazard Indices for Portland Harbor Human Health Scenarios

Exposure Scenario	Receptor	RME Scenarios				CT Scenarios			
		Estimated Cancer Risk		Cumulative Hazard Index		Estimated Cancer Risk		Cumulative Hazard Index	
		Min	Max	Min	Max	Min	Max	Min	Max
Direct Exposure to Beach Sediment	Dockside Worker	5.E-07	**9.E-05**	2.E-03	7.E-02	4.E-08	**6.E-06**	5.E-04	1.E-02
	Transient	1.E-07	6.E-07	4.E-02	1.E-01	8.E-09	4.E-08	6.E-03	1.E-02
	Adult Recreational Beach User	5.E-07	**4.E-06**	8.E-03	3.E-02	2.E-08	2.E-07	2.E-03	6.E-03
	Child Recreational Beach User	**2.E-06**	**4.E-05**	8.E-02	4.E-01	2.E-07	**2.E-06**	1.E-02	5.E-02
	Combined Adult/Child Recreational Beach User	**2.E-06**	**5.E-05**	NA	NA	2.E-07	**2.E-06**	NA	NA
	Tribal Fisher	**2.E-06**	**2.E-05**	2.E-02	8.E-02	1.E-07	**2.E-06**	3.E-03	3.E-02
	Low-Frequency Fisher	4.E-07	**4.E-06**	7.E-03	3.E-02	1.E-08	1.E-07	8.E-04	3.E-02
	High-Frequency Fisher	5.E-07	**6.E-06**	1.E-02	5.E-02	2.E-08	3.E-07	2.E-03	3.E-02
	Breastfeeding Infant	7.E-09	1.E-06	1.E-02	1.E+00	5.E-10	9.E-08	2.E-03	2.E-01
Direct Exposure to Groundwater Seep	Transient	3.E-09	3.E-09	6.E-03	6.E-03	4.E-10	4.E-10	1.E-03	1.E-03
Direct Exposure to In-water Sediment	Diver in Dry Suit	3.E-08	**1.E-05**	2.E-04	2.E-01	NA	NA	NA	NA
	Diver in Wet Suit	9.E-08	**3.E-05**	7.E-04	6.E-01	3.E-09	6.E-07	6.E-05	1.E-02
	In-water Worker	7.E-08	**2.E-05**	1.E-03	1.E+00	5.E-09	4.E-07	2.E-04	6.E-02
	Tribal Fisher	1.E-06	**3.E-04**	3.E-03	**3.E+00**	6.E-08	**6.E-06**	3.E-04	9.E-02
	Low-Frequency Fisher	2.E-07	**6.E-05**	1.E-03	1.E+00	5.E-09	4.E-07	9.E-05	2.E-02
	High-Frequency Fisher	3.E-07	**8.E-05**	2.E-03	**2.E+00**	9.E-09	9.E-07	2.E-04	4.E-02
	Breastfeeding Infant	5.E-10	**3.E-04**	7.E-04	**5.E+00**	4.E-11	**3.E-06**	3.E-04	1.E-01
Direct Exposure to Surface Water	Diver in Dry Suit	1.E-08	**2.E-06**	6.E-05	2.E-03	NA	NA	NA	NA
	Diver in Wet Suit	1.E-08	**1.E-05**	8.E-05	6.E-03	8.E-10	5.E-07	1.E-05	7.E-04
	Transient	6.E-07	7.E-07	4.E-02	4.E-01	7.E-08	1.E-07	1.E-02	8.E-02
	Adult Recreational Beach User	2.E-08	2.E-08	1.E-04	1.E-04	2.E-09	2.E-09	3.E-05	3.E-05
	Child Recreational Beach User	4.E-08	5.E-08	1.E-03	1.E-03	8.E-09	9.E-09	2.E-04	2.E-04
	Combined Adult/Child Recreational Beach User	6.E-08	7.E-08	NA	NA	9.E-09	1.E-08	NA	NA
Surface Water as Hypothetical Drinking Water Source	Domestic User, Adult	**6.E-06**	**3.E-04**	3.E-02	7.E-01	1.E-06	**3.E-05**	2.E-02	3.E-01
	Domestic User, Child	**4.E-06**	**7.E-04**	1.E-01	**2.E+00**	**2.E-06**	**2.E-04**	5.E-02	8.E-01
	Domestic User, Combined Adult/Child	**9.E-06**	**9.E-04**	NA	NA	**3.E-06**	**2.E-04**	NA	NA

Table ES-1 (continued). Ranges of Estimated Cumulative Excess Lifetime Cancer Risks and Hazard Indices for Portland Harbor Human Health Scenarios

		RME Scenarios				CT Scenarios			
		Estimated Cancer Risk		Cumulative Hazard Index		Estimated Cancer Risk		Cumulative Hazard Index	
Exposure Scenario	**Receptor**	**Min**	**Max**	**Min**	**Max**	**Min**	**Max**	**Min**	**Max**
Tribal Fish Ingestion	Tribal Adult Consumer	**2.E-02**	**2.E-02**	**4.E+02**	**4.E+02**	**5.E-03**	**5.E-03**	**9.E+01**	**9.E+01**
Multi-Species Diet	Tribal Child Consumer	**3.E-03**	**3.E-03**	**8.E+02**	**8.E+02**	**8.E-04**	**8.E-04**	**2.E+02**	**2.E+02**
Whole Body Tissue	Combined Tribal Adult/Child Consumer	**2.E-02**	**2.E-02**	NA	NA	**5.E-03**	**5.E-03**	NA	NA
Approximate number of meals per month: 23	Breastfeeding Infant	**2.E-02**	**2.E-02**	**9.E+03**	**9.E+03**	**5.E-03**	**5.E-03**	**2.E+03**	**2.E+03**
Tribal Fish Ingestion	Tribal Adult Consumer	**1.E-02**	**1.E-02**	**3.E+02**	**3.E+02**	**2.E-03**	**2.E-03**	**5.E+01**	**5.E+01**
Multi-Species Diet	Tribal Child Consumer	**2.E-03**	**2.E-03**	**6.E+02**	**6.E+02**	**4.E-04**	**4.E-04**	**1.E+02**	**1.E+02**
Fillet Tissue	Combined Tribal Adult/Child Consumer	**1.E-02**	**1.E-02**	NA	NA	**3.E-03**	**3.E-03**	NA	NA
Approximate number of meals per month: 23	Breastfeeding Infant	**1.E-02**	**1.E-02**	**8.E+03**	**8.E+03**	**2.E-03**	**2.E-03**	**1.E+03**	**1.E+03**
Fish Ingestion	Adult Consumer	**7.E-05**	**6.E-02**	**2.E+00**	**3.E+03**	**7.E-05**	**2.E-02**	**2.E+00**	**1.E+03**
Single-Species Diet	Child Consumer	**3.E-05**	**2.E-02**	**4.E+00**	**5.E+03**	**3.E-05**	**8.E-03**	**4.E+00**	**2.E+03**
Whole Body Tissue	Combined Adult/Child Consumer	**9.E-05**	**7.E-02**	NA	NA	**8.E-05**	**2.E-02**	NA	NA
Approximate number of meals per month: 2 - 19	Breastfeeding Infant	**8.E-05**	**7.E-02**	**3.E+01**	**6.E+04**	**7.E-05**	**2.E-02**	**3.E+01**	**2.E+04**
Fish Ingestion	Adult Consumer	**7.E-06**	**4.E-02**	5.E-01	**2.E+03**	**7.E-06**	**1.E-02**	5.E-01	**7.E+02**
Single-Species Diet	Child Consumer	**3.E-06**	**1.E-02**	1.E+00	**4.E+03**	**3.E-06**	**5.E-03**	9.E-01	**1.E+03**
Fillet Tissue	Combined Adult/Child Consumer	**9.E-06**	**4.E-02**	NA	NA	**8.E-06**	**2.E-02**	NA	NA
Approximate number of meals per month: 2 - 19	Breastfeeding Infant	**6.E-06**	**2.E-02**	**7.E+00**	**5.E+04**	**6.E-06**	**2.E-02**	**7.E+00**	**2.E+03**
Fish Ingestion	Adult Consumer	**1.E-03**	**1.E-02**	**8.E+01**	**6.E+02**	**4.E-04**	**3.E-03**	**2.E+01**	**1.E+02**
Multi-Species Diet	Child Consumer	**6.E-04**	**5.E-03**	**1.E+02**	**1.E+03**	**1.E-04**	**1.E-03**	**3.E+01**	**3.E+02**
Whole Body Tissue	Combined Adult/Child Consumer	**2.E-03**	**1.E-02**	NA	NA	**4.E-04**	**4.E-03**	NA	NA
Approximate number of meals per month: 2 - 19	Breastfeeding Infant	**2.E-03**	**1.E-02**	**2.E+03**	**1.E+04**	**4.E-04**	**4.E-03**	**3.E+02**	**3.E+03**
Fish Ingestion	Adult Consumer	**1.E-03**	**9.E-03**	**6.E+01**	**5.E+02**	**2.E-04**	**1.E-03**	**9.E+00**	**7.E+01**
Multi-Species Diet	Child Consumer	**4.E-04**	**4.E-03**	**1.E+02**	**1.E+03**	**6.E-05**	**6.E-04**	**2.E+01**	**1.E+02**
Fillet Tissue	Combined Adult/Child Consumer	**1.E-03**	**1.E-02**	NA	NA	**2.E-04**	**2.E-03**	NA	NA
Approximate number of meals per month: 2 - 19	Breastfeeding Infant	**1.E-03**	**1.E-02**	**2.E+03**	**1.E+04**	**2.E-04**	**2.E-03**	**2.E+02**	**2.E+03**
Shellfish Ingestion (clam or crayfish)	Adult Consumer	9.E-07	**7.E-04**	7.E-02	**4.E+01**	9.E-07	**7.E-04**	6.E-02	**4.E+01**
Approximate number of meals per month: 0.4 - 2.5	Breastfeeding Infant	1.E-10	**7.E-04**	5.E-04	**8.E+02**	1.E-10	**7.E-04**	4.E-04	**8.E+02**

Notes:

Values presented are for exposure areas assessed in the BHHRA that lie within the Study Area.

Bolded cells exceed the EPA target cancer risk level of 1×10^{-6} or the target hazard index of 1.

Highlighted cells exceed the EPA target cancer risk level of 1×10^{-4} or the target hazard index of 1.

For tissue ingestion, the RME scenario represents the 95 percent upper confidence limit/maximum exposure point concentration. The CT scenario represents the mean exposure point concentration.

The exposure medium shown for the breastfeeding infant represents the exposure medium for the adult.

Ranges for tissue ingestion include all consumption rates.

NA = Not applicable because a CT scenario was not evaluated or because hazard indices were not calculated for the combined adult/child scenario.

Hazard indices presented are the ranges for cumulative hazard indices per exposure area and exposure scenario. Endpoint-specific hazard indices were calculated for cumulative hazard indices greater than 1.

For tissue ingestion, number of meals per month is calculated based on an 8 ounce serving for adults a 3.4 ounce serving for children.

For both cancer risks and noncancer hazards, the maximum estimates are for fish consumption and represent the highest consumption rate, the 95% UCL or maximum tissue concentrations, and localized exposure areas. The following summarizes the assumptions associated with the highest risk estimates:

- **Fish ingestion rate.** The highest ingestion rates used in this BHHRA for adult tribal fishers and adult fishers are 175 g/day (CRITFC 1994) and 142 g/day (EPA 2002b), respectively. These are equivalent to 23 and 19 meals per month, respectively, based on an 8-ounce serving size, every month of the year exclusively of fish caught within the Study Area.
- **Exposure duration.** Fish consumption is assumed to occur at that same rate every month of every year for 30 years for adult fishers and 70 years for tribal fishers.
- **Whole body tissue.** Only whole body tissue (i.e., the entire fish) is consumed.
- **Single species.** For non-tribal fishers, only one species (i.e., common carp) is consumed.
- **Source of fish.** 100 percent of the fish consumed is caught/harvested from the same location.

In addition to the uncertainty associated with the exposure assumptions listed above, there are uncertainties associated with the cooking and preparation methods for fish consumption and background contributions to the Study Area. Possible effects of cooking methods, which can reduce concentrations of lipophilic chemicals in fish tissue, were not considered. PCB concentrations have been shown to be reduced with various cooking methods though due to the variability in the measured rates of reduction there is uncertainty in assigning a rate of reduction of PCBs associated with cooking and preparation methods. Assumptions made during this BHHRA introduce uncertainty to the actual risks that may exist within the Study Area. The contribution of background sources is another important consideration. On a regional scale, fish consumption results in risk estimates exceeding cumulative risks of 10^{-4} or HIs of 1 based on fish tissue data collected from the Willamette and Columbia Rivers outside of the Study Area (EVS 2000, EPA 2002c). However, concentrations are higher at the Site than in the regional tissue.

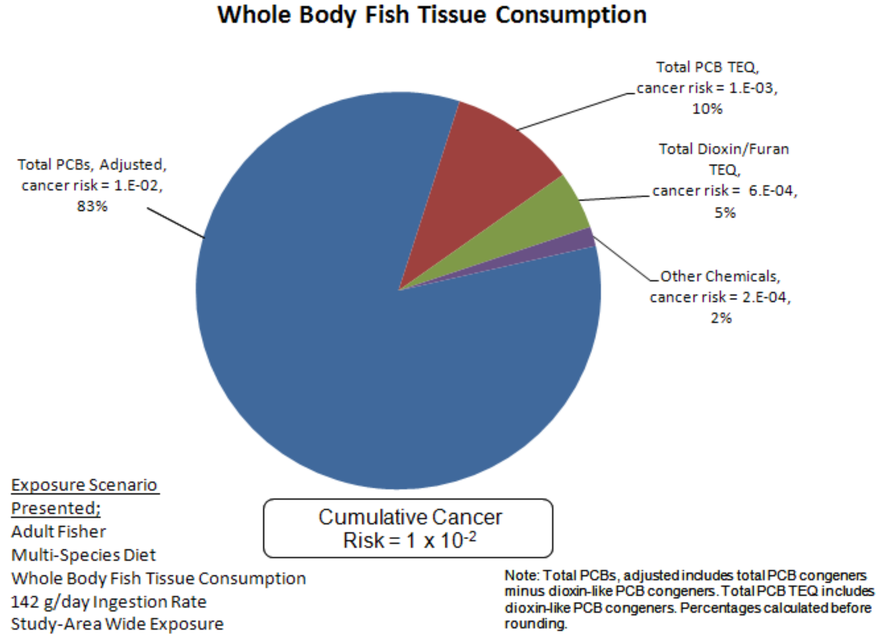
Chemicals were identified as contaminants potentially posing unacceptable risks[3] if they resulted in a cancer risk greater than the EPA point of departure of 1×10^{-6} or a HQ greater than 1 under any of the exposure scenarios for any of the exposure point concentrations evaluated in the BHHRA, regardless of the uncertainties. There were 28 chemicals identified as contaminants potentially posing unacceptable risks for the

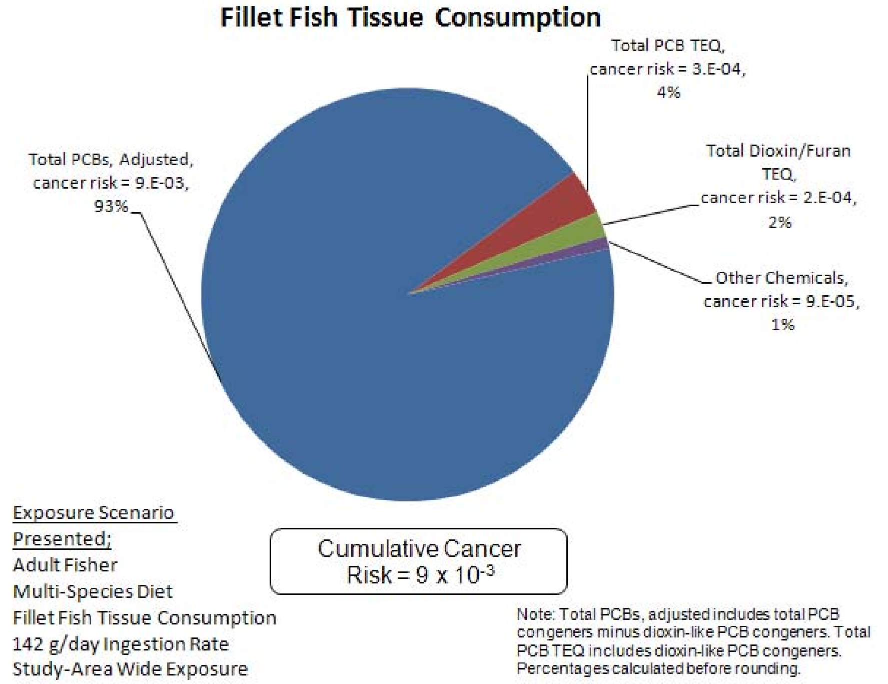
[3] Prior deliverables and some of the tables and figures attached to this document may use the term “Chemicals posing potentially unacceptable risks,” which has the same meaning as “Contaminant posing potentially unacceptable risks” and refers to “contaminants” as defined in 42 USC 9601(33).

exposure scenarios listed above. Only a subset of these contaminants were associated with cancer risks exceeding 1 x 10^{-4} or HQs exceeding 1, and an even smaller number of contaminants contributed to most of the relative percentage of total risk. Of the 33 contaminants identified as potentially posing unacceptable risks, four of the chemicals (alpha-, beta-, and gamma-hexachlorocyclohexane and heptachlor) were identified on the basis of N-qualified data only. The use of an "N" qualifier indicates that the identity of the analyte is not definitive. These four chemicals are not recommended for further evaluation of potential risks to human health. The remaining 29 contaminants identified as potentially posing unacceptable risks to human health are evaluated further in the Human Health Risk Management Recommendations.

As shown in Figure ES-1, PCBs contribute the majority of the total cancer risk for the fish tissue consumption pathway (both whole body and fillet tissue) on a Study Area-wide exposure area basis, and are the primary contributor to risk under this exposure scenario. Dioxins and furans are the secondary contributor to risk. PCBs contribute approximately 93 percent of the cumulative cancer risk, and dioxins/furans contribute approximately 5 percent of the cumulative cancer risk for Study Area-wide whole body fish tissue consumption. For fillet tissue consumption, PCBs contribute approximately 97 percent of the cumulative cancer risk, and dioxins/furans contribute approximately 2 percent for Study Area-wide exposure. The remaining COPCs for Study Area-wide fish consumption account for less than 2 percent of the cumulative cancer risk. PCBs and dioxins/furans also resulted in the highest HQs for Study Area-wide fish tissue consumption.

Figure ES-1. Relative Contribution of Individual Analytes to Cumulative Study Area-Wide Risk For The Non-Tribal Adult Fish Consumption Scenario, Whole Body and Fillet Tissue



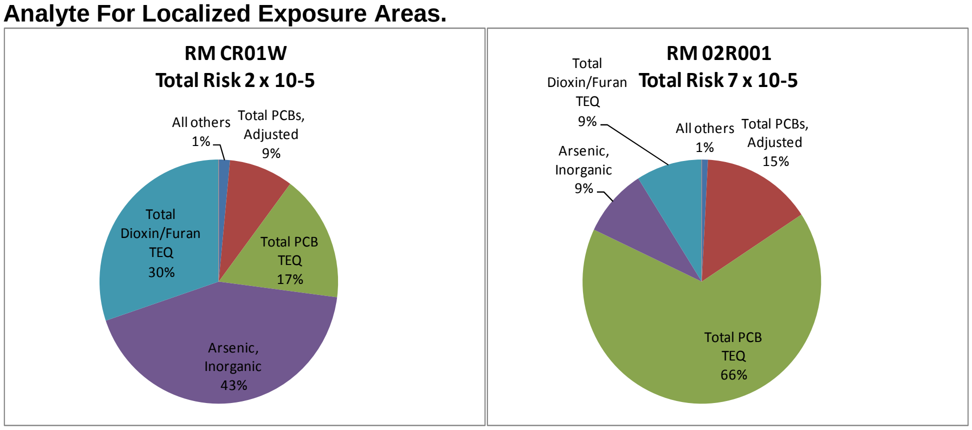


While tissue concentrations and risks are higher in Portland Harbor, in regional studies of fish tissue data from the Willamette and Columbia Rivers outside of the

Study Area (EVS 2000, EPA 2002c) both PCBs and dioxins/furans also resulted in cancer risks greater than 1 x 10^{-4} and/or HQs greater than 1 for fish consumption using exposure assumptions similar to those in the BHHRA.

In some cases in the Portland Harbor, contaminants contributing most to cumulative risks differ between localized exposure areas. For example, Figure ES-2 shows the relative contribution of contaminants to cumulative cancer risks from ingestion of crayfish tissue by an adult fisher at two different localized exposure areas. In the pie chart on the left, which shows relative risks from consumption of crayfish at sampling station CR01W, arsenic is the primary contributor to cancer risk (42% of total risk), followed by total dioxin/furan TEQ (30% of total risk). The pie chart on the right shows relative risks from consumption of crayfish at sampling station RM 02R001, where ingestion of PCBs in shellfish tissue contributes to approximately 81% of total cancer risks (total adjusted PCBs plus total PCB TEQ), followed by an almost equal contribution from arsenic and total dioxin/furan TEQ (approximate 9% contribution to total risks by each contaminant)

Figure ES-2. Example of Differing Relative Contributions to Cumulative Risk by Analyte For Localized Exposure Areas.



Figures show relative risks from adult fisher consumption of crayfish tissue at the 95% UCL/Max Exposure Point Concentrations

A detailed breakdown of risks by exposure scenario, contaminant, and exposure area is provided in the figures and tables in Section 5 of this BHHRA. In addition, Figure ES-3 and ES-4 provide a visual representation of the ranges of cancer risks (ES-3) and noncancer hazards (ES-4) by receptor. Fish tissue consumers have the highest estimated cancer and noncancer risks.

Figure ES-3. Ranges of Cancer Risks by Receptor Across All Exposure Media and Scenarios Evaluated

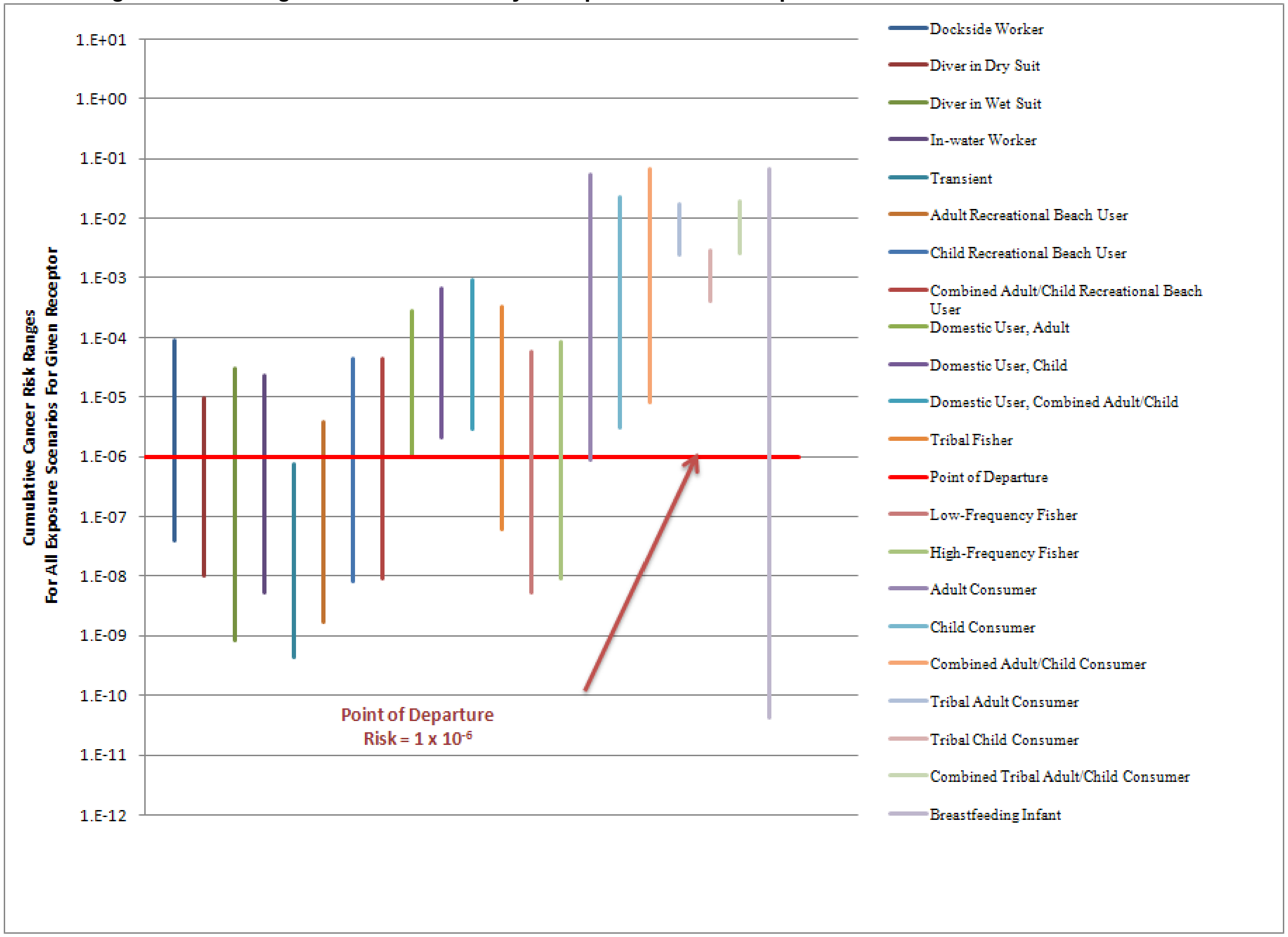
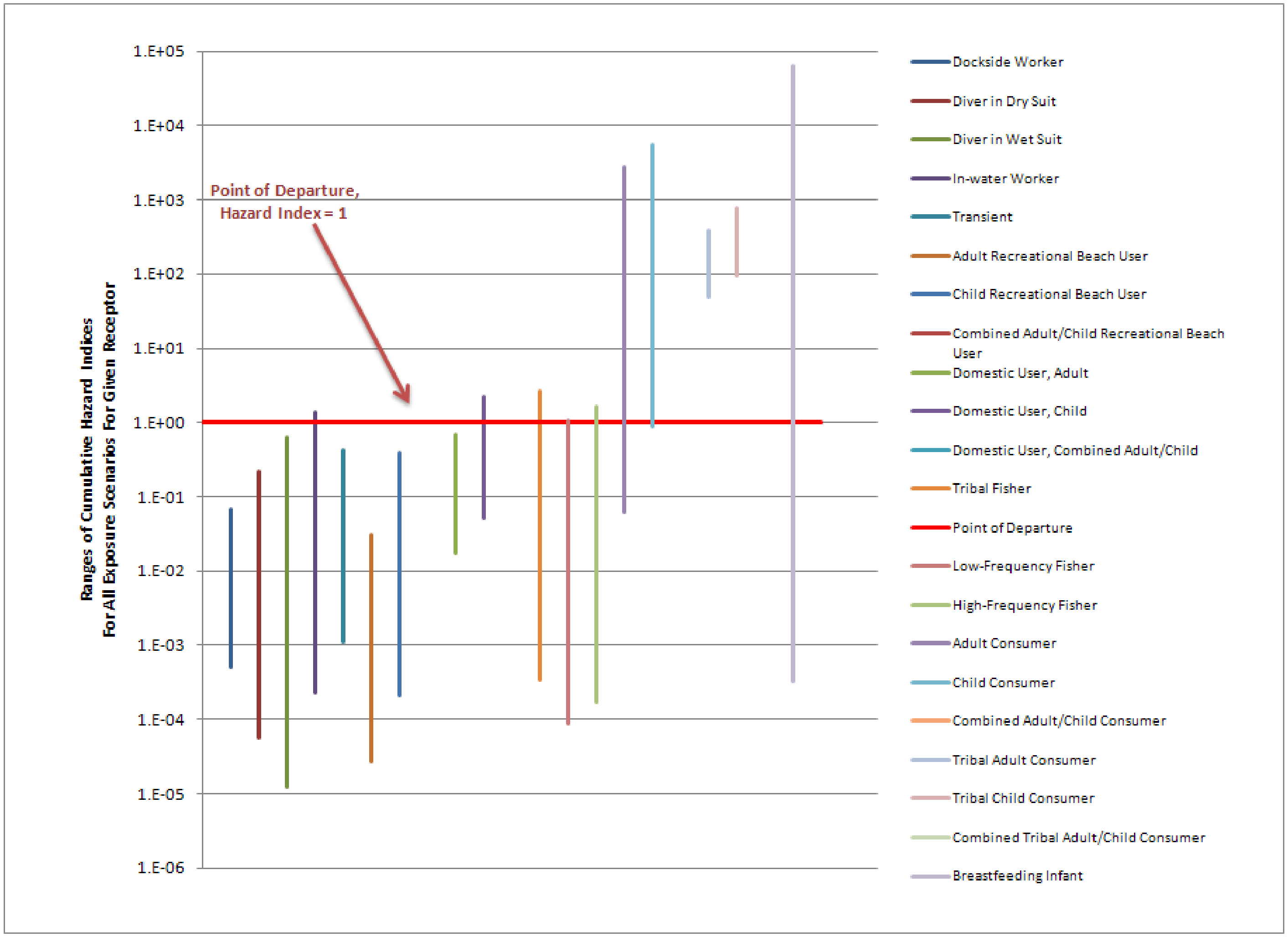


Figure ES-4. Ranges of Cumulative Noncancer Hazard Indices by Receptor Across All Exposure Media and Scenarios Evaluated



ES.6 SUMMARY OF BHHRA

The following presents the major findings of the BHHRA:

- Risks resulting from the consumption of fish or shellfish are generally orders of magnitude higher than risk resulting from direct contact with sediment, surface water, or seeps. Risks from fish and shellfish consumption exceed the EPA point of departure for cancer risk of 1 x 10^{-6}, as well as the target cancer risk range of 1 x 10^{-6} to 1 x 10^{-4} and target HI of 1. With the exception of two ½-mile river segments for the tribal fisher scenario and one location for the hypothetical use of untreated surface water as a drinking water source by a future resident, all of the direct contact scenarios result in risks within or below the EPA target cancer risk range of 1 x 10^{-6} to 1 x 10^{-4}. The direct contact scenarios also result in non-cancer hazards below the target HI of 1, with the exception of one ½-river mile segment for in-water sediment and one location for hypothetical use of untreated surface water as a drinking water source.

- Fish consumption results in the highest risks of the scenarios evaluated in the BHHRA. PCBs are the primary contributor to risk for fish consumption, and dioxins/furans are a secondary contributor for fish consumption for exposure occurring over the full length of the Study Area. Other contaminants potentially posing unacceptable risks at a Study Area-wide or localized scale for at least one fish consumption exposure scenario include the following contaminants:
  - antimony
  - arsenic
  - lead
  - mercury
  - selenium
  - zinc
  - benzo(a)anthracene
  - benzo(a)pyrene
  - dibenzo(a,h)anthracene
  - total carcinogenic PAHs
  - bis(2-ethylhexy) phthalate
  - hexachlorobenzene
  - total PCBs and PCB TEQ
  - total dioxin TEQ
  - aldrin
  - dieldrin
  - heptachlor epoxide
  - total chlordane

DO NOT QUOTE OR CITE
This document is currently under review by US EPA and its federal, state, and tribal partners, and is subject to change in whole or in part.

  - total DDD
  - total DDE
  - total DDT
  - PBDEs

- Risks from PCBs based on consumption of fish within the Study Area exceed the EPA target risk range of 1 x 10^{-6} to 1 x 10^{-4}, with a maximum estimated risk of 7 x 10^{-2} (combined adult and child receptor). The maximum cumulative hazard index from fish consumption is 5,000 (child receptor), primarily from exposure to PCBs in whole body tissue. The maximum cumulative hazard index from consumption of fillet fish tissue is 4,000 (child receptor), also primarily from exposure to PCBs.

The body of information available regarding fish consumption rates, both nationally and regionally, indicates that the fish ingestion rates used in the BHHRA address a range of exposures that might occur for consumers of locally caught fish in Portland Harbor, including high fish consuming populations.
Concentrations of bioaccumulative chemicals are higher at the Site than in regional tissue. However, on a regional basis, risks from exposure to bioaccumulative chemicals in tissue exceed EPA target risk levels. For example, the PCB concentrations detected in resident fish from the Willamette and Columbia Rivers are approximately 20 to 100 times higher than the EPA target fish tissue concentration, when adjusted for the ingestion rates used in this BHHRA and based on a target risk level of 1 x 10^{-6}. Regional efforts are underway to reduce fish tissue concentrations. Sources contributing to regional tissue concentrations are unknown. The contribution of background sources of contaminants potentially posing unacceptable risks is an important consideration in risk management decisions. For example, arsenic concentrations in beach sediment contribute approximately 50% of cumulative risk from exposure to this medium for the highest-risk scenarios, yet arsenic concentrations detected in beach sediment within the Study Area are comparable to Oregon DEQ-established background levels.

1.0 INTRODUCTION

This Baseline Human Health Risk Assessment (BHHRA) presents the Lower Willamette Group's (LWG's) evaluation of risks to human health for the Portland Harbor Superfund Site (Site) in Portland, Oregon. This BHHRA is intended to provide an assessment of human health risks for the Site and to support risk management decisions for the Site.

Portland Harbor encompasses the authorized navigation channel in the Lower Willamette River (LWR) in Portland, Oregon, from the confluence with the Columbia to about River Mile (RM) 11.8. Portland Harbor has been the focus of numerous environmental investigations completed by the LWG and various other governmental and private entities. Major LWG data collection efforts occurred during three sampling rounds in the Remedial Investigation/Feasibility Study (RI/FS) Study Area (RM 1.9 to 11.8) to characterize the physical system of the river and to assess the nature and extent of contamination in sediment, surface water, transition zone water, stormwater, and biota. This BHHRA incorporates the results of these environmental investigations and builds from the initial Human Health Risk Assessment (HHRA) performed as part of the Portland Harbor RI/FS Comprehensive Round 2 Site Characterization Summary and Data Gaps Analysis Report (Round 2 Report) (Integral et. al. 2007).

The LWG has worked with the United States Environmental Protection Agency (EPA) to develop the methods and assumptions used in this BHHRA. At the direction of EPA, this BHHRA incorporates assumptions to provide a health protective assessment of risks associated with contaminants present at the Site, which is consistent with EPA guidance on risk assessment (1989). For many of the exposure scenarios evaluated in this BHHRA, upper-bound literature values are used to quantify exposure due to the lack of site-specific exposure information. In some cases, the maximum detected concentrations are used to quantify long-term exposures, which may not be representative of ongoing exposures in the Study Area. Therefore, the results of the BHHRA have a margin of conservatism built into the risk conclusions consistent with EPA guidance (1989).

This BHHRA is being conducted as part of the Remedial Investigation Report (RI Report) to evaluate potential adverse health effects caused by hazardous substance releases at the Site, consistent with the requirements of the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA). The BHHRA will be used to support the development of contaminant thresholds to be used as preliminary remediation goals (PRGs) for sediment. The BHHRA PRGs are provided along with PRGs developed under the baseline ecological risk assessment (BERA) for the Site. The PRGs will provide preliminary estimates of the long-terms goals to be achieved by any cleanup actions in Portland Harbor. During the feasibility study (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.

1.1 OBJECTIVES

The general objective of a HHRA is to assess the potential risks to human health from exposure to chemicals present in or entering into environmental media (i.e., water or sediment) or bioaccumulating in the food chain. The overall objective of this BHHRA for the Site is to evaluate whether exposure to contaminants in sediment, surface water, groundwater seeps, or biota may result in unacceptable risks to human health. To achieve the overall objective, the following are specific objectives of this BHHRA:

- Identify contaminants of potential concern (COPCs)[4] for human health
- Identify potential exposure pathways to populations who may contact COPCs
- Characterize potentially exposed populations and estimate the extent of their exposure to COPCs
- Quantitatively characterize the noncarcinogenic and carcinogenic risks to the populations resulting from potential exposure to COPCs and identify contaminants potentially posing unacceptable risks
- Characterize uncertainties associated with this risk assessment
- Identify the contaminants and pathways that contribute the majority of the risk.

1.2 APPROACH

This BHHRA follows the approach that was documented in the Programmatic Work Plan (Integral et al. 2004) and subsequent interim deliverables. It also reflects numerous discussions and agreements on appropriate risk assessment techniques for the Site among interested parties, including the EPA, Oregon Department of Environmental Quality (DEQ), Oregon Department of Human Services (ODHS), and Native American Tribes.

Most of the exposure scenarios, including potential exposure pathways and potentially exposed populations, were originally identified in the Programmatic Work Plan. Most of the assumptions used to estimate the extent of exposure for these scenarios were also identified in the Programmatic Work Plan. Additional assumptions for estimating the extent of exposure were provided in the Exposure Point Concentration Calculation Approach and Summary of Exposure Factors Technical Memorandum (Kennedy/Jenks Consultants 2006) and the Human Health

[4] Prior deliverables and some of the tables and figures attached to this document may use the term “Chemicals of potential concern,” which has the same meaning as “Contaminants of potential concern” and refers to “contaminants” as defined in 42 USC 9601(33).

Toxicity Values Interim Deliverable (Kennedy/Jenks Consultants 2004a). Exposure scenarios that were not included in the Programmatic Work Plan were evaluated in this BHHRA based on direction from EPA. Specific agreements with and direction from EPA related to the approach for this BHHRA are documented in Attachment F1.

The approach of this BHHRA is based on EPA (1989, 1991b, 2001a, 2004, 2005a) and EPA Region 10 (2000a) guidance and direction from EPA. The approach is also consistent with DEQ guidance for HHRAs (DEQ 2000a, 2010).

1.3 SITE BACKGROUND

The LWR extends from the Willamette's convergence with the Columbia River at river mile (RM) 0 upstream to the Willamette Falls at RM 26. Portland Harbor generally refers to a heavily industrialized reach of the LWR between RM 0 and RM 11.8, the extent of the navigation channel. Additional information on the environmental setting of Portland Harbor, including historical and current land use, regional geology and hydrogeology, surface water hydrology, the in-water physical system, habitat, and human access and use is provided in Section 3 of the RI Report. The approximate 10-mile portion of Portland Harbor from RM 1.9 to 11.8 is referred to as the Study Area (Map 1-1). Because the Site boundaries have not yet been defined[5], this BHHRA focused on the Study Area.

Portland Harbor and the Willamette River have served as a major industrial water corridor for more than a century. Industrial use of the Study Area and adjacent areas has been extensive. The majority of the Study Area is currently zoned for industrial land use and is designated as an "Industrial Sanctuary" (City of Portland 2006a). Much of the shoreline in the Study Area includes steeply sloped banks covered with riprap or constructed bulkheads, with human-made structures such as piers and wharves over the water in various locations. A comprehensive update of Portland's Willamette Greenway Plan and related land use policies and zoning (The River Plan) is underway, addressing all of the Willamette riverfront in Portland (City of Portland 2006b). The Willamette Greenway Plan addresses the quality of the natural and human environment along the Willamette River and generally includes all land adjacent to the river, public lands near the river, and land necessary for conservation of significant riparian habitat. (The Willamette Greenway Plan, adopted by City Council November 5, 1987, Ordinance 160237.) The Greenway Plan is intended to "protect, conserve, enhance, and maintain the natural, scenic, historical, economic, and recreational qualities of lands along Portland's rivers." (Portland City Code Chapter 33.440). The Plan supports industrial uses within Portland Harbor while at the same time looks to increase public access to the river. As a result, recreational use within the Study Area may increase at certain locations in the future.

[5] The Site boundaries will be defined by EPA in the Record of Decision for the Site.

There are numerous potential human uses of Portland Harbor. Worker activities occur at the industrial and commercial facilities in the Study Area. However, due to the sparse beach areas and high docks associated with most of the facilities, worker exposure to the in-water portion of the Study Area may be limited in shoreline areas. Commercial diving activities also occur in the LWR.

In addition, the LWR provides many natural areas and recreational opportunities, both within the river itself and along the riverbanks. Within the Study Area, Cathedral Park, located under the St. Johns Bridge, includes a sandy beach area and a public boat ramp and is used for water skiing, occasional swimming, and waterfront recreation. Recreational beach use also may occur within Willamette Cove, which is a riverfront natural area, in Swan Island Lagoon, and on the southern end of Sauvie Island, which is within the Study Area. Swan Island Lagoon includes a public boat ramp. Additional LWR recreational beach areas exist on the northern end of Sauvie Island and in Kelley Point Park, both of which are outside of the Study Area.

Fishing is conducted throughout the LWR basin and within the Study Area, both by boaters and from locations along the banks. The LWR also provides a ceremonial and subsistence fishery for Pacific lamprey (particularly at Willamette Falls) and spring Chinook salmon for Native American Tribes. Many areas in the LWR are also important currently for cultural and spiritual uses by local Native Americans.

Transients have been observed along the LWR, including some locations within the Study Area. The observation of tents and makeshift dwellings during RI sampling events confirms that transients were living along some riverbank areas. Transients are expected to continue to utilize this area in the future.

The RI/FS being completed for the Site is designed to be an iterative process that addresses the relationships among the factors that may affect chemical distribution, risk estimates, and remedy selection. Three rounds of field investigations have been completed as part of the RI/FS. Round 1 was conducted in 2002 and focused primarily on chemical concentrations in fish and shellfish tissue and in beach sediment. Round 2 was conducted in 2004 and 2005 and focused on chemical concentrations in sediment cores, in-water surface sediment, surface water, transition zone water, and additional shellfish tissue and beach sediment. Round 3 was conducted in 2006 and 2007 and focused on chemical concentrations in additional surface water, sediment, and fish and shellfish tissue. These Round 1, Round 2, and Round 3 sampling efforts, while initially focused on RM 3.5 to 9.2, which is the Administrative Order on Consent-defined initial study area (ISA), extended well beyond the ISA to RM 0 downstream and to RM 19 upstream.

1.4 ORGANIZATION

In accordance with guidance from EPA (1989), which is consistent with DEQ guidance (2000a, 2010), the BHHRA incorporates the four steps of the baseline risk assessment process: data collection and evaluation, exposure assessment, toxicity assessment, and risk characterization (which includes an uncertainty assessment).

This BHHRA is organized as follows:

- Section 2, Data Evaluation – This section evaluates the available data for the Study Area and identifies the COPCs for further evaluation in the BHHRA.
- Section 3, Exposure Assessment – This section presents potentially complete routes of exposure and potential receptor populations for further evaluation in the BHHRA, which are summarized in the conceptual site model (CSM).
- Section 4, Toxicity Assessment – This section evaluates the potential hazard and toxicity of the COPCs selected for quantitative evaluation in this BHHRA.
- Section 5, Risk Characterization – This section presents the cancer risks and noncancer hazards and identifies the contaminants potentially posing unacceptable risks to human health.
- Section 6, Uncertainty Analysis – This section discusses the uncertainties that are inherent in performing a HHRA, and the uncertainties specific to this BHHRA.
- Section 7, Summary – This section summarizes the findings of this BHHRA and identifies chemicals and pathways that contribute the majority of the risk within the Study Area.
- Section 8, Conclusions – This section provides the conclusions for this BHHRA.
- Section 9, References – This section lists the references used in this BHHRA.

2.0 DATA EVALUATION

Data collection and evaluation included the gathering and analysis of data relevant to human exposures and the identification of those contaminants that are the focus of this BHHRA. Data needs for the BHHRA were identified through the data quality objective (DQO) process described in Section 7 of the Programmatic Work Plan (Integral et al. 2004).

This section presents the data that were used in this BHHRA and the results of the selection of COPCs in sediment, water, and tissue. The LWG sampling events and non-LWG sampling events included in the site characterization and risk assessment (SCRA) dataset are described in detail in Section 2.0 of the RI Report. The BHHRA dataset used in this risk analysis and described in this section is a subset of data from the sampling events that comprised the SCRA dataset as of September 2008. Additional information on the BHHRA dataset and details on the use of the data in the BHHRA are provided in Attachment F2. In addition, a risk evaluation of potential exposures to polybrominated diphenyl ethers (PBDEs) in in-water sediment, fish tissue, and shellfish tissue was performed at the direction of EPA using a subset of data from the sampling events that comprised the SCRA dataset as of February 2011. The data for the PBDE analysis are discussed in Attachment F3, and the PBDE risk assessment used the general data evaluation methodology discussed in this section.

2.1 AVAILABLE DATA

The risk characterization BHHRA dataset includes only those matrices relevant for direct human health exposure pathways that were quantitatively evaluated: surface sediment (0 to 30.5 centimeter (cm) in depth), clam and crayfish tissue, fish tissue, surface water and groundwater seeps. Other matrices included in the SCRA dataset (e.g., subsurface sediment) were not evaluated in the BHHRA because they were not relevant to the exposure scenarios evaluated (see Section 3). Although the BHHRA focused on the Study Area, data from outside the Study Area, from downstream to RM 1.0, including Multnomah Channel, and upstream to RM 12.2, were also used to assess risk, per an agreement with EPA. The BHHRA dataset is divided into samples within the Study Area and outside of the Study Area, and summarized by matrix in Tables 2-1 and 2-2. The dataset is described briefly in the following subsections, and described in more detail in Section 2.0 of the RI Report.

2.1.1 Beach Sediment

Areas where potential exposure to beach sediment could occur were identified and designated as human use areas in the Programmatic Work Plan. Human use areas were designated based on current conditions. Beaches are relatively dynamic environments; if beach conditions change in the future, additional risk evaluation of the human use areas may be required. Composite sediment samples were collected during Round 1 from each beach that had been designated as a potential human use area within the ISA. Additional human use areas within the Study Area but downstream of the ISA were sampled during Round 2 as part of the sampling of shorebird habitat. All of the Round 1 beach samples and the six Round 2 beach samples that were collected from potential human use areas located downstream of the ISA were included in the BHHRA dataset. The designated potential human use areas and associated beach sediment samples are shown in Map 2-1. Table 2-3 presents a summary of the beach composite sediment samples included in the BHHRA dataset.

2.1.2 In-Water Sediment

In-water surface sediment chemistry data in the BHHRA dataset include LWG collected data (from Rounds 1, 2, and 3) and non-LWG collected data. Tables 2-3 and 2-4 present a summary of the surface sediment samples both within the Study Area and outside of the Study Area that are included in the BHHRA dataset. All non-LWG data included in the BHHRA dataset (see Section 2.0 of the RI Report) met the data quality requirements for risk evaluation (Category 1/QA2), as agreed to between LWG, EPA, and EPA's partners in the Programmatic Work Plan (Integral et al. 2004).

All in-water surface sediment data included in the BHHRA dataset were collected from the top 30.5 cm in depth, outside of the navigation channel of the river. Samples from within the Study Area were located throughout its entire length (RM 1.9 to RM 11.8), and samples outside of the Study Area extended downstream to RM 1.0, including Multnomah Channel, and upstream to RM 12.2. Surface sediment samples that were collected from areas that have been characterized in the SCRA as capped or dredged were not included in the BHHRA dataset because these samples are no longer representative of the current conditions in the Study Area. A more detailed description of the in-water sediment dataset used in this BHHRA is provided in Attachment F2; a description of samples that have been characterized as capped or dredged in the SCRA is provided in Appendix A of the RI Report.

2.1.3 Surface Water

Surface water data were collected by the LWG during Rounds 2 and 3, as described in Appendix A of the RI Report. All Round 2 and Round 3 surface water data between RM 1.9 and 11.8, as well as samples collected from Multnomah Channel, were included in the BHHRA dataset. The use of the surface water dataset in evaluating different human exposure scenarios is discussed in subsequent sections and in Attachment F2. Surface water sampling was performed in seven separate events between 2004 and 2007 to capture the seasonal water flow conditions on the LWR. Tables 2-5 and 2-6 present a

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summary of the surface water samples included in the BHHRA dataset from within and outside of the Study Area.

Amongst all seven sampling events, 37 surface water locations were sampled between RM 1.9 and RM 11.8, and were included in the BHHRA dataset. Surface water samples in the BHHRA dataset were collected from 32 single point stations and 5 transect locations (at RM 2.0, Multnomah Channel, RM 3.9, RM 6.3, and RM 11). Surface water samples were collected with either a peristaltic pump or an XAD-2 Infiltrex™ 300 system (XAD). Single point samples included near-bottom and near-surface samples, as well as vertically integrated water column samples. Transect samples included horizontally integrated near-bottom and near-surface samples, cross-sectional equal discharge increment samples (i.e., samples horizontally integrated across the entire width of the river into a single sample for either near-surface or near-bottom horizontally integrated samples), and vertically integrated samples from the east, west, and middle sections of a transect on the river. Additional information on the surface water sampling methods is available in Section 5.3 of the RI Report.

2.1.4 Groundwater Seep

A seep reconnaissance survey was conducted during Round 1 to document readily identifiable groundwater seeps along approximately 17 miles of riverbank from RM 2 to 10.5 (GSI 2003). Twelve potential groundwater seeps were observed at or near a potential human use beach area. Of these, only three sites were identified where it was likely for upland contaminants of interest (COIs)[6] to reach groundwater seeps or other surface expressions of groundwater discharging to human use beaches (GSI 2003): City of Portland storm sewer Outfall 22B, Willbridge, and McCormick and Baxter (at Willamette Cove).

Of the three potential groundwater seep areas, only the Outfall 22B discharge was evaluated in this BHHRA. At this location, groundwater infiltrates into the outfall pipe, which subsequently discharges to a beach. The beach where Outfall 22B discharges was identified as a potential transient use area, so exposure to the groundwater seep in that beach by transients is considered a potentially complete pathway. The groundwater seep identified at Willbridge is in a beach restricted to industrial use, and exposure to groundwater seeps is considered an incomplete pathway for workers. The groundwater seep identified during the seep survey (GSI 2003) in Willamette Cove, downgradient of the McCormick and Baxter Superfund Site, was capped during remedial activities in 2004.

The stormwater pipeline that discharges at Outfall 22B provides a conduit for surface discharge of groundwater containing COIs that infiltrates into the pipe upland of the

[6] Prior deliverables and some of the tables and figures attached to this document may use the term "Chemicals of interest," which has the same meaning as "Contaminants of interest" and refers to "contaminants" as defined in 42 USC 9601(33).

beach. Samples of the discharge at Outfall 22B have periodically been collected for analysis, both during stormwater events and outside of stormwater events. In order to represent potential exposure from the groundwater seep, samples taken during stormwater events were not included in the BHHRA dataset. The data from Outfall 22B met the data quality requirements for risk evaluation (Category 1/QA2), and the results of this sampling were included in the SCRA database. Samples taken since 2002 were used in the BHHRA. Table 2-5 presents a summary of the samples from Outfall 22B that were included in the BHHRA dataset. The BHHRA Outfall 22B dataset is further described in Attachment F2. The sampling events for this data are described in Appendix A of the RI Report.

2.1.5 Fish Tissue

Target fish species for human consumption were identified in the Programmatic Work Plan (Integral et al. 2004). Resident fish samples were collected during Rounds 1 and 3 by the LWG. In addition, adult white sturgeon (*Acipenser transmontanus*), adult spring Chinook salmon (*Oncorhynchus tshawytscha*), and adult Pacific lamprey (*Lampetra tridentate*) were collected in the summer of 2003 through a cooperative effort of the ODHS, Agency for Toxic Substances and Disease Registry (ATSDR), Oregon Department of Fish and Wildlife (ODFW), the City of Portland and EPA Region 10. (This sampling effort is referred to as the "ODHS Study" in the rest of this BHHRA). Table 2-7 presents a summary of the fish tissue samples included in the BHHRA dataset.

2.1.5.1 Resident Fish Tissue

Smallmouth bass (*Micropterus dolomieui*), black crappie (*Pomoxis nigromaculatus)*, common carp (*Cyprinus carpio carpio)*, and brown bullhead (*Ameiurus nebulosus*) were the resident fish species collected and analyzed to support the BHHRA. The sampling design was based on the reported home ranges of the target fish, so the sampling approach differed based on species. For Round 1 data collection, the tissue compositing scheme for each sample was reviewed and approved by EPA in November and December 2002 prior to laboratory analysis. For Round 3 data collection, the tissue compositing scheme for each sample was reviewed and approved by EPA in October 2007 prior to laboratory analysis.

During Round 1, smallmouth bass samples were collected from eight locations between RM 2 and 9, each corresponding to approximately one river mile. Smallmouth bass were collected and composited based on river mile locations due to their small home range relative to the other fish collected during Round 1. Three whole body replicate composite samples were collected at three of the eight river mile locations. At each of the remaining five river mile locations, one whole body composite sample and one fillet composite sample were collected. All Round 1 results from within the Study Area were included in the BHHRA dataset.

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During Round 3, smallmouth bass were collected from 18 stations between RM 2 and 12, each corresponding to approximately one river mile, and either the west or east portion of the river. One composite sample was collected from each station, for which fillet tissue and remainder tissue (body without fillet) were analyzed separately. All Round 3 results were included in the BHHRA dataset.

During Round 1, black crappie, common carp, and brown bullhead samples were collected and composited for two fishing zones, each approximately three river miles in length (RM 3-6 and RM 6-9). Three whole body and three fillet replicate composite samples were collected at each of the two fishing zones for common carp and brown bullhead. Two whole body and two fillet replicate composite samples were collected within each of the fishing zones for black crappie. All Round 1 results from within the Study Area were included in the BHHRA dataset.

During Round 3, common carp samples were collected for three fishing zones, each approximately four river miles in length (RM 0-4, RM 4-8, and RM 8-12). Three common carp composite samples were collected from each fishing zone and analyzed separately as fillet tissue and remainder tissue. All Round 3 results were included in the BHHRA dataset.

For smallmouth bass, black crappie, and common carp, all fillet samples were analyzed as fillet with skin, except for the analysis of mercury, which was performed using fillet without skin. For brown bullhead, all fillet samples were analyzed as fillet without skin.

2.1.5.2 Salmon, Lamprey, and Sturgeon

The tissue data collected during the ODHS Study were the only non-LWG fish tissue data of acceptable data quality for risk evaluation (Category 1/QA2). Although these data were not collected as part of the RI, they were evaluated by the LWG and used in this BHHRA.

The adult Chinook salmon samples were collected at the Clackamas fish hatchery. Whole body, fillet with skin, and fillet without skin composite samples were analyzed. Each composite sample included three individual fish. Five whole body composite samples, including one split, three fillet with skin, and three fillet without skin composite samples were analyzed. The fillet without skin composite samples were only analyzed for dioxin, furan, and polychlorinated biphenyl (PCB) congeners and mercury.

The adult Pacific lamprey samples were collected at the Willamette Falls. Only whole body composite samples were analyzed. Each composite sample included 30 individual fish. Four whole body composite samples were analyzed.

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The adult sturgeon samples were collected between RM 3.5 and 9.2. Only fillet without skin samples were analyzed. Each sample was an individual fish. Six fillet samples, including one split, were analyzed.

2.1.6 Shellfish Tissue

Shellfish tissue in the BHHRA dataset included field-collected samples for crayfish and clam (*Corbicula* sp.) tissue. Crayfish samples were collected during Rounds 1 and 3 and clam samples were collected during Rounds 1, 2, and 3. Although data from laboratory bioaccumulation samples were also available from Round 2, these data were not used because field-collected tissue samples provide for a more direct evaluation of potential human exposure than laboratory bioaccumulation samples. No field-collected, non-LWG shellfish tissue data of acceptable data quality for risk evaluation (Category 1/QA2) were identified. Tables 2-7 and 2-8 present a summary of the shellfish tissue samples included in the BHHRA dataset, from both inside and outside the Study Area, respectively.

For crayfish, samples were collected from 24 stations during Round 1. The Round 1 crayfish stations were selected based on habitat areas. Crayfish were collected from 9 stations during Round 3. The Round 3 crayfish stations were based on data needs identified by the EPA and habitat areas. Crayfish were collected and composited from individual stations commensurate with their limited home ranges. Only whole body composite samples were collected for crayfish. During Round 1, two replicate composite samples were collected at three of the 24 stations. At each of the remaining stations, a single composite sample was collected. During Round 3, a single composite sample was collected at each station.

For clams, samples were collected from 3 stations during Round 1, 33 stations during Round 2, and 10 stations during Round 3. Clams were collected and composited from individual stations that were selected based on habitat areas and biomass availability. A single composite sample was collected at each station in Rounds 1 and 2. In Round 3, two composite samples were collected from each of five stations, and a single composite sample was collected from each of the remaining five stations. Depuration is a common method for cleansing shellfish that is often done prior to human consumption to eliminate the sediment present in the gastrointestinal (GI) tract of the shellfish. The Round 1 and Round 2 field-collected clams were not depurated prior to analysis, and the data therefore may over predict human health risks from this exposure pathway for consumers that do depurate clams prior to consumption. In Round 3, five samples were depurated prior to analysis (depurated samples were from stations where two samples were collected; one sample from each Round 3 station was not depurated). Additional discussion of the potential effects of depuration on human health risks is included in Section 6. All LWG field-collected clam samples were included in the BHHRA dataset.

2.2 USE OF DATA

Prior to using the data in the BHHRA, data reduction was conducted consistent with the Guidelines for Data Reporting, Data Averaging, and Treatment of Non-Detected Values for the Round 1 Database (Kennedy/Jenks Consultants et al. 2004), the Exposure Point Concentration Calculation Approach and Summary of Exposure Factors (Kennedy/Jenks Consultants 2006), and Proposed Data Use Rules and Data Integration for Baseline Human Health Risk Assessment (BHHRA), submitted to EPA in a May 28, 2008 email communication with EPA. Data reduction and data use rules applied to the combining of surface water data collected by different methods, the handling of non-detects, the summing of chemical groups, and the calculation of exposure point concentrations (EPCs). These rules are described in detail in Attachment F2.

2.3 CHEMICAL SCREENING CRITERIA

EPA guidance (1989) recommends considering criteria to limit the number of chemicals that are included in a quantitative risk assessment while also ensuring that all contaminants that may contribute significantly to the overall risk are addressed. According to EPA guidance, the screening procedure is used to focus quantitative risk assessment efforts on contaminants that could be of concern under health-protective exposure assumptions. For purposes of the BHHRA, the only screening criterion used to select COPCs was a comparison with risk-based concentrations, as described in the Programmatic Work Plan (Integral et al. 2004). The risk-based concentrations used to select COPCs are described below for the respective BHHRA media. When specified below, COPCs were selected for a medium based on a subset of data determined to represent exposure to a specific human population. Potentially exposed human populations are discussed as part of the exposure assessment in Section 3, and include but are not limited to: transients, divers, recreational beach users, and fishers.

2.3.1 Sediment

Sediment data were quantitatively evaluated in the BHHRA for direct exposure scenarios. As a health-protective initial approach, the current EPA Regional Screening Levels (RSLs) for soil (EPA 2010a) were used as the basis for screening values for sediment. For chemicals that do not have EPA RSLs, EPA RSLs for surrogate chemicals with similar chemical structures were used if available (e.g., pyrene for phenanthrene). As required by EPA Region 10 (see e-mail from Dana Davoli to Laura Kennedy, October 17, 2008, in Attachment F1), for trichloroethylene, the geometric mid-point of the slope factor range from EPA 2001 (0.089 per mg/kg-day) was used for evaluating cancer risks for both inhalation and oral exposures. This value was also used to calculate an acceptable soil screening level of 7.7 mg/kg.

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For carcinogenic chemicals, the EPA RSLs were used as the screening values. For noncarcinogenic chemicals, the EPA RSLs were divided by 10 to account for potential cumulative effects from multiple chemicals, as required by EPA Region 10 (2007a), and these modified RSLs were used as the screening values. For chemicals that exhibit both carcinogenic and noncarcinogenic effects, the lower screening value was used for selecting COPCs.

EPA RSLs have been developed for both residential and industrial exposure scenarios for soil. Residential soil EPA RSLs are based on exposure assumptions of 350 days per year. For cancer endpoints, the residential EPA RSLs are calculated using an age-adjusted soil ingestion factor that takes into account the difference in daily soil ingestion rates, body weight, and exposure duration for children from 1 to 6 years old and others from 7 to 31 years old (total exposure over 30 years). For noncancer endpoints, the residential EPA RSLs are calculated using exposure factors for children from 1 to 6 years old and chronic toxicity criteria. Industrial soil EPA RSLs are based on exposure assumptions of 250 days per year for 25 years. Both residential and industrial EPA RSLs are based on a target cancer risk of 1×10^{-6} for carcinogenic chemicals or a hazard quotient of 1 for noncarcinogenic chemicals. Dividing EPA RSLs for noncarcinogenic chemicals by 10 is equivalent to using a hazard quotient of 0.1. Because the potential exposure to sediments that may occur is anticipated to be less than the exposure that was assumed to occur with soil in developing the EPA RSLs, the soil RSLs represent conservative screening values for protection of human health. Because uses of Portland Harbor include both recreational and industrial activities, COPCs were selected using both residential and industrial EPA RSLs, consistent with the EPA comments on the Round 2 Comprehensive Report provided on January 15, 2008 (EPA 2008b).

For beach sediment, residential soil EPA RSLs were used to select COPCs in areas where exposures could occur during recreational, transient, or fishing activities. In areas where occupational exposures could occur, COPCs were selected using industrial soil EPA RSLs. The designated potential uses for beaches in the Study Area are presented in Map 2-1.

The extent of direct contact (i.e., ingestion and dermal contact) with in-water sediment that could occur under site-specific exposure scenarios would be significantly less than with upland soil or beach sediment. Therefore, COPCs for in-water sediment were identified using only the industrial soil EPA RSLs.

2.3.2 Surface Water and Groundwater Seep

Surface water and groundwater seep data were quantitatively evaluated in the BHHRA for direct exposure scenarios. A discussion of potential sources of contaminants to surface water is provided in the RI. As a health-protective initial approach, EPA RSLs for residential tapwater (EPA 2010a) were used as the screening values for surface water and the groundwater seep to select COPCs for direct exposure scenarios. For chemicals that do not have EPA RSLs, EPA RSLs for surrogate chemicals with similar chemical structures were used if available (e.g., pyrene for phenanthrene). As required by EPA Region 10 (EPA 2007a), the EPA Region 6 Human Health Medium-Specific Screening Levels for trichloroethylene (EPA 2008a), rather than the EPA RSLs, were used in this BHHRA. For carcinogenic chemicals, the EPA RSLs were used as the screening values. For noncarcinogenic chemicals, the EPA RSL was divided by 10 to account for potential cumulative effects from multiple chemicals, and this modified EPA RSL was used as the screening value, as required by EPA Region 10.

Residential tapwater EPA RSLs are based on domestic use of water, including ingestion, and represent conservative screening values for direct contact scenarios where water may not be used for domestic purposes, such as surface water contact during beach recreation. EPA RSLs are based on a target cancer risk of 1×10^{-6} for carcinogenic chemicals or a hazard quotient of 1 for noncarcinogenic chemicals. Dividing EPA RSLs for noncarcinogenic chemicals by 10 is equivalent to using a hazard quotient of 0.1.

2.3.3 Tissue

EPA Region 10 has not accepted any criteria for screening tissue from Portland Harbor; therefore, per an agreement with EPA, risk-based concentrations were not used for screening the tissue data, and all chemicals detected in fish and shellfish in the BHHRA dataset were selected as COPCs for tissue.

2.3.4 Hypothetical Future Exposure to Untreated Surface Water for Domestic Use

Even though no current or future uses of the LWR within Portland Harbor as a domestic water source have been identified, under OAR 340-041-0340 Table 340A, domestic water supply is a designated beneficial use of the Willamette River, with adequate pretreatment. Because the Willamette River is capable of serving as a potential drinking water source, the expectation is that this resource will be protected to achieve such use with adequate pretreatment. Although surface water within the Study Area is not currently used as a domestic water source, nor are there future plans for domestic water use within the Study Area, surface water data were quantitatively evaluated in the BHHRA as a hypothetical future domestic water source at the direction of EPA (see Section 2.4.5 below). The same criteria and screening values used for data to assess direct contact with surface water and the groundwater seep were used to select COPCs for surface water as a hypothetical future domestic water source. As with the surface water and groundwater seep screening, the noncarcinogen RSLs were divided by 10 to account for potential multiplicative effects, and the modified RSLs were used as the screening values.

In addition to the EPA RSLs, EPA maximum contaminant levels (MCLs) for drinking water (EPA 2003a) were used as screening criteria for the selection of COPCs for the hypothetical future use of untreated surface water for domestic purposes. If the maximum detected concentration for a contaminant in the dataset selected to represent hypothetical exposure to untreated surface water for domestic use exceeded either the EPA RSL or the EPA MCL, the contaminant was selected as a COPC for this scenario.

2.4 IDENTIFICATION OF CONTAMINANTS OF POTENTIAL CONCERN

COPCs for human health were selected according to the approach described in the Programmatic Work Plan (Integral et al. 2004) using the screening criteria described in Section 2.3 and were quantitatively evaluated in this BHHRA. The process used to select the COPCs for quantitative evaluation in this BHHRA is described in the following subsections.

2.4.1 Sediment

Humans can be exposed to both beach sediment and in-water sediment. Because the exposure scenarios for beach versus in-water sediment are different, COPCs were selected for both beach and in-water sediment exposures.

2.4.1.1 Beach Sediment

Beach sediment data were evaluated in the BHHRA for potential risks to human health through direct contact. The selection of COPCs for beach sediment evaluated sediment data from potential human use areas where direct contact with human receptors could occur (only reasonably accessible beach sediments, such as those with access from contiguous upland areas or by boat). The locations of the beach sediment data evaluated in the BHHRA are shown in Map 2-1.

For contaminants that were detected in beach sediment, the detected concentrations were compared to risk-based screening levels described in Section 2.3.1. The maximum detected concentration of each contaminant from all samples collected in recreational, transient, or fishing beach areas was compared to the screening level based on the residential soil EPA RSL. The maximum detected concentration of each contaminant from all samples collected in industrial beach areas was compared to the screening level based on the industrial soil EPA RSL. If the maximum detected concentration of a contaminant was greater than the screening level, that contaminant was selected as a COPC for beach sediment. The contaminants selected as COPCs for beach sediment and the rationale for selection are presented in Tables 2-9 and 2-10.

Contaminants selected as COPCs for beach sediment were quantitatively evaluated in this BHHRA. Contaminants with maximum detected concentrations less than the

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screening values were not selected as COPCs and were not evaluated further in this BHHRA for direct contact with beach sediment.

2.4.1.2 In-Water Sediment

In-water sediment data were evaluated in the BHHRA for potential risks to human health through direct contact and not based on the potential for bioaccumulation. The potential for bioaccumulation is evaluated separately in this BHHRA as part of the fish and shellfish tissue assessments. The selection of COPCs for in-water sediment evaluated all surface sediment data in the BHHRA dataset within the Study Area, excluding the navigation channel and beach composite samples. The sample locations of the in-water sediment data evaluated in the BHHRA are shown in Map 2-2.

For chemicals that were detected in in-water sediment, the maximum detected concentration of each chemical from surface sediment samples was compared to the screening level based on the industrial soil EPA RSL, as described in Section 2.3.1. If the maximum detected concentration of a contaminant was greater than the screening level, that chemical was selected as a COPC for in-water sediment. The contaminants selected as COPCs for in-water sediment and the rationale for selection are presented in Table 2-11.

Contaminants selected as COPCs for in-water sediment were quantitatively evaluated in this BHHRA. Chemicals with maximum detected concentrations less than the EPA RSLs were not selected as COPCs and were not evaluated further in this BHHRA for direct contact with in-water sediment.

2.4.2 Surface Water

Direct contact with surface water was evaluated in the BHHRA for potential risks to human health. The selection of COPCs for quantitative evaluation in the BHHRA in surface water was based only on potential for direct human contact and not based on the potential for bioaccumulation. The potential for bioaccumulation is evaluated separately in this BHHRA as part of the fish and shellfish tissue assessments. Surface water data gathered during the RI were used to identify the COPCs in surface water for quantitative evaluation in the BHHRA. Because the exposure scenarios for divers are different from those of transients and beach users, COPCs were selected separately for both divers and transient/beach user exposures. For divers, COPCs were selected from all available surface water samples taken within the Study Area, as described in Section 2.1.3. Near-bottom and near-surface sample results, as well as vertically integrated transect results, were combined according to the rules described in Attachment F2 prior to selecting COPCs. For transients and beach users, COPCs were selected from surface water samples taken from areas where direct contact with transient or beach users could occur, including both single point sampling stations where vertically integrated samples were collected and transect samples. This included one sample from Swan Island Lagoon. A summary of

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samples used for each surface water COPC screening is provided in Table 2-12. In addition, the sample locations of the surface water data evaluated for transients and recreational beach user exposure scenarios are shown in Map 2-3. The sample locations of the surface water data evaluated for diver exposures are shown in Map 2-4.

For chemicals that were detected in each surface water dataset, the detected concentrations were compared to screening values based on the residential tapwater RSLs. If the maximum detected concentration of a contaminant in surface water was greater than the screening value, that contaminant was selected as a COPC for surface water and was quantitatively evaluated in the BHHRA. Chemicals that were detected only at concentrations less than the RSLs were not selected as COPCs for quantitative evaluation. The contaminants selected as COPCs for surface water and the rationale for selection are presented in Table 2-13 for divers, and Table 2-14 for transients and beach users.

2.4.3 Groundwater Seep

Direct contact with the groundwater seep at Outfall 22B, shown in Map 2-5, was evaluated in the BHHRA for potential risks to human health. The selection of COPCs for quantitative evaluation in the BHHRA was based only on potential for direct human contact with the groundwater seep, and not based on the potential for bioaccumulation.

For chemicals that were detected in the groundwater seep, the detected concentrations were compared to screening values based on the residential tapwater EPA RSLs. If the maximum detected concentration of a contaminant in the groundwater seep was greater than the screening value, that contaminant was selected as a COPC for the groundwater seep and was quantitatively evaluated in the BHHRA. Chemicals that were detected only at concentrations less than the EPA RSLs were not selected as COPCs for quantitative evaluation. The contaminants selected as COPCs for the groundwater seep and the rationale for selection are presented in Table 2-15.

2.4.4 Fish and Shellfish Tissue

Fish and shellfish tissue were evaluated in the BHHRA for potential risks to human health through ingestion. Because EPA Region 10 has not accepted any criteria for screening tissue from Portland Harbor, all chemicals detected in fish and shellfish tissue in the BHHRA dataset were considered to be COPCs and evaluated further in the BHHRA. Map 2-6 shows the general location of all fish for a particular composite of the smallmouth bass and common carp tissue data evaluated for ingestion scenarios in this BHHRA. Samples for brown bullhead and black crappie were each composited for RM 3-6 and RM 6-9, and are not shown on a map. The sample locations of the shellfish tissue data (both crayfish and clam) evaluated for ingestion scenarios are shown in Map 2-7. Shellfish were also composited over areas representing their assumed home range, and the sample locations on Map 2-7 represent the general spatial distribution of composited samples. The contaminants detected in each individual species were selected as COPCs only for ingestion of that species. For the multi-species diet scenarios (discussed in Section 3), analytes detected in any of the target resident fish species (see Section 2.1.5) were selected as COPCs. Since no screening took place to determine COPCs for tissue, the tissue COPCs are presented in the exposure point concentration summary tables, discussed in Section 3.

2.4.5 Hypothetical Future Exposure to Untreated Surface Water for Domestic Use

There is no known current or anticipated future use of surface water within the Study Area for a drinking water supply. Even though no current or future uses of the LWR within Portland Harbor as a domestic water source have been identified, under OAR 340-041-0340 Table 340A, domestic water supply is a designated beneficial use of the Willamette River, with adequate pretreatment. Because the Willamette River is capable of serving as a potential drinking water source, the expectation is that this resource will be protected to achieve such use with adequate pretreatment. Potential sources of contaminants to surface water are discussed in the RI. Because future use of the LWR as a domestic water supply would require adequate pretreatment, the evaluation of untreated surface water as a drinking water source is designated a hypothetical scenario. The inclusion of the assessment of domestic use of untreated surface water from the Study Area was done at the direction of EPA.

Surface water as a hypothetical future domestic water source was evaluated in the BHHRA for potential risks to human health. The selection of COPCs for quantitative evaluation in the BHHRA in surface water was based only on potential for hypothetical contact from domestic uses, and not based on the potential for bioaccumulation. The potential for bioaccumulation is evaluated separately in this BHHRA as part of the fish and shellfish tissue assessments. Surface water data gathered during the RI were used to identify the COPCs for quantitative evaluation in the BHHRA. At the direction of EPA, results from surface water samples collected near-bottom and near-surface within the water column were combined according to the rules described in Attachment F2. The combined near-bottom and near-surface samples, vertically integrated single point samples, and vertically integrated transect samples were used to select the COPCs. These samples are presented in Table 2-12, and shown in Map 2-8. Filter and column data collected from samples collected by XAD were combined before selection of COPCs, according to the rules described in Attachment F2. No further data reduction was performed on the hypothetical future domestic water dataset prior to COPC selection.

For chemicals that were detected in this dataset, the detected concentrations were compared to screening values based on the RSLs for tap water and on EPA MCLs for drinking water (EPA 2003a). If the maximum detected concentration of a

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contaminant in surface water was greater than either of the screening values, that contaminant was selected as a COPC for surface water and was quantitatively evaluated in the BHHRA. Chemicals that were detected only at concentrations less than both screening values were not selected as COPCs for quantitative evaluation. The maximum detected concentration did not exceed the MCL for any chemical (Table 2-16). Maximum concentrations exceeded other RSLs [e.g., tap water screening levels for arsenic and 2-(4-Chloro-2-methylphenoxy)propanoic acid (MCPP)]. The contaminants selected as COPCs for surface water as a hypothetical domestic water source, and the rationale for selection, are presented in Table 2-16.

3.0 EXPOSURE ASSESSMENT

The objectives of the exposure assessment are to identify potential exposure pathways for individuals who may come in contact with COPCs at the Study Area, to characterize potentially exposed populations, and to estimate the extent of exposure.

The exposure assessment in this BHHRA followed EPA guidance and incorporated the reasonable maximum exposure (RME) methods recommended by EPA. As stated in EPA guidance (EPA 1989), the RME is a conservative exposure level that is still within the range of possible exposures. The exposure assessment also used average values, which represent central tendency (CT) exposures, for some exposure scenarios. According to EPA (1989), an exposure assessment includes four primary tasks:

- Identify potentially exposed human populations that may come in contact with the COPC. This requires knowledge of (and/or making reasonable assumptions regarding) both current and future populations.
- Identify relevant exposure pathways for human populations by which potentially exposed populations may contact environmental media containing COPCs.
- Estimate EPCs at the points of potential human contact for all identified COPCs.
- Estimate daily intakes for exposure routes and potentially exposed populations. The daily intakes are derived using the EPCs and assumptions regarding such variables as exposure duration, consumption rates, skin absorption factors, and other parameters that describe human activities.

The exposure assumptions and methods for each task included in the exposure assessment are discussed below.

3.1 IDENTIFICATION OF POTENTIALLY EXPOSED HUMAN POPULATIONS

Potentially exposed and hypothetically exposed populations were identified based on consideration of current, future, and hypothetical future uses of the Study Area and EPA (1989) guidance. A pathway analysis for the Study Area is detailed in the Portland Harbor RI/FS Programmatic Work Plan (Integral 2004). The human populations identified below represent those populations that are anticipated to be maximally-exposed to contaminants within the Study Area under current and reasonably foreseeable or hypothetical future conditions. The evaluation performed for the selected populations is considered to be protective of other potentially exposed populations that are not evaluated quantitatively in this BHHRA. The populations for current, future, and hypothetical future uses of the Study Area include the following:

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- Dockside worker
- In-water worker
- Transient
- Diver
- Recreational beach user
- Non-tribal Fisher
- Tribal fisher
- Domestic water user

The above populations were identified based on human activities that are known to occur within the Study Area, as described in the Programmatic Work Plan, or were required by EPA for evaluation in this BHHRA. Divers, clam consumption by fishers, and domestic water user were included in this BHHRA as required by EPA. Infant consumption of human milk was included as a complete exposure pathway for all adult receptor populations that were assessed quantitatively for bioaccumulative chemicals (i.e., PCBs, dioxin/furans, and DDX), as required by EPA.

Potential risks were quantified for each of the receptor populations; however, certain individuals may participate in activities resulting in potential exposures under more than one category (e.g., recreational beach users may also be fishers). Potentially overlapping exposures are discussed in Section 3.3.7 of this BHHRA.

This BHHRA focused on potential exposures occurring within and immediately upstream and downstream of the Study Area in quantifying potential risks to humans.

Except for the hypothetical future exposure to untreated surface water for domestic water users, the exposure assessment assumes that future land and water use will be the same as current land use; therefore, the risks characterized are based only on current use. If land or water use changes in the future, exposures and risk may also change.

3.2 IDENTIFICATION OF EXPOSURE PATHWAYS

Exposure pathways are defined as the physical ways in which chemicals may enter the human body (e.g., ingestion, inhalation, dermal absorption). A complete exposure pathway consists of the following four elements:

- A source of chemical release
- A release or transport mechanism (or media in cases involving media transfer)
- An exposure point (a point of potential human contact with the contaminated exposure medium)

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- An exposure route (e.g., ingestion, dermal contact) at the exposure point.

If any of the above elements is missing, the pathway is considered incomplete and exposure does not occur.

As discussed in Sections 4, 5, and 6 of the RI Report, the affected media within the Study Area are sediment, water, and biota. Current and historical industrial activities and processes within, upstream and downstream of the Study Area may have led to chemical releases from either point or nonpoint sources to the Study Area. In addition to these releases, discharges to the river from outfalls and groundwater may be potential contaminant sources to the Study Area. Finally, releases that occur upstream and downstream of the Study Area and global, regional, and local emissions resulting in atmospheric deposition may be potential sources to the Study Area. These potential sources and release mechanisms are discussed in greater detail in Section 4 of the RI Report.

Chemicals in sediment and water may be accumulated by organisms in the water column or associated with the sediments. Edible fish and shellfish species feeding on these organisms and living within the Study Area may accumulate chemicals in their tissues through dietary exposures and direct exposure to sediment and water. The potential exposure pathways to human populations at the Study Area include:

- Ingestion of and dermal contact with beach sediment
- Ingestion of and dermal contact with in-water sediment
- Ingestion of and dermal contact with surface water
- Ingestion of and dermal contact with groundwater seep
- Ingestion of fish and shellfish
- Infant consumption of human milk.

Section 3.3 provides a more detailed discussion of potential exposures for the Study Area under current, reasonably foreseeable and hypothetical future conditions, and presents the rationale for including or eliminating pathways from quantitative evaluation. The identified receptors, exposure routes, and exposure pathways, and the rationale for selection are also summarized in Table 3-1.

3.2.1 Definition and Significance of Exposure Pathways

Exposure pathways are designated in one of the following four ways:

Potentially Complete: There is a source or release from a source, an exposure point where contact can occur, and an exposure route by which contact can occur.

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Pathways considered potentially complete are quantitatively evaluated in this BHHRA.

Potentially Complete and Insignificant: There is a source or release from a source, an exposure point where contact can occur, and an exposure route by which contact can occur; however, the pathway is considered a negligible contributor to the overall risk. Pathways considered potentially complete and insignificant were not evaluated further in this BHHRA.

Incomplete: There is no source or release from a source, no exposure point where contact can occur, or no exposure route by which contact can occur for the given receptor. Pathways considered potentially incomplete were not evaluated further in this BHHRA.

Potentially complete pathway, but evaluated under a different receptor category: These pathways may be complete for individuals in this receptor category due to overlapping exposure scenarios (e.g., some in-water workers may also be fishers), but are not evaluated for the identified receptor category because the pathways are not considered relevant for that receptor. These pathways are evaluated under different receptor categories where the pathways are considered potentially complete and significant. Overlapping exposures that may occur for the different receptor categories are discussed further in Section 3.3.7 of this BHHRA.

3.2.2 Conceptual Site Model

The conceptual site model (CSM) for human exposures based on the current understanding of the Study Area and requirements from EPA is presented in Figure 3-1. The CSM graphically depicts possible sources of COPCs based on current information, possible COPC-affected media, mechanisms of COPC transfer between media, and the processes through which human receptors may be exposed to chemicals. Additional information on potential sources of COPCs is provided in Section 5 of the RI Report. Potentially complete exposure pathways were identified in the Programmatic Work Plan or based on subsequent requirements from EPA. In-water workers exposure to river sediment, transients exposure to shoreline seeps, divers exposure to surface water and in-water sediment, infant exposure via consumption of human milk for all receptors with bioaccumulative COPCs, and hypothetical future exposures of domestic water users to surface water were included as potentially complete pathways per requirements from EPA. Pathways that are potentially or hypothetically complete and may result in significant exposure, or for which significance is unknown, were evaluated quantitatively in this BHHRA, per direction from EPA. Pathways included at the direction of EPA include clam consumption, exposure to surface water and in-water sediment by a commercial diver, and hypothetical exposure to untreated surface water by a domestic water user.

3.3 EXPOSURE SCENARIOS

The following sections provide a detailed discussion of the exposure scenarios that are quantitatively evaluated in this BHHRA. The following exposure scenarios were identified based on exposures that may generically occur throughout the Study Area and do not consider site-specific conditions that may limit exposure at a given location.

3.3.1 Direct Exposure to Beach Sediment

Ingestion of and dermal contact with beach sediment could occur within natural river beach areas used by human populations within the Study Area. These areas were identified as human use areas in the Programmatic Work Plan based on current and future uses within the Study Area. Human use areas were further classified based on the type of exposures that could occur at these beaches including recreational, fishers, tribal fishers, transient, or dockside worker use areas. These classifications are described in greater detail below. The human use areas in the Study Area and their associated classifications are shown in Map 2-1.

3.3.1.1 Dockside Workers

Dockside workers include industrial and commercial workers at facilities adjacent to the river who conduct specific activities within natural river beach areas, such as unloading ships or barges from the beach itself or conducting occasional maintenance activities from the water's edge. The actual activities that occur within natural river beach areas are site-specific and generally occur only very infrequently. Although exposure is anticipated to be infrequent, workers conducting activities within natural river beach areas may contact beach sediment within riverfront industrial and commercial sites at the Study Area. Exposure for a given worker would occur only within the defined dockside worker use area adjacent to the facility of that worker.

3.3.1.2 Transients

During past site tours, tents and makeshift dwellings were observed as evidence that individuals were occupying some riverbank areas. While the tents and makeshift dwellings were typically observed above the actual beach areas, transients may contact beach sediment within transient use areas, which are beach areas that are not active industrial sites and are not otherwise restricted from access. Although transients are anticipated to move throughout the Study Area, some may spend a majority of their time at relatively few of the possible areas. Exposure for a given transient was evaluated in this BHHRA on the basis of a single transient use area, although it is possible that transients move from one transient use area to others within or outside the Study Area. This BHHRA presented an evaluation of individual use areas not only because transients may inhabit single beach areas, but also because such an evaluation provides a range of possible risks for individuals that either move frequently or remain at a single location.

3.3.1.3 Recreational Beach Users

Both adults and children participate in recreational activities in beach areas within the Study Area. Areas currently used for recreational beach activities, as well as other areas in the Study Area where sporadic beach use may occur were identified as recreational use areas. Recreational beach users may contact beach sediment within recreational use areas at the Study Area. Some recreational beach users may primarily use a specific recreational use area while other recreational beach users may use various recreational use areas throughout and outside the Study Area.

3.3.1.4 Tribal Fishers

The LWR provides a ceremonial and subsistence fishery for Native American tribes. The extent to which tribal members fish within the Study Area, as well as the extent to which that fishing occurs from beach areas and the degree of sediment exposure that might occur while fishing are unknown. However, exposure assumptions provided by EPA were used to evaluate beach sediment exposure by tribal fishers.

3.3.1.5 Non-tribal Fishers

Fishers who fish from the water's edge within natural river beach areas could have direct exposure to beach sediment. In theory, fishing could occur at any beach area without restricted access. Therefore, all non-dockside worker use areas (i.e., all transient and recreational use areas) were considered potential human use areas where fishers could be exposed to beach sediment. Some fishers may primarily use a specific beach area for fishing activities while other fishers may use beach areas throughout and outside the Study Area.

For beach sediment exposure, two different fisher scenarios were included in this BHHRA to evaluate differences in the frequency of fishing activities. High-frequency fishers were assumed to fish recreationally, and at more frequent intervals than the low-frequency fisher (exposure frequency of 156 days per year for high frequency fishers compared to 104 days per year for low-frequency fishers). The extent to which fishing from beach areas actually occurs is unknown, as is the degree of sediment exposure that might occur while fishing.

3.3.1.6 Potentially Complete and Insignificant Exposure Pathways

This BHHRA did not identify any potentially complete and insignificant exposure pathways for beach sediment exposure.

3.3.1.7 Incomplete Exposure Pathways

Beach sediment exposures are considered incomplete exposure pathways for both in-water workers and divers based on the defined activities of these receptor populations in this BHHRA. In-water workers are those workers who conduct over water activities and thus are not directly exposed to beach sediments. Dockside workers are the worker population for which beach sediments exposures are considered potentially complete and were evaluated in this BHHRA. Divers conduct activities in the river that do not result in beach sediment exposures. The hypothetical future domestic water use scenario evaluates use of surface water for domestic water supply and thus beach sediment exposures were considered incomplete exposure pathways for this receptor population.

3.3.2 Direct Exposure to In-Water Sediment

Ingestion of and dermal contact with in-water sediment could occur through over-water activities (i.e., activities conducted from a boat or other vessel) that result in bringing sediment to the river's surface where exposure would be possible. Unlike the beach sediment exposure scenarios that are restricted to specific beach areas, potential exposure to in-water sediment could occur anywhere that over-water activities occur. As a result, direct exposure to in-water sediment was evaluated throughout the Study Area. At the direction of the EPA, exposure to in-water sediment by divers is also evaluated in this BHHRA.

3.3.2.1 In-Water Workers

While this population is referred to as "in-water" workers, these workers are not actually in the water. Rather, in-water workers are those workers who conduct over-water activities such as maintenance dredging and repair of in-water structures. Exposure to in-water sediment could occur while performing these specific activities, although most maintenance dredging activities are mechanical and are unlikely to result in significant sediment contact. Although likely occurring less frequently than mechanical dredging activities, other activities such as maintenance and cleaning of equipment or in off-loading sediments to disposal sites may result in a greater exposure potential.

3.3.2.2 Divers

In the Study Area, the majority of divers are expected to be commercial divers. To evaluate diver exposures, two different exposure scenarios are included in this BHHRA, one assuming that a wet suit is worn during diving and one assuming that a dry suit is worn during diving. The diver exposure scenarios were directed by EPA in a memorandum regarding the *Proposed Commercial Diver Exposure Scenario for the Portland Harbor Risk Assessment* (EPA 2008c). Both the wet suit and dry suit diver exposure scenarios assume that the diver is exposed to sediment through inadvertent ingestion of sediment and dermal exposure to sediment. As EPA stated in its approach, the use of a dry suit is expected to limit diver exposure, so it is assumed that the wet suit diver has more dermal exposure to sediment than the dry suit diver. Based on communications with commercial diving companies in the Portland area (Hutton 2008, Johns 2008, and Burch 2008), the standard of practice for commercial divers is the use of dry suits and helmets when diving in the LWR. However, based on the directive of the EPA, the wet suit diver scenario is also included in this BHHRA.

3.3.2.3 Tribal Fishers

The LWR provides a ceremonial and subsistence fishery for Native American tribes. The extent to which tribal members fish within the Study Area, as well as the extent to which that fishing occurs from boats or piers and the degree of sediment exposure that might occur while fishing are unknown. However, exposure assumptions provided by EPA were used to evaluate in-water sediment exposure by tribal fishers.

3.3.2.4 Non-tribal Fishers

Fishers who fish from boats or piers could be theoretically exposed to in-water sediment on anchors, hooks, or crayfish pots. For in-water sediment exposure, two different fisher scenarios were included in this BHHRA to evaluate differences in the frequency of fishing activities: high-frequency fishers and low-frequency fishers. The extent to which fishing actually occurs under these two scenarios is unknown, as is the degree of sediment exposure that might occur while fishing. However, exposure assumptions provided by EPA were used to evaluate in-water sediment exposure by fishers.

3.3.2.5 Potentially Complete and Insignificant Exposure Pathways

Recreational beach users could contact in-water sediment while swimming. However, any exposure to in-water sediment is expected to be minimal and the exposure would occur under water, so it cannot be quantitatively evaluated using EPA exposure models. In-water sediment exposures were considered potentially complete and insignificant exposure pathways for recreational beach users and were not quantitatively evaluated in this BHHRA.

3.3.2.6 Incomplete Exposure Pathways

In-water sediment exposures were considered incomplete exposure pathways for dockside workers and transients based on the defined activities of these receptor populations in this BHHRA. Dockside workers are those workers who conduct specific activities within natural river beach areas and thus are not directly exposed to in-water sediments. In-water workers are the worker population for which in-water sediments exposures are considered potentially complete and were evaluated in this BHHRA. Transients who conduct specific activities while occupying natural river beach areas are unlikely to contact in-water sediment. The hypothetical future domestic water use scenario evaluates use of surface water for domestic water supply and thus in-water sediment exposures are considered incomplete exposure pathways for this receptor population.

3.3.3 Direct Exposure to Surface Water

Direct exposure to surface water could potentially occur for many of the populations evaluated in this BHHRA. Two populations expected to potentially have the most frequent contact with surface water are transients and recreational beach users. At the direction of the EPA, exposure to surface water by divers and the hypothetical future use of untreated surface water as a domestic water source are also evaluated in this BHHRA.

3.3.3.1 Transients

Transients may have dermal contact with surface water during swimming, bathing or other activities, such as washing of clothing or equipment. In theory, transients may also use river water as a drinking water source. Exposure to surface water by transients would likely occur within transient use areas.

3.3.3.2 Divers

As described in Section 3.3.2.2, two different diver exposure scenarios are included in this BHHRA. The two exposure scenarios for divers differentiate between the use of either a wet suit or dry suit, as directed by the EPA (2008c). Both the wet suit and dry suit diver exposure scenarios assume that the diver is exposed to surface water through inadvertent ingestion of surface water and dermal exposure to surface water. As EPA stated in its approach, the use of a dry suit is expected to limit diver exposure, so a diver using a wet suit is assumed to have more dermal exposure to surface water.

3.3.3.3 Recreational Beach Users

The LWR is used by both adults and children for boating, water skiing, swimming, and other water activities that result in exposure to surface water. Of these activities, exposure to surface water would occur to the greatest extent while swimming in the river. Swimming would likely occur primarily within recreational beach areas.

3.3.3.4 Domestic Water User

As mentioned in Section 2.4.5, there is no known current use of surface water within the Study Area for a domestic water supply. However, because domestic water use is a designated beneficial use of the Willamette River following adequate pretreatment, the use of untreated river water as a domestic water source was assessed as a hypothetical future pathway for both adult and child residents, at the direction of EPA. In this scenario, exposure to untreated surface water could hypothetically occur from ingestion and dermal contact throughout the Study Area. At the direction of the EPA, volatilization of chemicals from untreated surface water to indoor air through household uses was identified as a potentially complete exposure pathway for hypothetical future domestic water use.

3.3.3.5 Potentially Complete and Insignificant Exposure Pathways

Surface water exposures through dermal absorption and ingestion were considered potentially complete and insignificant exposure pathways for dockside workers, in-water workers, tribal fishers, and fishers. It is unlikely that both dockside and in-water populations would have direct contact with surface water through industrial activities. It is also unlikely that tribal fishers and fishers would have significant direct contact with surface water through fishing activities. Any exposures to surface water by the dockside workers, in-water workers, tribal fishers, or fishers would be minimal; therefore, surface water exposures were considered potentially complete and insignificant exposure pathways for these receptor populations.

Volatilization of chemicals from surface water to outdoor air is unlikely to result in a significant exposure considering the amount of mixing with ambient air that would occur. Given the low levels of chemicals in outdoor air from volatilization from surface water, surface water exposures through inhalation of volatiles was considered a potentially complete and insignificant exposure pathway for all receptor populations who conduct outdoor activities.

3.3.3.6 Incomplete Exposure Pathways

This BHHRA did not identify any incomplete exposure pathways for surface water exposures.

3.3.4 Direct Exposure to Groundwater Seeps

Direct contact with groundwater would occur only within human use areas where groundwater comes to the surface (i.e., seeps) on the beach above the water line and is only considered a potentially complete exposure pathway for transients and recreational beach users. As described in Section 2.1.4, there was only one groundwater seep identified during the seep reconnaissance survey that has not been remediated and is located in a recreational or transient use area. That seep, which is the potential groundwater discharge from Outfall 22B, occurs within a potential transient use area. Therefore, only transients were evaluated for exposure to groundwater seeps in this BHHRA.

3.3.4.1 Transients

Transients may have direct contact with groundwater seeps, within riverfront beach areas that have been identified as transient use areas. While contact with seep water would be unintentional, dermal contact with or incidental ingestion of seep water may occur.

3.3.4.2 Potentially Complete and Insignificant Exposure Pathways

This BHHRA did not identify any potentially complete and insignificant exposure pathways for direct exposure to groundwater seeps.

3.3.4.3 Incomplete Exposure Pathways

Direct exposure to groundwater seeps were considered incomplete exposure pathways for all receptor populations who do not conduct activities at beaches where

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groundwater discharges above the water line. As discussed above, only one groundwater seep was identified, which is within a transient use area. Therefore, direct exposure to groundwater seeps is considered an incomplete exposure pathway for dockside and in-water workers, recreational beach users, tribal fishers, fishers, and divers. The hypothetical future domestic water use scenario evaluates use of surface water for domestic water supply and thus groundwater seep exposures were considered incomplete exposure pathways for this receptor population.

3.3.5 Fish Consumption

Certain chemicals may bioaccumulate in fish tissue, and human populations that consume fish may be exposed to COPCs bioaccumulating in the fish tissue. Fish may be caught throughout the Study Area. While the populations evaluated in this BHHRA are described as "fishers", the fish consumption evaluation in this BHHRA includes people who consume fish caught within the Study Area, not just those who catch the fish.

3.3.5.1 Non-tribal Fishers

A year-round recreational fishery exists within the Study Area. Current information suggests that spring Chinook salmon, steelhead, Coho salmon, shad, crappie, bass, and white sturgeon are the fish species preferred by local recreational fishers (DEQ 2000b, Hartman 2002, and Steele 2002). In addition to recreational fishing, the investigation by the Oregonian newspaper and the limited surveys conducted on other portions of the Willamette River indicate that immigrants from Eastern Europe and Asia, African-Americans, and Hispanics are most likely to be catching and eating fish from the lower Willamette (ATSDR 2002). These preliminary surveys also indicate that the most commonly consumed species are carp, bullhead catfish, and smallmouth bass (ATSDR 2002). However, other species may also be consumed. Conversations were conducted with transients about their consumption of fish or shellfish from the Willamette River as part of a project by the Linnton Community Center (Wagner 2004). Transients reported consuming a large variety of fish, and several transients said they ate whatever they could catch themselves or get from other fishers. However, the frequency and amount of consumption was not reported, and many of the transients indicated they were in the area temporarily. Site-specific information is not available for fish consumption rates for specific species, so a range of ingestion rates and various diets were evaluated in this BHHRA for both adult and child consumers.

3.3.5.2 Tribal Fishers

Four (Yakama, Umatilla, Nez Perce, and Warm Springs) of the six Native American tribes involved in the Portland Harbor RI/FS participated in a fish consumption survey that was conducted on the reservations of the participating tribes and completed in 1994 (Columbia River Inter-tribal Fish Commission (CRITFC) 1994). The results of the survey show that tribal members surveyed generally have higher fish ingestion rates than the general public. Fish species, especially salmon and Pacific lamprey, are an important food source as well as an integral part of the tribes' cultural, economic, and spiritual heritage. Ingestion of fish by both adult and child tribal members was evaluated in this BHHRA.

3.3.5.3 Potentially Complete but Evaluated Under a Different Receptor Category

Fish could be consumed by dockside workers, in-water workers, recreational beach users, and divers; however, fish consumption by these receptor populations is evaluated under the fisher receptor category. Long-term, ongoing fish consumption by transients would not occur; therefore, the fisher receptor category would be protective of fish consumption by transients.

3.3.5.4 Incomplete Exposure Pathways

The hypothetical future domestic water use scenario evaluates use of surface water for domestic water supply and thus fish consumption was considered an incomplete exposure pathway for this receptor population.

3.3.6 Shellfish Consumption

Like fish, shellfish may bioaccumulate certain chemicals in their tissue. Populations that consume shellfish may be exposed to COPCs that accumulate in the shellfish tissue. In the Programmatic Work Plan, crayfish was identified as the species to use to evaluate shellfish consumption. Additionally, as required by EPA, consumption of clams is also evaluated in this BHHRA. Harvest and possession of Asian clams, which is the clam species that was found in the LWR during sampling events, is illegal in the State of Oregon because Asian clams are on the prohibited species list of the ODFW rules regarding the importation, possession, confinement, transportation and sale of nonnative wildlife (OAR 635–056–0050).

3.3.6.1 Fishers

In theory, shellfish consumption could occur throughout the Study Area wherever shellfish are found. However, it is not known to what extent shellfish consumption occurs.

The Linnton Community Center project (Wagner 2004) reported that some transients reported eating clams and crayfish; however, many of the individuals indicated that they were in the area temporarily, move from location to location frequently, or have variable diets based on what is easily available. The Superfund Health Investigation and Education (SHINE) program in the Oregon Department of Human Services (DHS) stated that is unknown whether or not crayfish are harvested commercially within Portland Harbor (ATSDR 2006). ODFW has records for crayfish collection in the Columbia and Willamette Rivers, but these records do not indicate whether the collection actually occurs within the Study Area. Based on ODFW's data for 2005 to

2007, no commercial crayfish landings were reported for the Willamette River in Multnomah County. DHS had previously received information from ODFW indicating that an average of 4300 pounds of crayfish were harvested commercially from the portion of the Willamette River within Multnomah County each of the five years from 1997-2001. In addition to this historical commercial crayfish harvesting, DHS occasionally receives calls from citizens who are interested in harvesting crayfish from local waters who are interested in fish advisory information. According to a member of the Oregon Bass and Panfish club, crayfish traps are placed in the Portland Harbor Superfund Site boundaries and collected for bait and possibly consumption (ATSDR 2006). Even if collection does occur within the Study Area, it is not known whether those crayfish are consumed by humans or used as bait.

Because site-specific information is not available for shellfish consumption, a range of ingestion rates was evaluated in this BHHRA for adult shellfish consumers.

3.3.6.2 Potentially Complete but Evaluated Under a Different Receptor Category

Shellfish could potentially be consumed by dockside workers, in-water workers, recreational beach users, and divers; however, shellfish consumption by these receptor populations is evaluated under the adult shellfish consumer receptor category. Long-term, ongoing shellfish consumption by transients would not occur; therefore, the adult shellfish consumer receptor category would be protective of shellfish consumption by transients.

3.3.6.3 Incomplete Exposure Pathways

The hypothetical future domestic water use scenario evaluates use of surface water for domestic water supply and thus shellfish consumption was considered an incomplete exposure pathway for this receptor population.

3.3.7 Potentially Overlapping Exposure Scenarios

Exposure can potentially occur under more than one scenario for an individual. Examples of these overlapping scenarios include: an in-water worker who is also a high-frequency fisher and recreational beach user, a transient who is also a fisher, a tribal fisher who is also a recreational beach user, and others. The potentially overlapping scenarios are indicated in Figure 3-1. It is likely that one or more of the exposure scenarios potentially affecting an individual will pose a much higher level of risk than the other scenario(s), such that combining the effects of the scenarios will not influence risk management decisions for the Study Area. Risks from potentially overlapping scenarios are discussed in Section 5 of this BHHRA.

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3.4 CALCULATION OF EXPOSURE POINT CONCENTRATIONS

EPCs were calculated for media and pathways that were evaluated quantitatively in this BHHRA. The process to estimate EPCs for tissue and beach sediment was previously described in the Programmatic Work Plan, and the Round 1 tissue EPCs were previously presented in *Round 1 Tissue Exposure Point Concentrations* (Kennedy/Jenks Consultants 2004b) and *Salmon, Lamprey, and Sturgeon Tissue Exposure Point Concentrations for Oregon Department of Human Services* (Kennedy/Jenks Consultants 2004c), both of which were approved by EPA. The process for deriving EPCs for in-water sediment, surface water, and groundwater seeps was previously described in *Exposure Point Concentration Calculation Approach and Summary of Exposure Factors* (Kennedy/Jenks Consultants 2006), which was approved by EPA.

EPCs were calculated for the 95% upper confidence limit on the arithmetic mean (95% UCL) and the arithmetic mean for each exposure area. In some exposure areas, the maximum concentration was used instead of the 95% UCL. Therefore, the EPCs are referred to as the 95% UCL/max and mean throughout this BHHRA.

Prior to calculating EPCs for sediment, surface water, tissue, and groundwater seeps, data were reduced, as needed, to address reporting of multiple results for the same constituent in the same sample and to reduce laboratory duplicates and field splits of samples to derive one value for use. Data reductions performed within the SCRA database followed the rules described in *Guidelines for Data Reporting, Data Averaging, and Treatment of Non-Detected Values for the Round 1 Database Technical Memorandum* (Kennedy/Jenks Consultants et al. 2004). Additional data reductions and data use rules specific to the BHHRA were approved by EPA and are detailed in Attachment F2.

Chemicals that were not detected at concentrations above the detection limit were designated as non-detects. Non-detects may represent concentrations that are zero, or may represent concentrations greater than zero but less than the detection limit. For purposes of calculating mean EPCs, non-detected values were used in the calculations at one half their detection limit. For both mean and 95% UCL/max EPCs, non-detects whose detection limit was greater than the maximum detected concentration for an exposure area were removed from the dataset prior to calculation of the EPCs. For the purposes of calculating 95% UCL/max EPCs, the following rules were applied to datasets for tissue (based on species and tissue type), sediment, surface water, and the groundwater seep:

1. If a chemical was not detected in any sample for a given medium within the Study Area, it was assumed to not be present, so an EPC was not calculated for that chemical in that medium

2. If a chemical was detected at least once within the Study Area in samples for a given medium, the non-detect concentrations were used in the EPC

calculations in accordance with the methods used in the software ProUCL Version 4.00.02 (EPA 2007b). ProUCL software output for the 95% UCLs calculated in this BHHRA are provided in Attachment F4.

In risk characterization, some toxicity values are based on exposure to chemical mixtures and not to individual chemicals. The risks from these chemicals, which were identified in *Human Health Toxicity Values Interim Deliverable* (Kennedy/Jenks Consultants 2004a), were evaluated for the combined exposure to the chemicals and not on an individual chemical basis. For chemicals that were evaluated as mixtures in the BHHRA, the concentrations of the individual isomers or congeners that comprise the mixtures were summed to calculate the EPCs for the mixtures, as described in Attachment F2. The chemicals evaluated as mixtures are described in Attachment F2 as well, and include: PCBs, endosulfans, chlordanes, DDTs, DDDs, DDEs, and 2,3,7,8-TCDD TEQs.

3.4.1 Beach Sediment

Sediment data collected from human use areas during Round 1 and 2 were used to estimate the EPCs for beach sediment. There were no additional beach sediment data collected from human use areas for Round 3. Within the Study Area, EPCs were estimated for exposure areas based on the types of populations potentially exposed. Since potentially complete exposure pathways for sediment involve direct contact with beach sediments, only beach sediment data were used in estimating EPCs for direct exposure pathways.

One composite sample was collected from each beach area. Therefore, the results from the composite sample were used for both the 95% UCL/max and the mean EPCs for that beach. The process to estimate EPCs for each receptor population is described below.

3.4.1.1 Dockside Workers

Dockside workers could potentially be exposed to beach sediment in dockside worker use areas, which are shown in Map 2-1. Beach sediment data from these areas were used to estimate the EPCs for dockside workers. For dockside workers, the exposure area is considered to be the industrial site (i.e., facility within a property boundary) where the worker is employed. To estimate an EPC for each industrial site, beach sediment data from the composite sample collected from the beach associated with that industrial site were used. If the beach area extends across multiple industrial sites, the same EPC was used to evaluate exposure of dockside workers at each of the adjacent industrial sites. Beach sediment EPCs for exposures of dockside workers are presented in Table 3-2.

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3.4.1.2 Transients

Transients could potentially be exposed to beach sediment in transient use areas, which are shown in Map 2-1. Transients may move throughout the Study Area, while some may spend a majority of their time at only one of the identified areas. Therefore, EPCs for transients were estimated for each beach area within the transient use areas to represent a range of possibilities for transients residing in the Study Area. Beach sediment EPCs for exposures by transients are presented in Table 3-3.

3.4.1.3 Recreational Beach Users

Recreational beach users could potentially be exposed to beach sediment in recreational use areas, which are shown in Map 2-1. Beach sediment data from these areas were used to estimate the EPCs for recreational beach users. For recreational beach users, the exposure area is considered to be one river beach area, which represents a conservative assumption for the BHHRA because the beach user could be exposed to multiple recreational beach areas within and outside of the Study Area during the exposure time period. EPCs were estimated for individual beaches within the recreational beach use areas. Beach sediment EPCs for exposures by recreational beach users are presented in Table 3-3.

3.4.1.4 Fishers

Fishing could occur from beaches with unrestricted access, which are the potential transient and/or recreational use areas. Beach sediment data from these areas were used to estimate the EPCs for non-tribal and tribal fishers, as shown on Map 2-1. Fishers are likely to fish from multiple beach areas within and outside of the Study Area during the exposure time period. The exposure area for fishers was considered to be one individual beach in order to provide a range of risk estimates for individual beaches within the Study Area. EPCs were estimated for individual beaches within the recreational and transient use areas and are the same as the EPCs for transients and recreational beach users. Beach sediment EPCs for exposures by fishers are presented in Table 3-3.

3.4.2 In-Water Sediment

In-water sediment data of appropriate data quality collected within the Study Area were used to estimate EPCs for in-water sediment. Direct contact would only occur with near-shore surface sediment, so only surface sediment data (less than 30.5 cm in depth) collected outside of the navigation channel were used in estimating the EPCs.

If a contaminant was detected at least once in surface sediment within the Study Area, an EPC was calculated for that contaminant, and any non-detect concentrations were included in the EPC calculations in accordance with the ProUCL Version 4.00.02 guidance (EPA 2007b). In-water sediment EPCs were estimated for in-water workers, fishers, and divers and are presented in Table 3-4.

3.4.2.1 In-Water Workers

For in-water workers, exposure could occur anywhere within the Study Area that docks or pilings are being constructed or where other in-water activities are occurring (such as maintenance dredging of private slips or berths). While these activities would not necessarily be restricted to a given area, exposure would most likely be localized to in-water sediment adjacent to facilities where these activities occur. Most of these activities would be between the shore and the navigation channel. As a result, sediment samples in near-shore (i.e., excluding the central navigation channel) half-river mile segments along both sides of the river were used to develop in-water sediment EPCs. In addition to calculating EPCs for exposure within the Study Area, EPCs were also calculated for the downstream reach of the river from RM 1.0 – 1.9, the downtown reach of the river from RM 11.8 – 12.2, and for samples within Multnomah Channel, per an agreement with EPA.

In accordance with EPA guidance (1989), the 95% UCL was used for the 95% UCL/max EPC for in-water workers for exposure areas with at least 5 detected concentrations for a given analyte. For analytes with less than 5 detected concentrations, the maximum detected concentration for that exposure area was used as the 95% UCL/max EPC. Uncertainties associated with estimating EPCs for small datasets (i.e., less than 10 detected concentrations) and in using the maximum detected concentration as the EPC are discussed in Section 6. The arithmetic mean of detected concentrations was used for the mean EPC. The 95% UCLs were calculated for each dataset following EPA guidance (EPA 2002a and EPA 2007b). ProUCL version 4.00.02 (EPA 2007b) was used to test datasets for normal, lognormal, or gamma distributions and to calculate the 95% UCLs. Data were tested first for normality, then for gamma distributions, and finally for lognormal distributions, as recommended by ProUCL guidance (EPA 2007b). If the data did not exhibit a discernable distribution, a non-parametric approach (e.g., Chebyshev) was used to generate a UCL. The 95% UCLs were calculated using the method recommended by ProUCL guidance (EPA 2007b) for the data distribution, sample size, and skewness. In-water sediment EPCs for exposures by in-water workers are presented in Table 3-4.

3.4.2.2 Fishers

Fishers include adult non-tribal and tribal fishers. The fisher scenario is based on long-term exposure. For repeated exposures over an entire lifetime, direct contact with in-water sediment would occur over a very wide area. Even though exposure would occur over a wide area, in-water sediment EPCs for the fisher were derived on a half-mile segment on each side of the river, as was done for the in-water workers, as requested by EPA in its comments, dated February 24, 2005 on draft *Exposure Point Concentration Calculation Approach and Summary of Exposure Factors*. Deriving exposure areas based on a half-mile segment on each side of the river provides a range of possibilities for risk management and for risk communication to fishers making fishing location choices. In addition to calculating EPCs for exposure within the Study Area, EPCs were also calculated for the downstream reach of the river from RM 1.0 – 1.9, the downtown reach of the river from RM 11.8 – 12.2, and for samples within Multnomah Channel, per an agreement with EPA. Both the mean and 95% UCL/max EPCs were calculated as described for the in-water worker EPCs. In-water sediment EPCs for exposures to fishers are presented in Table 3-4.

3.4.2.3 Divers

Commercial divers could conduct diving activities anywhere within the Study Area, though exposure would most likely be to in-water sediment adjacent to facilities where commercial diving is required for purposes such as marine construction, underwater inspections, and routine operation and maintenance. It is assumed that all other diving done by a diver is done outside of the Study Area. Therefore, in-water sediment EPCs for the diver were derived for half-mile segments on each side of the river, as was done for the in-water workers, and as directed by EPA in the memorandum dated September 15, 2008 (EPA 2008c). In addition to calculating EPCs for exposure within the Study Area, EPCs were also calculated for the downstream reach of the river from RM 1.0 – 1.9, the downtown reach of the river from RM 11.8 – 12.2, and for samples within Multnomah Channel, per an agreement with EPA. Both the 95% UCL/max and mean EPCs were calculated as described for the in-water worker EPCs. In-water sediment EPCs for exposures to divers are presented in Table 3-4.

3.4.3 Surface Water

Surface water data of appropriate data quality collected within the Study Area were used to estimate EPCs. Both integrated and non-integrated water column surface water samples were collected within the Study Area and were used in estimating the surface water EPCs. The specific samples used to estimate EPCs for each receptor were dependent upon the exposures of that receptor to surface water within the Study Area. A summary of surface water samples used to calculate EPCs for each receptor is provided in Table 3-5. Surface water EPCs were estimated for transient, recreational beach user, diver, and hypothetical future domestic water user exposure scenarios.

3.4.3.1 Transients

Transient exposures to surface water could occur throughout the year at transient use areas within the Study Area. As a result, data from all seven of the completed seasonal sampling events were used in estimating the surface water EPCs for transients. Data from the four transect stations within the Study Area were used to estimate surface water EPCs for exposures at transient use areas throughout the Study Area. Results of near-bottom and near-surface horizontally integrated transect samples from the same sample location and sampling event were combined prior to calculation of EPCs, as were vertically integrated transect samples from the east, middle, and west portions of the river. Rules for combining transect samples are

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described in Attachment F2. Surface water samples were also collected at Willamette Cove, which is a quiescent transient use area that may not be adequately characterized by the transect samples. Year-round data from this surface water sample location were used to estimate surface water EPCs for exposures in Willamette Cove. Surface water EPCs for exposures by transients are presented in Table 3-6.

Given that transients can live along many parts of the river, EPCs were calculated for each transect, as well as for the combination of all four transects. In addition to calculating EPCs for exposure within the Study Area, EPCs were calculated for one transect station outside of the Study Area, at Multnomah Channel. For the 95% UCL/max EPC, the 95% UCL was used for the EPC for exposure areas with at least 5 detected concentrations for a given analyte. For analytes with less than 5 detected concentrations in a given exposure area, the maximum detected concentration was used as the EPC. Uncertainties associated with estimating EPCs for small datasets (i.e., less than 10 detected concentrations) and in using the maximum detected concentration as the EPC are discussed in Section 6. The 95% UCLs were calculated as described for in-water sediment. The arithmetic mean of the detected concentrations for each exposure area was used for the mean EPC.

3.4.3.2 Recreational Beach Users

Recreational beach user exposures to surface water could occur during summer months at recreational use areas within the Study Area. The only summer sampling event for recreational use areas occurred in July 2005. As a result, only data from the low-water sampling event in July 2005 were used in estimating the surface water EPCs for recreational beach users. The uncertainty associated with using data from only the low-water summer sampling event is discussed further in Section 6. Data collected from recreational beaches in July 2005 included three transect locations and three single-point locations (Cathedral Park, Willamette Cove, and Swan Island Lagoon). Data from the three transect stations were used to estimate surface water EPCs for exposures at non-quiescent recreational beach use areas throughout the Study Area, and data from single-point surface water samples were used to estimate EPCs for exposure at quiescent recreation beach areas. Because only one sample was collected from each quiescent area during low-water periods, the results for the single sample were used as both the 95% UCL/max EPC and the mean EPCs for each area. Only three transect samples were collected in July 2005 during the low-water period, so the maximum concentrations were used as the 95% UCL/max EPCs and the arithmetic mean of detected concentrations were used as the mean EPCs. Surface water EPCs for exposures by recreational beach users are presented in Table 3-7.

3.4.3.3 Divers

Diver exposures to surface water could occur throughout the year at all areas within the Study Area. Therefore, for divers, all of the surface water data collected in the Study Area, including both transect data and data collected from single point stations, were used to estimate EPCs. In addition to calculating EPCs for exposure within the

Study Area, EPCs were calculated for one transect station outside of the Study Area, at Multnomah Channel. Transect data were used to estimate EPCs for diver exposures as described for transient exposures (Section 3.4.3.1). Surface water data available as single point samples from Round 2 in several areas of the Study Area, and as near-bottom and near-surface samples from Round 3 sampling, were also used to estimate EPCs. For the Round 3 surface water samples collected as single point samples, the near-bottom and near-surface samples were combined for use in estimating EPCs, as described in Attachment F2. As with diver exposure to in-water sediment, diver exposure to surface water is expected to be in localized areas adjacent to facilities where commercial diving is required for purposes such as marine construction, underwater inspections, and routine operation and maintenance. Therefore, samples from single point stations were used to calculate EPCs for near-shore half-river mile segments along both sides of the river, consistent with the approach for in-water sediment EPCs and per direction from EPA. Surface water EPCs for exposures by divers are presented in Table 3-8.

3.4.3.4 Domestic Water User

The hypothetical use of untreated surface water as a domestic water source could occur within the Study Area throughout the year. As a result, data from all seven of the completed seasonal sampling events were used in estimating the surface water EPCs for the domestic water user. EPCs were determined for individual transect stations and for single point stations with vertically integrated samples. This dataset included samples from the four transect stations within the Study Area and single point vertically integrated samples from Cathedral Park, Willamette Cove, and Swan Island Lagoon. In addition, EPA required that data from near-bottom and near-surface surface water stations where both samples were collected be averaged and used in the domestic water dataset. Study Area-wide EPCs included all vertically integrated samples. Transect data were used to estimate EPCs for hypothetical domestic water use as described for transient exposures (Section 3.4.3.1). For single point stations, fewer than five samples were taken from each station, so the maximum detected concentration was used as the 95% UCL/max EPC and the mean of detected concentrations was used as the mean EPC. Surface water EPCs for hypothetical use of untreated surface water as a domestic water source are presented in Table 3-9.

3.4.4 Groundwater Seeps

Direct contact with groundwater would occur only within human use areas where groundwater comes to the surface (i.e., seeps) on the beach above the water line. Each groundwater seep where direct contact could occur represents an exposure area for groundwater. The only groundwater seep where direct contact could occur within the Study Area is within the potential transient use area located on the west side of the river at RM 7 (Map 2-5). Outfall 22B, which is a potential conduit of groundwater discharge and results in the water present on that beach, was sampled twice between 2002 and 2007 at times that did not involve stormwater influence. If a chemical was detected in only one of the two samples, that result was used as both the 95% UCL/max and mean EPCs for that contaminant. If a contaminant was detected in both samples, the maximum concentration was used as the 95% UCL/max EPC, and the arithmetic mean of the detected concentrations was used as the mean EPC. Groundwater seep EPCs are presented in Table 3-10.

3.4.5 Fish and Shellfish Tissue

Fish and shellfish tissue EPCs were derived from tissue sampling results of the LWG Round 1, Round 2, and Round 3 investigations and the ODHS study. Fish tissue EPCs are presented in Tables 3-11 through 3-21, and shellfish tissue EPCs are presented in Tables 3-22 though 3-25. The EPCs derived from Round 1 data were originally presented in *Round 1 Tissue Exposure Point Concentrations* (Kennedy/Jenks Consultants 2004b), which was approved by EPA. These EPCs were derived for fish species and crayfish that were evaluated for human consumption. Since Round 1, new data have been collected for clam, crayfish, smallmouth bass, and common carp. No new data have been collected since Round 1 for use in the calculation of brown bullhead and black crappie EPCs. The EPCs derived for adult salmon, adult lamprey, and adult sturgeon using the results of the ODHS study were originally presented in *Salmon, Lamprey, and Sturgeon Tissue Exposure Point Concentrations for Oregon Department of Human Services* (Kennedy/Jenks Consultants 2004c). These EPCs were derived for salmon whole body, fillet with skin, and fillet without skin composite samples, lamprey whole body composite samples, and sturgeon fillet without skin samples.

Crayfish and clams were collected and composited at each sampling location. EPCs were calculated for crayfish at individual locations, as well as for the entire Study Area per the Programmatic Work Plan. EPCs were calculated for clams for approximately one river mile on each side of the river, as well as for the entire Study Area, as required by EPA in its comments on the Round 2 Report. EPCs were also calculated for crayfish and clams collected between RM 1.0 and 1.9 and between RM 11.8 and 12.2, per an agreement with EPA. EPCs for clams were calculated for both depurated and undepurated samples.

Smallmouth bass were collected and composited over a river mile. EPCs were calculated for smallmouth bass at each river mile as well as for the entire Study Area per the Programmatic Work Plan. EPCs were calculated for both whole body and fillet samples.

Common carp, black crappie, and brown bullhead were collected and composited within river segments designated as fishing zones. For Round 1 data collection, there were two fishing zones that extended over three-mile segments: RM 3-6 and RM 6-9. For Round 3 data collection, which included additional common carp collection but not black crappie or brown bullhead, there were three fishing zones that extended over four-mile segments: RM 0-4, RM 4-8, and RM 8-12. EPCs for common carp, black crappie, and brown bullhead were calculated for each fishing zone in which they were sampled, as well as for the entire sampling area to represent Study Area-wide exposure. EPCs were calculated for both whole body and fillet samples.

Adult salmon were collected at the Clackamas fish hatchery, adult lamprey were collected at Willamette Falls, and sturgeon were collected at locations throughout the Study Area. EPCs were calculated for adult salmon, adult lamprey, and sturgeon using available data to be representative of the entire Study Area. EPCs were calculated for both whole body and fillet samples for adult salmon. Only whole body data were available for adult lamprey and only fillet data were available for sturgeon, so the EPCs for adult lamprey were calculated for whole body samples and the EPCs for sturgeon were calculated for fillet samples.

In calculating the EPCs for fish and shellfish, if only one sample was collected within a given exposure area, that result was used as both the 95% UCL/max and mean EPC for that contaminant. If more than one sample was collected, either the 95% UCLs or maximum concentrations were used as the 95% UCL/max EPCs, depending on the number of reported concentrations. If detected concentrations for at least five samples were available, the 95% UCLs were calculated as described for in-water sediment. If less than five detected concentrations were available, the maximum detected concentration was used as the 95% UCL/ max EPC. EPCs for Study Area-wide exposure were calculated from the Study Area-wide data set. Uncertainties associated with estimating EPCs for small datasets (i.e., less than 10 detected concentrations) and in using the maximum detected concentration as the EPC are discussed in Section 6. The arithmetic mean of detected concentrations was used as the mean EPC, assuming that all non-detects were one-half the detection limit.

EPCs for multi-species fish tissue consumption scenarios were calculated using a weighted average of site-wide EPCs for each COPC, based on the percent of each species consumed in the diet.

3.5 PROCESS TO CALCULATE INTAKES

EPA (1989) defines exposure as “the contact with a chemical or physical agent” and defines the magnitude of exposure as “the amount of an agent available at human exchange boundaries (i.e., the lungs, gut, and skin) during a specified time period.” Exposure assessments are designed to determine the degree of contact a person has with a chemical. Thus, estimating human exposure to a chemical requires information regarding the concentration of the chemical in the environmental media (sediment, water, tissue) with which a person will come into contact and the extent of contact the person will have with the media.

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Chemical-specific intake or dose was quantified in this BHHRA by estimating the chronic daily intake (CDI) for noncarcinogens, or the lifetime average daily intake (LADI) for carcinogens. CDI and LADI, expressed in terms of the mass of substance taken into the body per unit body weight per unit time (mg/kg/day), were calculated using equations based on exposure parameters that represent the duration of exposure, frequency of exposure, and other factors that affect overall chemical dose. Consistent with EPA guidance (1989), exposure assessments were based on the RME expected to occur under both current and future land use conditions, as well as hypothetical future conditions. Exposure assessments using CT values, which are more representative of average exposures, were also conducted. Rationale and/or references for each of the RME and CT values for exposure pathways that were quantitatively assessed for each exposure scenario for different populations are presented in exposure factor Tables 3-26 through 3-30 and discussed in the following sections.

Intakes were quantified using standard exposure equations (EPA 1989). These equations take the general form:

$$\text{CDI or LADI} = \frac{EPC \times IR \times EF \times ED}{BW \times AT}$$

Where:

- CDI = Chronic daily intake
- LADI = Lifetime average daily intake
- EPC = Exposure point concentration
- IR = Intake rate
- EF = Exposure frequency
- ED = Exposure duration
- BW = Body weight
- AT = Averaging time.

The detailed intake equations, as well as the specific exposure parameters and associated units, are dependent on the exposure scenario evaluated; please see Tables 3-26 to 3-30 for additional details. For exposure areas outside of the Study Area, the same intake equations and exposure parameters were used as used for exposure areas within the Study Area.

3.5.1 Population-Specific Assumptions

Assumptions about each population evaluated in this BHHRA were used to select exposure parameters to calculate the pathway-specific chemical intakes. Currently, site-specific values are not available for all populations and pathways. Therefore,

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default values were used where site-specific values are not available. Where default values are not available, best professional judgment based on knowledge of human uses of the Study Area, or requirements from EPA, were used.

Exposure parameters that were used in this BHHRA to calculate the CDIs and LADIs for most receptors were previously included in *Exposure Point Concentration Calculation Approach and Summary of Exposure Factors* (Kennedy/Jenks Consultants 2006), which was approved by EPA. For divers, the exposure parameters were provided by EPA in a directive dated September 15, 2008. For hypothetical future domestic water use, EPA default exposure parameters for residential drinking water were used as required by EPA in its comments on the Round 2 Report. The exposure parameters are discussed below and presented in Tables 3-26 to 3-30. These values represent potential exposures for application at appropriate areas and/or areas agreed upon with EPA and its partners within the Study Area. Except where specifically noted, the exposure assumptions used in the BHHRA were applied uniformly to all of the Study Area, and may or may not be applicable at specific locations within the Study Area depending on factors not specifically addressed in the BHHRA (e.g., accessibility, habitat). The actual exposure at a given location may be less than that assumed for the population and Study Area as a whole due to location-specific conditions.

3.5.1.1 Dockside Worker

For the dockside worker, exposure to beach sediment is the only exposure pathway determined to be potentially complete and evaluated in this BHHRA. Industrial land use was assumed only for portions of the Study Area that are zoned for industrial use and with river-front areas that include natural river beach or bank areas. Activities at Portland Harbor industrial sites do not occur frequently in these areas, which are the only areas where direct exposure to beach sediment might occur. It is unlikely that workers are in direct contact with beach sediment through typical industrial activities on a daily basis.

Because it is unlikely that significant beach sediment exposure would occur for a dockside worker on a daily basis, exposure assumptions for the dockside worker were developed using EPA default exposure values for an industrial worker for most parameters except for exposure frequency. For exposure frequency, it was assumed that a worker would contact sediment one day per week while working at the industrial site, rather than the EPA default value or 5 days per week. Therefore, the default exposure frequency of 250 days per year, which represents 5 days per week for 50 weeks, was changed to 50 days per year (i.e., 1 day per week for 50 weeks) for RME. Table 3-26 summarizes RME and CT exposure values for the dockside worker and the reference or rationale for each value.

3.5.1.2 In-Water Worker

For the in-water worker, exposure to in-water surface sediment is the only exposure pathway determined to be potentially complete and evaluated in this BHHRA. In-water workers could contact in-water sediment while performing specific activities such as replacement of fender piles or maintenance dredging. Exposure factors for in-water sediment were developed for Terminal 4 based on in-depth interviews with several workers who conduct or oversee activities that could result in contact with in-water sediment. According to the Army Corps of Engineers (Siipola 2004), the Port of Portland conducts the most frequent dredging within the Study Area, so the exposure factors for workers at Terminal 4 are considered protective of in-water workers throughout the Study Area for potential in-water sediment exposures. For the RME scenario, in-water workers are assumed to contact in-water sediment for 10 years during 25 years of employment at a given facility with 10 days of sediment contact per year. For the CT scenario, in-water workers are assumed to contact in-water sediment for 4 years during 9 years of employment at a given facility with 10 days of sediment contact per year. The in-water worker exposure factor intake rates for in-water sediment are the same as the dockside worker for beach sediment, which in turn are the same as default exposure factors for soil for an industrial worker. Table 3-27 summarizes RME and CT exposure values for the in-water worker and the reference or rationale for each value.

3.5.1.3 Transients

Transient land use is assumed only for portions of the Study Area with riverfront access and that are not also active industrial sites. Transients may be exposed to beach sediment, surface water, and groundwater seeps while utilizing river beaches within transient use areas. EPA does not have recommended exposure parameters for transient scenarios, so the exposure frequency and duration for transients are based on best professional judgment. However, by definition, transient exposures are assumed to occur over a short duration of time. At the request of EPA, it was assumed that transients might remain at a single beach for up to two years for the RME scenario. For intake rates for transients, EPA required that the soil ingestion rate and soil adherence factor used for beach sediment be increased above those EPA default values recommended for residential soil exposures and that residential tap water ingestion rates be used for surface water. A higher soil ingestion rate (200 mg/day instead of 100 mg/day) and soil adherence factor (0.3 mg/cm^2 instead of 0.07 mg/cm^2) were used as it is expected that transients living on a beach would have more contact with beach sediment than a residential adult might have with residential soil and dust. Transients may have limited access to washing facilities and could therefore more frequently transfer sediments from hand to mouth while eating, smoking, etc. Tables 3-26 and 3-28 summarize RME and CT exposure values for the transient scenario for beach sediment and water (surface water and groundwater seeps respectively), and the reference or rationale for each value.

3.5.1.4 Recreational Beach User

Recreational beach use is assumed only for portions of the Study Area where recreational exposures are reasonably likely to occur. Recreational beach users may have direct contact with beach sediment within river beach areas and with surface water while swimming or during other water activities. EPA does not have recommended exposure parameters for recreational beach use scenarios, so the exposure frequency and duration for recreational beach users are based on best professional judgment. Beach use was assumed to be more frequent (5 days per week) in the summer with less frequent use in the spring/fall (1 day per week) and even less use in the winter (1 day per month). The temperature of river water would limit swimming activities during much of the year. Therefore, exposure to surface water was only evaluated for the summer months when swimming might occur (2 days per week). For beach sediment intake, the recommended default values for residential soil were generally used but the adherence factor for children was more than 10 times greater than the value for residential soil. For surface water intake, the recommended default values for swimming scenarios were used. The recreational beach user includes both adults and children. Tables 3-26 and 3-28 summarize RME and CT exposure values for beach sediment and surface water, respectively, for adult and child recreational beach users. A reference or rationale is included for each value.

3.5.1.5 Non-Tribal Fishers

Exposure assessments for the non-tribal fisher scenarios evaluated potential exposure to COPCs through direct contact with beach and in-water sediment and through consumption of fish and shellfish. Direct contact with beach sediment only occurs in river beach areas where fishing activities occur. Non-tribal fishers could theoretically contact in-water sediment on anchors, hooks, or crayfish pots while fishing from boats or piers at the Study Area. For fish and shellfish consumption, it is assumed that exposure could occur throughout the Study Area and is continuous year-round as fishers may catch fish at the Study Area and then freeze them for later use.

This BHHRA evaluated both a non-tribal fisher exposure scenario and a tribal fisher exposure scenario, which is discussed in Section 3.5.1.6. The non-tribal fisher scenario included two different fishing frequencies for sediment exposures, three different ingestion rates for fish consumption exposures, and two different ingestion rates for shellfish consumption exposures. Non-tribal fish consumption was evaluated for both adults and children while sediment exposure was evaluated for adults only, with the assumption that fishing is done primarily by adults but both adults and children may consume the fish that are caught.

3.5.1.5.1 Beach Sediment Exposure

Beach sediment exposure would only occur for fishers during bank fishing at natural river beach areas within the Study Area. EPA specified the exposure frequencies and durations for the fishers used in this BHHRA. High-frequency fishers were assumed

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to fish from the same beach area three days per week for the entire year (156 days/year) for 30 years for the RME. Low-frequency fishers were assumed to fish from the same beach area for two days per week for the entire year (104 days/year)for 30 years for the RME. Exposure assumptions for beach sediment contact for fishers are presented in Table 3-26.

3.5.1.5.2 In-Water Sediment Exposure

At the request of EPA, the exposure frequencies and durations for beach sediment for each fisher scenario were assumed to represent the fishing activity at the Study Area regardless of whether that fishing occurs from a beach or a boat. A factor of 25 percent was used to represent the percent of time spent fishing in a single area within the Study Area.

Based on the exposure scenarios for in-water sediment (i.e., contact with sediment on anchors, hooks, or crayfish pots), the extent of contact with in-water sediment is expected to be less than what would occur with residential soil. Ingestion rates for soil are based on exposure to soil during yard work and to indoor dust (EPA 1997a). These ingestion rates are not applicable to the in-water sediment exposure scenarios; however, incidental ingestion rates are not available for sediment. It is assumed that the incidental ingestion rate for in-water sediment is 50% of the ingestion rate for residential incidental soil scenarios. For dermal contact, hands and forearms are the only body parts that could be exposed to in-water sediment on a regular basis (i.e., on a year-round basis). It is assumed that the entire surface area of both hands and forearms would be exposed to in-water sediment. The adherence and absorption factors are assumed to be the same as those for beach sediment. Exposure assumptions for in-water sediment contact for fishers are presented in Table 3-27.

3.5.1.5.3 Fish Consumption

The fish consumption scenario included three different fish ingestion rates, as well as single species and multiple species diets of resident fish species. Study Area-specific fish consumption information is not available for the fish consumption scenarios. Therefore, to evaluate the potential range in consumption patterns that may exist, three ingestion rates were used to calculate intakes for adults and three were used for children. EPA specified the ingestion rates used in this BHHRA. For adults, the fish ingestion rates were 17.5 grams per day (g/day), 73 g/day, and 142 g/day. These rates correspond to approximately 2 meals per month, 10 meals per month, and 19 meals per month, based on an 8-ounce serving size, every month of the year, consisting exclusively of fish caught within the Study Area. It should be noted that the current fish consumption advisory, based on PCBs, for the LWR recommends that children and expectant mothers do not eat resident fish from the Portland Harbor, and that healthy adults eat no more than one 8-ounce meal per month of resident fish from the Portland Harbor (ODHS 2007). However, it is unclear to what extent this advisory is followed by people who consume fish from the Study Area.

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Two of these rates, 17.5 g/day and 142 g/day, represent the 90th and 99th percentile ingestion rates for diets including uncooked freshwater and estuarine finfish and shellfish by individuals (consumers and non-consumers) of age 18 and over in the United States (EPA 2002b). The 90th and 99th percentile ingestion rates for uncooked freshwater and estuarine finfish and shellfish for consumers-only are 200 g/day and 506 g/day, respectively (EPA 2002b). Because these rates are from a national dietary study, they may not be representative of site-specific consumption patterns. Relative to the ingestion rate of 142 g/day, an adult consuming fish and shellfish tissue at a rate of 200 g/day would need approximately 70 percent of their total fish and shellfish diet to be fish caught within the Study Area, and an adult consuming fish and shellfish tissue at a rate of 506 g/day would need approximately 28 percent of their total fish and shellfish diet to be fish caught within the Study Area. If a different proportion of fish were caught within the Study Area versus outside of the Study Area, exposure to chemicals within the Study Area would change accordingly. Additional uncertainties associated with these ingestion rates are discussed in Section 6. The other ingestion rate used in this BHHRA, 73 g/day, is from a creel study conducted in the Columbia Slough and is the 95 percent upper confidence limit on the average for ingestion of fish where 75 percent of the mass of the total fish is consumed (Adolfson 1996). While this study may be more representative of consumption patterns for the Study Area, the study was limited in scope and the reported ingestion rates were estimated based on numerous assumptions. These ingestion rates were used for both the mean and 95% UCL/max risk calculations.

Limited information is available about fish consumption by children. The child scenario evaluated in this BHHRA is for 0 to 6-year olds. The national dietary study does not include consumption information for this age range. However, this age range was evaluated in the CRITFC Fish Consumption Survey (CRITFC 1994). In that survey, the ratio of the child 95th percentile ingestion to the adult 95th percentile ingestion rate, which is the comparison specified by EPA, was 0.42. This ratio was applied to the three adult ingestion rates to estimate the child ingestion rates. The corresponding rates that were used for children were 7 g/day, 31 g/day, and 60 g/day. Exposure assumptions for fish consumption are presented in Table 3-29.

For the fish consumption scenarios, risks were evaluated separately for consumption of each individual target resident fish species (smallmouth bass, black crappie, brown bullhead, and common carp) assuming only one species was consumed in each scenario. For these individual species scenarios the ingestion rates for the entire diet (regardless of species) were used with concentration data on each individual resident species (for both whole body and fillet tissue). EPCs were calculated for fishing zones (common carp, black crappie and brown bullhead) and mile reach (smallmouth bass) as well as for the entire Study Area, as described in Section 3.4.5. In addition to the individual species diet, a multiple species diet was also evaluated by using the fish ingestion rates for the scenarios with the concentration data of all resident species (for whole body and fillet tissue) for the Study Area (i.e., a multiple species diet assuming that each of the 4 fish target species represents 1/4 of a person's diet). The following

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scenarios were evaluated for each of the above ingestion rates using both the 95% UCL/max and mean EPCs described in Section 3.4.5 for both whole body and fillet samples (because these scenarios were not classified as CT or RME):

	River Mile	Fishing Zone	Entire Study Area
Smallmouth bass	X		X
Black crappie		X	X
Common carp		X	X
Brown bullhead		X	X
Multiple species			X

The uncertainties associated with the fish consumption scenarios are discussed in Section 6 of this BHHRA.

3.5.1.5.4 Shellfish Consumption

Site-specific shellfish consumption information is not available. For shellfish, only adult consumption was evaluated. It should be noted that there is currently a fish consumption advisory for wood-treating chemicals in a portion of the Study Area recommending that crayfish not be eaten (ODHS 2007). Ingestion rates of 3.3 g/day and 18 g/day were used to calculate intakes from shellfish consumption. These values represent the 50th percentile (3.3 g/day) and 95th percentile (18 g/day) ingestion rates for shellfish consumption from freshwater and estuarine systems for individuals of age 18 and older in the United States (EPA 2002b). These ingestion rates were used with 95% UCL/max and mean EPCs for crayfish and clams described in Section 3.4.5 (because these scenarios were not classified as CT or RME). Exposure assumptions for shellfish consumption are presented in Table 3-29. The uncertainties associated with the shellfish consumption scenario are discussed in Section 6 of this BHHRA.

3.5.1.6 Tribal Fishers

For thousands of years, the Willamette River has been an important ceremonial and subsistence fishery (i.e., salmon, lamprey, and sturgeon) for Native American tribes of the region. Native Americans continue to rely on the Willamette River. For example, tribal members conduct a ceremonial spring Chinook harvest and continue to harvest lamprey at Willamette Falls annually.

3.5.1.6.1 Beach Sediment Exposure

Beach sediment exposure would only occur for tribal fishers during bank fishing at natural river beach areas within the Study Area. EPA provided the exposure frequencies and durations for the tribal fishers used in this BHHRA. Tribal fishers were assumed to fish from the same beach area five days per week for the entire year

(260 days/year) for an entire lifetime (70 years) for the RME. Although it is not known how much sediment contact actually occurs during fishing activities, default intake values for residential soil were used. Exposure assumptions for beach sediment contact for tribal fishers are presented in Table 3-26.

3.5.1.6.2 In-Water Sediment Exposure

At the request of EPA, the exposure frequencies and durations for beach sediment were assumed to represent the fishing frequency at the Study Area regardless of whether that fishing occurs from a beach or a boat. Therefore, a factor of 25 percent was used to represent the percent of time exposed to in-water sediment while fishing in a single area within the Study Area.

Contact with sediment on anchors or hooks represents the most likely exposure route for contact with in-water sediments for tribal fishers. Ingestion rates for soil are based on exposure to soil during yard work and to indoor dust (EPA 1997a). These ingestion rates are not applicable to the in-water sediment exposure scenarios; however, incidental ingestion rates are not available for sediment. It is assumed that the incidental ingestion rate for in-water sediment is 50% of the ingestion rate for residential soil scenarios. For dermal contact, hands and forearms are the only body parts that could be exposed to in-water sediment on a regular basis. It is assumed that the entire surface area of both hands and forearms would be exposed to in-water sediment. The adherence and absorption factors are assumed to be the same as those for beach sediment. Exposure assumptions for in-water sediment contact for tribal fishers are presented in Table 3-27.

3.5.1.6.3 Tribal Fish Consumption

A multi-species diet that includes resident fish as well as salmonids, lamprey, and sturgeon was evaluated for tribal fish consumption. While site-specific fish consumption information is not available for the tribal fish consumption scenario, a fish consumption survey was conducted on the reservations of four of the participating Tribes (CRITFC 1994). The 95th percentile fish ingestion rate for consumers only from the CRITFC Fish Consumption Survey, which is 175 g/day, was used to calculate intakes for adult tribal fish consumers. On October 23, 2008, the Oregon Environmental Quality Commission approved a fish consumption rate of 175 g/day, referenced from the CRITFC (1994) survey, as the basis for ODEQ to revise state water quality standards. To date, the water quality standards have not yet been revised using the fish consumption rate of 175 g/day. This rate corresponds to approximately 23 meals per month every month of the year of fish caught exclusively within the Study Area. The CRITFC survey reported that none of the respondents fished the Willamette River for resident fish and approximately 4 percent fished the Willamette River for anadromous fish. The 95th percentile fish ingestion rate of 73 g/day for children from the CRITFC Fish Consumption Survey was used for child tribal fish consumers. Exposure assumptions for tribal fish consumption are presented in Table 3-29.

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A multi-species diet was evaluated using the fish consumption data from the CRITFC Fish Consumption Survey (CRITFC 1994) with concentration data from the target resident species as well as from sturgeon, salmon and lamprey caught as a part of the ODHS sampling effort. The fish consumption information from the CRITFC survey was used to determine the ingestion rate for each fish species, as shown below:

Species	Grams per day[a]	Percent of diet
Salmon	67	38.4
Lamprey	12.3	7.0
Sturgeon	8.6	4.9
Smelt	12.5	7.2
Whitefish	23.2	13.3
Trout	25.1	14.3
Walleye	9.9	5.7
Northern Pikeminnow	3.7	2.1
Sucker	7.3	4.2
Shad	5.2	3.0
Total Ingestion Rate	175	100

(a) Grams per day are based on the weighted mean data in Table 18 of the CRITFC Fish Consumption survey.

For adult tribal consumers, the ingestion rates for anadromous salmonids (67 g/day), lamprey (12.3 g/day), and sturgeon (8.6 g/day) were used with the respective 95% UCL/max and mean EPCs for those species to calculate intakes. For the remaining species, each of the 95% UCL/max and mean EPCs calculated for the entire Study Area for smallmouth bass, black crappie, common carp, and brown bullhead were used with an ingestion rate of 21.7 g/day (i.e., the ingestion rate for the sum of the species that are not anadromous salmonid, sturgeon or lamprey, 86.9 g/day, divided by 4). The combined intakes from anadromous salmonids and lamprey, from sturgeon, and from the remaining fish species in the above table were used to estimate risks from fish consumption. The intakes for child tribal fish consumers were calculated using the same dietary percentages as the adult tribal fish consumers, but with a total ingestion rate of 73 g/day.

Adult salmon, adult lamprey, and sturgeon have life histories such that significant exposure to contaminants can occur outside of the Study Area. The uncertainties in estimating the proportion of contaminants in sturgeon, salmon and lamprey and associated risks that result from contaminants at the Study Area are discussed in Section 6.

3.5.1.7 Divers

Divers could contact in-water sediment and surface water while performing specific commercial diving activities such as marine construction, underwater inspections, and routine operation and maintenance. As previously discussed in Section 3.3.2.2, exposure factors for divers were provided as a directive from EPA in a memorandum dated September 15, 2008 (EPA 2008c). The EPA developed two exposure scenarios to differentiate exposures by divers wearing wet suits from exposures by divers wearing dry suits. For both the RME wet suit and dry suit scenarios, divers were assumed to contact in-water sediment and surface water for 25 years of employment with 5 days of exposure frequency per year. For the CT scenario, which only includes wet suit divers, divers were assumed to contact in-water sediment and surface water for 9 years of employment with 2 days of exposure frequency per year. The event duration for exposure to sediment and surface water for both diver scenarios was 4 hours per diver for the RME and 2 hours per diver for the CT exposure. Whole body exposure was assumed for the skin surface area for the wet suit diver scenario (RME and CT), so that the surface area for the exposed skin was 18,510 square centimeters (cm^2). For the skin surface area for the dry suit diver scenario (RME only), it was assumed that only the head and neck would be exposed, equivalent to a skin surface area of approximately 2,510 cm^2. The sediment dermal adherence factors for both diver exposure scenarios were the same as those for the in-water fishers. The sediment ingestion rates for both diver exposure scenarios were the same as the in-water fishers (RME of 50 mg/day and CT of 25 mg/day), though the sediment contact frequency term was not used for divers. The water ingestion rates for both diver exposure scenarios were the same as those used for the recreational beach swimmers. Tables 3-27 and 3-28 summarize exposure assumptions for the wet suit and dry suit divers for in-water sediment and surface water, respectively, and the reference or rationale for each value.

3.5.1.8 Domestic Water Users

Surface water within the Study Area is not currently used as a domestic water source and there are no known plans to use it as a domestic water source in the future. However, the designated beneficial uses of the Willamette River include domestic water supply, assuming adequate pretreatment of the water prior to consumption. EPA specified that the BHHRA evaluate use of untreated river water as a domestic water supply. This scenario is considered hypothetical because pretreatment of surface water for domestic use would be required under current state laws.

To evaluate this hypothetical scenario, default EPA intake parameters for residential drinking water were used for both adult and child exposures. Exposure duration was assumed to be 350 days per year for both adult and child residents. The water ingestion rates used for both adult and child were those recommended for residential ingestion of drinking water (EPA 1989). The event duration and skin surface area were the recommended values for adults and children while showering or bathing (EPA 2004). Event frequency was once per day for both adult and child. None of the chemicals selected as COPCs for the domestic water use scenario were volatile, and therefore the inhalation exposure route was not evaluated for this scenario.

Table 3-30 summarizes the exposure assumptions for the hypothetical domestic water use by adult and child residents, and the reference or rationale for each value.

3.5.2 Chemical-Specific Exposure Factors and Assumptions

In calculating chemical intakes, certain assumptions were made that were specific to a given chemical or class of chemicals. These chemical-specific assumptions had an effect on both EPCs and intake calculations, and are described below.

3.5.2.1 Exposure Point Concentrations

Calculations of EPCs are described in Section 3.4 and the resulting EPC values are presented in Tables 3-2 through 3-25. Inorganic arsenic EPCs were estimated from total arsenic concentrations, as described below. In addition, PCBs were summed in several different ways, as described below.

Arsenic was analyzed as total arsenic, but the toxicity values for arsenic are only relevant for inorganic arsenic, which is most significant for tissue. In previous fish tissue studies in the lower Columbia and Willamette Rivers, the percent of inorganic arsenic relative to total arsenic ranged from 0.1% to 26.6% with an average percent inorganic arsenic of 5.3% in the resident fish samples from the Willamette River (Tetra Tech 1995, EVS 2000). Shellfish may have a higher percentage of inorganic arsenic, as measured in studies on the Lower Duwamish River. The Columbia River Basin Fish Contaminant Survey (EPA 2002c) concluded that a “value of 10% is expected to result in a health protective estimate of the potential health effects from arsenic in fish.” Therefore, it was assumed that 10% of total arsenic in tissue was in the form of inorganic arsenic for purposes of this BHHRA. The total arsenic concentrations were multiplied by 10% and the resulting value was used in calculating the tissue EPCs for arsenic. Uncertainties associated with the assumption that 10% of the total arsenic is in the inorganic form in fish and shellfish are discussed further in Section 6.

PCBs were analyzed as Aroclors and congeners in tissue. For PCBs analyzed as Aroclors, the summed concentration of individual Aroclors was used in calculating the EPCs, as described in Attachment F2. For PCBs analyzed as congeners, EPCs were calculated using both the total PCB value (sum of individual congeners) and an adjusted total PCB value. The adjusted total PCB value was calculated by subtracting the concentration of the coplanar PCB congeners from the total PCB concentration. This was done because the coplanar PCB congeners were evaluated separately (as

TCDD toxic equivalents [TEQs]) for cancer risks. Further explanation of how PCB congeners were summed is provided in Attachment F2.

3.5.2.2 Dermal Absorption Factors for Sediment

EPA's Supplemental Guidance for Dermal Risk Assessment (2004) provides chemical-specific values for dermal absorption from contaminated soil. These chemical-specific dermal absorption factors were used in the intake equations for dermal contact with sediment and are presented in Table 3-31. However, as noted in EPA guidance (2004), the amount of chemical absorbed from sediment may differ from that absorbed from soil due to differences in the relative importance of numerous chemical, physical, and biological factors. A default dermal absorption value was used for semi-volatile organic compounds (SVOCs) that do not have chemical-specific values. Per EPA guidance (2004), only those compounds or classes of compounds for which dermal absorption factors exist were evaluated quantitatively for the dermal contact exposure pathway. For compounds without dermal absorption factors, which are certain metals and perchlorate for the sediment COPCs, dermal intake was assumed to be zero. The uncertainties associated with chemicals lacking dermal absorption factors are discussed in Section 6.

3.5.2.3 Dermal Absorption Factors for Surface Water and Groundwater Seeps

One of the parameters in the intake equations for dermal contact with surface water or groundwater seeps is the absorbed dose per event (DA_{event}). This parameter was derived per EPA guidance (2004) using chemical-specific factors, which are presented in Table 3-32 for scenarios involving direct contact with surface water or groundwater seeps and in Table 3-33 for the hypothetical domestic water use scenario. The chemical-specific factors used in the calculation of DA_{event} were obtained from Appendix B (Screening Tables and Reference Values for the Water Pathway) of EPA's Supplemental Guidance for Dermal Risk Assessment (2004). The uncertainties associated with calculating DA_{event} for chemicals with factors outside of the predictive domain are discussed in Section 6.

3.5.2.4 Oral Bioavailability Factors for Sediment

Consistent with EPA guidance (1989), the chemical intake equations calculate the amount of chemical at the human exchange boundaries, not the amount of chemical available for absorption. Therefore, the estimated intakes calculated in this BHHRA are not the same as the absorbed dose of a chemical. However, the toxicity of an ingested chemical depends on the degree to which the chemical is absorbed from the gastrointestinal tract into the body. Per EPA guidance (1989, 2007c), if the exposure medium in the risk assessment differs from the exposure medium assumed by the toxicity value, an adjustment for bioavailability may be appropriate. For purposes of this BHHRA, oral bioavailability factors were not used to adjust the estimated exposures from COPCs in sediment. The uncertainties associated with not considering bioavailability in this BHHRA are discussed in Section 6.

4.0 TOXICITY ASSESSMENT

Toxicity values provide a quantitative estimate of the potential for adverse effects resulting from exposure to a chemical. Toxicity values are used in risk assessment to quantify the likelihood of adverse effects occurring at different levels of exposure to a chemical.

Toxicity values were identified for the COPCs that were selected in Section 2.4. The cancer and noncancer toxicity values are shown in Tables 4-1 and 4-2, respectively. The following sections discuss the toxicity values and describe how they were selected.

4.1 CARCINOGENIC TOXICITY VALUES

Slope factors (SFs) are used to quantify the dose-response potency of potential carcinogens. SFs are derived from either human epidemiological or animal studies by applying a mathematical model to the dataset to extrapolate from the high doses in studies to the lower exposure levels expected for human contact in the environment (EPA 1989). The SF is an upper-bound estimate or maximum likelihood estimate of the probability of a response over a lifetime.

Slope factors are available for oral and inhalation exposure pathways. The inhalation exposure pathway was not quantitatively evaluated in this BHHRA, so inhalation unit risk values were not selected as toxicity values. Dermal SFs were derived from the oral SFs, as described in Section 4.7. The oral and dermal cancer slope factors are presented in Table 4-1. In accordance with EPA (2005a) guidance, the weight of evidence for carcinogenicity for each COPC is also presented in Table 4-1.

4.2 NONCARCINOGENIC TOXICITY VALUES

A chemical that exhibits adverse effects other than cancer or mutation-based developmental effects is believed to have a threshold (i.e., a dose below which no adverse effect is expected to occur). Reference doses (RfDs) are typically used as toxicity values for chemicals with noncarcinogenic effects. A chronic RfD is defined as a daily dose to which humans, including sensitive subpopulations, may be exposed throughout their lifetimes without adverse health effects.

Reference doses are available for oral and inhalation exposure pathways. The inhalation exposure pathway was not quantitatively evaluated in this BHHRA, so inhalation reference concentrations were not selected as toxicity values. Dermal reference doses were derived from oral reference doses, as described in Section 4.7. Reference doses for oral and dermal exposure pathways are presented in Table 4-2.

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4.3 SOURCES OF TOXICITY VALUES

The following hierarchy of sources of toxicity values is currently recommended for use at Superfund sites (EPA 2003b):

- Tier 1 – EPA's Integrated Risk Information System (IRIS) database (EPA 2010b) is the preferred source of information because it normally represents the official EPA scientific position regarding the toxicity of the chemicals based on the data available at the time of the review. IRIS contains RfDs and SFs that have gone through a peer review and EPA consensus review.
- Tier 2 - EPA's Provisional Peer Reviewed Toxicity Values (PPRTVs) are toxicity values derived for use in the Superfund Program when such values are not available in IRIS. PPRTVs are derived after a review of the relevant scientific literature using the methods, sources of data and guidance for value derivation used by the EPA IRIS Program. The PPRTV database includes RfDs and SFs that have undergone internal and external peer review. The Office of Research and Development/National Center for Environmental Assessment/Superfund Health Risk Technical Support Center (STSC) develops PPRTVs on a chemical-specific basis when requested by EPA's Superfund program.
- Tier 3 - Tier 3 includes additional EPA and non-EPA sources of toxicity information. Priority is given to those sources of information that are the most current, the basis for which is transparent and publicly available, and which have been peer reviewed. Tier 3 sources may include, but need not be limited to, the following sources:
  - The California Environmental Protection Agency (Cal EPA) Toxicity Criteria Database (Cal EPA 2008) includes toxicity values that have been peer reviewed.
  - The ATSDR Minimal Risk Levels are similar to RfDs and are peer reviewed.
  - Health Effects Assessment Summary Table (HEAST) toxicity values are currently under review by the STSC to derive PPRTVs. The toxicity values remaining in HEAST are considered Tier 3 values.

Toxicity values were retrieved from the most current version of the Regional Screening Levels for Chemical Contaminants at Superfund Sites (EPA 2010a, values updated November 2010. These values follow the above hierarchy, and present toxicity values from IRIS for both noncarcinogenic and carcinogenic effects were selected when available. If a toxicity value is not available from IRIS, toxicity values from the PPRTV database are presented, if available. In the absence of toxicity values from either IRIS or the PPRTV database, toxicity values from Tier 3 sources are presented, if available. The sources of the cancer or noncancer toxicity value are indicated in Tables 4-1 and 4-2. The dates shown in Tables 4-1 and 4-2 indicate the date of release of the Regional Screening Levels for Chemical Contaminants at Superfund Site table (EPA 2010a).

For trichloroethylene, EPA provided the draft toxicity value equal to the geometric mid-point of the slope factor range (EPA 2001b) to use as the oral cancer slope factor. Recommendations were not provided for evaluating oral exposures for noncancer endpoints for trichloroethylene.

4.4 CHEMICALS WITH SURROGATE TOXICITY VALUES

For some chemicals, if a toxicity value was not available from the above hierarchy, a structurally similar chemical was identified as a surrogate. The reference dose or slope factor for the surrogate chemical was selected as the toxicity value and the surrogate chemical was indicated in Tables 4-1 and 4-2. The following chemicals have toxicity values from surrogate chemicals:

- Butyltin ion. Toxicity values were identified from the recommended hierarchy for dibutyltin compounds and tributyltin compounds. Toxicity of alkyltin compounds depends on the number of alkyl side-chains, with monoalkyl tin being the least and trialkyl tin the most toxic (National Library of Medicine [NLM] 2004). Therefore, dibutyltin is thought to be more similar to butyltin than tributyltin in toxicity, and is more toxic than butyltin. As a health protective approach, the toxicity value for dibutyltin compounds was selected as a surrogate for butyltin ion.

- Dibutyltin ion. The available toxicity value for dibutyltin is for dibutyltin compounds. However, the BHHRA sample results were for dibutyltin ion. The dibutyltin compounds toxicity value was selected as a surrogate for dibutyltin ion.

- Tributyltin ion. The available toxicity value for tributyltin is for tributyltin compounds. However, the BHHRA sample results were for tributyltin ion. The tributyltin compounds toxicity value was selected as a surrogate for tributyltin ion.

- Acenaphthylene. IRIS classifies acenaphthylene as a category D carcinogen (not classifiable as to human carcinogenicity), and therefore, is considered a noncarcinogenic polycyclic aromatic hydrocarbon (PAH). Acenaphthene is the noncarcinogenic PAH most similar in structure and carbon number to acenaphthylene. Therefore, the acenaphthene toxicity value was selected as a surrogate for acenaphthylene.

- Benzo(e)pyrene. IRIS classifies benzo(e)pyrene as a category D carcinogen (not classifiable as to human carcinogenicity), and therefore, is considered a noncarcinogenic PAH. Of the noncarcinogenic PAHs most similar in structure and carbon number to benzo(e)pyrene, pyrene has the lowest toxicity value and is therefore, considered the most toxic. As a health protective approach, the pyrene toxicity value was selected as a surrogate for benzo(e)pyrene.

- Benzo(g,h,i)perylene. IRIS classifies benzo(g,h,i)perylene as a category D carcinogen (not classifiable as to human carcinogenicity), and therefore, is considered a noncarcinogenic PAH. Of the noncarcinogenic PAHs most similar in structure and carbon number to benzo(g,h,i)perylene, pyrene has the lowest toxicity value and is therefore, considered the most toxic. As a health protective approach, the pyrene toxicity value was selected as a surrogate for benzo(g,h,i)perylene.

- Dibenzothiophene. Toxicity values were not available for dibenzothiophene. The chemical with the most similar structure with available toxicity values is fluorene. The toxicity value for fluorene was selected as a surrogate for dibenzothiophene.

- Dibenzofuran. Toxicity values were not available for dibenzofuran. The chemical with the most similar structure with available toxicity values is fluorene. The toxicity value for fluorene was selected as a surrogate for dibenzofuran.

- Di-n-octyl phthalate. Toxicity values were not available for di-n-octyl phthalate. The chemical with the most similar structure with available toxicity values is dibutyl phthalate. The toxicity value for dibutyl phthalate was selected as a surrogate for di-n-octyl phthalate.

- Perylene. IRIS classifies perylene as a category D carcinogen (not classifiable as to human carcinogenicity), and therefore, is considered a noncarcinogenic PAH. Of the noncarcinogenic PAHs similar in structure and carbon number to perylene, pyrene has the lowest toxicity value and is therefore, considered the most toxic. As a health protective approach, the pyrene toxicity value was selected as a surrogate for perylene.

- Phenanthrene. IRIS classifies phenanthrene as a category D carcinogen (not classifiable as to human carcinogenicity), and therefore, is considered a noncarcinogenic PAH. Of the noncarcinogenic PAHs similar in structure and carbon number to phenanthrene, pyrene has the lowest toxicity value and is therefore, considered the most toxic. As a health protective approach, the pyrene toxicity value was selected as a surrogate for phenanthrene.

- Retene. Retene is a PAH classified by IRIS as a category D carcinogen (not classifiable as to human carcinogenicity). Of the noncarcinogenic PAHs similar in structure and carbon number to retene, pyrene has the lowest toxicity value and is therefore, considered the most toxic. As a health protective approach, the pyrene toxicity value was selected as a surrogate for retene.

- Endrin aldehyde. Endrin aldehyde can occur as an impurity of endrin or as a degradation product (ATSDR 1996). The toxicity value for endrin was selected as a surrogate for endrin aldehyde.

- Endrin ketone. Endrin ketone can occur as an impurity of endrin or as a degradation product (ATSDR 1996). The toxicity value for endrin was selected as a surrogate for endrin ketone.

- 4-Nitrophenol. IRIS has toxicity values for 2-methylphenol and 4-methylphenol, but not 4-nitrophenol. The toxicity value for 4-methylphenol was selected as a surrogate for 4-nitrophenol.

4.5 CHEMICALS WITHOUT TOXICITY VALUES

Only two COPCs, titanium and delta-hexachlorocyclohexane (delta-HCH), did not have available SF and RfD toxicity values or appropriate surrogate chemicals from sources included in the hierarchy. Titanium is a naturally occurring element and has been characterized as having extremely low toxicity (Friberg et al. 1986). An STSC review concluded that the other hexachlorocyclohexane isomers could not be used as surrogates for delta-HCH due to differences in toxicity (EPA 2002d). In this BHHRA, the potential risks from titanium and delta-HCH are discussed qualitatively in the uncertainty assessment in Section 6.

SFs and RfDs were not identified for lead because lead was evaluated through comparison with benchmark concentrations that are based on blood lead levels. Benchmark concentrations for child exposure scenarios were predicted by the Integrated Exposure Uptake Biokinetic (IEUBK) model. Benchmark concentrations for adult exposure scenarios were predicted by the Adult Lead Methodology (ALM). Uncertainties associated with using these benchmark concentrations are discussed in Section 6.4.4.

4.6 TOXICITY VALUES FOR CHEMICAL MIXTURES

Some toxicity values are based on exposure to chemical mixtures and not to individual chemicals. As a result, the risks were evaluated for the combined exposure to the chemicals and not on an individual chemical basis. The chemicals that were evaluated for toxicity as mixtures are indicated in Tables 4-1 and 4-2, and are discussed below.

- Chlordane. The chlordane toxicity values were derived for technical chlordane, which is composed of a mixture of chlordane isomers. The chlordane isomers analyzed in Round 1, Round 2, and Round 3 samples were alpha-chlordane, trans-chlordane, cis-nonachlor, trans-nonachlor, and oxychlordane. These isomers were summed in a total chlordane concentration. The SF and RfD for technical chlordane were used to evaluate total chlordane.

- DDD, DDE, and DDT. Technical DDT includes 2,4'-DDT and 4,4'-DDT, as well as 2,4'-DDE, 4,4'-DDE, 2,4'-DDD, and 4,4'-DDD. DDD, DDE, and DDT have separate SFs included in IRIS. While the SFs were derived for the 4,4' isomers, the SFs were used to evaluate the sum of the 2,4' and 4,4' isomers because toxicity values are not available for the 2,4' isomers. The DDT RfD was derived for a mixture of the 2,4' and 4,4' isomers and was used to evaluate the noncancer endpoint of DDT. An RfD is not available for the DDD or DDE isomers, so the DDT RfD was selected as a surrogate toxicity value and was used to evaluate the noncancer endpoint of DDD and DDE.

- Endosulfan. The toxicity value (RfD) for endosulfan was derived from studies using technical endosulfan, which includes alpha-endosulfan, beta-endosulfan and endosulfan sulfate. These compounds were summed in a total endosulfan concentration. The RfD for technical endosulfan was used to evaluate total endosulfan.

- PCBs. The PCB cancer SF was derived for PCB mixtures based on administered doses of Aroclors to rats. The cancer SF was applied to total PCBs, measured either as congeners or Aroclors. The PCB SF was applied to the total PCB congener concentration after subtracting the total dioxin-like PCB congener concentration. Dioxin-like PCB congener concentrations were evaluated separately using the 2,3,7,8-tetrachlorodibenzo-p-dioxin (2,3,7,8-TCDD) SF, as described below for dioxins and furans. This approach may double-count a portion of the toxicity of the dioxin-like PCBs, as discussed in Section 6.3.6. The Aroclor 1254 RfD was used to evaluate the noncancer endpoint for total PCBs, measured either as total unadjusted congeners or Aroclors.

- Dioxins and furans. Toxic Equivalency Factors (TEFs) from the World Health Organization (WHO) (Van den Berg 2006) were used to evaluate carcinogenic effects of dioxin and furan congeners and dioxin-like PCB congeners (see Table 4-3). Concentrations of congeners are multiplied by their TEFs to estimate the toxicity of these congeners relative to 2,3,7,8-TCDD; the resulting concentrations are then summed into a total 2,3,7,8-TCDD TEQ. The 2,3,7,8-TCDD SF was used to evaluate the cancer endpoint of the TEQ for dioxin and furan congeners and for dioxin-like PCB congeners. The 2,3,7,8-TCDD RfD was used with the same approach to evaluate the noncancer endpoint of the TEQ for dioxin and furan congeners and for dioxin-like PCB congeners.

- Carcinogenic PAHs. Carcinogenic PAHs can be evaluated for toxicity based on their potency equivalency factor (PEF), which estimates toxicity relative to benzo(a)pyrene (EPA 1993). The toxicity values for individual PAHs shown in Table 4-1 incorporate their respective PEFs. Risk from both individual and total carcinogenic PAHs was assessed in this BHHRA.

4.7 DERMAL TOXICITY ASSESSMENT

Most toxicity values are based on oral, not dermal, exposures. For oral exposures, toxicity values are often expressed as the amount of substance administered, whereas dermal exposures are expressed as absorbed dose. EPA has developed a simplified method for oral-to-dermal extrapolations (EPA 2004). These extrapolations involve an adjustment to the oral toxicity value based on the GI absorption factor of the specific chemical in the same administration vehicle (e.g., corn oil, food) as used in the critical toxicity study to derive an estimated dermal dose.

As recommended by EPA guidance (EPA 2004), an adjustment to the oral toxicity factor to account for the estimated absorbed dose was applied in this BHHRA when the following conditions are met:

- The toxicity value derived from the critical study is based on an administered dose (e.g., through diet or by gavage)

- A scientifically defensible database demonstrates the GI absorption of the chemical is less than 50% in a medium similar to the one used in the critical study.

If both of these conditions are met, the oral toxicity factor was adjusted to reflect the absorbed dose in this BHHRA. For carcinogenic effects, the oral slope factor was divided by the GI absorption factor to estimate the dermal slope factor. Hexavalent chromium was the only COPC for which the oral slope factor was adjusted to reflect the absorbed dose. For noncarcinogenic effects, the oral reference dose was multiplied by the GI absorption factor to estimate the dermal reference dose. The COPCs for which the oral reference dose was adjusted to reflect the absorbed dose are the metals: antimony, barium, cadmium, trivalent chromium, hexavalent chromium, manganese, mercury, silver, and vanadium.

If both conditions for adjustment are not met, the oral toxicity value was used as a surrogate for the dermal toxicity value in the BHHRA. Dermal toxicity factors are presented in Tables 4-1 and 4-2.

5.0 RISK CHARACTERIZATION

Risk characterization integrates the information from the exposure assessment and toxicity assessment, using a combination of qualitative and quantitative information. With this information, risk characterization estimates the potential health risk, based on the dose of a chemical, that a person may receive under certain site-specific exposure conditions and based on the toxicity of that chemical. Consistent with DEQ (DEQ 2000a) and EPA guidance (EPA 1989), noncarcinogenic and carcinogenic effects were evaluated separately. To characterize potential noncarcinogenic effects, comparisons were made between projected intakes of substances and toxicity values (Section 5.1.1). To characterize potential carcinogenic effects, projected intakes and chemical-specific, dose-response data were used to estimate the probability that an individual will develop cancer over a lifetime of exposure (Section 5.1.2).

5.1 RISK CHARACTERIZATION ESTIMATES

This section describes how noncancer hazards and cancer risks were estimated in this BHHRA.

5.1.1 Noncancer Hazard Estimates

The potential for adverse effects resulting from exposure to chemicals with noncarcinogenic effects is generally addressed by comparing the CDI or absorbed dose for a specific COPC to its RfD. This comparison was made by calculating the ratio of the estimated CDI (or absorbed dose) to the corresponding RfD to yield a hazard quotient (HQ):

$$HQ = \frac{CDI}{RfD}$$

HQs for individual chemicals were summed to yield cumulative hazard indices (HIs) that provide an estimate of total hazard. Per EPA guidance (1989), HQs should only be summed for chemicals with common toxicological endpoints. Toxicological endpoints for COPCs are summarized in Table 5-1. Endpoint-specific HIs (e.g., neurological or immune system effects) were calculated for each exposure area in this BHHRA where the cumulative HI was greater than 1. The Columbia River Fish Contaminant Study performed a similar analysis for fish tissue collected from the Columbia River Basin (EPA 2002c). Toxicity endpoints were retrieved from EPA's Integrated Risk Information System (EPA 2010b), and may differ from the endpoints used in CRITFC due to updates in the IRIS database since the CRITFC study.

Estimated HIs were compared to a target HI of 1, below which remedial action at a Superfund site is generally not warranted (EPA 1991a).

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5.1.2 Cancer Risk Estimates

Potential cancer risks were assessed by multiplying the estimated LADI or absorbed dose of a carcinogen by its SF. This calculated risk is expressed as the probability of an individual developing cancer over a lifetime as a result of exposure to the potential carcinogen, and is a health protective estimate of the incremental probability of excess individual lifetime cancer risk.

$$Risk = LADI * SF$$

Initially, potential cancer risks were estimated separately for each chemical. The separate potential cancer risk estimates were summed across chemicals for each exposure area to obtain the cumulative excess lifetime cancer risk for the exposure scenario.

Cancer risks were calculated using this same linear model, even though risk estimates for some scenarios exceed 1 x 10^{-2}, in which case, EPA guidance (EPA 1989) states that risks should be calculated using an exponential model. Where cancer risks exceeded 1 x 10^{-2}, the exponential model was used. Estimated total cancer risks were compared to 1 x 10^{-4}, 1 x 10^{-5}, and 1 x 10^{-6} cancer risk targets based upon the following language in EPA's National Contingency Plan (NCP): "For known or suspected carcinogens, acceptable exposure levels are generally concentration levels that represent an excess upper bound lifetime cancer risk to an individual of between 1 x 10^{-4} and 1 x 10^{-6}." The point of departure for cancer risks is 1 x 10^{-6}.

5.1.3 Combined Adult/Child Scenarios

Cancer risks were calculated separately for adult and child receptors for the recreational beach user and fisher scenarios. To assess risks to individuals exposed as both a child and an adult, cancer risks were also calculated for a combined adult and child receptor for the recreational beach user and fisher scenarios. The combined adult and child receptor was based on EPA guidance (1991b, 2010a), in which 6 years of exposure is assumed to occur as a child and 24 years of exposure is assumed to occur as an adult for a total of 30 years for the non-tribal fisher scenario and the RME exposure duration for the beach user scenario. For the tribal fisher scenario, the combined adult and child scenario assumed 6 years of exposure as a child and 64 years of exposure as an adult. For the CT exposure duration for the beach user scenario, the combined adult and child scenario assumed 6 years of exposure as a child and 9 years of exposure as an adult.

For chemicals not acting by a mutagenic mode of action (i.e., all chemicals evaluated in this BHHRA other than carcinogenic PAHs), the cancer risks for the combined adult and child receptor were calculated by adding the cancer risks for the adult to the cancer risks for the child. For the non-tribal fisher and the RME beach sediment exposure scenarios, the adult cancer risk was multiplied by a factor of 24/30 to account for the 24 years of exposure as an adult in the combined scenario versus the

30 years used in the adult only scenario and then added to the child cancer risk. For the tribal fisher scenario, the adult cancer risk was multiplied by a factor of 64/70 to account for the 64 years of exposure as an adult in the combined scenario versus the 70 years used in the adult only scenario and then added to the child cancer risk.

For chemicals acting by a mutagenic mode of action (i.e., carcinogenic PAHs), the cancer risks were calculated for the combined adult and child receptor by incorporating EPA's guidance (2005b) on early life exposures to carcinogens. Specifically, age dependent adjustment factors (ADAFs) were used to account for the increased carcinogenic potency during early life exposures. For ages 0 to 2 years, an ADAF of 10 was used. For ages 2 to 6 years and 6 to 16 years, an ADAF of 3 was used. For ages over 16 years, an ADAF of 1 was used. The ADAFs were incorporated into the risk calculations through the use of age adjusted factors. The exposure factors used for the ages 0 to 2 and 2 to 6 years were the same as the child receptor and the exposure factors used for the ages 6 to 16 years and over 16 years were the same as the adult receptor.

The cancer risk estimates for the combined adult and child receptor are presented in the beach sediment and fish consumption risk characterization results below.

5.1.4 Infant Consumption of Human Milk

For bioaccumulative chemicals, exposure to the mother can lead to the presence of those chemicals in human milk, which can pose a risk to breastfeeding infants. Per agreement with EPA and DEQ, risks to infants through the consumption of human milk were included for all receptors where PCBs, dioxins, and/or DDX were identified as COPCs. Risks were assessed in accordance with DEQ guidance (2010).

To assess risks to infants, infant risk adjustment factors (IRAFs) were applied to the mother's risk. For cancer risks, the combined adult and child risks were used for the mother cancer risk for receptors where both adult and child exposures could occur. For receptors where only adult exposure was evaluated, the adult cancer risk was used for the mother cancer risk. For noncancer hazards, the adult hazard quotient was used for the mother hazard quotient.

The IRAFs used to assess risks were from DEQ guidance (2010). Specifically, IRAFs of 1 were used for PCB, PCB TEQ, and dioxin TEQ cancer risks. An IRAF of 0.007 was used for DDX cancer risks. IRAFs of 2 were used for PCB TEQ, dioxin TEQ, and DDX noncancer hazards. An IRAF of 25 was used for PCB noncancer hazards.

The risks to infants through consumption of human milk are presented in the risk characterization results below.

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5.1.5 Cumulative Risk Estimates

Noncancer HQs and cancer risks were calculated for all individual contaminants for which EPCs were available, as described above. In some cases, contaminants were analyzed by different methods, so there were multiple EPCs for that contaminant. In calculating the cumulative risks, only the risk associated with the EPC for one method was included in the sum to avoid double-counting the risks from a given contaminant.

For example, total PCBs were analyzed both as congeners and as Aroclors. In sediment, the Aroclor dataset was larger, so the risk from total PCBs as Aroclors was included in the cumulative risk estimate for sediment. For tissue, the congener analysis provides better detection limits. Therefore, the risk from total PCBs as congeners was included in the cumulative risk estimate for tissue, if congener data were available. If congener data were not available for tissue, the risk from total PCBs as Aroclors was used in estimating the cumulative risk for tissue.

In surface water and most of the groundwater seep samples, metals were analyzed as both total and dissolved. Because total concentrations are typically higher, the EPCs for total metals were included in the cumulative risk estimates as a conservative approach.

The individual risks from the EPCs for all of the analytical methods are presented in the risk characterization result tables (Tables 5-2 through 5-98). The tables also indicate which results were included in the cumulative risks when multiple EPCs were available for a given chemical.

5.2 RISK CHARACTERIZATION RESULTS

This section presents the results of the risk characterization for each of the scenarios described in Section 3. Uncertainties associated with the assumptions in each exposure scenario are discussed in detail in Section 6. Risks from exposures to PBDEs in in-water sediment and tissue were assessed separately, and are presented in Attachment F3. If actual exposures for each scenario were less than the exposures assumed in the risk calculations, the estimated risks would also decrease correspondingly.

5.2.1 Beach Sediment Risk Characterization Results

Potential risks from exposure to beach sediment through incidental ingestion and dermal absorption were estimated for the dockside workers, transients, recreational beach users, fishers and tribal fishers. There were multiple uncertainties associated with the direct exposure to beach sediment scenarios such as the spatial scale of the individual beaches and the exposure parameters, which are further described in the following sections. Beaches with cumulative cancer risks greater than 1×10^{-6} and 1×10^{-5} are summarized by exposure point and receptor in Maps 5-1-1 and 5-1-2.

There were no beach areas associated with cancer risk levels greater than 1 x 10^{-4} or HIs greater than 1.

5.2.1.1 Dockside Worker

Risks for the dockside worker were estimated separately for each beach designated as a potential dockside worker use area, which are shown in Map 2-1. The results of the risk evaluation for dockside worker exposure to beach sediment are presented in Tables 5-2 through 5-3.

The dockside worker RME scenario for beach sediment results in exceedances of a cumulative cancer risk level of 1 x 10^{-6} at beaches 06B025 (9 x 10^{-5} risk) and B004 (2 x 10^{-6} risk). There are no exposure areas that result in an exceedance of 1 x 10^{-4} cancer risk for the dockside worker RME scenario. The maximum cumulative cancer risk for an individual exposure area occurs at 06B025 and is primarily due to incidental ingestion of beach sediment containing benzo(a)pyrene. In addition to benzo(a)pyrene, other chemicals contributing to a calculated individual cancer risk greater than 1 x 10^{-6} for at least one exposure area include: benzo(a)anthracene, benzo(b)flouranthene, dibenzo(a,h)anthracene, and indeno(1,2,3-cd)pyrene. The HIs for the dockside worker RME scenario do not exceed 1.

The dockside worker CT scenario for beach sediment results in one exceedance of 1 x 10^{-6} cumulative cancer risk (at beach 06B025, 6 x 10^{-6} risk), which is primarily due to the incidental ingestion of sediment containing benzo(a)pyrene. There are no exposure areas that result in an exceedance of 1 x 10^{-4} cancer risk for the dockside worker CT beach sediment scenario. The dockside worker CT scenario results in no exceedances of a HI of 1. Figures 5-1 shows risks to the dockside worker from exposure to beach sediment per beach, and shows the relative contribution of individual chemicals to total risk.

5.2.1.2 Transients

Risks for the transients were estimated separately for each beach designated as a potential transient use area, which are shown in Map 2-1. The results of the risk evaluation for transient exposure to beach sediment are presented in Tables 5-4 through 5-5.

The transient RME scenario for beach sediment results in no exceedances of 1 x 10^{-6} cancer risk and no exceedances of a HI of 1. The transient CT scenario for beach sediment results in no exceedances of 1 x 10^{-6} cancer risk and no exceedances of a HI of 1.

5.2.1.3 Recreational Beach Users

Risks for the recreational beach users were estimated separately for each beach designated as a potential recreational use area, which are shown in Map 2-1. Cancer risks and noncancer hazards were evaluated for both adult and child recreational beach users. In addition, carcinogenic risks were calculated for a combined child and adult scenario. The results of the risk evaluation for recreational beach user exposure to beach sediment are presented in Tables 5-6 through 5-11.

5.2.1.3.1 Adult Recreational Beach Users

The adult recreational beach user RME scenario for beach sediment results in cumulative cancer risk exceedances of 1 x 10^{-6} at the following beaches: 04B024 (risk is 3 x 10^{-6}), 06B030 (risk is 4 x 10^{-6}), B003 (risk is 3 x 10^{-6}), and B005 (risk is 2 x 10^{-6}). There are no exceedances of 1 x 10^{-4} cancer risk for the adult recreational beach user RME scenario. The maximum cumulative cancer risk from RME occurs at Beach 06B030 and is primarily due to incidental ingestion of beach sediment containing arsenic. The adult recreational beach user RME scenario for beach sediment resulted in no HIs greater than 1. Figures 5-2 and 5-3 show the relative risk contribution of individual COPCs for each beach, as well as total risk by river mile for adult recreational beach user exposure to beach sediment.

Arsenic is a naturally occurring metal. The concentration for arsenic in soil recognized by DEQ to represent background levels in Oregon is 7 milligrams per kilogram (mg/kg) (DEQ 2007). At this background concentration, the calculated risk from arsenic would exceed 1 x 10^{-6} for the adult recreational beach user RME scenario. When a background concentration of 7 mg/kg is subtracted from detected concentrations of arsenic in beach sediment, resulting cumulative risks for the adult recreational beach user RME scenario exceed 10^{-6} at beaches 04B024 and B003. Beaches with risk exceedances of 1 x 10^{-6} excluding risks from background arsenic are shown for all exposure scenarios for beach sediment in Maps 5-2-1 and 5-2-2. In addition to risks from exposure to arsenic in beach sediment, risks from exposure to total cPAHs in beach sediment exceed 1 x 10^{-6} at two beach locations: 04B024 (2 x 10^{-6}) and B003 (2 x 10^{-6}) At each of these beaches, benzo(a)pyrene is the cPAH with the highest contribution to total risks from cPAHs.

The adult recreational beach user CT scenario for beach sediment results in no exceedances of 1 x 10^{-6} cumulative cancer risk and no exceedances of an HI of 1.

5.2.1.3.2 Child Recreational Beach Users

The child recreational beach user RME scenario for beach sediment results in cumulative risk exceedances of 1 x 10^{-6} at all 15 of the exposure areas. There are no exceedances of 1 x 10^{-4} cancer risk for the child recreational beach user RME scenario. The maximum cumulative cancer risk from RME occurs at beaches B003, and 04B024 (4 x 10^{-5}) and is primarily due to dermal absorption of soil containing arsenic and benzo(a)pyrene. The child recreational beach user RME scenario resulted in no HIs greater than 1.

The cumulative risk exceedances are due in part to arsenic, which is naturally occurring. At the DEQ background soil concentration of 7 mg/kg, the calculated risk from arsenic would exceed 1 x 10^{-6} for the child recreational beach user RME scenario. When a background arsenic concentration of 7 mg/kg is subtracted from detected arsenic concentrations in beach sediment from potential human use areas, resulting cumulative risks for the child recreational beach user RME scenario exceed 1 x 10^{-6} at five beaches, as shown in Map 5-2-1. These exceedances are due to exposure to arsenic at one beach, and exposure to benzo(a) pyrene or total cPAHs at the other four. Cancer risks above 1 x 10^{-6} from exposures to cPAHs in beach sediment range from 2 x 10^{-8} to 4 x 10^{-5}, due primarily to contributions from benzo(a)pyrene. Figures 5-4 and 5-5 show the relative risk contribution of individual COPCs for each beach, as well as total risk by river mile for child recreational beach user exposure to beach sediment.

The child recreational beach user CT scenario for beach sediment results in an exceedance of 1 x 10^{-6} cumulative cancer risk at two beaches (risk of 2 x 10^{-6} at 04B024 and B003). There are no exceedances of an HI of 1.

5.2.1.3.3 Combined Child/Adult Recreational Beach Users

Cancer risks were calculated for the combined child and adult recreational beach users to incorporate early life exposures in accordance with EPA (2005b) and DEQ (2010) guidance. Cumulative risks per exposure area for RME scenarios ranged from 2 x 10^{-6} to 5 x 10^{-5}. For the CT scenarios, risks ranged from 2 x 10^{-7} to 2 x 10^{-6}. The highest risk was at Beach 04B024, primarily due to exposures to benzo(a)pyrene in beach sediment.

5.2.1.4 Tribal Fishers

Risks for the tribal fishers were estimated separately for each beach designated as a potential transient or recreational use area, which are shown in Map 2-1. The results of the risk evaluation for tribal fisher exposure to beach sediment are presented in Tables 5-12 through 5-13.

The tribal fisher RME scenario for beach sediment results in exceedances of 1 x 10^{-6} cumulative cancer risk at 18 of 18 exposure areas. There are no exceedances of 1 x 10^{-4} cancer risk for the tribal fisher RME scenario. The maximum cumulative cancer risk occurs at beaches 06B030, B003 and 04B024 (2 x 10^{-5}) and is primarily due to incidental ingestion of sediment containing arsenic or benzo(a)pyrene. The tribal fisher RME scenario for beach sediment resulted in no HIs greater than 1. Figures 5-6 and 5-7 show the relative risk contribution of individual COPCs for each beach, as well as total risk by river mile for tribal fisher exposure to beach sediment.

The tribal fisher CT scenario for beach sediment results in exceedances of 1 x 10^{-6} cumulative cancer risk at one of the 18 exposure areas (beach 06B030) primarily due to incidental ingestion of sediment containing arsenic. There are no exceedances of 1 x 10^{-4} cancer risk or HI of 1 for the tribal fisher CT scenario.

The cumulative risk exceedances of 1 x 10^{-6} are primarily due to arsenic, which is naturally occurring. At the DEQ background soil concentration of 7 mg/kg, the

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calculated risk from arsenic would exceed 1 x 10^{-6} for the tribal fisher RME scenarios. When a background arsenic concentration of 7 mg/kg is subtracted from detected arsenic concentrations in beach sediment from potential human use areas, resulting cumulative risks for the tribal fisher RME scenario exceed 1 x 10^{-6} at eight beaches, due primarily to exposure to benzo(a)pyrene and total cPAHs, as shown in Map 5-2-1. Risks from exposure to cPAHs in sediment at these eight beaches range from 2 x 10^{-6} to 1 x 10^{-5}. Excluding background arsenic concentrations, exposure to beach sediment results in risks exceeding 1 x 10^{-6} from exposure to arsenic at one beach location. The maximum cumulative risk to tribal fishers from potential exposure to beach sediment excluding background contribution from arsenic is 1 x 10^{-5}, which occurs at beaches 04B024 and B003.

5.2.1.5 Fishers

Risks for the high- and low- frequency fishers were estimated separately for each beach designated as a potential transient or recreational use area, which are shown in Map 2-1. The results of the risk evaluation for high-frequency fisher exposure to beach sediment are presented in Tables 5-14 through 5-15. The results of the risk evaluation for low-frequency fisher exposure to beach sediment are presented in Tables 5-16 through 5-17.

5.2.1.5.1 High-Frequency Fishers

The high-frequency fisher RME scenario for beach sediment results in exceedances of 1 x 10^{-6} cumulative cancer risk at 9 of 18 exposure areas (see Table 5-14). There are no exceedances of 1 x 10^{-4} cancer risk for the high-frequency fisher RME scenario. The maximum cumulative cancer risk occurs at beaches 04B024 and 06B030 (6 x 10^{-6}) and is primarily due to incidental ingestion of sediment containing arsenic. In addition to arsenic, benzo(a)pyrene is the only other individual analyte resulting in a cancer risk greater than 1 x 10^{-6} at some exposure areas. The high-frequency fisher RME scenario for beach sediment resulted in no HIs greater than 1.

The cumulative risk exceedances of 1 x 10^{-6} are primarily due to arsenic, which is naturally occurring. At the DEQ background soil concentration of 7 mg/kg, the calculated risk from arsenic would exceed 1 x 10^{-6} for the high-frequency fisher RME scenarios. When a background arsenic concentration of 7 mg/kg is subtracted from detected arsenic concentrations in beach sediment from potential human use areas, resulting cumulative risks for the high-frequency fisher RME scenario exceed 1 x 10^{-6} at three beaches, as shown in Map 5-2-1. The maximum cumulative risk to high-frequency fishers from potential exposure to beach sediment excluding background contribution from arsenic is 3 x10^{-6}, which occurs at beaches 04B024 and B003.

The high-frequency fisher CT scenario for beach sediment results in no exceedances of 1 x 10^{-6} cumulative cancer risk and no exceedances of an HI of 1.

5.2.1.5.2 Low-Frequency Fishers

The low-frequency fisher RME scenario for beach sediment results in exceedances of 1 x 10^{-6} cumulative cancer risk at six of 18 exposure areas (see Table 5-16). There are no exceedances of 1 x 10^{-4} cancer risk for the low-frequency fisher RME scenario. The maximum cumulative cancer risk occurs at beaches 06B030 and 04B024 (4 x 10^{-6}), and is primarily due to incidental ingestion of sediment containing arsenic. Besides arsenic, there are no individual analytes resulting in a cancer risk greater than 1 x 10^{-6}. The low-frequency fisher RME scenario for beach sediment resulted in no HIs greater than 1.

The cumulative risk exceedances of 1 x 10^{-6} are primarily due to arsenic, which is naturally occurring. When a background arsenic concentration of 7 mg/kg is subtracted from detected arsenic concentrations in beach sediment from potential human use areas, resulting cumulative risks for the low-frequency fisher RME scenario exceed 1 x 10^{-6} at three beaches, as shown in Map 5-2-1. The RME cumulative risk to low-frequency fishers from potential exposure to beach sediment, excluding background contributions from arsenic, is 2 x10^{-6} at all three of these beaches.

The low-frequency fisher CT scenario for beach sediment results in no exceedances of 1 x 10^{-6} cumulative cancer risk and no exceedances of an HI of 1.

5.2.1.6 Breastfeeding Infants of Adults Exposed to Beach Sediment

Risks and hazards to breastfeeding infants from exposure to bioaccumulative compounds in human milk were assessed for scenarios resulting in bioaccumulative compounds as COPCs. In the case of the beach sediment exposure scenarios, only the dockside worker exposures include bioaccumulative compounds as COPCs. The assessment of risks to infants entails applying a compound-specific infant risk adjustment factor (IRAF) to risks and hazards to the adult mother, in accordance with DEQ guidance (2010). Cumulative cancer risks to an infant consuming human milk from a dockside worker range from 5 x 10^{-10} to 1 x 10^{-6} across both CT and RME scenarios. Noncancer hazards range from 6 x 10^{-3} to 1 across both CT and RME scenarios. Risks to breastfeeding infants of dockside workers exposed to beach sediment are shown in Tables 5-18 through 5-19.

5.2.1.7 Summary of Beach Sediment Risk Characterization

Direct contact with beach sediment resulted in cumulative cancer risks ranging from 8 x 10^{-9} to 9 x 10^{-5}. Cumulative HIs for direct exposure to beach sediment were at or below the EPA target HI of 1 for all exposure scenarios. The highest cumulative cancer risks at industrial use beaches were for the dockside worker scenario, and the highest cumulative cancer risks at residential use beaches were for the tribal fisher scenario. Two chemicals resulted in a cancer risk greater than 1 x 10^{-6} for at least one of the scenarios evaluated for direct contact with beach sediment: arsenic and PAHs. Arsenic occurs both naturally and as a result of environmental releases. A summary of risks from beach sediment per beach is shown in Maps 5-1-1 and 5-1-2, and risks after subtracting an assumed background arsenic concentration of 7 mg/kg from the EPCs are shown in Maps 5-2-1 and 5-2-2. Table 5-20 provides a summary of risks from exposure to beach sediment, per receptor and exposure area.

5.2.2 In-Water Sediment Risk Characterization Results

Potential risks from exposure to in-water sediment through incidental ingestion and dermal absorption were estimated for the in-water workers, fishers, tribal fishers, and divers. There were multiple uncertainties associated with the direct exposure to in-water sediment scenarios such as the spatial scale of the exposure areas and the exposure parameters, which are further described in the following sections. Risks were estimated separately for in-water sediment for each of the ½-mile river segment exposure areas (east (E) and west (W)) and for Study Area-wide exposure. In addition to calculating risks from in-water sediment exposure within the Study Area (which includes exposure areas from RM 1.9 to RM 11.8, including Swan Island Lagoon), risks from in-water sediment exposure were calculated for three river segments outside of the Study Area: the downstream reach (RM 1.0-1.9), the downtown river segment (RM 11.8 – 12.2), and Multnomah Channel. The exposure area from RM 11.5 to 12.0 encompasses samples from both inside and outside of the Study Area. However, Study Area-wide risks were calculated only for samples within the Study Area. Cumulative risk exceedances for in-water sediment scenarios are summarized by exposure area in Maps 5-3-1 through 5-3-2. In addition, risks from exposures to PBDEs in in-water sediment were evaluated separately and are presented in Attachment F3, following the general methodology discussed in this BHHRA.

5.2.2.1 In-Water Worker

The results of the risk evaluation for in-water worker exposure to in-water sediment are presented in Tables 5-21 through 5-22.

The in-water worker RME scenario for in-water sediment results in cumulative cancer risk greater than 1×10^{-6} at RM segments 4.5E, 6W, and 7W. There are no exceedances of 1×10^{-4} cancer risk for the in-water worker RME scenario. The maximum cumulative cancer risk for an individual exposure area occurs at RM 7W (2×10^{-5}) and is primarily due to incidental ingestion of sediment containing dioxins/furans. The only other individual contaminant resulting in a cancer risk greater than 1×10^{-6} within the Study Area is benzo(a)pyrene. The HIs for in-water worker RME scenario do not exceed 1.

The in-water worker RME scenarios do not result in an exceedance of 1×10^{-6} cumulative cancer risk or an HI greater than 1 for exposure to in-water sediment from river segments assessed outside of the Study Area.

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The in-water worker CT scenario for in-water sediment results in no exceedances of 1 x 10^{-6} cancer risk and no exceedances of an HI of 1.

5.2.2.2 Tribal Fisher

The results of the risk evaluation for tribal fisher exposure to in-water sediment are presented in Tables 5-23 through 5-25.

The tribal fisher RME scenario for in-water sediment results in exceedances of 1 x 10^{-6} cumulative cancer risk in 33 of 40 river mile segments within the Study Area, and from Study Area-wide exposure (see Table 5-23). The tribal fisher RME scenario for in-water sediment results in cumulative cancer risk greater than 1 x 10^{-4} at RM 6W and RM 7W. RM 7W is the location of the maximum cumulative cancer risk (3 x 10^{-4}). Risk at RM 7W is primarily due to incidental ingestion of sediment containing dioxins/furans (risk from dioxins/furan exposure is 3 x 10^{-4}); risk at RM 6W is primarily due to dermal contact with sediment containing benzo(a)pyrene (risk from benzo(a)pyrene exposure is 1 x 10^{-4}). In addition to these two contaminants, the following individual analytes also result in an individual cancer risk greater than 1 x 10^{-6} in at least one exposure area: arsenic, PCBs, benzo(b)fluoranthene, dibenzo(a,h)anthracene, benzo(a)anthracene, indeno(1,2,3-cd)pyrene.

Exposure areas including river mile segments outside of the Study Area that result in risks above 1 x 10^{-6} from the tribal fisher RME scenario for in-water sediment are: RM 12W (includes samples from RM 12.0W – 12.2W), Multnomah Channel, and RM 1.5E (includes samples from RM 1.5E – RM 1.9E), RM 1E, and RM1W. Tribal fisher exposure to in-water sediment from river segments outside of the Study Area do not result in HIs greater than 1.

The tribal fisher CT scenario for in-water sediment results in exceedances of 1 x 10^{-6} cumulative cancer risk at two of the 40 river mile segments (RM 6W and RM 7W). There are no exceedances of 1 x 10^{-4} cancer risk for the tribal fisher CT scenario. The maximum cumulative cancer risk occurs at RM 6W (6 x 10^{-6}) and is primarily due to exposure to sediment containing benzo(a)pyrene. The tribal fisher CT scenario for in-water sediment results in no HIs greater than 1.

There are no risks greater than 1 x 10^{-6} or HIs greater than 1 for CT tribal fisher exposure to in-water sediment from river segments assessed outside of the Study Area.

5.2.2.3 Fisher

To evaluate differences in fishing frequencies, risks were evaluated for both high-frequency and low-frequency fishers. High-frequency fishers were assumed to fish from the same 1/2-mile river segment three days per week for the entire year (156 days/year) for the default residential exposure duration (30 years) for the RME. Low-frequency fishers were assumed to fish from the same 1/2-mile river segment for two days per week for the entire year (104 days/year) for the default residential exposure duration (30 years) for the RME. The results of the risk evaluation for high-frequency fisher exposure to in-water sediment are presented in Tables 5-26 through 5-28. The results of the risk evaluation for low-frequency fisher exposure to in-water sediment are presented in Tables 5-29 through 5-30.

5.2.2.3.1 High-Frequency Fisher

The high-frequency fisher RME scenario for in-water sediment results in exceedances of 1 x 10^{-6} cumulative cancer risk in 17 of 40 river mile segments within the Study Area and from Study Area-wide exposure (see Table 5-26). There are no exceedances of 1 x 10^{-4} cancer risk for the high-frequency fisher RME scenario. The maximum cumulative cancer risks occur at RM 7W (8 x 10^{-5}) and RM 6W (5 x 10^{-5}). At RM 7W, risk is primarily due to incidental ingestion of sediment containing dioxins/furans. At RM 6W, risk is primarily due to dermal contact with sediment containing benzo(a)pyrene. In addition to these chemicals, the following individual analytes also result in a cancer risk greater than 1 x 10^{-6} in at least one exposure area: arsenic, PCBs, benzo(b)fluoranthene, dibenzo(a,h)anthracene, benzo(a)anthracene, and indeno(1,2,3-cd)pyrene.

For river mile segments outside of the Study Area, RM 12W is the only exposure area that results in risk above 1 x 10^{-6} for the high-frequency fisher RME scenario for in-water sediment. Risk at RM 12W is 2 x 10^{-6}, primarily due to exposure to benzo(a)pyrene. There are no exposure areas outside of the Study Area resulting in an HI greater than 1.

The high-frequency fisher CT scenario for in-water sediment results in no exceedances of 1 x 10^{-6} cumulative cancer risk and no exceedances of an HI of 1 for exposure areas assessed inside and outside of the Study Area.

5.2.2.3.2 Low-Frequency Fisher

The low-frequency fisher RME scenario for in-water sediment results in exceedances of 1 x 10^{-6} cumulative cancer risk at 12 of 40 river mile segments within the Study Area, and from Study Area-wide exposure (see Table 5-29). There are no exceedances of 1 x 10^{-4} cancer risk for the low-frequency fisher RME scenario. The maximum cumulative cancer risks occur at RM 7W (6 x10^{-5}) and RM 6W (3 x10^{-5}). At RM 7W, risk is primarily due to incidental ingestion of sediment containing dioxins/furans. At RM 6W, risk is primarily due to dermal contact with sediment containing benzo(a)pyrene. In addition to these chemicals, the following individual analytes also result in a cancer risk greater than 1 x 10^{-6} in at least one exposure area: PCBs, dibenzo(a,h)anthracene, benzo(a)anthracene, benzo(b)fluoranthene, and indeno(1,2,3-cd)pyrene. The low-frequency fisher RME scenario for in-water sediment results in no HIs greater than 1.

There are no risks greater than 1 x 10^{-6} or HIs greater than 1 for the low-frequency fisher RME scenario for exposure to in-water sediment from river segments assessed outside of the Study Area.

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The low-frequency fisher CT scenario for in-water sediment results in no exceedances of 1 x 10^{-6} cumulative cancer risk and no exceedances of an HI of 1 for exposure areas inside and outside of the Study Area.

5.2.2.4 Diver

Risks were evaluated for commercial divers wearing either a wet suit or a dry suit. The results of the risk evaluation for commercial wet suit diver exposure to in-water sediment are presented in Tables 5-31 through 5-32. The results of the risk evaluation for a commercial dry suit diver exposure to in-water sediment are presented in Table 5-33.

5.2.2.4.1 Diver in Wet Suit

The commercial diver in a wet suit RME scenario for in-water sediment results in exceedances of 1 x 10^{-6} cumulative cancer risk in 10 of 40 ½-mile river mile segments within the Study Area and for Study Area-wide exposure (see Table 5-31). There are no exceedances of 1 x 10^{-4} cancer risk for this scenario. The maximum cumulative cancer risk (3 x 10^{-5}) occurs at RM 6W and RM 7W. At RM 6W, the risk is primarily due to dermal adsorption of sediment containing benzo(a)pyrene. At RM 7W, the risk is primarily due to dermal absorption of sediment containing dioxins and furans. In addition to these two chemicals, the following individual analytes also result in a cancer risk greater than 1 x 10^{-6} in at least one exposure area: PCBs, benzo(b)fluoranthene, dibenzo(a,h)anthracene, benzo(a)anthracene, and indeno(1,2,3-cd)pyrene. The commercial diver in a wet suit RME scenario for in-water sediment results in no HIs greater than 1.

There are no exposure areas outside of the Study Area that result in risks above 1 x 10^{-6} or HIs greater than 1 for this scenario.

The commercial diver in a wet suit CT scenario for in-water sediment results in no exceedances of 1 x 10^{-6} cumulative cancer risk and no exceedances of an HI of 1 for exposure areas assessed inside and outside of the Study Area (see Table 5-32).

5.2.2.4.2 Diver in Dry Suit

The commercial diver in a dry suit RME scenario for in-water sediment results in exceedances of 1 x 10^{-6} cumulative cancer risk in two of 40 river mile segments within the Study Area (see Table 5-33). The maximum cumulative cancer risks occur at RM 7W (1 x 10^{-5}) and RM 6W (6 x 10^{-6}). At RM 7W, risk is primarily due to incidental ingestion of sediment containing dioxins/furans. At RM 6W, risk is primarily due to dermal contact with sediment containing benzo(a)pyrene. No other analytes result in a cancer risk greater than 1 x 10^{-6} for this scenario. The commercial diver in a dry suit RME scenario for in-water sediment results in no HIs greater than 1. There are no river mile segments outside of the Study Area that result in risk above 1 x 10^{-6} or an HI greater than 1. A CT scenario was not evaluated for a commercial diver in a dry suit, per direction from EPA.

5.2.2.5 Breastfeeding Infants of Adults Exposed to In-Water Sediment

Risks to infants consuming breastmilk from adults exposed to in-water sediment were calculated for all adult receptors for which bioaccumulative compounds were COPCs. This included all receptors assessed in this BHHRA for direct exposure to in-water sediment. These risk results are shown in Tables 5-34 through 5-44. The highest cumulative cancer risk to breastfeeding infants of adults exposed to in-water sediment occurs at RM 7W, due to consumption of dioxin/furans in human milk of a tribal fisher exposed to in-water sediment. The highest noncancer hazard to an infant also occurs at RM 7W (HI is 5).

5.2.2.6 Summary of In-Water Sediment Risk Characterization

Direct contact with in-water sediment resulted in cumulative cancer risks ranging from 5×10^{-9} to 3×10^{-4} across all scenarios. The only HI that was greater than 1 was for the tribal fisher and high frequency fisher RME scenario due to dioxin/furans, which occurred at the ½-mile exposure area at RM 7 west (W). The highest cumulative cancer risks and HIs from direct contact with in-water sediment were for the tribal fisher scenario. Four contaminants resulted in a cancer risk greater than 1×10^{-6} or hazard quotient greater than 1 for at least one of the in-water sediment scenarios: PCBs, dioxins, arsenic, and PAHs. A summary of in-water sediment risks by receptor and analyte are shown in Table 5-45.

5.2.3 Surface Water Risk Characterization Results

Potential risks from exposure to surface water through ingestion and dermal absorption were estimated for transients, recreational beach users, and divers. In addition, potential risks were estimated for a hypothetical future use of surface water as a domestic water source. There were multiple uncertainties associated with the direct exposure to surface water scenarios such as the exposure parameters, which are further described in the following sections, and contributions from background sources.

5.2.3.1 Transients

Risks to transients from surface water were evaluated for drinking water and bathing scenarios. The risks were evaluated for year-round exposure to surface water for four individual transect stations, for the four transects grouped together (to represent Study Area-wide exposure), and for Willamette Cove. In addition to these exposure areas within the Study Area, risk was evaluated for exposure to surface water for a transect in Multnomah Channel, which is outside of the Study Area. The results of the risk evaluation for transient exposure to surface water are presented in Tables 5-46 through 5-47.

The transient RME and CT scenarios for surface water result in no exceedances of 1×10^{-6} cancer risk and no exceedances of an HI of 1 inside or outside of the Study Area.

5.2.3.2 Recreational Beach Users

Risks to recreational beach users from surface water were evaluated for swimming scenarios, using data from summer months. Risks were evaluated for exposure to surface water for three transects grouped together (to represent Study Area-wide exposure) and for exposure to surface water for three individual quiescent areas during summer months. Risks for both adults and children were evaluated, as well as cancer risks to a combined child and adult receptor, in order to incorporate early-life exposures. The results of the risk evaluation for adult recreational beach user exposure to surface water are presented in Tables 5-48 through 5-49. The results of the risk evaluation for child recreational beach user exposure to surface water are presented in Tables 5-50 through 5-51. The results of the combined child and adult receptor are presented in Tables 5-52 through 5-53.

The adult, child, and combined recreational beach user RME and CT scenarios for surface water result in no exceedances of 1 x 10^{-6} cancer risk and no exceedances of an HI of 1.

5.2.3.3 Diver

Risks to commercial divers from surface water were evaluated for year-round exposure to four individual transect stations, and to single-point sampling stations within the Study Area grouped together on a ½-river mile basis, per side of river (E, W). In addition to these exposure areas within the Study Area, risk was evaluated for exposure to surface water for a transect in Multnomah Channel, which is outside of the Study Area. Risks were evaluated for commercial divers in wet suits and in dry suits. The results of the risk evaluation for commercial divers in wet suits exposure to surface water are presented in Tables 5-54 through 5-55. The results of the risk evaluation for commercial divers in dry suits are presented in Table 5-56.

5.2.3.3.1 Diver in Wet Suit

The commercial diver in a wet suit RME scenario for surface water results in exceedances of 1 x 10^{-6} cumulative cancer risk in one exposure area (RM 6W). There are no exceedances of 1 x 10^{-4} cancer risk for the commercial diver in a wet suit RME scenario. The maximum cumulative cancer risk occurs at RM 6W (1 x 10^{-5}) and is primarily due to dermal contact with surface water containing benzo(a)pyrene. There are no other analytes resulting in a cancer risk greater than 1 x 10^{-6}. The commercial diver in a wet suit RME scenario for surface water resulted in no HIs greater than 1. There are no exceedances of 1 x 10^{-6} risk or an HI of 1 for surface water exposure to river segments assessed outside of the Study Area.

The commercial diver in a wet suit CT scenario for surface water results in no exceedances of 1 x 10^{-6} cumulative cancer risk and no exceedances of an HI of 1 for exposure inside or outside of the Study Area.

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5.2.3.3.2 Diver in Dry Suit

The commercial diver in a dry suit RME scenario for surface water results in exceedances of 1 x 10^{-6} cumulative cancer risk in one exposure area (RM 6W). This exposure area is the location of the maximum cumulative cancer risk (2 x 10^{-6}) and is primarily due to dermal contact with surface water containing benzo(a)pyrene. There are no individual analytes resulting in a cancer risk greater than 1 x 10^{-6}. The commercial diver in a dry suit RME scenario for surface water resulted in no HIs greater than 1. There are no exceedances of 1 x 10^{-6} risk or an HI of 1 for surface water exposure to river segments assessed outside of the Study Area.

The commercial diver in a dry suit was not evaluated for CT exposure, as directed by EPA.

5.2.3.4 Domestic Water User

There is no known or anticipated future use of surface water within the Study Area for a domestic water supply. Because the designated beneficial use of the Willamette River is as a domestic water supply with adequate pretreatment, EPA directed that surface water be evaluated as a future domestic water source for both adult and child residents. For purposes of this BHHRA, untreated surface water was used to assess risks from future domestic water uses, so the risks are considered hypothetical. Risks were calculated for year-round exposure to surface water for the four transect stations within the Study Area and single point vertically integrated samples from Cathedral Park, Willamette Cove, and Swan Island Lagoon. In addition, Study Area-wide risk was calculated by combining the data from all vertically integrated samples to estimate Study Area-wide exposure. The results of the risk evaluation for surface water as a hypothetical future domestic water source are presented in Tables 5-57 through 5-58 for adult residents, Tables 5-59 through 5-60 for child residents, and Tables 5-61 through 5-62 for combined child and adult residents.

5.2.3.4.1 Adult Resident

The adult resident RME scenario for hypothetical future use of untreated surface water as a domestic water source results in cumulative risk exceedances of 1 x 10^{-6} at all 20 of the 20 exposure areas, and for Study Area-wide exposure (see Table 5-57). There is one exceedance of 1 x 10^{-4} cancer risk for the adult resident RME future hypothetical domestic water scenario, which occurs at RM 6.1 (cumulative risk is 3 x 10^{-4}, primarily due to benzo(a)pyrene in drinking water). Risks from untreated surface water exposure to both total and dissolved arsenic exceed 1 x 10^{-6} for all exposure areas. The adult resident RME scenario results in no HIs greater than 1.

Arsenic is a naturally occurring metal, and background concentrations in surface water may contribute to risk resulting from the hypothetical future use of untreated surface water as a domestic water source. Background concentrations for some chemicals in surface water were calculated using data collected from upstream of the Study Area, as described in Section 6 of the RI Report. The 95% UCL concentration of total arsenic in surface water upstream of the Study Area is 0.402 ug/l, and the 95th percentile value is 0.485 ug/l, which are both above the EPA tap water RSL for arsenic of 0.045 ug/l but below the EPA MCL of 10 ug/l. The 95% UCL/max EPCs for total arsenic for the hypothetical future use of untreated surface water for domestic use within the Study Area range from 0.32 to 0.60 ug/l, which include both maximum concentrations for an exposure area and 95% UCLs for an exposure area. EPCs at 17 out of 21 locations within the Study Area exceed 0.402 ug/l (the 95% UCL concentration of total arsenic in surface water upstream of the Study Area), and seven out of 21 of the EPCs exceed 0.485 ug/l (the 95th percentile value of total arsenic in surface water upstream of the Study Area). These concentrations are similar to the upstream arsenic concentration statistics. The 95% UCL concentration of total arsenic upstream of the Study Area (0.402 ug/l) results in a cancer risk of 7 x 10^{-6} for the adult resident exposure scenario.

The adult resident CT scenario for hypothetical use of untreated surface water as a future domestic water source results in cumulative risk exceedances of 1 x 10^{-6} at 17 of the 20 exposure areas, and for Study Area-wide exposure (see Table 5-58). There are no exceedances of 1 x 10^{-4} cancer risk for the adult resident CT future hypothetical domestic water scenario. The maximum cumulative cancer risk for the CT scenario is 3 x 10^{-5}, which occurs at RM 6.1. This exceedance is due to the hypothetical ingestion of untreated surface water containing benzo(a)pyrene. The adult resident CT scenario results in no HIs greater than 1.

5.2.3.4.2 Child Resident

The child resident RME scenario for hypothetical future use of untreated surface water as a domestic water source results in cumulative risk exceedances of 1 x 10^{-6} at all 20 of the 20 exposure areas, and for Study Area-wide exposure (see Table 5-59). There is one exceedance of 1 x 10^{-4} cancer risk for the child resident RME future hypothetical domestic water scenario, which occurs at RM 6.1 (cumulative risk is 7 x 10^{-4}, primarily due to benzo(a)pyrene in drinking water). The child resident RME scenario results in HIs greater than 1 at two locations: RM 2.9 (Multnomah Channel) and RM 8.5. The HI at both of these locations is 2, due primarily to exposures to MCPP in drinking water.

The child resident CT scenario for hypothetical use of surface water as a future domestic water source results in cumulative risk exceedances of 1 x 10^{-6} at all 20 of the 20 exposure areas, and for Study Area-wide exposure (see Table 5-60). There is one exceedance of 1 x 10^{-4} cancer risk for the child resident CT future hypothetical domestic water scenario, which occurs at RM 6.1 (cumulative risk is 2 x 10^{-4}, primarily due to benzo(a)pyrene in drinking water). The child resident CT scenario results in no HIs greater than 1.

5.2.3.4.3 Combined Child and Adult Resident

Cancer risks for a combined child and adult resident were calculated to incorporate early life exposures, per EPA (2005) and DEQ (2010) guidance. The maximum cancer risk for the combined child and adult receptor is 9 x 10^{-4}, occurring at RM 6.1, primarily from exposures to benzo(a)pyrene in drinking water. Risks from RME and CT scenarios exceed 1 x 10^{-6} for all exposure areas evaluated.

5.2.3.5 Summary of Surface Water Risk Characterization

Direct contact with surface water resulted in cumulative cancer risks ranging from 8 x 10^{-10} to 9 x 10^{-4} across all scenarios, including hypothetical future use as a domestic water source. The only HIs that were greater than 1 were for hypothetical future use as a domestic water source by a child resident under the RME scenario. The HI was 2 at Multnomah Channel and RM 8.5, due primarily to ingestion of MCPP in surface water. Eight contaminants resulted in a cancer risk greater than 1 x 10^{-6} or hazard quotient greater than 1 for at least one of the surface water scenarios, including: benzo(a)pyrene, benzo(a)anthracene, benzo(b)fluoranthene, dibenzo(a,h)anthracene, indeno(1,2,3-cd)pyrene, MCPP, arsenic, hexavalent chromium, and total PAHs. A summary of risks from exposure to surface water is provided in Table 5-63.

5.2.4 Groundwater Seep Risk Characterization Results

Only one groundwater seep was identified in a transient or recreational use area where upland COIs were potentially discharging. The seep identified is actually the potential groundwater discharge that could occur from Outfall 22B, which discharges into a transient use area. As a result, risks to transients from potential exposure to groundwater seeps were evaluated at that beach (07B024).

5.2.4.1 Transients

Risks to transients from the groundwater seep were evaluated for direct contact scenarios. There were multiple uncertainties associated with the exposure parameters for the direct exposure to groundwater seeps scenario. To evaluate the risks from exposure to the groundwater seep without stormwater influence, outfall data from stormwater sampling events was excluded from the dataset. The results of the risk evaluation for transient exposure to the groundwater seep are presented in Tables 5-64 through 5-65.

The transient RME and CT scenarios for the groundwater seep results in no exceedances of 1 x 10^{-6} cancer risk and no exceedances of an HI of 1.

5.2.4.1 Summary of Groundwater Seep Risk Characterization

There were no cancer risk or noncancer hazard exceedances from exposure to the groundwater seep. A summary of groundwater seep risks is provided in Table 5-66.

5.2.5 Fish Consumption Risk Characterization Results

Potential risks from fish consumption were estimated for fisher and tribal fisher scenarios. There were multiple uncertainties associated with the fish consumption scenarios such as assumptions regarding fish consumption rates, tissue type and fish species consumed, EPCs, and the use of cooking and preparation methods[7]. Uncertainties associated with this scenario are discussed further in Section 6.

5.2.5.1 Tribal Fishers

Risks to tribal fishers who consume fish caught within the Study Area were evaluated for a multi-species diet that includes salmon, lamprey, and sturgeon, in addition to resident fish species. A single ingestion rate for the multi-species diet was used to evaluate risks to the tribal fish consumer. Risks were evaluated using both 95% UCL/max and mean Study Area-wide tissue concentrations for both fillet and whole body tissue (see Section 3.4.5). Risks were higher for whole body tissue than for fillet tissue; however, fillet tissue was not analyzed for PCB or dioxin/furan congeners in all resident species. The results of the risk evaluation for adult tribal fish consumption are presented in Tables 5-67 through 5-70. The results of the risk evaluation for child tribal fish consumption are presented in Tables 5-71 through 5-74, and the results of the risk evaluation for the combined child and adult tribal consumers of fish are presented in Tables 5-75 through 5-76.

5.2.5.1.1 Tribal Adult, Fish Consumption

The risks ranged from a cumulative cancer risk of 2 x 10^{-2} for the 95% UCL/max EPCs of whole body tissue to a cumulative cancer risk of 2 x 10^{-3} for the mean EPCs of fillet tissue. For all scenarios, estimated risks are above a 1 x 10^{-4} cumulative cancer risk and are primarily due to PCBs and dioxins/furans. Figure 5-8 shows the relative risk contribution of individual COPCs for both whole body and fillet tissue diets of an adult tribal consumer, and Figure 5-9 shows a comparison of total risk per tissue type.

The cumulative HIs ranged from 400 for the 95% UCL/max EPCs of whole body tissue to 50 for the mean EPCs of fillet tissue. For the whole body tissue, 95% UCL/max EPC scenario, the PCB HQ is approximately 26 times higher than any other HQ. The toxicity endpoint for PCBs is immunological and skin. The immunological- and skin-specific HIs for tribal adult consumption are the highest endpoint-specific HIs, and exceed the next highest HI by a factor of 10. Additional endpoints that exceed an HI of 1 for the tribal adult 95% UCL/max consumption scenario are reproduction, central nervous system (CNS), and blood.

The multi-species diet evaluated in this BHHRA included resident fish as well as salmon, sturgeon, and lamprey. Because salmon, sturgeon, and lamprey spend time outside the Study Area, the risks from ingestion of salmon, sturgeon, and lamprey cannot be conclusively associated with sources within the Study Area. However, resident fish accounted for approximately 95 percent of the cumulative risk in the whole body diet. Of the four resident fish species included in the multi-species diet,

[7] For the purposes of the risk calculations, reference to "uncooked" fish tissue is the same as not accounting for reductions in contaminant concentrations from cooking or other food preparation.

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risks from ingestion of smallmouth bass and common carp were the primary contributors to the cumulative risk.

5.2.5.1.2 Tribal Child, Fish Consumption

The risks ranged from a cumulative cancer risk of 3 x 10^{-3} for the 95% UCL/max EPCs of whole body tissue to a cumulative cancer risk of 4 x 10^{-4} for the mean EPCs of fillet tissue. For all scenarios, risks are above a 1 x 10^{-4} cumulative cancer risk and are primarily due to PCBs and dioxins/furans.

The cumulative HIs ranged from 800 for the 95% UCL/max EPCs of whole body tissue to 100 for the mean EPCs of fillet tissue. The PCB HQ for the whole body tissue diet is approximately 26 times higher than any other HQ. The immunological- and skin- specific HIs for tribal child consumption are the maximum endpoint-specific HIs, and exceed the next highest HI by a factor of 10. Additional health endpoints that exceed an HI of one for the tribal child 95% UCL/max consumption scenario are reproduction, CNS, liver, and blood.

The multi-species diet evaluated in this BHHRA included resident fish as well as salmon, sturgeon, and lamprey. Because salmon, sturgeon, and lamprey spend time outside the Study Area, the calculated risks from ingestion of salmon, sturgeon, and lamprey cannot be conclusively associated with sources within the Study Area. However, resident fish accounted for approximately 95 percent of the cumulative risk associated with this scenario.

5.2.5.1.3 Combined Tribal Child and Adult, Fish Consumption

Cancer risks were calculated for the combined child and adult tribal fisher scenarios in order to incorporate early life exposures (EPA 2005, DEQ 2010). Cumulative cancer risks from fish consumption for the combined child and adult tribal fisher ranged from 3 x 10^{-3} (fillet tissue consumption, mean scenario) to 2 x 10^{-2}, (whole body tissue consumption, 95% UCL/Max scenario) primarily due to ingestion of PCBs in tissue. The results of the combined tribal child and adult cancer risks for consumption of fish tissue are presented in Tables 5-75 and 5-76.

5.2.5.1.4 Breastfeeding Infant of Tribal Adult Who Consumes Fish

Risks and hazards to an infant consuming human milk of a tribal adult who consumes fish were calculated for bioaccumulative compounds, consistent with EPA (2005) and DEQ (2010) guidelines. These risks are presented in Tables 5-77 and 5-78. Cancer risks range from 2 x 10^{-3} to 2 x 10^{-2}, and noncancer hazards range from 1,000 to 9,000.

5.2.5.1.5 Summary of Risks from Tribal Consumption of Fish

A summary of risks from tribal consumption of fish is provided in Table 5-79. Both cancer risks and noncancer hazards exceed the target risk values of 1 x 10^{-6} and 1, respectively, for all tribal receptors.

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5.2.5.2 Non-tribal Fishers

Risks for the non-tribal fish consumption scenarios were estimated for both single- and multi-species diets consisting only of resident fish species (smallmouth bass, black crappie, brown bullhead, and common carp). Risks were estimated separately for each exposure area (based on species home range) and for Study Area-wide exposure. Consumption of smallmouth bass was evaluated on a river mile basis, and consumption of common carp, brown bullhead, and black crappie was evaluated on a fishing zone basis (fishing zones were designated from RM 3-6 and from RM 6-9 for black crappie and brown bullhead, and from RM 3-6, RM 6-9, RM 0-4, RM 4-8, and RM 8-12 for common carp). In addition to evaluating risks using mean and 95% UCL/max tissue concentrations for both whole body and fillet tissue, each fish consumption scenario was evaluated using three different ingestion rates for adult and child consumers. The results of the risk evaluation for fish consumption by an adult are presented in Tables 5-80 through 5-119. The results of the risk evaluation for fish consumption by a child are presented in Tables 5-120 through 5-159. The results of the risk evaluation for fish consumption by a combined child and adult receptor are presented in Tables 5-160 through 5-169. In addition, Maps 5-4-1 through 5-7-3 show exposure areas with risk exceedances from 95% UCL/max EPCs for single species diets, at the 17.5 g/day, 73 g/day, and 142 g/day ingestion rates for adults.

5.2.5.2.1 Adult, Fish Consumption

Risks to adult fish consumers were evaluated for ingestion rates of 142 g/day, 73 g/day, and 17.5 g/day. These rates correspond to approximately 19 meals per month, 10 meals per month, and two meals per month, based on an 8-ounce serving size, every month of the year exclusively of resident fish caught within the Study Area.

The highest risk for all adult consumer scenarios was equal to a cumulative cancer risk of 6 x 10^{-2}. This was for the scenario based on the 95% UCL/max EPC, 142 g/day ingestion rate, and a fish diet comprised solely of whole-body common carp. The lowest risk was equal to a cumulative cancer risk of 7 x 10^{-6} for the 95% UCL/max and mean EPCs, 17.5 g/day ingestion rate, and a fish diet comprised solely of black crappie fillet tissue. For all tissue consumption scenarios, PCBs are the primary contributor to cumulative cancer risks. The highest cumulative HI from fish tissue ranged from 3,000 for the 95% UCL/max EPC, 142 g/day ingestion rate, common carp fillet tissue scenario to 0.5 for the mean EPC, 17.5 g/day ingestion rate, black crappie fillet tissue-only scenario. For the 95% UCL/max EPC, multi-species, whole body tissue scenario, the PCB HQ is approximately 30 times higher than the HQ for any other chemical. In general, the immunological-specific HIs for adult consumption scenarios are the highest of all endpoint-specific HIs, and exceed the next highest HIs by a factor of 10 to 100. Additional health endpoints that exceed an HI of 1 for the 95% UCL/max EPCs at the 17.5 g/day ingestion rate are reproduction, CNS, liver, skin, and blood.

Figures 5-10 through 5-17 show a summary of risk results for adult consumption of tissue for single species diets. These figures illustrate the relative contribution of

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individual COPCs to total risk for both whole body and fillet tissue consumption, per river mile, per fishing area, and per species.

In general, risks from consuming whole body tissue were greater than risks from consuming fillet tissue; however, fillet tissue was not analyzed for PCB or dioxin/furan congeners in black crappie or brown bullhead, and therefore PCB TEQ and dioxin/furan TEQ risks could be not evaluated in fillet tissue for those species. Smallmouth bass and common carp diet scenarios generally resulted in higher risks than the other diets evaluated. Black crappie diet scenarios generally resulted in the lowest risks of the diets evaluated.

5.2.5.2.2 Child, Fish Consumption

Risks to child consumers were evaluated for 60 g/day, 31 g/day, and 7 g/day ingestion rates. The risks for all child consumer scenarios ranged from a cumulative cancer risk of 2 x 10^{-2} for the 95% UCL/max EPC, 60 g/day ingestion rate, common carp whole body tissue-only scenario to a cumulative cancer risk of 3 x 10^{-6} for the mean EPC, 7 g/day ingestion rate, black crappie fillet tissue-only scenario. For all tissue consumption scenarios, PCBs are the primary contributor to cumulative cancer risks.

The highest endpoint-specific HIs ranged from 5,000 for the 95% UCL/max EPC, 60 g/day ingestion rate, common carp whole body tissue-only scenario to 0.9 for the mean EPC, 7 g/day ingestion rate scenario for black crappie fillet tissue-only scenario. For the 95% UCL/max EPC, multi-species, whole body tissue diet scenario, the PCB HQ is approximately 30 times higher than the HQ for any other chemical. In general, the immunological-specific HIs for child consumption scenarios exceed the next highest HIs by a factor of approximately 10. Additional health endpoints that exceed an HI of 1 for the child 95% UCL/max consumption scenarios at the 31 g/day ingestion rate are reproduction, CNS, liver, skin, and blood.

In general, risks from whole body tissue were greater than risks from fillet tissue. Smallmouth bass and common carp diet scenarios generally resulted in higher risks than the other diets evaluated. Black crappie diet scenarios generally resulted in the lowest risks of the diets evaluated.

5.2.5.2.3 Combined Child and Adult, Fish Consumption

Cancer risks were calculated for a combined child and adult consumer of fish, to account for early life exposures, for all fish consumption scenarios evaluated in this BHHRA. Results for the evaluation of combined child and adult cancer risks from fish consumption are presented in Tables 5-160 through 5-169. Cancer risks for the combined child and adult consumer of fish are generally the same order of magnitude as adult-only risks. The highest cumulative cancer risk for the combined child and adult consumer is 7 x 10^{-2}, which occurs at the child ingestion rate of 60 g/day and the adult ingestion rate of 142 g/day, due to consumption of whole body carp from the fishing zone covering RM 4 through RM 8.

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5.2.5.2.4 Breastfeeding Infant of Adult Who Consumes Fish

Risk and hazards to infants consuming human milk from adults consuming fish collected from the Study Area were assessed for bioaccumulative compounds for all adult fish consumption scenarios, in accordance with EPA (2005) and DEQ (2010) guidance. Cancer risks to infants were calculated by applying an IRAF to the combined child and adult cancer risk from fish consumption. Noncancer hazards were calculated by applying an IRAF to the adult HQ for each fish consumption scenario. Results of the risk and hazard calculations for breastfeeding infants of adult consumers of fish are provided in Tables 5-170 through 5-179. The highest cancer risk to a breastfeeding infant of an adult consumer of fish is 7 x 10^{-2}, due primarily to PCBs in breastmilk. The highest noncancer hazard is 60,000, also due primarily to PCBs in breastmilk.

5.2.5.3 Consideration of Regional Tissue Concentrations

PCBs and dioxins/furans have been detected in fish tissue collected in the Willamette and Columbia Rivers, outside of the Study Area. In the Columbia River Basin Fish Contaminant Survey, the basin-wide average concentrations of total PCBs in resident fish ranged from 0.032 to 0.173 parts per million (ppm) for whole body samples and from 0.033 to 0.190 ppm for fillet with skin samples (EPA 2002c). In the Middle Willamette River (RM 26.5 to 72), the average concentrations of total PCBs in resident fish ranged from 0.086 to 0.146 ppm for whole body samples and from 0.026 to 0.071 ppm for fillet with skin samples (EVS 2000). These concentrations are lower than the concentrations detected in the Study Area where average concentrations ranged from 0.16 to 2.8 ppm in whole body samples and from 0.17 to 2.5 ppm in fillet with skin samples (for PCBs as total congeners). The fish species included in the studies were different than those collected within the Study Area, so the concentrations may not be directly comparable. Sources contributing to the PCBs and dioxins/furans detected in fish collected outside of the Study Area are unknown and may not be relevant to the Study Area.

In addition, the LWG collected upstream fish tissue samples at RM 20 and 28 during Round 1. The data for the upstream fish tissue samples are described in further detail in Section 5.5 of the RI Report. While there are a limited number of samples and species in the upstream fish tissue dataset, the results from the upstream fish tissue are consistent with the results from the Columbia and mid-Willamette River studies.

The EPA established a target fish tissue concentration of 0.0015 ppm for PCBs to allow a monthly fish consumption rate of more than 16 meals per month (EPA 2000c). The highest fish ingestion rates used in this BHHRA, 142 g/day for adult fishers and 175 g/day for adult tribal fishers, equate to over 19 and 23 meals per month, respectively, assuming an eight-ounce meal size.

The target fish tissue concentration established by EPA is based on a target cancer risk level of 1 x 10^{-6}. The regional PCB concentrations detected in resident fish from

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the Willamette and Columbia Rivers are approximately 20 to 100 times higher than the EPA target fish tissue concentration. These concentrations from outside of the Study Area are equivalent to cancer risks ranging from 2 x 10^{-5} to 1 x 10^{-4} relative to the EPA target fish tissue concentration, indicating that regional concentrations of PCBs exceed the lowest target cancer risk level of 1 x 10^{-6} for fish consumption rates higher than 16 meals per month. For noncancer endpoints, the EPA established a target tissue concentration is 0.0059 ppm. Concentrations detected in resident fish from the Willamette and Columbia Rivers are up to 30 times higher than this target tissue concentration. Regional efforts are underway to reduce concentrations in fish tissue.

5.2.5.4 Summary of Fish Consumption Risk Characterization

Consumption of individual species by the fisher resulted in cumulative cancer risks ranging from 7 x 10^{-6} to 6 x 10^{-2} for the adult consumer and from 3 x 10^{-6} to 2 x 10^{-2} for the child consumer. The maximum endpoint-specific hazard index (HI) for both adult and child fish consumption scenarios was for the immunological endpoint, primarily due to consumption of PCBs in tissue. The highest HI for the immunological endpoint occurs from child consumption of whole body common carp tissue from river miles (RM) 4-8. The range of HIs for the immunological endpoint across all single-species exposure scenarios evaluated for non-tribal consumers is from 0.9 to 3,000 for the adult fish consumer and from 0.7 to 5,000 for the child fish consumer.

Fish consumption risks were also evaluated for adult and child tribal fishers based on the 95th percentile ingestion rate from the CRITFC Consumption Study (1994). The tribal fish consumption risks assumed a multi-species diet consisting of resident fish species (common carp, black crappie, brown bullhead, and smallmouth bass) as well as sturgeon, lamprey, and salmon. Risks from the tribal fish diet were based on consumption of either whole body or fillet with skin tissue. It was assumed that all fish consumed were caught within the Study Area. Consumption of fish by the tribal fisher resulted in cumulative cancer risks ranging from 2 x 10^{-3} to 2 x 10^{-2} for the tribal adult fisher and from 4 x 10^{-4} to 3 x 10^{-3} for the tribal child consumer. The maximum endpoint-specific HIs for both the tribal adult and tribal child fishers were for the immunological endpoint, primarily due to consumption of PCBs in fish tissue. The range of immunological HIs for all tribal fisher fish consumption scenarios was from 50 to 400 for the tribal adult and from 100 to 800 for the tribal child.

Twenty-four contaminants resulted in a cancer risk greater than 1 x 10^{-6} or hazard quotient greater than 1 for at least one of the fish consumption scenarios evaluated in the draft BHHRA. The contaminants identified as posing potentially unacceptable risks were: PCBs, dioxins, six metals (antimony, arsenic, lead, mercury, selenium, and zinc), bis 2-ethylhexyl phthalate (BEHP), PAHs, hexachlorobenzene, and eleven pesticides (aldrin, dieldrin, heptachlor epoxide, total chlordane, total DDD, total DDE, total DDT, alpha-, beta, and gamma-hexachlorocyclohexane, and heptachlor). Of these, PCBs resulted in the highest cancer risks and hazard quotients.

A summary of risks from fish consumption is provided in Tables 5-180 and 5-181.

5.2.6 Shellfish Consumption Risk Characterization Results

5.2.6.1 Adult, Shellfish Consumption

Potential risks from shellfish consumption were estimated for the adult fisher scenarios. Risks to adult shellfish consumers were evaluated for clam and crayfish diets. For crayfish, risks were evaluated for each sample station and for Study Area-wide exposure. For clam, risks were evaluated on a river-mile basis and for Study Area-wide exposure separately for depurated and undepurated tissue, as agreed upon with EPA. Risks were estimated for an 18 g/day ingestion rate, which equates to approximately two and a half 8-ounce meals per month, and for a 3.3 g/day ingestion rate, which is just less than an 8-ounce meal every 2 months. Risks were calculated using both the 95% UCL/max and mean tissue concentrations of shellfish tissue. The results of the risk evaluation for shellfish consumption are presented in Tables 5-182 to 5-193. Cumulative risk exceedances for shellfish scenarios are summarized by exposure point in Maps 5-8-1 through 5-8-4.

Estimated risks from shellfish consumption within the Study Area ranged from a high cumulative cancer risk of 7×10^{-4}, which was for the 95% UCL/max EPCs, 18 g/day ingestion rate undepurated clam tissue scenario, to a cumulative cancer risk of 9×10^{-7}, which was for the mean EPC, 3.3 g/day ingestion rate crayfish tissue scenario. Estimated risks from shellfish consumption in areas assessed outside of the Study Area ranged from 2×10^{-6} to 8×10^{-5}. Clam samples were not all analyzed for the same chemicals, and the uncertainties associated with the resulting risks are discussed in Section 6. Study Area-wide risks from ingestion of undepurated clam tissue are two to three times higher than Study Area-wide risks from ingestion of depurated clam tissue, as shown in Table 5-182 and Table 5-183. Depurated clam tissue samples were collected from five locations at the northern and southern edges of the Study Area, while undepurated clam tissue samples were collected from 22 locations throughout the Study Area. For all high ingestion rate scenarios, risks are above a 1×10^{-6} cumulative cancer risk and are primarily due to PCBs.

Figures 5-18 through 5-21 show the relative contribution of individual COPCs to total risks from clam and crayfish consumption, as well as a summary of total risks per exposure point for the different ingestion rates.

The cumulative HIs from shellfish consumption ranged from 40 for the 95% UCL/max EPCs, 18 g/day ingestion rate, undepurated clam tissue scenario to 0.06 for the mean EPCs, 3.3 g/day ingestion rate, crayfish tissue scenario. Noncancer hazards above an HI of 1 are primarily due to PCBs. Study Area-wide HIs from ingestion of undepurated clam tissue are one to two times higher than Study Area-wide risks from ingestion of depurated clam tissue. These results are shown in Table 5-182 and Table 5-183.

5.2.6.2 Breastfeeding Infant of Adult Who Consumes Shellfish

Risk and hazards to infants consuming human milk from adults consuming shellfish were assessed for bioaccumulative compounds for all adult shellfish consumption scenarios, in accordance with EPA (2005) and DEQ (2010) guidance. Cancer risks and noncancer hazards to infants were calculated by applying an IRAF to the adult cancer risk and noncancer results from shellfish consumption, as shown in Tables 5-194 through 5-197. The highest cancer risk to a breastfeeding infant of an adult consumer of shellfish is 7 x 10^{-4}, from human milk consumption of an adult who consumed undepurated clam tissue at the 18 g/day ingestion rate. The risk is primarily from PCBs in breastmilk. The highest cumulative hazard quotient from bioaccumulative chemicals is 800 due primarily to PCBs in breastmilk.

5.2.6.1 Summary of Risks from Consumption of Shellfish

A summary of risks from consumption of Shellfish is provided in Table 5-198 by receptor and analyte. Cancer risks and noncancer hazards exceed the targets of 1 x 10^{-6} and 1, respectively, for all scenarios evaluated.

5.2.7 Evaluation of Cumulative and Overlapping Scenarios

As shown in the conceptual site model (Figure 3-1), multiple exposure scenarios may exist for a given population. For example, recreational beach users are potentially exposed to both beach sediment and surface water. The risks for each of the exposure scenarios that are considered potentially complete and significant for a given population were summed to estimate the cumulative risks for that population. The cumulative risks are presented in Table 5-199 for 95% UCL/max exposures, and in Table 5-200 for mean exposures. Additionally, cumulative risks for divers exposed to both in-water sediment and surface water are presented on a ½-river mile basis, per side of river, in Table 5-201 for RME exposures and Table 5-202 for CT exposures.

As discussed in Section 3, certain individuals may be exposed to COPCs within the Study Area through multiple exposure scenarios; for example, a recreational beach user might also be a fisher. This BHHRA quantitatively estimated risks for the individual exposure scenarios. Due to multiple exposure locations over different scales for both RME and CT scenarios, as well as ranges of ingestion rates and multiple diets for fish consumption, there are numerous potential combinations of overlapping scenarios. As a result, this BHHRA did not quantitatively evaluate all possible overlapping scenarios. However, risks from fish consumption are generally at least an order of magnitude higher than risks from other exposure scenarios, so if an individual consumes fish, the contribution from other exposure scenarios is not likely to contribute significantly to the overall risks for that individual.

5.2.8 Risk Characterization of Lead

A great deal of information on the health effects of lead has been obtained through decades of medical observation and scientific research. By comparison to most other environmental toxicants, the degree of uncertainty about the health effects of lead is quite low. Because age, health, nutritional state, body burden, and exposure duration influence the absorption, release, and excretion of lead, EPA has not established standard toxicity endpoints for lead. Instead, the concentration of lead in the blood is used as an index of the total dose of lead, regardless of the route of exposure (EPA 1994). As a result, blood lead levels, rather than intakes, are used to evaluate potential risks associated with exposure to lead. The Centers for Disease Control (CDC) has identified a blood lead level of 10 micrograms per deciliter (μg/dl) as the level of concern above which significant health risks may occur (CDC 1991). An acceptable risk for lead exposure typically equates to a predicted probability of no more than 5 percent greater than the 10 μg/dl level (EPA 1998).

Lead was identified as a COPC for in-water sediment, fish and shellfish. The following discusses the evaluation of risks from lead for each of those media.

5.2.8.1 In-Water sediment

Lead was identified as a COPC for in-water sediment because the maximum detected concentration exceeds the RSL for industrial soil of 800 mg/kg. The RSL was developed to be protective of the fetus of a pregnant woman exposed to lead. The only receptors for in-water sediment exposures are adults. Therefore, the fetus of a pregnant in-water worker or fisher is the most sensitive scenario for exposure to lead in in-water sediment, and the RSL is protective of that scenario. While maximum detected concentrations were used in identifying COPCs, EPCs were used to calculate risks. The maximum EPC for one of the in-water sediment exposure areas (2,200 mg/kg) is greater than the RSL. The adult lead model (ALM, Version 5/19/05, EPA 2003c) was used to estimate the probability of exceeding a target blood level for lead of 10 μg/dl from exposure to in-water sediment. Exposure parameters from Table 3-27 were used to develop site-specific ALM input parameters. For scenarios modeling exposure to in-water sediment, the exposure factors from Table 3-27 were adjusted with the assumption of a 25% sediment contact frequency. For ALM parameters without site-specific values, the model defaults for the West Region from Phases 1 and 2 of the National Health and Nutrition Evaluation Survey (NHANES III) (EPA 2002e) were used. The site-specific ALM blood lead concentration estimates for receptors potentially exposed to in-water sediment within the Study Area are presented in Tables F5-1 and F5-2 of Attachment F5.

Using the maximum EPC of 2,200 mg/kg, the maximum estimated probability of exceeding a fetal blood lead level of 10 μg/dL for any in-water sediment exposure scenario is one percent, which is for the RME in-water worker and RME high-frequency fisher scenarios. Because the maximum EPC for lead results in a probability of exceeding protective blood lead levels in the fetus of a pregnant woman that is less than 5 percent, lead is not considered a chemical potentially posing unacceptable risks for in-water sediment. All other EPCs for lead were below the

RSL. The uncertainty associated with the evaluation of lead is discussed further in Section 6.

5.2.8.2 Fish

Lead was identified as a COPC for fish consumption because it was detected in fish tissue. The Columbia River Basin Fish Contaminant Survey (EPA 2002c) determined fish tissue concentrations for lead that are unlikely to result in blood lead levels exceeding 10 μg/dl for the fetus of a pregnant adult, and for children. These concentrations were developed using the ALM (EPA 2003c) and the Integrated Exposure Uptake Biokinetic Model for Lead in Children (IEUBK, EPA 2007d), in combination with the fish ingestion rates from the CRITFC Fish Consumption Survey (CRITFC 1994). The concentrations of concern were developed using health protective exposure assumptions and were considered unlikely to underestimate risks from fish consumption.

Adults

The following equations from the ALM were used in the Columbia River Basin Fish Contaminant Survey (EPA 2002c) to develop tissue concentrations to be protective of fetuses of tribal adults:

$PbB_a = PbB_o + BKSF * (PbF * IR_F * AF_F * EF_F)/AT$

$PbBf = PbB_a * 0.9$

Probability that fetal blood lead is less than 10 μg/dl using the z-value where:

$p' = \Phi z \; [(\ln(PbBf)-\ln(10)) / \ln(GSD)]$

Where:

PbB_a = Central tendency of adult blood lead level
PbB_o = Adult baseline blood lead level
PbBf = Fetal blood lead level
GSD = Geometric standard deviation
BKSF = Biokinetic slope factor
PbF = Lead fish tissue concentration
IR_F = Fish tissue ingestion rate
AF_F = Absolute gastrointestinal ingestion factor for ingested lead in tissue
EF_F = Exposure frequency of fish ingestion
AT = Averaging time

The EPA (2003c) ALM approach was used to determine protective fish tissue concentrations for the fetuses of both adult fishers and adult tribal fishers in the Study Area, using updated default ALM assumptions for the West Region, which are based on current EPA guidance (EPA 2003c). Differences in default parameter values from the EPA (2003c) application of the ALM to the ALM application for this BHHRA include a change in PbB_o from 2.2 μg/dl to 1.4 μg/dl, and a change in AF_F from 0.1 to 0.12.

The evaluation of risks from lead is based on geometric mean levels and associated probabilities, so median values are generally used as inputs to the equations. The mean estimate of national per capita fish consumption of 7.5 g/day was used as the consumption rate for adults (EPA 2000b). The median fish ingestion rate for tribal fishers is 39.2 g/day, as stated in the CRITFC Fish Consumption Survey (CRITFC 1994) and used by the EPA (2002c) in calculations of protective lead tissue concentrations. The ALM inputs and results for estimating protective lead tissue concentrations for fetuses of adult fishers and adult tribal fishers consuming fish in the Study Area are provided in Table F5-3 of Attachment F5.

Using the above equations, the ALM predicts that fetal blood lead levels will exceed 10 μg/dl less than 5 percent of the time for adult fishers at a lead fish tissue concentration of 5.25 mg/kg. The maximum fish tissue EPC for lead in the Study Area is 1,100 mg/kg, detected in a smallmouth bass whole body tissue sample. This is above the protective concentration of 5.25 mg/kg. However, this maximum EPC is orders of magnitude greater than all other resident fish EPCs and may be attributable to lead in the gut of the fish due to the ingestion of a metallic object (e.g., sinkers) (Integral 2008). There are no other resident fish tissue EPCs which exceed a protective lead concentration of 5.25 mg/kg. Therefore, while lead is considered a preliminary chemical potentially posing unacceptable risks for fish ingestion by an adult fisher, the uncertainties associated with the maximum detected concentration and evaluations of lead are discussed further in Section 6.

The protective lead tissue concentration for fetuses of tribal adults, using the above methods, is 1.01 mg/kg. The maximum fish tissue lead EPC for an adult tribal fisher is 23 mg/kg. However, the tribal fisher tissue ingestion scenario is for a multi-species diet consisting of both resident and anadromous species. There are no detected concentrations in anadromous species exceeding 1.01 mg/kg. Over 99% of the lead in the maximum lead EPC for tribal fishers is attributable to the Study Area-wide EPC for lead in smallmouth bass, which is influenced by the maximum EPC mentioned above for adult fishers. Therefore, while lead is considered a preliminary chemical potentially posing unacceptable risks for fish ingestion by an adult tribal fisher, the uncertainties associated with the maximum detected concentration and evaluations of lead are discussed further in Section 6.

Children

The EPA (2002c) used the IEUBK model in the Columbia River Basin Fish Contaminant Survey to determine risks from ingestion of lead in tissue in tribal children. The same IEUBK methodology was applied to assess risks to children from ingestion of lead in fish tissue for this BHHRA.

To assess risks to children from ingestion of lead in fish tissue, a protective tissue concentration of lead in fish tissue was calculated using the IEUBK model with all exposure parameters set to default levels and with the addition of a fish ingestion rate based on the child consumption scenario for this BHHRA. The default exposure parameters for the IEUBK model, provided as Table F5-4, are the same model parameters used by the EPA (2002c) because site-specific values for soil lead concentration, house dust lead concentration, lead concentration in air and drinking water are not readily available. The ratio of child to adult consumption rates of 0.42, described in Section 3.5.1.5, was applied to the consumption rate for adults of 7.5 g/day to obtain a consumption rate for children of 3.15 g/day. In accordance with the methodology used by the EPA (2002c), fish ingestion was specified in the IEUBK model as the percentage of meat in diet consisting of locally caught fish and the lead concentrations in the fish. The protective fish tissue concentration for a child consumer, using the above method, is 2.6 mg/kg lead in fish tissue. The protective fish tissue concentration of 2.6 mg/kg is the fish tissue concentration resulting in predicted geometric blood lead level of 4.6 µg/dl and the probability of achieving a blood lead level greater than 10 µg/dl is no more than 5 percent.

The Columbia River Basin Fish Contaminant Survey (EPA 2002c) determined that 0.5 mg/kg is a protective tissue concentration for tribal children consuming tissue at a rate of 16.2 g/day, which is the 65th percentile consumption rate from their survey. Within the Portland Harbor Study Area, the maximum lead tissue EPC for the tribal child consumption scenario is 23 mg/kg, which is greater than the estimated protective concentration. Over 99% of this concentration is attributable to the contribution from the Study Area-wide smallmouth bass EPC. There are no anadromous species with detected lead concentrations exceeding 0.5 mg/kg. Therefore, while lead is considered a preliminary chemical potentially posing unacceptable risks for fish tissue for a tribal child consumer, the uncertainties associated with the maximum detected concentration and evaluations of lead are discussed further in Section 6.

5.2.8.3 Shellfish

Lead was identified as a COPC for shellfish consumption because it was detected in shellfish tissue. Shellfish consumption was only evaluated for adult scenarios. Therefore, the tissue concentration of concern for fetuses is the only tissue concentration relevant for shellfish consumption. The CRITFC approach to assessing risks from lead using the ALM was applied to the shellfish ingestion scenario for the site. Using the ALM equations applied to adult fishers in the previous section, the mean shellfish ingestion rate of 3.3 g/day, and the maximum shellfish exposure point concentration of 1,320 µg/kg, the ALM predicts that fetal blood lead levels will exceed 10 µg/dl less than 5 percent of the time. Therefore, lead is not considered a chemical potentially posing unacceptable risks for shellfish consumption. The ALM parameter values and results used to assess risk from adult exposure to lead via ingestion of shellfish are shown in Attachment F5.

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5.3 SUMMARY OF RISK CHARACTERIZATION

The ranges of estimated potential risks resulting from the different exposure scenarios evaluated in this BHHRA are summarized in Table 5-203. The ranges included in Table 5-203 for different scenarios reflect differences in CT vs. RME scenarios, differences in tissue EPCs (mean vs. 95% UCL/max), level of fish consumption (17.5 g/day [EPA 2002b], 73 g/day [Adolfson 1996], and 142 g/day [EPA 2002b]), location of sediment (for beach scenarios), tissue type (whole body vs. fillet or depurated vs. undepurated), and species of fish consumed. There were multiple uncertainties associated with the different scenarios such as the spatial scale of EPCs, sediment and surface water exposure parameters, tissue consumption rates, tissue type and fish and shellfish species consumed, fish and shellfish cooking and preparation methods, and contributions from background.

In general, the risks from fish consumption are higher than any of the other exposure scenarios evaluated in this BHHRA. These risks can be summarized as follows:

- The range of cumulative risks from all fish consumption scenarios is 3×10^{-6} to 7×10^{-2}, and the cumulative HIs range from 0.5 to 5,000. The highest HI for a breastfeeding infant of a fish consumer is 60,000.

- Cumulative cancer risks from consumption of shellfish range from 9×10^{-7} to 7×10^{-4}, and the cumulative HIs range from 0.06 to 40. The highest HI for a breastfeeding infant of a shellfish consumer is 800.

- For beach sediment, cumulative cancer risks range from 8×10^{-9} to 9×10^{-5}, and the cumulative HIs range from 5×10^{-4} to 1.

- For in-water sediment, cumulative cancer risks range from 3×10^{-9} to 3×10^{-4}, and the cumulative HIs range from 6×10^{-5} to 3. The highest HI for a breastfeeding infant of an in-water sediment receptor is 5 (for the tribal fisher).

- For direct contact to surface water, cumulative cancer risks range from 8×10^{-10} to 9×10^{-4}, and the cumulative HIs range from 1×10^{-5} to 2.

- For groundwater seeps, cumulative cancer risks range from 4×10^{-10} to 3×10^{-9}, and the cumulative HIs range from 1×10^{-3} to 6×10^{-3}.

Chemicals that resulted in a cancer risk greater than 1×10^{-6} or an HQ greater than 1 under any of the exposure scenarios for any of the exposure point concentrations evaluated in this BHHRA are presented in Table 5-204.

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6.0 UNCERTAINTY ANALYSIS

Uncertainty is associated with every step of a risk assessment, from the sampling and analysis of chemicals in environmental media to the assessment of exposure and toxicity and the risk characterization. In general, the approach and methodologies used in a risk assessment are designed to err on the side of conservatism, i.e., protection of health. In a deterministic risk assessment, conservative assumptions can compound to result in an estimate of risk that is at the upper end of the probable risk range.

Uncertainty can have two components: 1) variability in data or information, and 2) lack of knowledge. An uncertainty analysis conducted as part of a risk assessment focuses on issues of variability and knowledge uncertainty associated with each of the inputs and models used to derive the risk estimates.

Variability arises from true heterogeneity in exposure variables or responses, such as dose-response differences within a population or differences in contaminant levels in the environment. The values of some variables used in an assessment change with time and space, or across the population whose exposure is being estimated. Although variability can be better understood, it cannot be reduced through further study. Use of RME and CT scenarios provide an estimate of high-end and average exposures that may reasonably occur. The difference between the RME and CT risk estimates provides an initial evaluation of the degree of variability in exposure between individuals.

The second factor that generates uncertainty is a lack of knowledge about factors such as adverse effects or chemical concentrations. Uncertainty may be reduced by increasing knowledge about a factor through additional study, although it is impossible to gather enough data to eliminate uncertainty. In addition, at some point, there are diminishing returns associated with the collection of additional data; the cost of data collection is substantial and disproportional to the reduction in uncertainty. A substantial amount of uncertainty is often inherent in environmental sampling as well as in the scientific models used in risk assessment.

This section includes a detailed analysis of uncertainties associated with each step of the BHHRA. However, a deterministic risk assessment alone cannot quantify the degree of conservatism in risk estimates, and this BHHRA does not include a probabilistic risk assessment, per agreement with EPA. This uncertainty analysis addresses variability and/or uncertainty in the inputs to the risk estimates, focusing on those inputs likely to have the greatest effects on the results of the risk analyses. A summary of uncertainties associated with this BHHRA and discussed in this section are provided in Table 6-1.

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6.1 DATA EVALUATION

As discussed in Section 2, data collected during the RI, as well as data of confirmed quality that meet the DQOs for risk assessment, were used in this BHHRA to estimate risks. Sediment, surface water, groundwater seep, and biota data were collected for use in this BHHRA. Use of the EPA's DQO planning process (EPA 2000e) minimized the uncertainty associated with the data collected during the RI; however, some amount of uncertainty is inherent in environmental sampling. The following data evaluation uncertainties have been identified.

6.1.1 Use of Target Species to Represent All Types of Biota Consumed

Because it is not practical to collect samples of every resident species consumed by humans within the Study Area, target resident species were selected to represent the diet of all biota consumed by humans, as recommended by EPA guidance (2000a). Four target species were collected to represent resident fish tissue diet (smallmouth bass, black crappie, common carp, and brown bullhead), and two species were collected to represent shellfish diet (crayfish and clam). Factors in selecting the target species included: consumption by humans, home range, potential for bioaccumulation, trophic level of species, and abundance.

The range of concentrations detected in the target species generally coincides with the range of concentrations detected in other species that were collected. Furthermore, the concentrations of PCBs, which is the chemical group with the greatest contribution to risk, are generally highest in smallmouth bass and common carp, both of which were included in this BHHRA. Therefore, the use of target resident species to represent all biota consumed should not impact the conclusions of this BHHRA, and may in fact overestimate risks, especially if non-resident species are consumed.

6.1.2 Source of Chemicals for Anadromous and Wide-Ranging Fish Species

For non-resident fish species, salmon, lamprey, and sturgeon were chosen as target species to represent a portion of the tribal fish tissue diet. Due to the life cycles of these species, these fish spend some portion of their lives outside of the Study Area. The time spent outside the Study Area may be significant for bioaccumulation of chemicals due to the growth, development, and feeding that occurs, as well as the relative amount of time spent within the Study Area versus outside of the Study Area.

The Washington Department of Ecology analyzed returning fall Chinook salmon, as fillet tissue with skin, collected from three coastal rivers (Queets, Quinault, and Chehalis Rivers) in 2004 (Ecology 2007). PCBs as Aroclors were detected at concentrations ranging from 5.0 µg/kg to 6.3 µg/kg in the Ecology study relative to the maximum detected concentration of 20 µg/kg for salmon fillet tissue with skin collected from the Lower Willamette. The dioxin TEQ concentrations ranged from 0.09 picograms per gram (pg/g) to 0.23 pg/g in the Washington coastal rivers relative to the maximum detected concentration of 2 pg/g for salmon fillet tissue with skin collected from the Lower Willamette. A comparison of the tissue concentrations from the Ecology study and the Lower Willamette are presented in Table 6-2. While the Chehalis River passes through some developed areas and therefore may have localized sources, both the Queets and Quinault Rivers are located almost entirely within Olympic National Forest and wilderness areas, so the potential for contribution from localized sources should be minimal. These results indicate that sources of chemicals outside of the Study Area may contribute to bioaccumulation of chemicals in anadromous fish species.

There is a high degree of uncertainty as to the source of chemicals detected in non-resident fish species and whether those chemicals are actually due to exposures within the Study Area. However, approximately 95 percent of the cumulative risk from fish consumption is due to chemical concentrations detected in resident fish, even though resident fish only account for 50 percent of the mass of fish consumed. Therefore, using the results of the BHHRA to focus on potential sources of chemicals potentially posing unacceptable risks in resident fish species should address sources of chemicals potentially posing unacceptable risks within the Study Area that contribute to concentrations in non-resident fish species as well. As a result, the uncertainty associated with the source of chemicals to non-resident fish species should not impact the conclusions of this BHHRA.

6.1.3 Use of Either Whole Body or Fillet Samples to Represent All Fish Consumption

Chemicals bioaccumulate differently in different parts of an organism. Organic compounds tend to accumulate more in the fatty tissues, and heavy metals more in muscle tissues. The chemicals with the greatest contribution to the cumulative cancer risk and with the highest noncancer HQ are PCBs, which are organic compounds that accumulate preferentially in fatty tissue. Diets consisting of different fish parts result in varying levels of risk to the consumer. Using only whole body or fillet tissue with skin to evaluate risk from all types of fish tissue diets is a conservative representation of actual consumption of fish. Depending on the species and chemical, the difference in concentrations between fillet and whole body tissue can be minimal or more than a factor of 10, as discussed in Attachment F6. Since PCBs contribute to the vast majority of risks from tissue consumption on a Study Area-wide scale and on a localized scale for most exposure areas, this uncertainty could have a significant impact on the conclusions of this BHHRA. Alternatively, chemicals such as methyl mercury preferentially accumulate in muscle tissue, which means concentrations of mercury in fillet tissue would likely be higher than concentrations of mercury in whole body tissue.

Based on the Columbia Slough consumption survey (Adolfson 1996), the majority of fishers are most likely to consume only the fillet portion of the fish, which may not include skin. Based on the CRITFC Fish Consumption Survey (CRITFC 1994), tribal fish consumers are also most likely to consume only the fillet portion of the fish, which may not include skin. However, some individuals may consume other portions of the fish, and the whole body diet is the most conservative estimate of potential cumulative risk from tissue consumption, as organic chemicals have the greatest contribution to risk. For an individual who consumes primarily fillet tissue, it would be appropriate to focus on risk results from fillet tissue consumption, recognizing that the risks are based on fillet with skin tissue and that risks associated with fillet without skin would likely be even lower for organic chemicals.

While it is not known to what extent consumption of non-fillet portions of fish occurs, this BHHRA evaluated risks associated with consumption of only fillet tissue or only whole body tissue. This approach provides the potential range of risks associated with the different diets, and the risks from consumption of fillet tissue without skin would likely be even lower than those presented in this BHHRA. If an individual consumes mostly fillets, but occasionally other portions of the fish, the risks to that individual should fall within the range of risks estimated in this BHHRA. Because it is unlikely that a diet consists entirely of whole body tissue, the evaluation of risks associated with consumption of only whole body tissue provides a health protective approach.

6.1.4 Use of Undepurated Tissue to Represent Clam Consumption

Clam tissue throughout most of the Study Area was analyzed as undepurated samples, and a limited number of clam samples were depurated before analysis. A common practice in the preparation of clam tissue for consumption includes depuration, although undepurated clam may also be consumed. The amount of COPC-containing particles within the gut of bivalves can vary widely; however, studies have demonstrated that the sediment content in the gut of bivalves could represent up to 39% of the total body load of metals (Wallner-Kersanach et al. 1994). With the exception of a few metals, average chemical concentrations were higher in undepurated clam tissue collected at the Study Area than in depurated clam tissue collected at the Study Area. However, depurated clam tissue accounted for only five of the 22 clam samples collected for the BHHRA dataset, and the depurated samples were collected from edges of the site (northern and southern stretches). Therefore, there are uncertainties associated with comparing depurated and undepurated tissue in the BHHRA dataset. These concentrations are shown in the EPC tables in Section 3 (Tables 3-24 and 3-25). Using analytical concentrations of undepurated tissue to represent tissue consumption throughout most of the Study Area provides a health-protective approach to assessing risk from clam tissue consumption.

6.1.5 Use of Different Tissue Types to Assess the Same Chemical

For resident tissue samples from the Round 1 sampling event, mercury was analyzed in fillet tissue without skin. For resident tissue samples from the Round 3 sampling event, mercury was analyzed in fillet tissue with skin. The BHHRA resident species included in the Round 3 tissue sampling were smallmouth bass and common carp. These fillet datasets were combined for Study Area analysis. For the reasons presented in Section 6.1.3, the comparability of analytical data from fillet tissue with skin and fillet tissue without skin creates uncertainty in the BHHRA. Because mercury preferentially accumulates in muscle tissue, one would expect mercury concentrations to be slightly higher in fillet tissue samples without skin. However, for the smallmouth bass, mercury concentrations were generally higher in fillet tissue with skin, and in common carp, mercury concentrations were generally higher in fillet tissue without skin. A comparison of mercury tissue concentrations is provided in Table 6-3. The uncertainty associated with the use of different tissue types to assess risks from mercury should not impact the conclusions of this BHHRA.

6.1.6 Detection Limits That Are Above Analytical Concentration Goals (ACGs)

Uncertainty exists in the evaluation of chemicals that were not detected for which the method detection limits (DLs) exceed the ACGs. Site-specific ACGs were established for each media. However, ACGs for some chemicals are exceptionally low, and in some instances, not attainable with present laboratory methods. DLs for chemicals that were analyzed but never detected were compared to the appropriate ACG for each media. For sediment, maximum DLs exceed both ACGs and method reporting limits (MRLs) for four analytes (see Table 6-4).

In tissue, maximum DLs exceed ACGs and MRLs for eight analytes (see Table 6-5). Five chemicals were never detected in tissue, but their DLs were below ACGs. It should be noted that DLs were above ACGs for PAHs, and PAHs were not detected in Round 1 fish tissue. However, fish metabolize and excrete PAHs, and thus there is less likelihood for PAHs to bioaccumulate in fish. PAHs were detected in Round 3B fish tissue, as well as in Round 1, 2, and 3B shellfish tissue, indicating that data were sufficient to estimate risk from PAHs in both fish and shellfish tissue. As discussed in Attachment F2, when a non-detected result was greater than the maximum detected concentration for a given exposure area, that result was removed from the dataset prior to calculation of an EPC. When a non-detected result was less than the maximum detected concentration, it was included in the dataset for calculation of EPCs according to the rules presented in Attachment F2. These data rules apply to non-detected PAHs in Round 1 fish tissue. In addition, DLs for PCB congeners were elevated for some smallmouth bass tissue samples, which may add uncertainty to PCB TEQ estimates. However, the risks from total PCBs (due to detected congeners) were higher than the risks from the PCB TEQ for those exposure areas with elevated detection limits. Because the PCB congeners were detected in other smallmouth bass tissue samples, the elevated DLs were incorporated in the PCB TEQ estimates at one half the DL. Therefore, while the elevated detection limits contribute to uncertainty, using the elevated detection limits in this BHHRA should not significantly affect the risk results.

In the groundwater seep sample, maximum DLs exceed both ACGs and MRLs for one analyte (see Table 6-6). In surface water samples, five analytes plus PCB Aroclors exceed ACGs; two analytes plus PCB Aroclors exceed MRLs (see Table 6-7). However, for surface water PCB congener data were used instead of Aroclor data, as discussed in Attachment F2.

Chemicals that were not detected were not quantitatively evaluated further in this BHHRA. If chemicals were present at concentrations above the ACGs but below the DLs, those chemicals could contribute to unacceptable risks. However, given the number of chemicals that were detected at concentrations above their respective ACGs and the magnitude of difference between detected concentrations and ACGs, it is unlikely that exclusion of chemicals that were not detected would impact the conclusions of this BHHRA.

6.1.7 Removal of Non-Detected Results Greater Than the Maximum Detected Concentration for a Given Exposure Area

As discussed in Attachment F2, if a given non-detected result was greater than the maximum detected concentration for an exposure scenario and exposure area, that result was removed from the dataset prior to calculation of EPCs. These results are discussed in Attachment F2 and presented in tables F2-7 through F2-13. Inclusion of non-detected data greater than the maximum detected concentrations would likely have resulted in higher risk estimates in the risk characterization of the BHHRA.

6.1.8 Using N-Qualified Data

As discussed in Section 2.2.3 of the RI report, some data were qualified using the “N” qualifier, which indicates that the identity of the analyte is not definitive. The use of the N qualifier is generally a result of the presence in the sample of an analytical interference such as hydrocarbons or, in the case of pesticides, PCBs. Pesticide data and SVOCs analyzed by EPA Method 8081A were most commonly N-qualified as a result of analytical interference. N-qualified data were used in the BHHRA for calculating tissue EPCs (for hexachlorobenzene and several pesticides) that resulted in cancer risk estimates exceeding 1×10^{-6} or HIs exceeding 1. Alpha-hexachlorocyclohexane, beta-hexachlorocyclohexane, and gamma-hexachlorocyclohexane were identified as contaminants potentially posing unacceptable risks in fish tissue based on EPCs that were calculated using only N-qualified data. Heptachlor epoxide was identified as a contaminant potentially posing unacceptable risks in clam tissue based only on N-qualified data. While these contaminants were identified as contaminants potentially posing unacceptable risks based on the results of the BHHRA, it is important to note that there is uncertainty in both the identity and concentration of these contaminants. These contaminants were not detected in abiotic media at levels posing risk to human health. Attachment F6 discusses how EPCs and risk estimates would change for adult consumption of whole body fish tissue and shellfish tissue if N-qualified data were not included in the BHHRA dataset.

6.1.9 Using One-Half The Detection Limit for Non-Detect Results in Summed Analytes

When an individual analyte that is part of a summed analyte (i.e. total PCB congeners, total endosulfans, etc.) was determined to be present in a given medium according to the rules for non-detects discussed in Section 2, but was not detected for a specific sample, one-half of the detection limit was used to calculate the summed analyte result, as described in Attachment F1. This value is assumed to represent a conservative estimate for the concentrations below the detection limit, and introduces uncertainty into the summed analyte calculations. In general, the detection limits for non-detect results were low relative to detected concentrations. In addition, by only including those contaminants that were determined to be present in a given medium, the uncertainty associated with the use of non-detect results was minimized. However, in cases where the detection limits were above analytical concentration goals and the chemical was detected infrequently, use of one-half the detection limit could impact the risk results.

6.1.10 Contaminants That Were Not Analyzed in Certain Samples

Per the sampling and analysis plan that was approved by EPA, certain fish tissue samples were analyzed for a subset of the analytes. For example, Round 1 fillet tissue samples were not analyzed for PCB, dioxin, or furan congeners. In Round 3B, smallmouth bass and common carp fillet tissue samples were analyzed for PCB, dioxin, and furan congeners. In samples where congeners were analyzed, the risks from the total TEQ, which is not included through other analytes (i.e., risks from total PCBs are included through PCBs as Aroclors) comprise approximately 1 to 70 percent of the cumulative risks. Therefore, the risks from consumption of black crappie and brown bullhead fillet tissue, which were only analyzed in Round 1, likely underestimate the actual risks. However, a range of risks was calculated for fish consumption scenarios, which included samples that were analyzed for congeners, so the lack of analysis of contaminants in certain samples should not impact the conclusions of this BHHRA.

In addition, not all clam samples were analyzed for the same number of contaminants, due to lack of available tissue mass for some composites collected during the Round 2 sampling efforts. Missing analytes and associated sample identifications for clam tissue collected in Round 2 are shown in Table 6-8. In Round 3B, additional clam samples were collected and analyzed for additional contaminants. The Round 2 and Round 3B clam tissue data were combined and evaluated on a river-mile basis in the BHHRA. Therefore, EPCs were available for almost all COPCs in each exposure area. Lack of analytical values for COPCs in all samples within an exposure area may over or underestimate the risk for that exposure area. However, a range of risks

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was calculated for shellfish consumption scenarios, which included samples where all COPCs were analyzed, so the lack of analysis of contaminants in certain samples should not impact the conclusions of this BHHRA.

6.1.11 Chemicals That Were Not Included as Analytes

It is not possible to analyze for every chemical, and thus chemicals and chemical groups were chosen for analysis based on an investigation of known or probable sources and pollutants. Because chemicals expected to have the potential for significant contributions to risk are included in the risk assessment, chemicals not included as analytes introduce a low level of uncertainty to overall risk. The list of chemicals for analysis was determined in collaboration with EPA and its partners and was included in the sampling and analysis plan that was approved by EPA. Since then, there has been interest in two groups of chemicals that were not included as analytes in this BHHRA: polybrominated diphenyl ethers (PBDEs) and volatile organic compounds (VOCs) in tissue. Risks have subsequently been assessed for exposures to PBDEs in in-water sediment and resident fish tissue, as presented in Attachment F3.

VOCs were not analyzed in the BHHRA tissue or surface water datasets. Because of the nature of VOCs, they are not expected to accumulate in tissue to a degree high enough to pose significant risk via tissue consumption, especially given the other chemicals detected in tissue that are clearly primary contributors to the calculated risk (e.g., PCBs). Given the magnitude of concentrations and toxicities of other chemicals that were analyzed for and detected in surface water and tissue, VOCs are unlikely to contribute significantly to the overall risks. Therefore, the lack of analysis for VOCs should not impact the conclusions of this BHHRA.

As mentioned earlier in this section, it is impossible to analyze for every chemical, and there are a number of constituents that have not been historically considered as contaminants but are recently gaining attention as research provides documentation that they are ubiquitous in the environment. These chemicals are generally referred to as “emerging contaminants”, and are not considered in this BHHRA, with the exception of PBDEs, which are discussed in Attachment F3. In accordance with EPA guidance on risk assessment for superfund sites, this BHHRA assessed risks associated with CERCLA releases, and did not include studies focused on non-CERCLA releases, which include some recent studies on regional emerging contaminants. From a human health perspective, unregulated chemicals such as emerging contaminants may exist at the Site, but lack of knowledge and data regarding many of these chemicals precludes a human health risk assessment. Because emerging contaminants are not related to CERCLA releases for the Study Area, the lack of analysis for these chemicals should not impact the conclusions of this BHHRA.

6.1.12 Chemicals That Were Analyzed But Not Included in BHHRA

Not all chemicals analyzed for were included in the BHHRA. Specifically, not all conventional analytes or nutrient metals were analyzed for potential risk. Many conventional analytes are essential nutrients, and are not evaluated under the CERCLA program. The two conventionals that were included in this BHHRA are cyanide and perchlorate. The conventional analytes and metals that were excluded from assessment are listed here:

- Ammonia
- Calcium
- Calcium carbonate
- Carbon dioxide
- Chloride
- Ethane
- Ethylene
- Magnesium
- Methane
- Nitrate
- Nitrite
- Oxygen
- Phosphate
- Phosphorus
- Potassium
- Silica
- Sodium
- Sulfate
- Sulfide

Because of the lack of toxicity and/or essential nature of these analytes, exclusion of these chemicals from the BHHRA should not impact the conclusions of this BHHRA.

6.1.13 Data Not Included in BHHRA due to Collection Date

Data collected after June 2008 were not included in this BHHRA due to the collection date of the data relative to the RI/FS completion schedule. These data sets are discussed in the Portland Harbor RI Report, and include a number of in-water sediment samples. Because these data were not included in the BHHRA, there is uncertainty in the in-water sediment exposure scenarios. However, due to the large spatial coverage of the existing in-water sediment BHHRA dataset, this uncertainty is not expected to impact the overall conclusions of this BHHRA.

6.1.14 Compositing Methods for Biota and Beach Sediment Sampling

Compositing methods for biota and beach sediment sampling were designed to provide a conservative estimate of risk. Compositing schemes need to be developed to be representative of the medium sampled (grid pattern, stratified random, etc.) and to be representative of an exposure unit.

Fish were composited based on an estimate of the average home range for each species. The home ranges for common carp and brown bullhead may be as large as the Study Area and possibly even larger, and the home range for bass may be larger or smaller than the one mile assumed in the BHHRA. For example, bass may only reside on one side of a river mile reach instead of throughout the one mile reach on both sides of the river as assumed for the HHRA. Smallmouth bass were composited on a river mile basis, while black crappie, brown bullhead, and carp were composited on a fishing zone basis. Fishing zones for brown bullhead and black crappie were from RM 3-6 and RM 6-9; fishing zones for common carp were from RM 0-4, RM 4-8 and RM 8-12 as well. Uncertainty exists in this compositing scheme because the delineation of home range boundaries for the purposes of the risk evaluation are only an approximation of the home ranges of the fish samples actually collected. However, composite samples typically consisted of five individual fish, replicate composite samples were collected, and risks were evaluated both for individual sample locations as well as on a Study Area-wide basis. Therefore, the compositing method for biota is not expected to impact the conclusions of this BHHRA.

Beach sediment was composited on a beach by beach basis, resulting in one sample for each exposure area. Uncertainty exists in this compositing scheme because the results of the risk evaluation are dependent on a single sample. Composite samples are generally assumed to represent the area from which the individual samples of the composite were taken, but an unrepresentative individual sample (e.g., one representing extremely localized or ephemeral contamination) used in the composite could significantly bias the composite results. The compositing scheme for beaches results in risk evaluation based on a single sample at a single point in time. If a beach was found to pose an unacceptable risk, additional samples at that beach might be warranted. However, all of the beach sediment exposure scenarios ranged from 8×10^{-9} to 9×10^{-5}, which are below or within the target risk range of 10^{-4} to 10^{-6}.

6.1.15 Mislabeling of Smallmouth Bass Fish Sample

One smallmouth bass sample collected from the west side of RM 11 (LW3-SB11W-11) during the Round 3 sampling event was incorrectly recorded as LW3-SB11E-01 (RM 11 east) at the field lab. This fish became part of the final LW3-SB11E-C00B and LW3-SB11E-C00F composite samples, which are the body and fillet composites from RM 11 east. Fish SB11E-01 (actually from SB11W) accounted for 15% of both sample types on a mass basis. This results in uncertainty in the concentration of the smallmouth bass sample from the east side of RM 11, since a fish from outside RM 11E was included in the composite. However, since smallmouth bass exposure areas are on a river mile basis, the data from RM 11E and RM 11W were included in the same EPC calculations, and the effects of this uncertainty are not expected to impact the conclusions of this BHHRA.

6.1.16 Use of DEQ Risk-Based Concentrations for Screening Values

EPA RSLs were used to screen chemicals detected in in-water sediment for the identification of COPCs. RSLs are not available for petroleum hydrocarbons, so DEQ risk-based concentrations (RBCs) for occupational surface soil exposure DEQ 2003) were used. DEQ does not have specific RBCs for lube oil, motor oil, or residual range hydrocarbons, so the screening value for generic oil was used as a surrogate. There is uncertainty associated with applying the screening value for generic oil to heavier oils, as lighter range petroleum hydrocarbons tend to be more toxic than heavier-range petroleum hydrocarbons. However, the maximum detected concentrations of these three oils in in-water sediment also does not exceed the screening value for the lighter range hydrocarbons detected within the Study Area (diesel, gasoline), so the uncertainty associated with the COPC screening values for heavier oils are not expected to impact the conclusions of this BHHRA.

6.1.17 Selection of Tissue COPCs Based On Detection of An Analyte

The selection of fish and shellfish tissue COPCs was based on whether an analyte was detected in each species/tissue type, and not based on a comparison with health-protective screening levels. There is uncertainty associated with identification of tissue COPCs based on detections alone, and this could potentially impact the conclusions of this BHHRA.

6.2 EXPOSURE ASSESSMENT

Uncertainties that arise during the exposure assessment typically have some of the greatest impacts on the risk estimates. The following subsections address uncertainties associated with exposure models, exposure scenarios, exposure factors, and EPCs used in the risk estimates.

6.2.1 Model Applicability

The standard exposure models used to estimate risks may result in uncertainty. The exposure models rely on identification of exposure scenarios and selection of appropriate exposure factors for those scenarios. Uncertainty in the applicability of the exposure scenarios will result in uncertainty in the risk estimates. Site-specific exposure scenarios were developed to provide a conservative estimate of risk within the Study Area, using conservative exposure factors to represent both reasonable maximum and central tendency exposures that could hypothetically occur within the Study Area. While uncertainties associated with the exposure models could impact the conclusions of this BHHRA, the models used are consistent with applicable risk assessment guidance and are a source of uncertainty in all risk assessments.

6.2.2 Subsurface Sediment Exposure

A complete exposure pathway needs to include retention or a transport medium, an exposure point, and an exposure route. Subsurface sediment was not considered an exposure medium for this BHHRA because it was assumed that any potential human contact with river sediment below 30 cm in depth was unlikely, and if it does occur, the frequency and extent would be minimal. Situations in which exposure to subsurface might occur include: potential scouring, natural hydraulic events that are not well understood, future development of near-shore and upland properties, maintenance of the federal navigation channel, ports, and docks, placement and maintenance of cable and pipe crossings, pilings and dolphins, anchoring and spudding of vessels, and exposure to propeller wash from vessels. All of these situations could provide minimal impact to subsurface in-water sediment as well as to surface sediment, and thus the assessment of risk from exposure to surface sediment

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would be adequately protective of potential exposure to subsurface sediment. However, the uncertainty associated with not directly assessing subsurface sediment exposure could underestimate risks from multiple exposure pathways for the Study Area. Due to the low levels of possible exposure to subsurface sediment, this uncertainty is not expected to impact the conclusions of this BHHRA.

6.2.3 Potential Exposure Scenarios

Some of the exposure scenarios evaluated in this BHHRA have limited documentation regarding the actual extent of exposure to receptors in the Portland Harbor. These scenarios were included in this BHHRA at the direction of EPA Region 10. The uncertainties associated with these scenarios are discussed in the following subsections.

6.2.3.1 Human Milk Consumption

The BHHRA evaluated risks to an infant consuming human breastmilk for receptors exposed to bioaccumulative compounds selected as COPCs. The evaluation of this pathway was performed consistent with DEQ guidance (2010), but there are a number of uncertainties associated with modeling infant exposure to contaminants through breastmilk based on exposure to the mother, which could potentially affect the outcomes of this BHHRA.

Risks to an infant consuming breastmilk from the adult receptors evaluated in this BHHRA resulted in risks above the EPA points of departure for cancer and noncancer endpoints. However, breastfeeding is still the healthiest way to feed a baby, even if the milk contains contaminants. Even though infants may receive a dose of contaminants from their mothers' milk, human milk also contains hundreds of healthy nutrients, vitamins, minerals, and immune system boosters. These natural, healthy substances more than compensate for any health risks from contaminants and may even help repair damage caused by contaminants before the baby was born. Breastfeeding has been shown to boost immunity and IQ and prevent many diseases. Calculated risk to infants from breastfeeding presented in this report should not discourage any mother from breastfeeding her infant (adapted from DEQ, 2010).

6.2.3.2 Shellfish Consumption

This BHHRA evaluated risks from shellfish consumption based on crayfish and clam tissue data. However, the harvest or possession of Asian clams, which is the species assessed in this BHHRA, is illegal.

A commercial crayfish fishery exists in the LWR. Crayfish landings must be reported to ODFW by water body and county. Per ODFW, the crayfish fishery in the LWR is not considered a large fishery (Grooms 2008). Based on ODFW's data for 2005 to 2007, no commercial crayfish landings were reported for the Willamette River in Multnomah County. DHS had previously received information from ODFW indicating that an average of 4300 pounds of crayfish were harvested commercially from the portion of the Willamette River within Multnomah County each of the five years from 1997-2001. In addition to this historical commercial crayfish harvesting, DHS occasionally receives calls from citizens who are interested in harvesting crayfish from local waters who are interested in fish advisory information. According to a member of the Oregon Bass and Panfish club, crayfish traps are placed in the Portland Harbor Superfund Site boundaries and collected for bait and possibly consumption (ATSDR 2006). It is not known to what extent non-commercial harvesting of crayfish occurs within the Study Area, if at all, or whether those crayfish are consumed and/or used for bait.

The only reported clam consumption was from a project conducted by the Linnton Community Center (Wagner 2004). As part of the project, conversations were conducted with transients about their consumption of fish or shellfish from the Willamette River. These conversations were not conducted by a trained individual nor were the conversations documented. The transients that were contacted reported consuming various fish species, as well as crayfish and clams. Many of the individuals indicated that they were in the area temporarily, move from location to location frequently, or have variable diets based on what is easily available. Assuming that clam consumption occurs, the Linnton Community Center project suggests that it does not occur on an ongoing basis within the Study Area.

The evaluation of risks from shellfish consumption in this BHHRA is a health protective approach.

6.2.3.3 Wet Suit Divers

Commercial diving companies in the Portland area were contacted to develop a better understanding of potential diver exposures within the Study Area. All of the diving companies that were contacted indicated that the standard of practice for commercial divers is the use of dry suits and helmets when diving in the LWR (Hutton 2008, Johns 2008, and Burch 2008). EPA Region 10 reported observing divers in wet suits and with regulators that are held with the diver’s teeth within the Study Area, so a wet suit diver and associated ingestion for the “in the mouth” regulator exposure scenarios were included at the direction of EPA. Evaluation was also performed of helmet diving with use of a neck dam, which allows polluted water leakage into the diving helmet. Commercial divers as recently as 2009 have been observed using techniques to don a diving helmet which increase exposure (Sheldrake personal communication with RSS, 2009, DEQ, 2008). The observed wet suit divers were performing environmental investigation and remedial activities, which are not activities evaluated as part of a commercial diver scenario. Also, it is not known whether the individuals who were observed diving in wet suits on specific occasions are diving within the Study Area on a regular basis, as they do not work for the commercial diving companies in the Portland area. Recreational diving also takes place in Portland Harbor (Oregon Public Broadcasting Think Out Loud, "Are you going to swim in that?" August 22, 2008). Therefore, including a wet suit diver scenario with associated ingestion from use of a recreational type regulator, rather than a full face mask or diving helmet, and full body dermal exposure in this BHHRA (in addition to a dry suit diver scenario) is a conservative approach.

6.2.3.4 Domestic Water Users

The domestic water user risks are based on the hypothetical use of untreated surface water drawn from the Study Area as a domestic water source. Surface water in the LWR within the Study Area is not currently used as a domestic water source. According to the City of Portland, the primary domestic water source for Portland is the Bull Run watershed, which is supplemented by a groundwater supply from the Columbia South Shore Well Field (City of Portland 2008). In addition, the Willamette River was determined not to be a viable water source for future water demands through 2030 (City of Portland 2008).

Under OAR 340-041-0340 Table 340A, domestic water supply is a designated beneficial use of the Willamette River, but only with adequate pretreatment and natural quality that meets drinking water standards. The use of the Willamette River as a domestic water source would only occur after adequate pretreatment to meet Safe Drinking Water Act standards and Oregon rules. As a result, the term hypothetical was used to describe the scenario, which was based on the use of untreated surface water.

Therefore, the evaluation of untreated surface water as a domestic water source, even under hypothetical future conditions, is a conservative approach and is not based on current knowledge of future planned uses of the Willamette River within the Study Area as a domestic water source or based on Oregon rules that require adequate pretreatment.

6.2.4 Potentially Complete and Insignificant Exposure Pathways

Exposure pathways that have been determined to be potentially complete and insignificant were not evaluated further in this BHHRA. As described in Section 3.2, these exposure pathways have a “source or release from a source, an exposure point where contact can occur, and an exposure route by which contact can occur; however, the pathway is considered a negligible contributor to the overall risk”. The exposure pathways identified as potentially complete and insignificant were related to Willamette River surface water exposures to populations evaluated in this BHHRA. The populations that are expected to have the most frequent contact with surface water (transients, recreational beach users, and hypothetical future residents) as well as the EPA directed evaluation of surface water exposure to divers were quantitatively evaluated in this BHHRA for ingestion and dermal absorption of chemicals from surface water. The populations for which surface water exposures were not evaluated were for dockside workers, in-water workers, tribal fishers, and fishers. For several other populations, only the inhalation exposure pathway was determined to be insignificant. These populations were transients, divers, recreational beach users, and hypothetical future residents.

This BHHRA identified and evaluated the exposure pathways that were expected to result in the most significant exposure to COPCs in the Study Area. The magnitude of exposures experienced by populations for these exposure pathways are much greater than that expected for the exposure pathways identified as “insignificant”. Thus, the assessment of risk to populations from exposure pathways that were quantitatively evaluated in this BHHRA would be adequately protective of exposed populations in the Study Area. However, the uncertainty associated with not directly evaluating “insignificant” exposure pathways could underestimate risks for the Study Area. Due to the low levels of possible exposure for these “insignificant” exposure pathways, this uncertainty is not expected to impact the conclusions of this BHHRA.

6.2.5 Exposure Factors

Assumptions about exposure factors typically result in uncertainty in any risk assessment. RME and CT values were used for some of the exposure scenarios to evaluate the overall impact that variability in each of the exposure assumptions has on the risk estimates. As discussed previously, most of the RME scenarios represent the reasonable maximum exposures that could occur in the Study Area under current and future conditions. In the case of the scenarios assessing the use of untreated surface water as a domestic water source, both the RME and CT scenarios represent hypothetical exposures. The other CT exposure scenarios represent the expected average or mean exposure for exposures that could occur in the Study Area in the present and future. The range of risk estimates between these two exposure scenarios provides a measure of the uncertainty surrounding these estimates.

For fish consumption, a range of ingestion rates were used to evaluate variability on the risk estimates (see discussion of exposure parameters for tissue ingestion scenarios below). As recommended by EPA guidance, these ingestion rates were used with EPCs calculating using both the mean and 95% UCL on the mean (or maximum concentrations for EPCs when sample size was less than 5), and thus the resulting risks in this BHHRA represent a range of possible human health risks, including estimates that might fall into the high end of those possible.

In addition to the variability, there is also uncertainty associated with the exposure factors that were used in this BHHRA.

The following exposure factor uncertainties have been identified and analyzed further to determine the potential effects on the risk estimates:

6.2.5.1 Exposure Parameters for Sediment Exposure Scenarios

The beach and in-water sediment exposure parameters used in this BHHRA were conservative estimates of potential uses for the Study Area.

Beach areas that are accessible to the general public were identified as potential human use areas, even though it is not known whether recreational beach use actually occurs at these locations. Even if beach use occurs, the extent to which the beach is used and the nature of the contact with sediments/beach is unknown. Future changes in land use may make some beach areas more or less accessible for humans, which increases uncertainty about future exposure. For in-water sediment, every ½-river mile segment on each side of the navigation channel was considered a potential exposure area for all in-water sediment exposure scenarios, regardless of the feasibility or practicality of use of the area. Information from this approach can be used to inform the public about relative risks throughout the river and can help focus the feasibility study, but likely over-estimates risk estimates for in-water sediment.

The exposure duration, frequency, and intake parameters for both beach and in-water sediment also have associated uncertainties. The scenarios assume exposure to the same beach or ½-river mile segment for an entire childhood, or 25 to 70 year exposure duration for adults, depending on the receptor. Frequency of exposure ranges from 94 days/year to 250 days/year. Default intake parameters for soil exposure were generally used; however, the adherence factor (dermal contact with sediment) for a child recreational beach user was more than 10 times greater than the default for soil.

Another uncertainty associated with exposure parameters for sediment is the dermal absorption factor, which does not exist for all COPCs. Per EPA guidance (2004), only those compounds or classes of compounds for which dermal absorption factors exist were evaluated quantitatively for the dermal contact exposure pathway. For compounds without dermal absorption factors, which for the sediment COPCs are certain metals and perchlorate, dermal intake was assumed to be zero. However, dermal absorption factors exist for the chemicals and chemical groups that are likely to pose the greatest concern for risk from dermal contact. So although the lack of dermal absorption factors for all COPCs may underestimate risk from dermal contact with sediment for certain metals and perchlorate, this uncertainty would not change the conclusions of this BHHRA.

Most of the uncertainties associated with the sediment exposure parameters are likely to overestimate the risks associated with direct exposure to sediment. However, all of the beach sediment exposure scenarios were below or within the target risk range of 1×10^{-4} to 1×10^{-6}, and with the exception of two segments specifically for the tribal fisher RME scenario, all of the in-water sediment exposure scenarios were also below or within the target risk range of 1×10^{-4} to 1×10^{-6}. For the tribal fisher RME scenario, the exposure parameters are especially conservative as it is unlikely that an individual would fish the same ½-river mile river segment for five days every week of every year for 70 years.

6.2.5.2 Exposure Parameters for Surface Water and Groundwater Seep Exposure Scenarios

Transients were assumed to be exposed to surface water through ingestion and dermal contact. Tap water ingestion rates were used to represent exposure to surface water via ingestion for transients. However, tap water ingestion rates are an estimate of ingestion of a drinking water source, and the use of untreated water from the Lower Willamette as a source of drinking water by transients on an ongoing basis for two years is assumed to be health protective. The tap water ingestion rate used in the risk evaluation was 2 L/day for the transient and assumes surface water will be ingested every day for two years. In addition, it was assumed that transients bathe directly in the Lower Willamette two days per week throughout the entire year for two years.

For the recreational beach users, exposure to surface water was assumed to occur through incidental ingestion and dermal contact while swimming in the Lower Willamette. The incidental ingestion rate of 50 milliliters per day (ml/day) used in this BHHRA is that recommended by EPA for a swimming scenario. The exposure scenario assumes that adults frequent the same quiescent water area 26 times per year for 30 years, and that children frequent the same area 94 times per year for six years.

In addition to the direct contact scenarios mentioned above, risks were assessed from exposure to surface water as a hypothetical future domestic water source. This scenario assumes untreated surface water is used as a domestic water source 350 days a year for 30 years (adult resident) or six years (child resident). The LWR within the Study Area is not currently used as a domestic water source, but could be used as such in the future.

Another exposure parameter resulting in uncertainty for the surface water and groundwater exposure parameters is the absorbed dose per event. This parameter was derived per EPA guidance (2004) using chemical-specific factors, but the factors for some of the COPCs fall outside of the predictive domain. Specifically, the dermal permeability coefficient (K_p) falls outside of the effective predictive domain (EPD) for a number of PAHs, including the following COPCs:

- Benzo(a)anthracene
- Benzo(a)pyrene
- Benzo(b)fluoranthene
- Indeno(1,2,3-cd)pyrene
- Dibenzo(a,h)anthracene

EPA guidance (EPA 2004) states that "Although the methodology [for predicting the absorbed dose per event] can be used to predict dermal exposures and risk to contaminants in water outside the EPD, there appears to be greater uncertainty for these contaminants." The range of uncertainty associated with the Kp value can be several orders of magnitude. For instance, the predicted Kp value recommended by EPA (2004) for benzo(a)pyrene is 0.7 centimeters per hour (cm/hr), while the range of predicted Kp values presented by EPA (2004) is 0.024 cm/hr (95% lower confidence level) to 20 cm/hr (95% upper confidence level). This uncertainty could result in over-estimation or under-estimation of risk from exposure to surface water. With the exception of arsenic, the only exceedances of 1 x 10^{-6} risk from surface water scenarios are the result of dermal exposure to PAHs in surface water. However, all of the surface water exposure scenarios were below or within the target risk range of 1 x 10^{-4} to 1 x 10^{-6}.

6.2.5.3 Exposure Parameters for Tissue Ingestion Scenarios

The exposure parameters for tissue ingestion were designed to provide a conservative estimate of risk. Fish tissue ingestion rates were developed using fish consumption data from a national study of fish consumption (CSFII, USDA), from a creel survey of Columbia Slough fishers north of the Study Area, and from the CRITFC Columbia River Fish Consumption Survey (CRITFC) study. The CRITFC Fish Consumption Survey provides fish consumption data for the Columbia River Basin for four of the six tribes who are parties to the Consent Decree for the Portland Harbor site. In addition, although the Columbia Slough Study was not done in Portland Harbor , the Columbia Slough is within one-half mile of the northern part of the Portland Harbor site, so fishers in the Portland Harbor site may have similar fishing practices and fish consumption rates as those fishing in the Slough.

Site-specific fish consumption information is not available for the fisher scenarios. As a result, nationwide fish consumption data were used to calculate target fish tissue levels. A consumption study conducted for the Columbia Slough was also used. The 99th percentile rate from the nationwide Continuing Survey of Food by Individuals, CSFII (United States Department of Agriculture [USDA] 1998) of 142 g/day (as calculated in USEPA Estimated Per Capita Fish Consumption in the United States, freshwater and estuarine fish and shellfish) was used as one ingestion rate for adult fishers in the BHHRA. The 90th percentile rate of 17.5 g/day from the same study was used also used as one of the ingestion rates for adult fishers in the BHHRA. Concerns have been expressed regarding the methodology used by EPA in this study to establish the fish consumption rates, which are also recommended as default AWQC subsistence fish consumption rates in EPA's WQC Human Health Methodology guidance (EPA 2000d). Criticisms of these rates have been raised because they are based on per capita consumption rates from the general population – that is, "fish consumption" rates that are estimated based on the combined consumption information from fish consumers and fish non-consumers alike. For example, the 90th percentile rate for fish consumers is 200 g/day, while the 90th percentile rate including data regarding fish non-consumers is about 18 g/day. Similarly, the 99th percentile value for fish consumers is about 506 g/day, while the 99th percentile is approximately 142 g/day when data including the lack of fish in the diet of non-consumers are added. There is a large difference in the percentiles of the dataset when information from people who do not consume fish are included. The consumer-only ingestion rates likely overestimate actual ingestion rates because people who do consume fish but did not on the 2 days of the study (e.g., many infrequent consumers) are not included in consumers only rate. At the same time, EPA guidance (1989) recommends using the 95th percentile, or even the 90th percentile, for RME contact values. The 95th UCL rate from the Columbia Slough study was used as the 73 g/day rate for adult consumers in the BHHRA. The Columbia Slough Study was a creel survey. As a result, the consumption rates used in the BHHRA may overestimate or underestimate actual fish consumption rates in the Study Area. This is due to many reasons, including but not limited to:

- Willingness of anglers to participate
- Communication. If a substantial number of anglers consist of 1st or 2nd generation ethnic minorities, then language may be a barrier.
- Discrepancy between individuals who catch fish and those who prepare meals. Men generally fish but women generally prepare seafood and are much more familiar with the mass of seafood consumed.
- Difficulty in translating from the items inspected in an angler's basket to portion sizes and amounts consumed, since this requires assumptions about edible portions and cleaning factors.
- Lack of a random or representative sample. Interviewers can only speak with who they encounter.
- Timing and seasonality of interviews.
- Weather conditions may bias the results of any day's interviews.

In addition to the uncertainties behind the rates of fish consumption, it was assumed that the frequency of consumption occurred at the same ingestion rate for 30 years for the adult fisher scenarios. Furthermore, 100% of the fish consumed was assumed to be caught within a 1 mile stretch on both sides of the river for bass and within a 3 mile stretch on both sides of the river for crappie, carp and bullhead trout over 30 years for localized exposures.. No reduction in concentrations of contaminants during food preparation and cooking was assumed, although reductions can occur depending on cooking and methods of preparation.

For the tribal fish consumption scenario, the 95th percentile rate from the CRITFC Fish Consumption Survey (CRITFC 1994) was used. The CRITFC Fish Consumption Survey was performed by interviewing four of the six tribes who are natural resource trustees for the Site. It is not clear how this would impact the fish consumption rate for tribal populations used in the BHHRA, which was based up on the CRITFC Fish Consumption Survey. Also, some published articles have suggested that the fish consumption rates in the CRITFC Fish Consumption Survey are biased low for tribal members because:

- Tribal members who have a traditional lifestyle (and likely a higher consumption rate) would have been unlikely to travel to the tribal offices that were used for administering the CRITFC fish consumption interviews.
- The fish consumption rates for some tribal members that were perceived as being outliers (consumption rates were too high) were dropped from the CRITFC data before the consumption rates were calculated.

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- Current fish consumption rates may be suppressed and, therefore, do not reflect the potential of the higher consumption rates if fishery resources improved or if contaminant concentrations in the water body decrease.

While the tribal fish consumption rates may or may not be biased low, there were additional conservative assumptions incorporated in the tribal fish consumption scenario. For example, fish consumption by an adult tribal fisher was assumed to occur at the same rate every day of every year for 70 years. As with the fisher scenarios, it was assumed that 100% of the fish consumed was caught at the same location for 70 years, and no reduction in concentration of contaminants occurred during food preparation or cooking. The CRITFC Fish Consumption Survey that was used as the basis for the tribal fish ingestion rate also indicated that none of the respondents fished the Willamette River for resident fish and at most, approximately 4% fished the Willamette River for anadromous fish. However, future use of the site by tribal members may change. Tribal members who have a traditional lifestyle and were unlikely to travel to tribal offices for the CRITFC Fish Consumption Survey also may be unlikely to travel to Portland Harbor to fish. It is unknown to what extent future tribal fishing habits may change if fishery resources improved or if COC concentrations in the water body decrease. ODEQ is proceeding with development of state water quality limits based on a tribal ingestion rate of 175 g/day.

The information suggesting that shellfish consumption may occur at the Study Area comes from a community project sponsored by the Linnton Community Center, as discussed in Section 3.3.6. However, it is not known to what extent shellfish consumption occurs. Because site-specific shellfish ingestion rates are not available, nationwide CSFII (USDA 1998) shellfish consumption data were used to calculate target tissue levels for clams and crayfish. The 95th percentile rate for shellfish consumption for freshwater and estuarine habitats combined from the nationwide survey was the source of the18 g/day ingestion rate, and the mean rate from the nationwide survey was the source of the 3.3 g/day ingestion rate. As with the fish ingestion rates for adult consumers, these shellfish ingestion rates are based on per capita consumption rates from the general population – that is, consumption rates that include shellfish consumers and non-consumers alike. Consumer-only rates were not calculated in the EPA document for shellfish alone, but it is likely that they are higher for consumers-only compared to the rate based on both consumers and non-consumers. In the nationwide survey, shrimp, which is not found within the Study Area, accounted for more than 80% of the shellfish consumed. Crayfish accounted for less than 1% of the shellfish consumed, and freshwater clams were not included in the nationwide survey. It is not known to what extent fishers substitute alternative local types of shellfish. However, for freshwater habitat only, which is the same as the Study Area, the mean nationwide shellfish consumption rate is 0.01 g/day; upper percentiles for freshwater shellfish consumption rates are not available (EPA 2002b).

Daily shellfish consumption rates used in this BHHRA represent mathematical artifacts to account for annual consumption rates. The daily consumption rates for shellfish represent approximately two and a half 8-ounce meals per month (18 g/day ingestion rate), and just less than one 8-ounce meal every two months (3.3 g/day ingestion rate). As with fish, 100 percent of the shellfish was assumed to be caught from the same one-mile stretch of river, on the same side of the river, for the 30 years, and no losses in chemical concentration were assumed from food preparation or cooking. It is unlikely that the Study Area supports *Corbicula* populations large enough to supply the quantity of tissue needed to satisfy the ingestion rates used in the BHHRA. During the Round 2 sampling event, the maximum mass of clam tissue data collected at a given sampling location was only 217.57 grams. At 18 g/day, this location would be depleted of clam tissue within 13 days. However, following EPA direction, bivalve consumption is treated as a potential future exposure pathway at the rates used in the BHHRA.

Most of the uncertainties associated with the fish and shellfish exposure parameters provide a conservative estimate of the risks associated with fish and shellfish consumption. Because noncancer hazards and cancer risks associated with consumption of fish and shellfish exceeded the NCP target noncancer hazard quotient of one and the cancer risk range of 1 x 10^{-4} to 1 x 10^{-6} as well as the point of departure of 1 x 10^{-6}, the uncertainties associated with fish and shellfish consumption could affect the decisions made in the FS. The upper and lower bound magnitude of uncertainty associated with exposure parameters for tissue ingestion scenarios was estimated for the BHHRA based on the data presented above, and is discussed in Attachment F6.

6.2.5.4 Assumptions about a Multi-Species Diet

Uncertainties exist in the assumptions about the multi-species diet composition. The non-tribal multi-species diet assumes equal proportions of all four resident fish species. The tribal multi-species diet consists of equal proportions of the four resident fish species, as well as dietary percentages of salmon, lamprey, and sturgeon that come from the CRITFC Fish Consumption Survey (CRITFC 1994). Variations from these compositions would result in different risk estimates. Because the risks from consumption of the individual species that make up the multi-species diet were evaluated separately, the range of risks from fish consumption scenarios encompasses the potential variations in the multi-species diet. The range of the magnitude of these risks was between 1 and 8. The derivation of these risk ranges is further discussed in Attachment F6. The magnitude in the difference of risk estimates based on diet composition shows that this uncertainty could result in over or under-estimation of actual risks from a multi-species diet.

6.2.6 Exposure Point Concentrations

The EPC is supposed to represent the arithmetic average of the concentration of a contaminant that will be contacted over the exposure duration; however, as a protective approach, a UCL on the arithmetic average is recommended for use as the EPC (EPA 1989). Given the uncertainties and variability associated with environmental data, a high amount of uncertainty is associated with calculating a representative EPC. The following EPC uncertainties have been identified and were analyzed further in the BHHRA to determine the potential effects on the risk estimates.

6.2.6.1 Using 5-10 Samples to Calculate the 95% UCL on the Mean

Using less than ten sample results to calculate a 95% UCL on the mean increases the uncertainty associated with the 95% UCL for certain calculation methods. EPCs for a number of exposure areas throughout the Study Area were based upon the 95% UCL on the mean concentration calculated using less than 10 samples. These EPCs are discussed and listed in Attachment F2 text and tables. They include EPCs for in-water sediment, surface water, and tissue. Calculating the 95% UCL on the mean using less than 10 samples could overestimate or underestimate actual exposures. The Study Area-wide fish tissue EPCs that were calculated as 95% UCL on the mean concentrations, using less than 10 samples, included the Study Area-wide EPCs for whole body brown bullhead and fillet common carp. The maximum EPCs for the individual exposure points for whole body brown bullhead and fillet common carp were up to two times higher than the Study Area-wide EPCs, as discussed in Attachment F6.

If maximum detected concentrations had been used as EPCs in place of 95% UCL on the mean concentrations for exposure areas with less than 10 samples, exposures would have likely resulted in an overestimate of actual risks.

6.2.6.2 Nondetects Greater than Maximum Detected Concentrations

Individual non-detected analytical results for which the detection limit was greater than the maximum detected concentration in a given exposure area were removed from the dataset prior to 95% UCL calculations. These sample identifications, detection limits, and associated maximum concentrations are discussed and listed by media and exposure area in Attachment F2 text and tables. A nondetect concentration means the actual concentration of the chemical could be as high as the detection limit, or it could be not present. However, if a detection limit exceeds the maximum detected concentration in a given exposure area, it is unknown whether the actual concentration is closer to zero or closer to the detection limit. Removal of these data prior to 95% UCL calculations decreases the need for assumptions about what the actual concentration may be, but it also decreases overall sample size for a given chemical and exposure area.

As discussed in Section 5.2.5, PCBs are the primary contributor to the cumulative risks for all of the fish tissue consumption scenarios, and dioxins are the secondary contributor. There were no cases for which nondetect concentrations exceeded the maximum detected concentration of PCBs and dioxins in fish tissue. It follows that the cases where nondetect concentrations exceeded the maximum detected concentrations did not impact the cumulative risk estimates. PCBs and dioxins were also the primary contributor to cumulative risk for shellfish tissue consumption and there were no cases where nondetect concentrations exceeded the maximum detected concentration of PCBs and dioxins in shellfish tissue. For surface water and in-water sediment the ratio of the nondetect concentrations exceeding the maximum detected concentrations were within two orders of magnitude. If the actual concentrations were closer to the detection limit, the risk estimates would still be less than 1×10^{-6}.

6.2.6.3 Using the Maximum Concentration to Represent Exposure

For cases with less than five detected samples for a given analyte and exposure area, the sample size was not sufficient to calculate a 95% UCL on the mean concentration for an EPC, and the maximum concentration was used. This includes EPCs calculated to represent Study Area-wide exposure. Using maximum detected concentrations of infrequently detected contaminants to represent individual exposure areas, and especially Study Area-wide exposure, results in an extremely conservative estimate of risk for the Study Area. In general, use of 95% UCL on the mean concentrations or maximum concentrations provided a protective approach and likely resulted in overestimates of the actual risks, especially for ongoing, repeated, long-term exposures. Use of the maximum concentration to represent exposure occurred for all media, and occurred most frequently for the fish and shellfish consumption scenarios. Contaminants and exposure points for which the maximum detected concentration was used instead of a 95% UCL on the mean are presented in the exposure point concentration tables in Section 3. In some cases, the maximum concentration for a contaminant was anomalously high, and may not be representative of tissue concentrations resulting from exposure to CERCLA-related contamination within the Study Area.

Generally, the ratios between the maximum and minimum detected concentrations are less than 3. For in-water sediments, the ratios are less than 4. When comparisons are made within an exposure area for biota, the majority of the ratios of the 95% UCL/maximum EPCs to the mean are equal to or less than 2, and the remaining ratios are less than 4. A more in-depth analysis of scenarios for which using the maximum concentration to represent exposure significantly affected the result of the risk estimate, and consequently which chemicals were designated as contaminants potentially posing unacceptable risks for a scenario, is provided in Attachment F6.

The conservatism of using the maximum detected concentration as the EPC for exposure areas with less than 5 detected results impacts the conclusions of this BHHRA.

6.2.6.4 Possible Effects of Preparation and Cooking Methods

Cooking and preparation methods of fish tissue can modify the amount of contaminant ingested by fish consumers. The EPA (1997b) states that “cleaning and cooking techniques may reduce the levels of some chemical pollutants in the fish”. PCBs, which were found to have the greatest contribution to the cumulative cancer risks and the highest noncancer HQs, tend to concentrate in fatty tissues. Therefore, trimming away fatty tissues, including the skin, may reduce the exposure to PCBs.

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The concentrations of PCBs in raw fillet tissue have been shown to decrease by approximately 50% by removing the skin (EPA 2000c). Cooking can also reduce the concentrations of PCBs up to 87%, depending on the method (Wilson et al. 1998). However, one study showed a net gain in PCB concentrations after cooking (EPA 2000c).

As per EPA directive, dose modifications to account for cooking or tissue preparation were not used in determining EPCs for fish ingestion. If included, the risk estimates may have been reduced by up to approximately 90% for some contaminants. Since PCBs contribute to the majority of risks from fish consumption, this uncertainty could significantly impact the results of this BHHRA. For other contaminants, particularly mercury, which accumulates in the muscle tissue of fish, cooking is not known to reduce the concentrations in tissue; however, mercury does not contribute to the cumulative cancer risks.

6.2.6.5 Assumptions about Arsenic Speciation

Arsenic in tissue was analyzed only as total arsenic. Toxicity data are only available for inorganic arsenic. The Columbia River Basin Fish Contaminant Survey (EPA 2002c) determined that a "value of 10% is expected to result in a health protective estimate of the potential health effects from arsenic in fish". Therefore, the EPC for inorganic arsenic was estimated as 10% of the total arsenic detected in tissue. In previous fish tissue studies in the lower Columbia and Willamette Rivers, the percent of inorganic arsenic relative to total arsenic ranged from 0.1% to 26.6% with an average percent inorganic arsenic of 5.3% in the resident fish samples from the Willamette River (Tetra Tech 1995, EVS 2000).

In clams, inorganic arsenic was found to range as high as 50% of total arsenic in tissue collected in the Duwamish River. However, the Duwamish River is an estuary while Portland Harbor is a freshwater river, so the species of clams in the Duwamish River are different from those in Portland Harbor. Since the actual percent of arsenic that is inorganic in clam tissue from the Study Area is unknown, this results in uncertainty in the estimate of inorganic arsenic EPCs for clam. The clam tissue data collected from the Study Area in Rounds 1 through 3 was evaluated to determine whether a higher percentage of inorganic arsenic might have a significant effect on overall risk from the consumption of clam tissue. The analysis found:

- All of the arsenic concentrations in clam tissue are within a factor of 2 of each other (i.e., the maximum concentration is approximately 2 times higher than the minimum concentration). In addition, the arsenic concentrations in clams are normally distributed. Both of these facts support the conclusion that the arsenic in clams is due to ubiquitous concentrations, not localized sources.
- Due to the narrow range of arsenic concentrations, the risks from consumption of clams are within a factor of 2 throughout the Study Area.
- If inorganic arsenic is assumed to be 50% of the total arsenic rather than the assumption of 10% used in the BHHRA, the cumulative risks from

consumption of clams only increase by a factor of 1.1 to 1.3 because there are other contaminants that are primary contributors to risks from consumption of clams.

Given all of the other uncertainties associated with risks from clam consumption, the inorganic arsenic assumption is a minor uncertainty with minimal effect on the overall risk estimates.

Although arsenic resulted in risks greater than 1×10^{-6} for some of the fish consumption scenarios, the contribution of arsenic to the cumulative risk was insignificant relative to that from PCBs. Therefore, the assumptions about inorganic arsenic are not likely to impact the conclusions of this BHHRA.

6.2.6.6 Polychlorinated Biphenyls

PCBs were analyzed as Aroclors in some media and as individual PCB congeners in others. This introduces some uncertainty when comparing cumulative risk across media. Congener analysis may provide a more accurate measure of PCBs in environmental samples than does the Aroclor analysis. Although most PCBs may have originally entered the environment as technical Aroclor mixtures, environmental processes, such as weathering and bioaccumulation, may have led to changes in the congener distributions in environmental media such that they no longer closely match the technical Aroclor mixtures used as standards in the laboratory analysis, leading to inaccuracies in quantitation.

The results for PCBs in whole body tissue samples analyzed for both PCBs as Aroclors and as individual PCB congeners were qualitatively compared to evaluate correlations associated with the use of Aroclor data. Windward (2005) analyzed fish tissue from the Lower Duwamish Waterway as PCB Aroclors and as individual PCB congeners. The PCB Aroclor data and PCB congener data were significantly correlated for both fillet and whole body tissue. It should be noted that the Lower Duwamish Waterway is not freshwater, and different species were assessed in the Lower Duwamish study compared to Portland Harbor. There is less uncertainty associated with using PCB congener data to calculate EPCs; however, these correlations suggest that PCB Aroclor data may be used in the place of congener data if congener data are not available.

When available, PCB congener data were included in cumulative risk sums for tissue because differences in bioaccumulation, in addition to weathering, results in even greater uncertainty in the PCB Aroclor analysis for tissue. However, for fillet tissue, Round 1 samples were analyzed for PCB Aroclors only, and Round 3 samples, which were collected for smallmouth bass and common carp, were analyzed for PCB congeners only. Because PCB congener data are available for smallmouth bass and common carp fillet tissue, cumulative risks for exposure to fillet tissue from ingestion include only the most recent tissue data for these two species. This introduces uncertainty to the cumulative risk estimates for exposure to fillet tissue when comparing risks across all four resident species.

PCB Aroclor data were included in cumulative risk sums for sediment because the PCB Aroclor dataset is larger than the congener dataset.

PCB congener data were included in the risk evaluation for surface water because the PCB Aroclor data was derived from the results of the congener analysis for the samples used in the risk characterization of this BHHRA. Total PCB congeners did not screen in as COPCs for any surface water scenarios. If PCB Aroclor data from the surface water dataset were used in the COPC screening, PCBs would still not be considered a COPC for any surface water scenarios.

When PCB congener data were used, the total PCB concentration was adjusted by subtracting the concentrations of coplanar PCBs from the total PCB concentration. This was done for purposes of estimating cancer risks because the coplanar PCBs were evaluated separately for the cancer endpoint.

6.2.6.7 Bioavailability of Chemicals

The toxicity values used in the risk assessment are generally based on laboratory studies in which the chemical is administered in a controlled setting via food or water. The actual absorption from environmental media may be lower than that observed in the laboratory. Studies have shown that conditions in environmental media (e.g., pH, organic carbon content) can affect the bioavailability of a chemical (Ruby et al. 1999, Pu et al. 2003, Saghir et al. 2007). If the bioavailability of a chemical in a given environmental medium is less than that in the laboratory study used to derive the toxicity value, the risk assessment will overestimate the risks associated with exposure to that chemical in that medium. A committee of the National Research Council recommended that consideration of bioavailability be incorporated in decision-making at sites (National Academy of Sciences 2003). While site-specific information on the bioavailability of chemicals in sediment is not available, it is important to recognize that there is uncertainty associated with not incorporating bioavailability into the risk estimates, especially related to sediment-associated chemicals.

6.2.6.8 Smallmouth Bass Exposure Areas

Smallmouth bass exposure areas were on a river mile basis. Uncertainties associated with the home range of smallmouth bass are discussed in Section 6.1.13. In Round 1, samples were composited on a per river mile basis (e.g., RM 2, RM 3). In Round 3, samples were composited on a per river mile basis, per side of river (e.g., RM 2E, RM 2W). The Round 1 and Round 3 results were combined and included in the EPC calculations for each river mile exposure area. Although studies have shown that smallmouth bass migrate from one side of the river to another in the lower Willamette

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(ODFW 2005), it is possible that some smallmouth bass may have a home range that is limited to a single side of the river.

Figure 6-1 displays the ratios of concentrations of DDT, DDE, DDD, cPAH, dioxin/furan TEQ, and PCB congeners detected in composite smallmouth bass samples collected at the east side of the river mile compared to concentrations for those detected in composite samples collected at the west side of the river mile. At RM 8, 9, and 10, the ratios are all less than 1, indicating concentrations on the east side of the river are generally less than concentrations on the west side of the river. For the remaining river miles, some ratios exceed one. East to west side concentration ratios for PCBs at river mile 11 are highest of any river mile evaluated. It should be noted, as previously discussed in Section 6.1.14, that a fish from RM 11W was included in the composite for RM 11E due to a mislabeling of the sample. Due to the low number of samples for each exposure area, the maximum detected concentration from either side of the river is almost always used as the 95% UCL/max EPC for the river mile exposure areas anyway, which eliminates the possibility of underestimating risk for a given river mile based on whether or not smallmouth bass migrate across the river. Furthermore, the river mile exposure area was determined based on the smallmouth bass home range. In addition, the area over which fishing occurs should also be considered. Given the exposure duration of 30 to 70 years, it is likely that fish would be collected over an area greater than a single river mile for localized exposures. Therefore, the characterization of risk for bass in this risk assessment is a health protective estimate that is unlikely to underestimate risks.

6.2.6.9 Surface Water EPCs for Recreational Beach Users

For recreational exposures to surface water, data from only the low water sampling event was used, in order to represent surface water conditions during the time of year when most frequent recreational use occurs (i.e. summer months). There is some uncertainty in the representativeness of this dataset for surface water conditions for recreational users.

Transient exposure to surface water can occur throughout the year, so data from sampling events during three seasons of the year were used for this scenario and can be used to assess the representativeness of the single low water sampling event. Arsenic was the only surface water COPC detected in recreational exposure areas. The Study Area-wide average total arsenic concentration for transient exposure to surface water, using year-round data, is 0.48 µg/l. The Study Area-wide average total arsenic concentration for recreational beach user exposure to surface water, using low flow data, is 0.51 µg/l. Given the similarity of these results, the uncertainty associated with the recreational beach user surface water dataset should not impact the conclusions of this BHHRA.

6.3 TOXICITY ASSESSMENT

The results of animal studies are often used to predict the potential human health effects of a chemical. Extrapolation of toxicological data from animal studies to humans is one of the largest sources of uncertainty in evaluating toxicity factors. Much of the toxicity information used in this BHHRA comes from EPA's Integrated Risk Information System (IRIS), which states the following on its website:

> In general IRIS values cannot be validly used to accurately predict the incidence of human disease or the type of effects that chemical exposures have on humans. This is due to the numerous uncertainties involved in risk assessment, including those associated with extrapolations from animal data to humans and from high experimental doses to lower environmental exposures. The organs affected and the type of adverse effect resulting from chemical exposure may differ between study animals and humans. In addition, many factors besides exposure to a chemical influence the occurrence and extent of human disease (EPA 2010b, http://www.epa.gov/iris/limits.htm).

Because of these uncertainties, toxicological data parameters are usually conservative to be more protective of human health due to safety factors EPA uses when estimating toxicity values. The safety factors used by EPA typically range from two to three orders of magnitude (100 to 1,000 times), depending on various aspects of the animal study. As a result, actual risks within the Study Area could be lower than the potential risk estimates calculated in this BHHRA. In addition to the uncertainty already included in the toxicity values, the following toxicity value uncertainties have been identified.

6.3.1 Early Life Exposure to Carcinogens

In 2005, EPA finalized the Supplemental Guidance for Assessing Susceptibility from Early-Life Exposure to Carcinogens (EPA 2005b). The guidance provides a process to evaluate risks from early-life exposure to carcinogens with a mutagenic mode of action. The only exposure scenarios with early-life exposures (i.e., child populations) are recreational beach users and fish consumption. Of these, the only scenario with potential exposure to chemicals with a mutagenic mode of action is the recreational beach user scenario for exposure to PAHs.

This BHHRA did not evaluate risks using the new EPA guidance as the exposure factors for the specific age classes in the separate child and adult scenarios. However, the guidance was used to assess risks associated with exposure to PAHs in the combined adult/child scenarios. Therefore, the combined adult/child scenario accounts for the additional potency associated with early life exposures.

6.3.2 Lack of Toxicity Values for Delta-hexachlorocyclohexane, Thallium, and Titanium

Delta-HCH was detected in tissue and in-water sediment. An SF or RfD toxicity value could not be identified for delta-HCH according to the hierarchy of sources of toxicity values recommended for use at Superfund sites (EPA 2003b). Also, an STSC review concluded that the other hexachlorocyclohexane isomers could not be used as surrogates for delta-HCH due to differences in toxicity (EPA 2002d). Potential risk from delta-HCH was not quantitatively evaluated because of the lack of availability of toxicity data for the chemical.

Thallium was detected in in-water sediment and surface water, and titanium was detected in in-water sediment. Thallium and titanium are naturally occurring elements, and although thallium may have a wide spectrum of effects on humans and animals (EPA 2009a), titanium has been characterized as having extremely low toxicity (Friberg et al 1986). An SF or RfD toxicity value could not be identified for titanium according to the hierarchy of sources of toxicity values recommended for use at Superfund sites (EPA 2003b), and consultation with EPA indicated no surrogate toxicity value was available. Therefore potential risk from exposure to titanium was not quantitatively evaluated in this BHHRA.

6.3.3 Use of Toxicity Values From Surrogate Chemicals for Some Chemicals that Lack Toxicity Values

For some chemicals, if a RfD or SF toxicity value was not available from the recommended hierarchy, a structurally similar chemical was identified as a surrogate. The RfD or SF for the surrogate was selected as the toxicity value and the surrogate chemical was indicated in Section 4. Uncertainty exists in using surrogate chemicals to represent the toxicity of chemicals for which toxicity values are not available. Using surrogate toxicity values could over- or under-estimate risk for a specific chemical.

Based on the results of the BHHRA, the chemicals that exceeded the minimum target cancer risks of 1×10^{-6} or hazard quotient of 1 did not rely on surrogate toxicity values. Therefore, the use of surrogate toxicity values should not impact the conclusions of this BHHRA.

6.3.4 Toxicity Values for Chromium

Chromium was analyzed as total chromium in all media. Toxicity values exist for trivalent and hexavalent chromium only. A reference dose for hexavalent chromium is 0.003 mg/kg-day versus 1.5 mg/kg-day for trivalent chromium, which is a factor of 500 times higher. The toxicity values for trivalent chromium were used in the toxicity assessment for the Study Area because hexavalent chromium reduces to trivalent chromium in an aqueous environmental medium if an appropriate reducing agent is available, and thus trivalent chromium is more prevalent in the environment

(ATSDR 2008). Likewise, screening values for trivalent chromium were used in the selection of total chromium as a COPC for in-water sediment, beach sediment, the groundwater seep, and surface water. This is an uncertainty because the trivalent chromium screening level is for insoluble salts.

For fish consumption, the highest HQ from chromium was 0.004, so even if a portion of the chromium were present as hexavalent chromium, the HQ would likely still be less than 1. Therefore, use of toxicity values for trivalent chromium should not impact the conclusions of this BHHRA.

Additionally, that EPA currently considers the carcinogenic potential of hexavalent chromium via oral exposure as "cannot be determined." A Tier 3 source of toxicity criteria, the New Jersey Dept. of Environmental Protection, has derived quantitative dose-response criteria for evaluating the cancer risks associated with oral exposures to hexavalent chromium, which is the value used in the BHHRA.

6.3.5 Toxicity Values for Polychlorinated Biphenyls and Applicability to Environmental Data

The toxicity values for PCBs were applied to both PCB congeners (not including coplanar congeners) and Aroclors. The RfD for PCBs is based on an immunotoxicity endpoint for Aroclor 1254 (EPA 2010b). Several other Aroclors have been detected in media within the Study Area, indicating the mixture of PCBs differs from that used in the study to develop the RfD. The cancer SF for PCBs was derived for PCB mixtures based on administered doses of Aroclors to rats. The PCB mixtures used in the studies included the coplanar PCB congeners (i.e., dioxin-like PCBs). These coplanar PCBs may have contributed significantly to the carcinogenicity observed in the study. The cancer risk from coplanar PCB congeners was evaluated separately, so including both the total PCB and coplanar PCB congener risks in the cumulative cancer risk results in an overestimate of the cancer risks. Although the potential double counting of PCB mass was corrected for in the PCB adjusted values (mass of dioxin-like PCB was subtracted), there was no correction for the potential double counting of toxicity of dioxin-like PCBs in the PCB TEQ cancer risk estimate and as part of the PCB adjusted value cancer risk estimate.

Based on the dose-response data from studies in rats, PCBs are classified as probable human carcinogens. However, the human carcinogenicity data are inadequate for classification of PCBs as human carcinogens. Several cohort studies have been conducted that analyzed cancer mortality in workers exposed to PCBs. The studies did not find a conclusive association between PCB exposure and cancer; however they were limited by small sample sizes, brief follow-up periods, and confounding exposures to other potential carcinogens. Therefore, using a cancer SF based on the dose-response observed in rats adds further uncertainties to the cancer risk estimates from PCBs as a dose-response has not been observed in humans.

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In addition to the uncertainties with toxicity values for total PCBs, there are uncertainties with the toxicity values for the PCB TEQ, which is evaluated using toxicity values for dioxin and dioxin-like compounds (e.g., dioxin-like PCBs). In their 2001 evaluation of the EPA dioxin reassessment, members of the EPA's Science Advisory Board (SAB) did not reach consensus on the classification of 2,3,7,8-TCDD as a carcinogen (EPA 2001d). The National Academy of Sciences (NAS 2006) discussed the primary uncertainties with the toxicity values for dioxin and dioxin-like compounds as follows:

- The estimation of risks at doses below the range of existing reliable data may result in an overestimate of risk. An estimate of risk for typical human exposures to dioxin and dioxin like compounds would be lower in a sublinear extrapolation model than in the linear model that was used to derive the 2,3,7,8-TCDD SF.

- The issue of appropriately assessing the toxicity of various mixtures of these compounds in the environment. The relative concentrations may change over an exposure period, even though the potency of the individual congeners remains constant. The estimated risk in a given sample depends on both potency and concentration.

The above uncertainties apply to risks from dioxins and furans, as well as risks from dioxin-like PCBs.

6.3.6 Adjustment of Oral Toxicity Values for Dermal Absorption

To evaluate dermal exposures in this BHHRA, an adjustment to the oral toxicity factor to account for the estimated absorbed dose was applied, as discussed in Section 4.7 of this BHHRA.

As recommended by EPA guidance (EPA 2004), an adjustment to the oral toxicity factor to account for the estimated absorbed dose was applied in this BHHRA when the following conditions are met:

- The toxicity value derived from the critical study is based on an administered dose (e.g., through diet or by gavage)

- A scientifically defensible database demonstrates the GI absorption of the chemical is less than 50% in a medium similar to the one used in the critical study.

If both conditions are not met, then a default oral absorption value of 100% is used so that no adjustment for GI absorption is made to evaluate toxicity from dermal exposures.

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The EPA (2004) recommends the adjustment of oral toxicity values to reflect dermal absorption using a cutoff value of 50% GI absorption to reflect the intrinsic variability in the analysis of the absorption studies. The cutoff value of 50% GI absorption obviates the need for small adjustments in the oral toxicity value that are not supported by the level of accuracy in the critical studies that are the source of the toxicity values.

The EPA (2004) guidance states that scientific literature indicates that organic chemicals are generally well absorbed across the GI tract. For inorganic chemicals, the literature indicates a wide range of GI absorption values. However, if EPA (2004) guidance does not provide a GI absorption value for an inorganic COPC, then the default GI absorption value of 100% was used. The EPA (2004) guidance states that this assumption of 100% absorption may contribute to underestimation of dermal risk for those inorganics that are poorly absorbed. The extent of this underestimation is proportional to the actual GI absorption, which would not exceed 50%. The inorganic COPCs for which the default value of 100% GI absorption was used includes the following metals: aluminum, arsenic, boron, cobalt, copper, iron, molybdenum, selenium, thallium, and zinc.

6.4 RISK CHARACTERIZATION

Uncertainties arise during risk characterization due to the methods used in calculating, summing, and presenting risks. The following subsections address uncertainties associated with the risk characterization of this BHHRA.

6.4.1 Endpoint-specific Hazard Indices

In deriving endpoint-specific HIs, only one health endpoint is used for each chemical, even though most chemicals have a myriad of health effects as exposures increase. As an example, a majority of the non-cancer impacts from the site are from PCBs and total TEQ. The endpoint used for deriving the RfD for PCBs is immunotoxicity, while the endpoint used for deriving the RfD for dioxin/furan TEQ and PCB TEQs is reproduction. If the reproductive endpoint for PCBs based upon the lowest observed adverse effects level (LOAEL) of 0.02 mg/kg/day is used with the same Uncertainty Factor as the immunological endpoint to derive an RfD for a reproduction endpoint for PCBs, the RfD for reproductive effects will be 4 times the RfD for immunological effects. Using this ratio, the endpoint-specific HI for reproduction for this exposure scenario for PCBs would be 5,000/4 = 1,250. The total HI for reproduction effects, combining HIs for total TEQ (500) and non-dioxin-like PCBs (1,250), would increase from 500 to 1,750. For the chemicals that have the largest non-cancer contribution in the HHRA, there is a possibility of under-predicting non-cancer health effects by using only one endpoint per chemical.

6.4.2 Risks from Cumulative or Overlapping Scenarios

Where multiple exposure scenarios exist for a given population (i.e., recreational beach users are potentially exposed to both beach sediment and surface water), the risks for each of the exposure scenarios that are considered potentially complete and significant for a given population were summed to estimate the cumulative risks for that population (see Tables 5-199 and 5-200). In calculating the cumulative risks, the maximum cancer risk for each RME scenario was used. This provides a conservative approach, as the same individual may not have the maximum exposure under more than one exposure scenario. However, due to the fact that risks from one scenario are usually orders of magnitude higher than any other scenario for a given receptor, risks from potential cumulative scenarios should not impact the conclusions of this BHHRA. However, the possible magnitude of uncertainty associated with risks from cumulative or overlapping scenarios is discussed further in Attachment F6.

In addition to cumulative exposure scenarios for a given population, an individual may be part of multiple populations (i.e., a dockside worker that is also a non-tribal fisher) and thus could have overlapping exposure scenarios. Because there are numerous possible combinations of overlapping scenarios due to variations in exposure points and exposure assumptions, a model was not developed to quantitatively evaluate overlapping scenarios in this BHHRA. However, because the risk from tissue ingestion is typically at least 10 times higher than other exposure pathways, if an individual consumes fish, the contribution from other exposure scenarios is not likely to contribute significantly to the overall risks for that individual. This BHHRA presents the risks for all of the exposure scenarios, so the risks for a given overlapping scenario could be calculated simply by summing the risks for each of the exposure scenarios that make up the overlapping scenario.

This BHHRA assessed potential risks from exposure to media within the Study Area. Upland sites were not included in this BHHRA. If exposure to upland sites were incorporated with exposures to media within the study, the overall estimate of cumulative risk would likely be higher than the risk estimates in this BHHRA.

6.4.3 Risks from Background

Concentrations of arsenic and mercury in samples collected within the Study Area were found to result in risks greater than 1 x 10^{-6} or an HQ of 1 for at least one of the exposure scenarios evaluated in this BHHRA. However, metals are naturally occurring chemicals and may be present in tissue, water or sediment due to background concentrations. For beach sediment, the exposure point concentrations ranged from 0.7 mg/kg to 9.9 mg/kg and are consistent with the default background soil concentration for arsenic of 7 mg/kg used by DEQ (DEQ 2007). Risks from background concentrations of arsenic in beach sediment and surface water are discussed in Section 5 of this BHHRA. In addition to naturally occurring metals, anthropogenic background may contribute to the overall risks.

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Neither natural nor anthropogenic background tissue concentrations were established for the Study Area. Natural and anthropogenic sources of both metals and organic chemicals are known to contribute to COC concentrations in abiotic media and biota in the Study Area.

Although background tissue concentrations for the Study Area were not established, in some cases, regional tissue concentrations correspond to risk estimates above the target risk thresholds established by EPA (i.e. cancer risk of 10^{-6} to 10^{-4})[8]. For example, in the Columbia River Basin Fish Contaminant Survey, HIs were greater than 100 and cancer risks were as high as 2×10^{-2} for the highest tribal fish consumption rate (389 g/day) (EPA 2002c). In this study, the fish species collected included five anadromous species (Pacific lamprey, smelt, coho salmon, fall and spring Chinook salmon, steelhead) and six resident species (largescale sucker, bridgelip sucker, mountain whitefish, rainbow trout, white sturgeon, walleye). All samples were composites; the size of the individual fish varied with species. However, concentrations of certain contaminants are higher in tissue collected within the Study Area than in the regional tissue, the sources of the regional tissue concentrations are unknown, and regional efforts are underway to reduce contaminant concentrations in tissue.

While risks were presented in this BHHRA without accounting for contributions from background, it is important to recognize that background concentrations may result in unacceptable risks based on the exposure assumptions used in this BHHRA. The proportion of the concentrations that are not due to releases from sources in the Study Area cannot be controlled by remedial actions in the Study Area. This could prevent remedial actions in the Study Area from achieving acceptable risk levels.

6.4.4 Risks from Lead Exposure

Because the maximum EPCs for lead are greater than the protective fish tissue concentrations associated with an acceptable probability of exceeding protective blood lead levels in the fetus of a pregnant woman ingesting tissue from the Study Area, lead is considered a chemical potentially posing unacceptable risk for fish tissue. However, this maximum EPC is orders of magnitude greater than all other fish EPCs and may be attributable to lead in the gut of the fish.

Protective tissue concentrations were estimated using the EPA Adult Lead Methodology (ALM) (EPA 2003c), based on agreements with the EPA to follow the same methodology used in the CRITFC (1994) survey to assess tissue exposures from lead. The ALM focuses on potential impacts to the fetus of a pregnant worker, and therefore, is only appropriate when considering fish consumption by pregnant women. The ALM was developed based on exposure to lead in soil and may not be appropriate to use for fish consumption. Furthermore, the ALM is highly sensitive to

[8] Regional tissue concentrations are discussed in the Risk Management Recommendations document for the Portland Harbor, provided by the LWG to EPA under separate cover.

the bioavailability of ingested lead. For purposes of developing the protective tissue concentrations, the default bioavailability of lead in soil was used. It is not known whether this is an appropriate assumption for lead in tissue.

While lead was identified as a contaminant potentially posing unacceptable risk for fish tissue, there is considerable uncertainty associated with that decision. The identification of lead as a contaminant potentially posing unacceptable risk was based on the maximum EPC, which may not be due to CERCLA activities, and is not representative of Study Area-wide lead concentrations. Furthermore, the identification of lead as a contaminant potentially posing unacceptable risk was based on the ALM, which was not developed for fish consumption.

For in-water sediment, blood lead levels were also estimated using the ALM. As discussed above, the methodology focuses on potential impacts to the fetus of a pregnant worker, and therefore, is only appropriate when evaluating exposures by pregnant women. Because lead was not identified as a contaminant potentially posing unacceptable risk for in-water sediment, the use of the ALM to evaluate risks from lead exposure for in-water sediment is not likely to impact the conclusions of this BHHRA.

6.4.5 Future Risks

This BHHRA estimated current and future risks for exposure within the Study Area, based on known and reasonably foreseeable future uses of the Study Area. In addition, this BHHRA assessed hypothetical scenarios at EPA's request. However, the LWR is a highly dynamic, industrialized water way, and if the land uses in certain areas of the Study Area were to change in the future in a manner that was not foreseen in this BHHRA, the assumptions and scenarios used to evaluate risks for the Study Area may not be applicable to risks from new exposures. Nevertheless, due to the conservative nature of the assumptions used in this BHHRA, the risk estimates in this BHHRA may still be protective of future uses of the Study Area that were not evaluated. The uncertainty related to future risks could result in either higher or lower risk estimates for the Study Area.

6.5 OVERALL ASSESSMENT OF UNCERTAINTY

A summary of the uncertainties and a qualitative classification of their magnitude, their impact on the health protectiveness of the assessment, and their significance to risk management decisions are presented in Table 6-1. For each of the uncertainties identified and discussed in this section, Table 6-1 provides a qualitative assessment (using High, Medium, and Low as descriptors) for each of these properties. In addition, the table presents whether an uncertainty is more likely to over-estimate or under-estimate actual risks from the Study Area. While there are numerous uncertainties identified for this BHHRA, and the cumulative effect of these uncertainties could be significant to the conclusions of the BHHRA, some of these

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uncertainties would be expected to have more of a significant effect on risk management decisions than other uncertainties. These are identified with a “High” descriptor under the “Significance to Risk Management” column in Table 6-1.

Risk assessments typically include conservative assumptions to minimize the chances of underestimating exposure and/or risks of adverse effects to human health, and therefore potentially underestimating the need for remedial actions. In this BHHRA, conservative assumptions were incorporated into the identification of exposure scenarios, the selection of exposure assumptions, the development of EPCs, and the use of toxicity values. Only a portion of the uncertainties in this BHHRA are quantifiable. Further analysis of the data and review of pertinent published literature provided a possible range of values for some of the uncertainties presented above. The magnitude of these ranges are provided in Attachment F6 and discussed in this Section.

While it is not probable that the maximum values of the uncertainties apply for every tissue consumption exposure scenario and contaminant , this magnitude of uncertainty indicates that risks may actually be less than 1×10^{-4} or HI of 1 for certain scenarios.

While conservative, the results of the BHHRA are intended to show the relative risks associated with the exposure scenarios, and which contaminants are contributing the highest percentage of the calculated risks.

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7.0 SUMMARY

The overall objective of this BHHRA was to evaluate whether exposure to contaminants in sediment, surface water, groundwater seeps, or biota may result in unacceptable risks to human health. The results of this BHHRA will be used in developing remedial action objectives and assist in risk management decisions for the Site. The results of this BHHRA have been used in developing risk management recommendations for the Site, submitted to the EPA under separate cover.

The populations evaluated in the risk characterization portion of the BHHRA were identified based on human activities that are known to occur now and/or which could occur in the future within the Study Area, as described in the Programmatic Work Plan, or were directed by EPA for evaluation in this BHHRA. The following are the populations and associated exposure scenarios that were quantitatively evaluated in this BHHRA:

- Dockside Worker – Direct exposure to beach sediment
- In-water Worker – Direct exposure to in-water sediment
- Recreational Beach User – Direct exposure to beach sediment and surface water
- Transient – Direct exposure to beach sediment, surface water, and groundwater seep
- Diver – Direct exposure to in-water sediment and surface water
- Tribal Fisher – Direct exposure to beach sediment or in-water sediment, and fish consumption
- Fisher – Direct exposure to beach sediment or in-water sediment, fish consumption, and shellfish consumption
- Domestic Water User – Hypothetical direct exposure to untreated surface water used as a domestic water source
- Infants - Consumption of human milk was quantitatively assessed as a complete exposure pathway for all adult receptor populations exposed to bioaccumulative chemicals that were identified as COPCs for a given scenario (i.e., PCBs, dioxin/furans, and DDX).

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7.1 SUMMARY OF RISKS

Cancer risks and noncancer hazards were calculated for each of the exposure scenarios listed above for potential exposure to the contaminants selected as COPCs. The following sections present a summary of the risks for each of the media quantitatively evaluated in this BHHRA, and a discussion of the relative magnitude of the risk estimates for each media.

7.1.1 Summary by Exposure Scenario

This section summarizes the risks for each of the media evaluated for potential risks in this BHHRA (beach sediment, in-water sediment, surface water, groundwater seep, fish tissue, and shellfish tissue). Table 5-196 presents a tabular summary of the risk estimates by exposure scenario. Figures 5-1 through 5-21 illustrate the contaminants contributing to risk for each exposure scenario by exposure point, and comparisons of risk across exposure points.

7.1.1.1 Fish Consumption

Fish consumption risks were calculated for the adult and child non-tribal fish consumers, based on three different ingestion rates representing a range of potential consumption scenarios. Fish consumption risks were also evaluated for both single species- and multi-species diets (common carp, black crappie, brown bullhead, and smallmouth bass) based on consumption of either whole body or fillet with skin tissue. Fish consumption was assumed to occur at the same ingestion rate for 30 years for an adult and for 6 years for a child. It was assumed that all fish consumed were resident fish caught within the Study Area (from RM 2 to 11 for smallmouth bass, between RM 0 to 12 for carp, from RM 3 to 9 for brown bullhead and black crappie) or within a single exposure area (within a one mile area on both sides of the river for bass and within a 3 mile stretch of both sides of the river for crappie, carp and bullhead trout).

Fish consumption risks were also evaluated for adult and child tribal fishers based on an upper-bound ingestion rate for a multi-species diet consisting of resident fish species (common carp, black crappie, brown bullhead, and smallmouth bass) as well as sturgeon, lamprey, and salmon. Risks from the tribal fish diet were based on consumption of either whole body or fillet with skin tissue. Fish consumption was assumed to occur at the same ingestion rate for 70 years for an adult and for 6 years for a child. It was assumed that all fish consumed were caught within the Study Area.

Consumption of individual species by the non-tribal fisher resulted in cumulative cancer risks ranging from 3 x 10^{-6} to 7 x 10^{-2} for the scenarios including adult fisher, child fisher, combined adult and child fisher, or breastfeeding infant of an adult fisher consuming fish. The cumulative HIs range from 0.5 to 5,000 for the child and adult non-tribal fish consumers. The highest HI was 60,000 for the breastfeeding infant of a non-tribal fish consumer. Risks from fish consumption by non-tribal fishers are primarily from exposure to PCBs.

Consumption of fish by the tribal fisher resulted in cumulative cancer risks ranging from 4 x 10^{-4} to 2 x 10^{-2} for the tribal adult consumer, tribal child consumer, and breastfeeding infant of tribal adult consumer. The highest HI was 400 for the tribal adult fisher, 800 for the tribal child consumer, and 9,000 for a breastfeeding infant of a tribal adult consuming fish. Risks from fish consumption by tribal fishers are primarily from exposure to PCBs.

There were multiple uncertainties associated with the fish consumption scenarios of which the following were of primary significance: lack of site-specific fish consumption information, the small area assumed for exclusive collection of fish or shellfish consumed, fish consumption rates, tissue type and fish species consumed, cooking and preparation methods, and contributions from background. Round 1 fillet tissue samples were not analyzed for PCB, dioxin, or furan congeners. Therefore, the risks from consumption of black crappie and bullhead fillet tissue, which were only analyzed in Round 1, likely underestimate the actual risks. However, a range of risks was calculated for fish consumption scenarios, which included samples that were analyzed for congeners, so the lack of analysis of contaminants in certain samples should not impact the conclusions of this BHHRA.

7.1.1.2 Shellfish Consumption

Current and potential future shellfish consumption rates for the site are not known. However, both crayfish and clams were evaluated for consumption risks. Two different ingestion rates based on the nationwide survey for shellfish consumption for freshwater and estuarine habitats combined were used to calculate risks from shellfish consumption. Shellfish consumption was assumed to occur at the same ingestion rate for 30 years. It was assumed that all shellfish consumed were caught within the Study Area or within a single exposure area for spatial scales smaller than the Study Area. Cumulative cancer risks from consumption of shellfish ranged from 9 x 10^{-7} to 7 x 10^{-4}. The cumulative HIs range from 0.06 to 40 for shellfish consumption. The highest HI was 800 for the breastfeeding infant of a shellfish consumer.

In addition to the uncertainty of whether shellfish consumption actually occurs on an ongoing basis, there were other uncertainties associated with the shellfish consumption scenarios of which the following were of primary significance: spatial scale of EPCs, shellfish consumption rates, shellfish species consumed, cooking and preparation methods, and contributions from background.

7.1.1.3 Direct Exposure to In-Water Sediment

Risks from in-water sediment exposure were estimated separately for each of the ½-mile river segment exposure areas on each side of the river, and for Study Area-wide exposure. Each ½-river mile segment was considered a potential exposure area, regardless of the use of the area. In-water sediment within the navigation channel was not included in the risk evaluation. Risks from in-water sediment exposure were evaluated for exposures by in-water workers, tribal fishers, fishers, and divers.

The cumulative cancer risks for all of the CT scenarios for direct exposure to in-water sediment were below 1 x 10^{-4}, and only the tribal fisher CT scenario had cancer risks above 1 x 10^{-6}. For the RME scenarios, cumulative cancer risks were greater than 1 x 10^{-6} but were below 1 x 10^{-4}, with the exception of cancer risks above 1 x 10^{-4} for in-water sediment by a tribal fisher at exposure areas RM 6W (risk is 2 x 10^{-4} due primarily to PAHs) and RM 7W (risk is 3 x 10^{-4} due primarily to dioxins). The highest HI is 3.

There were multiple uncertainties associated with the direct exposure to in-water sediment scenarios of which the following were of primary significance: degree of sediment contact that occurs during fishing scenarios, spatial scale of in-water sediment EPCs, exposure parameters, bioavailability of contaminants in sediment, and contributions from background. The uncertainties associated with exposure parameters and contributions from background were not quantified in this BHHRA.

7.1.1.4 Direct Exposure to Beach Sediment

Beaches were identified as potential human use areas associated with industrial upland sites (dockside workers), recreation (recreational users or fishers), and/or trespassing or transient use (transients). Even if such beach use occurs, the extent to which the beach is used and the nature of the contact with sediments/beach is uncertain. However, health protective assumptions were included in the risk analysis of this exposure pathway to provide an estimate of potential risks.

The only CT scenarios for exposure to beach sediment resulting in risks above 1 x 10^{-6} were the dockside worker (6 x 10^{-6}) and tribal fisher and child recreational beach user scenarios (2 x 10^{-6}). The cumulative cancer risks for all of the CT scenarios were below 1 x 10^{-4}. The RME scenarios for exposure to beach sediment resulting in cumulative cancer risks above 1 x 10^{-6} include: dockside worker, adult and child recreational beach user, tribal fisher and fisher. The maximum cancer risk from RME scenarios was 9 x 10^{-5} for the dockside worker exposure to beach sediment. None of the RME scenarios for exposure to beach sediment resulted in risks greater than 1 x 10^{-4}. None of the scenarios resulted in HIs exceeding 1. Risks above 1 x 10^{-6} resulting from exposures to beach sediment are due primarily to arsenic, which is likely present at naturally occurring background concentrations, and benzo(a)pyrene.

There were multiple uncertainties associated with the direct exposure to beach sediment scenarios of which the following were of primary significance: spatial scale of beach sediment EPCs, exposure parameters, bioavailability of contaminants in sediment, and contributions from background. The uncertainties associated with exposure parameters and contributions from background were not quantified in the BHHRA.

7.1.1.5 Direct Exposure to Surface Water

Risks were evaluated for direct surface water exposures by transients, divers and adult and child recreational beach users. The scenarios resulting in cumulative cancer risks greater than 1 x 10^{-6} were the diver in wet suit (1 x 10^{-5}) and the diver in dry suit (2 x 10^{-6}) at RM 6W due primarily to cPAHs. None of the direct surface water exposure scenarios resulted in HIs exceeding 1.

Surface water within the Study Area is not currently used as a domestic water source, nor are there plans to use surface water within the Study Area as a domestic water source in the future. However, risks were also evaluated for hypothetical exposure to untreated surface water used as a domestic water source by future residents. The maximum cumulative cancer risk for hypothetical exposure to untreated surface water was 9 x 10^{-4}, due primarily to cPAHs, and benzo(a)pyrene specifically. The child RME scenario for hypothetical exposure to surface water as a domestic water source was the only scenario with an exceedance of an HI of 1. The exceedance occurred at RM 8.5, primarily from exposure to MCPP (HQ for MCPP was 2).

7.1.1.6 Direct Exposure to Groundwater Seeps

Risks from exposures to groundwater seeps were evaluated for exposure by a transient for only one exposure point. The transient exposure scenario did not result in cumulative cancer risks greater than 1 x 10^{-6} or HIs greater than 1.

7.1.2 Comparison of Risks Between Exposure Scenarios

A comparison of risk ranges across media can help focus risk management decisions by identifying the media contributing most to the overall risk to human health at the Study Area. As discussed in Sections 5, the magnitude of risk varies greatly across the different scenarios. Figures 7-1 and 7-2 display the ranges of total cumulative cancer risk and endpoint-specific HIs, respectively, for each media type, based on mean exposure assumptions for each media evaluated in the BHHRA. As illustrated in Figures 7-1 and 7-2, the risk ranges for the scenarios assessing consumption of fish and shellfish tissue are orders of magnitude higher than risks for others scenarios, and exceed a cumulative cancer risk of 1 x 10^{-4} and a HI of 1. Figures 7-3 and 7-4 display the ranges of total cumulative cancer risk and cumulative HIs, respectively, based on RME assumptions, for each media type evaluated in the BHHRA. As illustrated in Figures 7-3 and 7-4, the risk ranges for scenarios assessing consumption of fish and shellfish tissue are orders of magnitude higher than risks for other scenarios. The only scenarios that exceed a cumulative cancer risk of 1 x 10^{-4} or a HI of 1 are the tissue consumption scenarios and the scenario for direct contact with in-water sediment by tribal and high frequency fishers.

7.1.3 Contaminants Potentially Posing Unacceptable Risks

Contaminants were identified as potentially posing unacceptable risks if they resulted in a cancer risk greater than 1 x 10^{-6} or an HQ greater than 1 under any of the

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exposure scenarios for any of the exposure point concentrations evaluated in this BHHRA, regardless of the uncertainties. Given the uncertainties in the analytical data discussed in Section 6, the preliminary COCs were assessed to select the final COCs for this BHHRA.

Four of the contaminants identified as potentially posing unacceptable risks (alpha-, beta, and gamma-hexachlorocyclohexane and heptachlor) were only detected in fish tissue as N-qualified data. Due to retention time issues in the analytical methods used for the Round 1 tissue samples, some of the pesticide tissue data were N-qualified, indicating that the identity of the chemical could not be confirmed. In subsequent sampling events, different analytical methods were used so that the identification of pesticides was not an issue in tissue samples collected in Rounds 2 and 3. EPA guidance (1989) does not recommend the use of data where there are uncertainties in the identification of contaminants, as is the case in the N-qualified data. Therefore, if a chemical was identified as potentially posing unacceptable risks based only on the use of N-qualified data, that chemical is not recommended for further evaluation for potential risks to human health.

The contaminants potentially posing unacceptable risks to human health based on the results of this BHHRA that are recommended for further evaluation for potential risks to human health are presented in Table 7-1.

7.2 PRIMARY CONTRIBUTORS TO RISK

In this BHHRA, there are certain exposure scenarios and contaminants that result in risks that are orders of magnitude higher than risks from other exposure scenarios and contaminants within the Study Area, and that exceed risk levels that generally warrant remedial action under CERCLA. One role of the BHHRA is to identify those contaminants that pose the greatest risks to current and future receptors, along with the media and exposures routes associated with those risks. This information is used to inform response actions. This section presents the primary contributors to human health risk at the Site. The exposure scenarios and chemicals discussed here represent a subset of the scenarios and contaminants evaluated in this BHHRA.

The focus on primary contributors to risk can assist with the development of the FS by focusing on those scenarios and contaminants associated with the greatest overall risk in the Study Area. While these scenarios and contaminants may be the focus of the remedial analyses, other exposure scenarios and contaminants potentially posing unacceptable risks may still be considered in remedial decisions for the Site.

Only those exposure scenarios and contaminants that resulted in a cancer risk greater than 1×10^{-6} or an HQ greater than 1 were considered in identifying the

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primary contributors to risk. Additional considerations in the selection of contributors included:

- The relative percentage of each contaminant's contribution to the total human health risk consistent with assumptions on exposure areas.
- Uncertainties associated with the exposure scenarios, such as the likelihood of future risk scenarios, number of assumptions made in estimating exposure, or level of uncertainty in estimates of exposure variables.
- Frequency of detection, both on a localized basis and Study Area-wide.
- Comparison of risks within the Study Area to risks based on measured regional contaminant concentrations for similar exposure scenarios, indicating background sources of chemicals in the region.
- Magnitude of risk exceedance above EPA's target range for managing cancer risk of 10^{-4} to 10^{-6} and noncancer hazard of one.

The chemicals potentially posing unacceptable risks and the primary contributors to risk based on the above criteria for the exposure scenarios evaluated in this BHHRA are discussed below.

7.2.1 Fish Consumption Scenarios

Twenty six COCs (PCBs, dioxins, six metals, Bis 2-ethylhexyl phthalate (BEHP), PAHs, hexachlorobenzene, and seven pesticides) were identified as potentially posing unacceptable risks for the fish-consumption scenarios (i.e., both fisher and tribal fisher) based on exceedances of a cancer risk of 1 x 10^{-6} or HQ of 1:

- <u>PCBs</u>: Total PCBs resulted in cancer risk estimates exceeding 1 x 10^{-4} and/or HQs exceeding 1 for fish consumption. Total PCB TEQ also resulted in cancer risk estimates exceeding 1 x 10^{-4} and/or HQs exceeding 1 for fish consumption. PCBs resulted in risk estimates that exceeded a cancer risk of 1 x 10^{-4} and/or HQ of 1 for both localized and Study Area-wide exposures. PCBs are considered a primary contributor to risk for the fish consumption pathway because of the magnitude of the risk exceedances above the EPA target range for managing risk, spatial scale of the risk exceedances, and relative contribution to cumulative risk.

- <u>Dioxins/furans</u>: Total dioxin TEQ resulted in cancer risk estimates exceeding 1 x 10^{-4} and/or HQs exceeding 1 for fish consumption. Total dioxin TEQ resulted in risk estimates that exceeded a cancer risk of 1 x 10^{-4} and/or HQ of 1 for both localized and Study Area-wide exposures. Dioxins are considered a primary contributor to risk for the fish consumption pathway because of the magnitude of the risk exceedances, spatial scale of the risk exceedances, and relative contribution to cumulative risk.

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- <u>Metals</u>: Antimony, arsenic, mercury, selenium, and zinc were associated with one or more fish consumption exposure scenarios that resulted in a risk estimate that exceeded a cancer risk of 1 x 10^{-6} or HQ of 1.
  - Arsenic resulted in cancer risk estimates that exceeded a cancer risk of 1 x 10^{-4} for Study Area-wide exposures.
  - Antimony exceeded an HQ of 1 at RM 10 for consumption of whole body smallmouth bass tissue only due to a single smallmouth bass sample with the anomalously high result discussed in Section 6.1.14.
  - Lead was identified as a contaminant potentially posing unacceptable risk based on exceedance of protective tissue concentrations derived using blood lead models. The risk exceedances for lead from fish consumption are due to only a single sample of smallmouth bass whole body tissue collected at RM 10 with the anomalously high result discussed in Section 6.1.14
  - Mercury resulted in risk estimates that exceeded a HQ of 1 for both localized and Study Area-wide exposures.
  - Selenium exceeded an HQ of 1 at RM 11 only for consumption of smallmouth bass fillet tissue, due to a single sample. Due to a limited number of detected concentrations of antimony and selenium (i.e., 5 detects out of 32 samples and 1 detect out of 23 samples, respectively), antimony and selenium also resulted in HQs greater than 1 Study Area-wide.
  - Zinc slightly exceeded an HQ of 1 (HQ = 2) for fish consumption based on a single sample of whole body common carp tissue collected from RM 4 to RM 8.

- <u>BEHP:</u> BEHP resulted in cancer risk estimates greater than 1 x 10^{-6} for consumption of whole body smallmouth bass and brown bullhead, based on both a localized and Study Area-wide basis, for all ingestion rates. BEHP resulted in cancer risk estimates greater than 1 x 10^{-4} and HQs greater than 1 at RM 4 for consumption of smallmouth bass at the 73 g/day and 142 g/day ingestion rates.

- <u>PAHs</u>: Benzo(a)anthracene, benzo(a)pyrene, dibenzo(a)anthracene, and total carcinogenic PAHs were identified as a contaminant potentially posing unacceptable risk for fish tissue consumption based on cancer risk estimates exceeding 1 x 10^{-6}. Cancer risk estimates for total carcinogenic PAH exceeded 1 x 10^{-6} for all ingestion rates for consumption of smallmouth bass and only the 73 g/day and 142 g/day ingestion rates for consumption of common carp. No cancer risk estimates exceeded 1 x 10^{-4}. For consumption of smallmouth bass, cancer risk estimates for total carcinogenic PAHs exceeded 1 x 10^{-6} for five rive mile segments and Study Area-wide. For consumption of common carp, cancer risk estimates for total carcinogenic PAHs exceeded 1 x 10^{-6} for two fishing zones and Study Area-wide. PAHs account for less than 1% of the cumulative cancer risks where they were detected.

- Pesticides: Aldrin, dieldrin, heptachlor epoxide, total chlordane, total DDD, total DDE, and total DDT were associated with one or more fish consumption exposure scenarios that resulted in a risk estimate that exceeded a cancer risk of 1 x 10^{-6} or HQ of 1. These pesticides did not result in cancer risks greater than 1 x 10^{-4}.
  - Aldrin was identified as a contaminant potentially posing unacceptable risk based on cancer risk estimates slightly above 1 x 10^{-6}, at only the 142 g/day ingestion rate for consumption of common carp (localized and Study Area-wide). Aldrin only contributes approximately 0.01% to the total Study Area-wide risk for the whole body common carp diet.
  - Dieldrin was identified as a contaminant potentially posing unacceptable risk based on an exceedance of 1 x 10^{-6} for consumption of all fish species (smallmouth bass, common carp, black crappie, and brown bullhead), all ingestion rates, and on a localized and Study Area-wide basis. For the multi-species whole body tissue diet, dieldrin contributes to less than 1% of the site-wide risk from tissue consumption.
  - Heptachlor epoxide was identified as a contaminant potentially posing unacceptable risk based on cancer risk estimates slightly above 1 x 10^{-6}, at only the 142 g/day ingestion rate for consumption of common carp, and for one fishing zone (RM 0 to RM 4). For this fishing zone, heptachlor epoxide contributes to 0.1% of cumulative risk from consuming whole body common carp.
  - Total chlordane was identified as a contaminant potentially posing unacceptable risk based on an exceedance of 1 x 10^{-6} for consumption of all fish species (smallmouth bass, common carp, black crappie, and brown bullhead), all ingestion rates, and on a localized and Study Area-wide basis.
  - DDD was identified as a contaminant potentially posing unacceptable risk based on an exceedance of 1 x 10^{-6} for consumption of all fish species (smallmouth bass, common carp, black crappie, and brown bullhead), all ingestion rates, and on a localized and Study Area-wide basis.
  - DDE was identified as a contaminant potentially posing unacceptable risk based on an exceedance of 1 x 10^{-6} for consumption of all fish species (smallmouth bass, common carp, black crappie, and brown bullhead), all ingestion rates, and on a localized and Study Area-wide basis. DDE also resulted in an HQ slightly greater than 1 at RM 7 for smallmouth bass.
  - DDT was identified as a contaminant potentially posing unacceptable risk based on an exceedance of 1 x 10^{-6} for consumption of all fish

species (smallmouth bass, common carp, black crappie, and brown bullhead), all ingestion rates, and on a localized and Study Area-wide basis.

Based on the magnitude of risk, the relative contribution to risk, and the frequency of detection, PCBs and dioxins/furans are considered the primary contributors to risk for fish consumption scenarios. The risks for PCBs and dioxins/furans exceed a cancer risk of 1 x 10^{-4} or an HQ of 1 for both the mean and maximum exposure scenarios for both localized and Study Area-wide exposures. Figure 7-5 illustrates the relative percentages of cancer risks for individual contaminants contributing to total cumulative risk for consumption of fish tissue by an adult fisher, based on Study Area-wide EPCs for a multi-species diet. Separate charts are shown for diets based on whole body fish consumption and fillet tissue consumption. As illustrated in the pie charts in Figure 7-5, PCBs are the primary contributor to risk for fish consumption and dioxins are a secondary risk contributor for fish consumption of both whole body and fillet tissue diets. A similar pattern is shown in Figure 7-6, which illustrates the relative percentage of cancer risk for consumption of fish tissue by an adult tribal fisher, based on Study Area-wide EPCs for a multi-species diet for both whole body and fillet tissue consumption. For both the fisher and tribal fisher, and for both whole body and fillet tissue diets, PCBs contribute over 90% of the overall cancer risk and result in an HQ that is up to 57 times higher than any other HQ from whole body tissue consumption, and up to 153 times higher than any other HQ from fillet tissue consumption by adults.

The contributions of background concentrations to these risk estimates may exceed the risk levels that generally warrant remedial action under CERCLA. While background concentrations have not been established for fish tissue, as discussed in Section 6.4.2, regional tissue concentrations may be associated with unacceptable risks from fish consumption, especially at higher ingestion rates. On a regional level, PCBs and dioxins/furans have been detected in fish tissue collected in the Willamette and Columbia Rivers, outside of the Study Area. In a risk assessment for the mid-Willamette (EVS 2000), PCBs were found to result in an HQ greater than 1 for both the 142 g/day and 17.5 g/day ingestion rates, and a cancer risk greater than 1 x 10^{-4} for the 142 g/day ingestion rate. Dioxins and furans were also found to result in a cancer risk greater than 1 x 10^{-4} for the 142 g/day ingestion rate (non-cancer endpoints were not evaluated for dioxins and furans). In the Columbia River Basin Fish Contaminant Survey (EPA 2002c), PCBs were found to result in cancer risks greater than 1 x 10^{-4} and HQs greater than 1 for the 142 g/day and 7.5 g/day[9] ingestion rates for the general public consumption of resident fish. Dioxins and furans were also found to result in a cancer risk greater than 1 x 10^{-4} for the 142 g/day ingestion rate (non-cancer

[9] The low ingestion rate used in the Columbia River Basin Fish Contaminant Survey is lower than the lowest ingestion rate used in this BHHRA, which was 17.5 g/day.

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endpoints were not evaluated for dioxins and furans). While the concentrations in the Study Area are higher than the regional tissue concentrations, the sources of PCBs and dioxins and furans in regional tissue data are unknown, and efforts are underway to reduce regional tissue concentrations, the regional tissue data indicate that CERCLA actions alone may not be adequate to achieve a target risk level of 1 x 10^{-6} for some of the assumptions evaluated in this BHHRA.

7.2.2 Shellfish Consumption Scenarios

Seventeen contaminants were identified as potentially posing unacceptable risks for shellfish consumption, based on exceedances of the cumulative cancer risk of 1 x 10^{-6} or HQ of 1, including PCBs, dioxins, arsenic, PAHs, pentachlorophenol, and five pesticides:

- <u>PCBs</u>: Total PCBs resulted in cancer risk estimates exceeding 1 x 10^{-4} and/or HQs exceeding 1 for shellfish consumption. Total PCB TEQ also resulted in cancer risk estimates exceeding 1 x 10^{-4} and/or HQs exceeding 1 for shellfish consumption. PCBs resulted in risk estimates that exceeded a cancer risk of 1 x 10^{-4} and/or HQ of 1 for both localized and Study Area-wide exposures. PCBs are considered a primary contributor to risk for the shellfish consumption pathway because of the magnitude of the risk exceedances, spatial scale of the risk exceedances, the relative contribution to cumulative risk, and the frequency of detection.

- <u>Dioxins/furans</u>: Total dioxin TEQ resulted in cancer risk estimates exceeding 1 x 10^{-4} and/or HQs exceeding 1 for shellfish consumption. Dioxins and furans resulted in risk estimates that exceeded a cancer risk of 1 x 10^{-4} and/or HQ of 1 for both localized and Study Area-wide exposures. Dioxins are considered a primary contributor to risk for the shellfish consumption pathway because of the magnitude of the risk exceedances, spatial scale of the risk exceedances, the relative contribution to cumulative risk, and the frequency of detection.

- <u>Arsenic:</u> Arsenic was identified as a contaminant potentially posing unacceptable risk based on cancer risk estimates that exceeded 1 x 10^{-6} for both clams and crayfish, at both ingestion rates, and on a localized and Study Area-wide scale. No cancer risk estimates exceeded 1 x 10^{-4}. Though arsenic was identified as a contaminant potentially posing unacceptable risk on both a localized and Study Area-wide spatial scale, the concentrations in shellfish tissue may be due in part to naturally occurring background concentrations.

- <u>cPAHs</u>: cPAHs were identified as a contaminant potentially posing unacceptable risk based on cancer risk estimates that exceeded 1 x 10^{-6} for both clams and crayfish, at both ingestion rates, and on a localized and Study Area-wide scale. Cancer risk estimates for total cPAHs across all exposure areas and exposure scenarios ranged from 2×10^{-8} to 5×10^{-4}, and exceeded 1×10^{-4} for the 18 g/day ingestion rate for clams collected at locations RM 5W and RM 6W. cPAHs are considered a primary contributor to risk for the shellfish consumption pathway at those locations because of the magnitude of the risk exceedances and relative contribution to cumulative risk.

- Pentachlorophenol: Pentachlorophenol was only detected in one out of 41 shellfish samples, which was a crayfish composite sample collected near RM 8. This one detection of pentachlorophenol resulted in a cancer risk estimate within the range of 1×10^{-6} to 1×10^{-4}.

- Pesticides: Aldrin, dieldrin, total DDD, total DDE, and total DDT were associated with one or more shellfish consumption exposure scenarios that resulted in a risk estimate that exceeded a cancer risk of 1×10^{-6} or HQ of 1. These pesticides were not associated with shellfish consumption scenarios that resulted in a cancer risk estimate above 1×10^{-4}.
    - Aldrin was identified as a contaminant potentially posing unacceptable risk based on cancer risk estimates above 1×10^{-6} for ingestion of clam tissue, for the18 g/day ingestion rate only, and for one location (near RM 8W) and Study Area-wide.
    - Dieldrin was identified as a contaminant potentially posing unacceptable risk based on cancer risk estimates above 1×10^{-6} for ingestion of clam tissue, for the 18 g/day ingestion rate only, and for one location (near RM 8W) and Study Area-wide.
    - Total DDD was identified as a contaminant potentially posing unacceptable risk based on cancer risk estimates above 1×10^{-6} for ingestion of clam tissue, for the 18 g/day ingestion rate only, and for one location (near RM 6W) and Study Area-wide.
    - Total DDE was identified as a contaminant potentially posing unacceptable risk based on cancer risk estimates above 1×10^{-6} for ingestion of clam tissue, for the 18 g/day ingestion rate only, and for three locations (near RM 6W, RM 7W, and RM 8W).
    - Total DDT was identified as a contaminant potentially posing unacceptable risk based on cancer risk estimates above 1×10^{-6} for ingestion of clam tissue, for the 18 g/day ingestion rate only, and for only two locations (near RM 6W and RM 7W).

Based on the magnitude of risk, the relative contribution to risk, and the frequency of detection, PCBs, dioxins/furans, and cPAHs are considered the primary contributors to risk for shellfish consumption. PCBs and dioxins/furans contribute approximately 58% of the cumulative cancer risk for clam consumption and approximately 91% for crayfish consumption for the Study Area. Total cPAHs contribute approximately 35% of the cumulative cancer risk for clam consumption (for undepurated samples) and approximately 5% for crayfish consumption for the Study Area. PCBs and dioxins/furans are considered primary contributors to risk on a Study Area-wide basis. cPAHs are considered primary contributors to risk on a localized basis (RM 5W and RM 6W). PCBs are the primary contributors to risk and dioxins/furans are the secondary contributors to risk for shellfish consumption.

7.2.3 In-Water Sediment Scenarios

The contaminants potentially posing unacceptable risk identified for in-water sediment are PAHs (primarily benzo[a]pyrene), arsenic, PCBs, and dioxins. PAHs and dioxins were identified as contaminants potentially posing unacceptable risk for all of the in-water sediment scenarios, and arsenic and PCBs were identified as contaminants potentially posing unacceptable risk for tribal fisher and high frequency fisher scenarios only. The contribution of the contaminants to the cumulative cancer risks varied by river mile. Risks from cPAHs across all exposure areas and exposure scenarios ranged from 1×10^{-10} to 2×10^{-4}. For the entire Study Area, total cPAHs and dioxins/furans through direct contact with sediment each contributed approximately 50% of the cumulative cancer risk. As previously discussed, cumulative cancer risks associated with arsenic may be due in part to naturally occurring background sediment concentrations. Cumulative cancer risks above 1×10^{-6} for PCBs are associated with only four ½-mile river segments, and for dioxins are associated with only two ½-mile river segments. Cumulative cancer risks above 1×10^{-6} for PAHs are associated with twenty-two ½-mile river segments. Carcinogenic PAHs are considered the primary contributors to risk contaminant for in-water sediment on a Study Area-wide basis due to relative magnitude of the cumulative risk and the number of the risk exceedances. PCBs and dioxins are considered primary contributors to risk on a localized basis (RM 8.5W [PCBs] and RM 7W [dioxins]).

7.2.4 Beach Sediment Scenarios

The contaminants potentially posing unacceptable risk identified for beach sediment are PAHs (primarily benzo[a]pyrene) and arsenic. Risks above 1×10^{-6} resulting from exposure to arsenic in beach sediment are likely due in part to naturally occurring background concentrations of arsenic. If the contribution of naturally occurring background concentrations of arsenic is subtracted from the cumulative risk, then the primary contributor to risk for beach sediment is benzo(a)pyrene. Risks above 1×10^{-6} resulting from exposure to benzo(a)pyrene was limited to a few locations, with the maximum cumulative cancer risk associated with beach location 06B025. Therefore, direct exposure to beach sediment containing benzo(a)pyrene at beach 06B025 is considered a primary contributor to risk for beach sediment.

7.2.5 Surface Water Scenarios

The primary contributor to risk for direct contact with surface water is exposure to PAHs in surface water by divers at RM 6.0 W, because this is the only scenario and location with risk exceedance of 1×10^{-6} or HI greater than 1. However, risk management during remedy selection should consider the limited spatial scale and high degree of uncertainty associated with the diver exposure assumptions.

Risks were also evaluated for hypothetical exposure to untreated surface water used as a domestic water source by future residents. Cumulative cancer risks were up to 3 x 10^{-4} for adults, and up to 7 x 10^{-4} for child residents primarily due to benzo(a)pyrene. The only HIs that were greater than 1 were for a child resident under the RME scenario at Multnomah Channel and RM 8.5, due primarily to ingestion of MCPP in surface water. Because this is a hypothetical scenario, it is not considered a primary contributor to risk for the Study Area.

7.2.6 Summary of Primary Contributors to Risk

As per EPA guidance for the role of risk assessment in remedy selection under CERCLA (EPA 1991a), EPA uses the general risk range of 1 x 10^{-6} to 1 x 10^{-4} as a “target range” within which the EPA manages risk during the remedy selection. Furthermore, if the cumulative cancer risk to an individual based on RME assumptions is less than 1 x 10^{-4} and the non-cancer HQ is less than 1, remedial action generally is not warranted at a site (EPA 1991a). DEQ guidance sets an acceptable risk level of 1 x 10^{-6} for individual chemicals and 1 x 10^{-5} for cumulative risks (OAR 340-122-0115). While chemicals potentially posing unacceptable risks were identified based on exceeding a cancer risk of 1 x 10^{-6} or HQ of 1, the only exposure scenarios with cancer risks exceeding 1 x 10^{-4} or HQ greater than 1 are fish consumption and shellfish consumption and direct exposure to in-water sediment for two ½-river mile segments.

The primary exposure scenario contributing to risk for the Study Area is fish consumption, and the contaminants contributing to that risk are PCBs and dioxins/furans. PCBs and dioxins/furans both resulted in cancer risks greater than 1 x 10^{-4} and HQs greater than 1 for fish consumption for both localized and Study Area-wide exposures. PCBs and dioxins/furans contribute approximately 98% of the cumulative cancer risk for fish consumption. Regionally, fish consumption also results in risk estimates exceeding cumulative risks of 1 x 10^{-4} or HQ of 1 based on data collected from the Willamette and Columbia Rivers outside of the Study Area (EVS 2000, EPA 2002c). In those studies, both PCBs and dioxins/furans resulted in cancer risks greater than 1 x 10^{-4} and/or HQs greater than 1 for fish consumption. The concentrations of PCBs in regional tissue are lower than in the Study Area, and the sources of PCBs in regional tissue are unknown. The secondary exposure scenario contributing to risk is consumption of shellfish; however, it is not known to what extent shellfish consumption actually occurs on an ongoing basis within the Study Area.

The identification of the primary contributors to human health risks can help provide focus to the FS by identifying a smaller number of chemicals and exposure scenarios that have the largest contribution to overall risk. To provide context for the significance of the remedial actions to the protection of human health, the uncertainties associated with the exposure assumptions and potential contribution of background sources of contaminants to the Study Area should be considered when evaluating primary contributors to human health risks during the FS.

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8.0 CONCLUSIONS

A summary of chemicals contributing to risk by exposure scenario is provided in Table 7-1, and risk ranges by exposure scenario are presented in Table 5-203. The following presents the major findings of this BHHRA:

- Fish consumption is the exposure scenario that is considered the primary contributor to risk for this site. Risks resulting from the consumption of fish are generally orders of magnitude higher than risks resulting from direct contact with sediment, surface water, or groundwater seeps. Risks from fish consumption are within or above the cumulative cancer risk range of 1 x 10^{-6} to 1 x 10^{-4} and exceed an HI of 1 for most exposure scenarios evaluated, including both RME and CT assumptions. Risk estimates for shellfish consumption scenarios were also within or above the cumulative cancer risk range of 1 x 10^{-6} to 1 x 10^{-4} and exceeded an HI of 1 for most exposure scenarios evaluated, including both RME and CT assumptions. The evaluation of shellfish consumption was completed at the direction of EPA. With the exception of two ½-mile river segments for the tribal fisher scenario and one location for the hypothetical use of untreated surface water as a drinking water source by a future resident, all of the direct contact scenarios result in risks within or below the EPA target cancer risk range of 1 x 10^{-6} to 1 x 10^{-4}. The direct contact scenarios also result in non-cancer hazards below the target HI of 1, with the exception of one ½-river mile segment for in-water sediment and one location for hypothetical use of untreated surface water as a drinking water source.

- For fish consumption, which is the pathway with the highest risk estimates, PCBs are the primary contributor to risk, and dioxins/furans are the secondary contributor to risk.

- The uncertainties associated with the tissue consumption scenarios should be considered during the FS. The fish tissue consumption risks in this BHHRA incorporate assumptions that may under- or more likely over-estimate the actual risks.

- The contribution of background sources is an important consideration in risk management decisions. For example, arsenic concentrations in beach sediment contribute approximately 50% of cumulative risk from exposure from this medium for the highest-risk scenarios, yet arsenic concentrations detected in beach sediment within the Study Area are comparable to Oregon DEQ-established background levels.

The results of the BHHRA will be used to produce risk-based PRGs and AOPCs for the FS, as well as to develop risk management recommendations for the Site. In

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addition, the BHHRA may be consulted by risk managers as they deliberate practical risk management objectives during the course of the FS.

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