PORTLAND HARBOR RI/FS

FINAL REMEDIAL INVESTIGATION REPORT

APPENDIX F

BASELINE HUMAN HEALTH RISK ASSESSMENT

FINAL

March 28, 2013

Produced for The Lower Willamette Group and
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Produced by Kennedy/Jenks Consultants

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LIST OF ACRONYMS

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
cPAH	carcinogenic polycyclic aromatic hydrocarbon
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
DDx	DDD, DDE, and DDT
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.

[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).

HEAST	Health Effects Assessment Summary Table
HHRA	human health risk assessment
HI	hazard index
HQ	hazard quotient
IEUBK	Integrated Exposure Uptake Biokinetic model3
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	semi-volatile organic compound
TCDD	tetrachlorodibenzo-p-dioxin
TEF	toxic equivalency factor
TEQ	toxic equivalent
TZW	transition zone water
UCL	upper confidence limit
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
carcinogenic polycyclic aromatic hydrocarbons	cPAHs represent the cumulative risks from benzo(a)anthracene, benzo(a)pyrene, benzo(b)fluoranthene, benzo(k)fluoranthene, chrysene, dibenzo(a,h)anthracene, and indeno(1,2,3-cd)pyrene
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

[2] Prior deliverables and some of the tables and figures attached to this document may use the terms "chemical of concern", "chemical of interest", or "chemical of potential concern", which has the same meaning as "contaminant of concern", "contaminant of interest", or "contaminant of potential concern", respectively, and refers to "contaminants" as defined in 42 USC 9601(33).

Term	Definition
empirical data	data quantified in a laboratory
exposure assessment	the part of a risk assessment that characterizes the chemical exposure of a receptor
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

Term	Definition
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
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

This document presents the baseline human health risk assessment (BHHRA) for the Portland Harbor Superfund Site (Site). The BHHRA, which was conducted as part of the Remedial Investigation Report (RI Report) being conducted under the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA, or Superfund), presents an analysis of potential for adverse health effects associated with both current or potential future human exposures to hazardous substance releases at Portland Harbor in the absence of any actions to control or mitigate these releases. Portland Harbor generally refers to a heavily industrialized reach of the Lower Willamette River (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. The results of the BHHRA are used to assist in determining the need for remedial action, provide a basis for determining concentrations of chemicals that can remain in place and still be protective of public health, and to provide a basis for comparing the effectiveness of various remedial alternatives.

The overall process used for the BHHRA is based on EPA guidance and followed the approach documented in the Programmatic Work Plan and subsequent interim deliverables as well as numerous discussions, directives, and agreements with EPA, Oregon Department of Environmental Quality (DEQ), Oregon Department of Human Services (ODHS), and Native American Tribes.

As noted above, the Superfund program uses risk assessment as a tool to evaluate the likelihood and degree of exposure to hazardous substance released at Portland Harbor, and the adverse health effect associated with those exposures. The Superfund risk assessment process is comprised of the following four basic steps: 1) Data collection and analysis; 2) Exposure assessment; 3) Toxicity assessment, and 4) Risk characterization. Although the Portland Harbor BHHRA contains a separate assessment of uncertainties, this discussion should be considered an integral part of each of the four steps.

ES.1 DATA COLLECTION AND ANALYSIS

The dataset used in the BHHRA consists of surface sediment (defined as 0 to 30.5 centimeters in depth), surface water, groundwater seep water, clam and crayfish tissue, and fish tissue. 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.

ES.2 EXPOSURE ASSESSMENT

Currently or potentially exposed populations were identified based on consideration of both current and potential future uses of the Study Area, and include populations

who may be exposed to contamination though a variety of activities. The specific populations and exposure pathways evaluated were:

- Dockside workers — direct exposure via incidental ingestion and dermal contact with beach sediments.
- In-water workers — direct exposures to in-water sediment.
- Transients — direct exposure to beach sediment, surface water for bathing and drinking water scenarios, and groundwater seeps.
- Recreational beach users — direct exposure to beach sediment and surface water while for swimming.
- Tribal fishers — direct exposure to beach or in-water sediments, and consumption of migratory and resident fish.
- Recreational and subsistence fishers — direct exposure to beach or in-water sediments, consumption of resident fish, and consumption of shellfish.
- Divers — direct exposure to in-water sediment and surface water.
- Domestic water user — direct exposure to untreated surface water potentially used as a drinking water source in the future.
- Infant consumption of human breast milk — exposure to certain persistent and bioaccumulative contaminants (polychlorinated biphenyls [PCBs], dichlorodiphenyldichloroethane, dichlorodiphenyldichloroethylene, and dichlorodiphenyltrichloroethane [DDx] compounds, dioxins and furans, and polybrominated diphenyl ethers [PBDEs]) via nursing infants of dockside and in-water workers, divers, and recreational, subsistence, and tribal fishers

Exposure to beach sediment was assessed per beach, and exposure to groundwater seeps were assessed per seep. Exposure to in-water sediment, surface water, and fish and shellfish tissue were assessed on both localized and Study Area-wide scales.

As required by EPA policy, the exposure assessment evaluated a reasonable maximum exposure (RME), which is defined as the maximum exposure that is reasonably expected to occur. In addition, estimates of central tendency (CT), which are intended to represent average exposures, were also evaluated. Assumptions about each population were used to select exposure parameters to calculate the pathway-specific chemical intakes. As site-specific values are not available to describe potential exposures for each populations and pathways, default values representative of the larger U.S. population were used. Where default values are not available, best professional judgment was used based on likely activity patterns.

ES.3 TOXICITY ASSESSMENT

The toxicity assessment is an evaluation of both the type of adverse effects resulting from exposure to a chemical, as well as a quantitative estimate of the specific exposure, or dose at which adverse effects might occur. In the Superfund program, two types of health effects are evaluated; the incremental risk of developing cancer, and non-cancer health effects, such as developmental disorders and learning problems, or other health effects specific to certain organs. Cancer and noncancer toxicity values are used to assess the likelihood of a specific adverse health outcome. Cancer risk is expressed as a probability, and the cancer potency is known as the cancer slope factor. The non-cancer health hazard is expressed as the ratio of the estimated chemical intake, or dose, to a reference dose (RfD). The chronic RfD is an exposure representing an estimate (with uncertainty spanning perhaps an order of magnitude or greater) of a daily exposure, including sensitive subgroups, that is likely to be without appreciable health risks over a lifetime.

ES.4 RISK CHARACTERIZATION

Both cancer risks and noncancer hazards were evaluated in the BHHRA. To characterize potential noncancer hazard, the hazard quotient (HQ) was calculated as ratio of the estimated chemical intake, or dose, as described in the exposure assessment, to the chemical-specific RfD. The HQs were then summed to yield cumulative hazard indices (HIs). When the HI is less than 1, the estimated exposure is unlikely to result in adverse health effects. Where the HI is greater than 1, endpoint-specific HIs were calculated. This is done because the additive effects of exposure to more than one chemical are most appropriately considered when they cause similar health effects. Cancer risks are calculated as the product of the estimated dose and the cancer slope factor, resulting in an estimate of the incremental probability that an individual will develop cancer over a lifetime due to exposure to the chemical.

CERCLA actions are generally warranted where the baseline risk assessment indicates that a cumulative site risk to an individual using RME assumptions for either current or future land use is greater than a 1×10^{-4} lifetime excess cancer risk, or the HI is greater than 1.

ES.5 MAJOR FINDINGS

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 and hazards from fish and shellfish consumption exceed the EPA point of departure for cancer risk of 1×10^{-4} and target HI of 1 when evaluated on a harbor-wide basis, and when evaluated on the smaller spatial scale by river mile.

- Consumption of resident fish species consistently results in the greatest risk estimates. Evaluated harbor-wide, the estimated RME cancer risks are 4×10^{-3} and 1×10^{-2} for recreational and subsistence fishers, respectively. Evaluated on a river mile scale, it is only at RM 5, where the estimated RME risk for recreational fishers is 9×10^{-5}, that the risk from consumption of resident fish is less than 1×10^{-4}. River miles associated with the highest estimated risk estimates are RM 4, RM 7, RM 11, and in Swan Island Lagoon. Evaluated harbor-wide and assuming a diet that consisted of migratory fish in addition to resident fish species, the estimated RME cancer risk for tribal consumers is 1×10^{-2} assuming fillet-only consumption, and 2×10^{-2} assuming whole-body consumption.

- Noncancer hazard estimates for consumption of resident fish species are greater than 1 at all river miles. Evaluated harbor wide, the estimated RME HI is 300 and 1,000 for recreational and subsistence fisher, respectively. The highest hazard estimates are at RM 4, RM 7, RM 11, and in Swan Island Lagoon. The highest noncancer hazards are associated with nursing infants of mothers who consume resident fish from Portland Harbor. When fish consumption is evaluated on a harbor-wide basis, the estimated RME HI is 4,000 and 10,000 for infants of recreational and subsistence fishers, respectively. Evaluated on a harbor-wide scale, the estimated RME HI for tribal consumers of migratory and resident fish is 600 assuming fillet-only consumption, and 800 assuming whole-body consumption. The corresponding HI estimates for nursing infants of mothers who consume fish are 8,000 and 9,000 respectively, assuming maternal consumption of fillet or whole-body fish.

- PCBs are the primary contributor to risk from fish consumption harbor wide. When evaluated on a river mile scale, dioxins/furans are a secondary contributor to the overall risk and hazard estimates. PCBs are the primary contributors to the noncancer hazard to nursing infants, primarily because of the bioaccumulative properties of PCBs and the susceptibility of infants to the developmental effects associated with exposure to PCBs.

- The greatest source of uncertainty in the risk and hazard estimates includes the lack of good site-specific information about consumption of resident fish from Portland Harbor. Because tribal fish consumption practices were evaluated assuming a combined diet consisting of both resident and migratory fish, it is not clear to what degree contamination in Portland Harbor contributes to those estimated risks. In addition, it is important to remember that the noncancer hazard estimates presented in the BHHRA are not predictions of specific disease, and the cancer estimates represent upper-bound values, and the EPA is reasonably confident that the actual cancer risks will not exceed the estimated risks presented in the BHHRA.

1.0 INTRODUCTION

This Baseline Human Health Risk Assessment (BHHRA) presents an evaluation of risks to human health at the Portland Harbor Superfund Site (Site) in Portland, Oregon. This BHHRA is intended to provide an analysis of baseline risks and help determine the need for action at the Site, and to provide risk managers with an understanding of the actual and potential risks to human health posed by the site and any uncertainties associated with the assessment.

Portland Harbor encompasses the Lower Willamette River (LWR) in Portland, Oregon, from the confluence with the Columbia to about River Mile (RM) 12. It 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 four sampling rounds in the LWR from RM 0.8 to 12.2 to characterize the physical system of the river and to assess the nature and extent of contamination in sediment, surface water, transition zone water, storm water, and biota.

The LWG has worked with the United States Environmental Protection Agency (EPA) to develop the methods and assumptions used in this BHHRA. Consistent with EPA guidance (1989), this BHHRA incorporates assumptions to provide a health protective assessment of risks associated with contaminants present at the Site. The risk assessment for Portland Harbor is a baseline risk assessment in that it evaluates human health risks and hazards associated with contamination in the absence of remedial actions or institutional controls.

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 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 human health risk assessment in the CERCLA process is to provide an analysis of potential baseline risks to human health from site-related contaminants and help determine the need for remedial actions, provide a basis for determining contaminant concentrations that can remain onsite and still be protective

of public health, and provide a basis for comparing the effectiveness of various remedial alternatives. To achieve the overall objectives, the general process of BHHRA is:

- Identify contaminants of potential concern (COPCs)[3]
- Identify potentially exposed populations and pathways of exposure to 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 generally 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.

Potential exposure pathways, populations, and exposure assumptions were originally identified in the Programmatic Work Plan and in subsequent direction from EPA. 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 Toxicity Values Interim Deliverable (Kennedy/Jenks Consultants 2004a). Specific documents related to the approach for this BHHRA are presented in Attachment F1. The BHHRA is based on EPA (1989, 1991b, 2001a, 2004, 2005a) and EPA Region 10 (2000a) guidance, and is also consistent with DEQ guidance (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

[3] 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).

generally refers to a heavily industrialized reach of the LWR between RM 0 and RM 12, 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[4], this BHHRA focused on the Study Area, while also including data collected within the portion of the LWR that encompasses RMs 0.8 to 12.2.

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 the 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.

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 adjacent to 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, Swan Island Lagoon, and on the southern end of Sauvie Island. 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.

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

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. Four rounds of field investigations have been completed as part of the RI/FS. A preliminary sampling effort was conducted in 2001 and 2002 prior to the RI/FS work plan. 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 28.4 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, risk characterization, as well as a discussion of overall uncertainties.

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 potentially exposed 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, References – This section lists the references used in this BHHRA.

2.0 DATA EVALUATION

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 and non-LWG sampling events included in the site characterization and risk assessment (SCRA) dataset are described in detail in Appendix A of the RI Report. The dataset used in this BHHRA represents a subset of data from the sampling events that comprised the SCRA dataset as of September 2008. 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). Only data that met Category 1/QA2 data quality objectives was used in the BHHRA. A risk evaluation of exposures to polybrominated diphenyl ethers (PBDEs) detected in in-water sediment, fish and shellfish tissue was conducted 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 BHHRA dataset includes only those matrices relevant for direct human exposure pathways: surface sediment, clam and crayfish tissue, fish tissue, surface water and groundwater seeps. Other matrices included in the SCRA dataset (such as subsurface sediment) were not evaluated in the BHHRA because human exposure was considered unlikely. Data from RM 1.0, including Multnomah Channel, and upstream to RM 12.2, were included in the risk assessment. The BHHRA dataset is summarized by matrix from within and outside of the Study Area, respectively 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 based only on current conditions, as identified in the Programmatic Work Plan. Because beaches are relatively dynamic environments, specific beach conditions may change in the future, and the evaluation presented in the BHHRA may no longer be appropriately descriptive of potential risks.

Composite sediment samples were collected during Round 1 from each beach that had been designated as a potential human use area within the Initial Study Area (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 were also included in the BHHRA dataset. The designated potential human use areas and

associated beach sediment samples are shown in Map 2-1, and Table 2-3 presents a summary of the composite sediment samples included in the BHHRA dataset.

2.1.2 In-Water Sediment

The in-water sediment BHHRA dataset includes samples collected outside of the navigation channel of the river and from less than 30.5 cm in depth. Beach sediment samples are excluded, as well as natural attenuation core samples, radioisotope samples, and samples collected from areas that were subsequently dredged. The in-water sediment dataset is comprised of samples from river mile (RM) 1 to RM 12.2, including Swan Island Lagoon, as well as samples from the mouth of Multnomah Channel. As described in Appendix A of the RI, samples collected from areas that have subsequently been capped or dredged were not included in the BHHRA dataset. Per an agreement with EPA, the screening of contaminants of potential concern (COPCs) used only the subset of data collected from RM 1.9 to RM 11.8 (and including Swan Island Lagoon and the mouth of Multnomah Channel), whereas the exposure assessment and risk characterization used both subsets of data containing samples from RM 1 to RM 12.2. A summary of in-water sediment samples included in the BHHRA dataset is presented in Tables 2-3 and 2-4 from within and outside of the Study Area, respectively.

2.1.3 Surface Water

Surface water samples were collected by the LWG in seven separate events during Rounds 2 and 3 between 2004 and 2007, and are representative of various seasonal water flow conditions. Surface water samples were collected between RM 1.9 and RM 11.8 from 32 single point stations and 5 transect locations (at RM 2.0, Multnomah Channel, RM 3.9, RM 6.3, and RM 11). One additional surface water sample was collected from RM 16, outside the boundaries of the Study Area. Surface water samples were collected using 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 horizontally integrated across the entire width of the river, 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. Tables 2-5 and 2-6 present a summary of the surface water samples included in the BHHRA dataset from within and outside of the Study Area, respectively.

2.1.4 Groundwater Seeps

A seep reconnaissance survey was conducted during Round 1 to document readily identifiable groundwater seeps along both sides of the river from RM 2 to 10.5 (GSI

2003). Twelve potential groundwater seeps were observed at or near potential human use beach areas. Of these, only three sites were identified in the survey where it was considered likely for upland contaminants of interest (COIs)[5] to reach groundwater seeps or other surface expressions of groundwater discharging to human use beaches: the City of Portland storm sewer Outfall 22B, Willbridge, and McCormick and Baxter at Willamette Cove. Of these locations, only the Outfall 22B discharge was evaluated in the BHHRA. Groundwater infiltrates into the outfall pipe, which subsequently discharges to a beach that has been identified as a potential transient use area. The groundwater seep at Willbridge is at a beach restricted to industrial use, the seep at Willamette Cove, located 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 beach. The sampling events at Outfall 22B are described in Appendix A of the RI Report. Although samples have periodically been collected for analysis of the discharge at Outfall 22B both during and outside of stormwater events, samples taken during stormwater events were not included in the BHHRA dataset because they were not considered representative of typical exposures. Samples collected since 2002 were used in the BHHRA, and Table 2-5 presents a summary of the samples that were included in the BHHRA dataset.

2.1.5 Fish Tissue

The target fish species to be evaluated for human consumption were identified in the Programmatic Work Plan (Integral et al. 2004), and consisted of both resident and non-resident species. Samples of resident fish species were collected by the LWG during Rounds 1 and 3. Samples of non-resident fish species 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. Table 2-7 presents a summary of the fish tissue samples included in the BHHRA dataset.

2.1.5.1 Resident Fish Tissue

Resident fish species evaluated in the BHHRA are smallmouth bass (*Micropterus dolomieui*), black crappie (*Pomoxis nigromaculatus)*, common carp (*Cyprinus carpio carpio)*, and brown bullhead (*Ameiurus nebulosus*). The sampling protocol for each species differed based on the reported home ranges of species sampled. The tissue compositing scheme for the Round 1 data collection effort was reviewed and approved by EPA in November and December 2002. The Round 3 data collection, the tissue compositing scheme was approved by EPA in October 2007. Smallmouth bass

[5] 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).

and carp collected during Round 3 were analyzed separately as fillet and the remaining body-without-fillet tissue, and whole body concentrations were calculated using the individual fillet and body-without-fillet results. Thus, for the risk assessment, the Round 3 smallmouth bass samples were reported both as fillet and whole body results.

Smallmouth bass samples were collected in Round 1 from eight locations between RM 2 and 9, and corresponding to their small home range (ODFW 2005), and composited based on each river mile. Three whole body replicate composite samples were collected at three of the eight locations, one whole body composite sample and one fillet composite sample were collected at the 5 remaining sample locations. Round 3 samples were collected from 18 stations between RM 2 and 12, each corresponding to approximately one river mile, either the west or east side of the river, or both. One composite sample was collected from each station, typically consisting of five individual fish.

Black crappie, common carp, and brown bullhead samples were collected during Round 1 and composited from two three-mile long fishing zones, RM 3-6 and RM 6-9. Three common carp and brown bullhead whole body and fillet replicate composite samples were collected from each zone. Two black crappie whole body and fillet replicate composite samples were collected within each zone. All results from within the Study Area were included in the BHHRA dataset.

During Round 3, common carp samples were collected from 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 body-without-fillet tissue. All Round 3 results were included in the BHHRA dataset.

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

2.1.5.2 Salmon, Lamprey, and Sturgeon

Adult white sturgeon (*Acipenser transmontanus*), adult spring Chinook salmon (*Oncorhynchus tshawytscha*), and adult Pacific lamprey (*Lampetra tridentate*) were collected during ODHS Study. Although these data were not collected as part of the RI, the data met Category 1/QA2 data quality requirement s and were evaluated by the LWG and used in this BHHRA.

Adult Chinook salmon samples were collected at the Clackamas fish hatchery. Each composite sample consisted of three individual fish. Five whole-body (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.

Adult Pacific lamprey samples were collected at the Willamette Falls. Four whole body composite samples, each consisting of 30 individual fish, were analyzed.

Adult sturgeon samples were collected between RM 3.5 and 9.2. Six fillet samples were analyzed without skin (including one split), each sample consisting of a single fish.

2.1.6 Shellfish Tissue

Crayfish samples were collected from 24 stations during Round 1 based on habitat areas and from 9 stations during Round 3 based on habitat areas and data needs identified by the EPA. Commensurate with their limited home range, crayfish were collected and analyzed as whole body composite samples from each individual station. During Round 1, two replicate composite samples were collected at three of the 24 stations; a single composite sample was collected at the remaining stations. During Round 3, a single composite sample was collected at each station.

Clams (*Corbicula* sp.) were collected from three stations during Round 1, 33 stations during Round 2, and 10 stations during Round 3, sampling locations were 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. Round 1 and Round 2 samples were analyzed undepurated. As previously noted, two samples were collected from five of the sampling stations in Round 3, one sample from each station was depurated prior to analysis, the other was analyzed undepurated. At the remaining stations, only undepurated samples were analyzed. Depuration is a common method for cleansing shellfish that is often done prior to their consumption by humans to eliminate the sediment present in the gastrointestinal tract of the shellfish. 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. 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.

2.2 DATA EVALUATION

Prior to using the data in the BHHRA, the data were evaluated for inclusion in the BHHRA 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. 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).

2.2.1 Excluded Data

The data used BHHRA meet Category 1/QA2 data quality objectives, as described in Section 2.2 of the RI Report. Data that were not of this quality were removed from the BHHRA dataset. General reductions of the SCRA dataset to create the BHHRA dataset included removal of rejected analytical results ("R" qualified results), and removal of analytical results of samples collected from locations that have been capped, dredged, or remediated. This included all samples flagged as capped, dredged or remediated, including data from task WLCMBI02: the McCormick & Baxter September 2002 Sampling.

2.2.2 Field Replicates

Field replicates within the BHHRA dataset were handled per agreements with EPA. When calculating a mean or an upper confidence limit (UCL), and when reporting data in general, replicates were included in the dataset as discrete samples. Replicates with unique coordinates were included as separate samples when mapping or spatially weighting data. Where replicates have the same coordinates, data associated with the first sample were used and data from the second or third replicates were excluded.

2.2.3 Co-elution of PAHs

Benzo(b+k)fluoranthenes and benzo(k+j)fluoranthenes co-eluted in certain surface water and in-water sediment samples. For the purposes of the BHHRA, benzo(b+k)fluoranthenes results were assumed to be completely benzo(b)fluoranthene, and benzo(k+j)fluoranthenes results were assumed to be completely benzo(k)fluoranthene. Analytical results for these samples were not presented as co-elutions in the BHHRA, but rather, were presented as results for their assumed analyte.

2.2.4 Treatment of PCB Surface Water Data

Polychlorinated biphenyls (PCBs) were analyzed as Aroclors in samples collected using a peristaltic pump, and as congeners in high-volume samples collected using the XAD-2 sampling method. Because detection limits for the peristaltic pump samples were higher than those using high-volume samples, the results for PCBs from the high-volume samples were used. Aroclor concentrations in the high-volume samples were estimated from the PCB congener data by the

analytical laboratory. Therefore, Aroclor data were not used, and only PCB congener data were used to assess PCBs in the BHHRA surface water dataset.

2.2.5 Combining XAD Column and Filtered Surface Water Data

The XAD water quality samples consisted of two components: chemicals retained on the column that are representative of the dissolved concentration, and chemicals retained on the filter that are representative of the concentration of the suspended particulate fraction. In order to create a whole water sample from the XAD results, the analytical results for column and filter fractions for a given chemical were combined to give a total concentration. The following rules were used to calculate a whole water concentration for individual samples:

- If an analyte was detected in both the filter and the column, the detected concentrations were summed.
- If an analyte was detected in either the filter or the column but not in both portions of the sample, only the detected concentration was used.
- If an analyte was not detected in both the filter and the column, the highest detection limit reported for either the filter or the column was used.

Surface water samples collected using the high-volume XAD-2 sampling method are identified with the letters "XAD." The results of the combined XAD-2 column and filter data were renamed "WSXAD-Combo," and are presented as such in the BHHRA.

2.2.6 Combining Horizontal and Vertical Surface Water Data

The surface water data described in Section 2.1.3 were vertically integrated prior to use in the BHHRA. Transect samples are presented as a vertically and horizontally integrated transect. Non-integrated samples were collected from both near-bottom and near-surface (NB/NS) depths within the water column at single-point sampling locations. Vertically-integrated transect samples were collected from the east, west, and middle (E/W/M) sections of the river, horizontally integrated samples were collected from NB or NS water depths. NB/NS and/or E/W/M samples from the same location and date were combined to provide an integrated value for the water column or transect. In these cases, single-point data from NB and NS were vertically combined, vertically-integrated data from E/W/M were horizontally combined; and horizontally-integrated data from NB/NS were vertically combined using the following rules:

- If an analyte was detected in each sample, the detected concentrations were averaged.
- If an analyte was detected in at least one sample, the mean concentration was calculated using one-half the detection limit for non-detect results.

- If all results were non-detect, the mean of the detection limits was calculated and used as the non-detected concentration ("U" qualified).
- In some instances, a field replicate sample was collected from the middle of the river without corresponding replicate samples from the east or west side of the river, indicated by "M2" in the Sample ID. The results from these samples were included in the dataset at their reported concentrations, without combining them with other results.

Sample IDs for the results of the horizontally or vertically combined integrated data were renamed to include "-Int" at the end of the ID name, and are presented as such in the BHHRA.

2.2.7 Combining Fillet and Body-Without-Fillet Tissue Data

Fillet and whole-body data for smallmouth bass and carp were collected using the same fish for the LWG Round 3 sampling event. Fillet and remaining body-without-fillet tissue were composited and analyzed separately. Whole-body concentrations were calculated by weighting the concentration in the fillets and remaining-bodies according to the fractional weight of each tissue relative to the whole fish, and summing the weighted concentrations, as follows. The average fillet weight and remaining-body weight were calculated for each composite sample and added together to obtain the average whole-body weight. The proportion of each was then obtained by dividing the average fillet and remaining-body weights by the average whole-body weight. These values were multiplied by the respective chemical concentration and then added together to obtain the chemical concentration in the whole body, according to the following rules:

- If the analyte was detected in both the fillet tissue and the body without fillet tissue, a weighted average was calculated using the detected values
- If the analyte was not detected in either of the tissue types, a weighted average was calculated using the full detection limits
- If the analyte was detected either the fillet or body-without-fillet sample, one-half the detection limit for the non-detect result was used to calculate the weighted average.

The combined fillet and body without fillet tissue data were considered whole body tissue results for carp and smallmouth bass and were used in the BHHRA as such.

2.2.8 Summation Rules for Analytes Evaluated as Summed Values

Certain contaminants were evaluated as the sum of similar individual congeners, isomers, and closely related degradation products of the parent compound rather than as individual chemicals. The chemicals evaluated as mixtures and for which analytes evaluated as sums in the BHHRA are as follows:

- Total PCBs were calculated as either the sum of nine Aroclor mixtures (1016, 1221,1232, 1242, 1248, 1254, 1260, 1262, 1268) or the sum of individual PCB congeners.
- Total endosulfan was calculated as the sum of α-endosulfan, β-endosulfan, and endosulfan sulfate.
- Total chlordane was calculated as the sum of *cis*- and *trans*-chlordane, oxychlordane, and *cis*- and *trans*-nonachlor.
- Total DDD was calculated as the sum of 2,4'-DDD and 4,4'-DDD.
- Total DDE was calculated as the sum of 2,4'-DDE and 4,4'-DDE
- Total DDT was calculated as the sum of 2,4'-DDT and 4,4'-DDT
- Total dioxin-like PCB congeners were calculated as the sum of PCBs 77, 81, 105, 114, 123, 126, 156, 157, 167, 169, and 189.
- Total PCBs-adjusted were calculated as the sum of total PCB congeners minus dioxin-like PCB congeners.
- Total xylenes were calculated as the sum of *m*-, *o*-, and *p*-xylene.

The individual components of each chemical mixture used in the BHHRA are presented in Table F2-2.

If an individual analyte of a chemical mixture was detected at least once within the Study Area in a given medium, it was considered present in that medium. The presence of an analyte in biota samples was assessed separately for each individual species and tissue. The presence of individual analytes in sediment, and surface water were also assessed separately based on the specific exposure scenario. Individual analytes that were a part of a chemical mixture but were determined not to be present are summarized in Table F2-3 by medium and species. Additionally, a minimum number of individual analytical results in the mixture was required for the summed analytical result to be calculated. For example, if a sample was only analyzed for a limited number of individual PCB congeners, or if a large number of individual congener results for a sample were rejected, a total PCB congener sum may not have been calculated. In addition, chemical mixtures for samples meeting the criterion for the minimum number of individual analytical results required to calculate a sum, but with a limited number of individual analytical results, were qualified with an “A.” Mixture sums that did not have a limited number of individual analytical results were qualified with a “T,” indicating a calculated total. Table F2-4 shows the minimum number of individual analytical results required to calculate a sum for each mixture, and the maximum number of individual analytical results that would result in an “A” qualifier, indicating a limited number of individual analytical results were available for a sample. Table F2-4 also lists the number of samples for each medium for which a summed total was calculated, and the number of samples for which a summed total was not calculated because of lack of individual analytical

results for the mixture. Sample IDs of samples for which a summed analytical result was not calculated are presented in Table F2-5.

Concentrations of the individual analytes that comprise a mixture were summed for each sample according to the following rules:

- If an analyte was detected in the sample, the detected concentration was used to calculate the sum
- If an analyte was not detected in a sample but was assumed to be present in the sample medium, one-half the detection limit was used to calculate the sum
- If all results were non-detect, the highest detection limit of the analytes assumed to be present in the medium was used as the detection limit for the sample, and the sample was flagged as a non-detect.

2.2.9 Total Dioxin/Furan and PCB TEQs

A toxicity equivalence procedure was used to assess the cumulative toxicity of complex mixtures of PCDD, PCDF, and PCB congeners. The procedure involves assigning individual toxicity equivalency factors (TEFs) to the PCDD, PCDF, and PCB congeners in terms of their relative toxicity to 2,3,7,8-tetrachlorodibenzo-*p*-dioxin (2,3,7,8-TCDD). The reported concentration of each congener in a sample is multiplied by its respective TEF to give the TEF-equivalent concentration. The resulting concentrations are then summed to give a TEQ. The World Health Organization (WHO) TEFs (Van den Berg et al. 2006), shown in Table 4-3, were used to calculate the total dioxin/furan and PCB TEQs. Dioxin/furan and PCB-TEQs were calculated according to the following rules

- Congeners reported as not detected in a given sample but determined to be present in the medium, one-half the detection limit multiplied by the TEF was used in the sum
- If all results in a sample were non-detect, the maximum toxicity-weighted detection limit was used for the TEQ, and the result was flagged as non-detect (U-qualified). The maximum toxicity-weighted detection limit was obtained by multiplying each detection limit by its respective TEF and selecting the maximum value.
- Dioxin/furan TEQs were not calculated for those samples where analytical results for all 12 dioxin/furan congeners were not available.

Values were not presented for total TEQ in the BHHRA. Rather, risks from total TEQ were estimated by summing the risks from the total PCB TEQ and the total dioxin/furan TEQ.

2.3 CHEMICAL SCREENING CRITERIA AND SELECTION OF CONTAMINANTS OF POTENTIAL CONCERN

Because of the large number of chemicals detected in environmental media, a risk-based screening approach was used to focus the risk assessment on those contaminants most likely to significantly contribute to the overall risk. COPCs were selected for quantitative evaluation in the BHHRA by comparing the SCRA analytical data to risk-based screening values. The specific risk-based concentrations used to select COPCs are described below for the each media.

2.3.1 Sediment

EPA's Regional Screening Levels (RSLs) for soil (EPA 2010a) were used as the screening values for beach and in-water sediments. RSLs are risk-based concentrations in soil, air and water, and have been developed for both residential and industrial exposure scenarios. Using default exposure assumptions, RSLs represent concentrations that equate to a target cancer risk of 1×10^{-6} or a hazard quotient of 1. As described in Region 10 guidance (2007a), RSLs based on a noncancer endpoint were divided by 10 to give a value equivalent to using a hazard quotient of 0.1. This was done to account for the additive nature of noncancer effects. RSLs based on noncancer endpoints were divided by 10 to account for potential cumulative effects from multiple chemicals, and these modified RSLs were used as the screening values. Consistent with the then current EPA Region 10 recommendations (EPA, 2008a), RSLs of 7.7 mg/kg in soil for residential land use and 14 mg/kg for occupational land use were calculated for trichloroethylene (TCE) using a cancer slope factor of 0.089 per mg/kg-day, which represents the geometric mid-point of the slope factor range from EPA 2001. EPA finalized its risk assessment for TCE in 2011 and the revised RSL is 0.9 mg/kg. Because TCE does not contribute substantially to the cumulative risk estimates for the in-water portion of Portland Harbor, the screening process was not re-evaluated. Chemicals for which no RSL was available were screened using RSLs for chemicals with a similar chemical structure.

Because uses of Portland Harbor include both recreational and industrial activities, COPCs were selected using both residential and industrial RSLs, consistent with the EPA comments on the Round 2 Comprehensive Report (EPA 2008b). Residential RSLs were used to select COPCs in beach sediment for those areas where exposures could occur during recreational, transient, or fishing activities in those areas considered reasonably accessible from contiguous upland areas or by boat. In-water sediment data collected within the navigation channel were not used in the COPC screen. In areas where occupational exposures could occur, and for in-water sediment, COPCs were selected using industrial RSLs.

If the maximum detected concentration of a contaminant at a specific use area was greater than its respective screening level, that contaminant was selected as a

COPC. The designated potential uses for beaches in the Study Area are presented in Map 2-1. COPCs for beach sediment and the rationale for selection are presented in Tables 2-9 and 2-10. COPCs for in-water sediment are presented in Table 2-11. Sample locations used to select COPCs for in-water sediment are shown on Map 2-2.

2.3.2 Surface Water

Surface water COPCs were selected for divers and transient/beach user exposures using EPA residential tapwater RSLs (EPA 2010a), COPCs for the potential use of surface water as a drinking water source were selected using the lower of either the tapwater RSLs or MCLs (EPA 2003a). TCE was evaluated using the EPA Region 6 Human Health Medium-Specific Screening Level (EPA 2008c). COPCs for evaluating exposure to divers and for drinking water were selected from the combined surface water data set described in Section 2.2.6. COPCs for transient and beach use scenarios were selected from surface water samples taken from areas where direct contact could occur. A summary of samples used for screening surface water for COPCs is provided in Table 2-12. Sample locations of surface water data evaluated for diver exposures are shown on Map 2-4 and in Table 2-13; sample locations for transient and recreational beach uses are shown on Map 2-3 and Table 2-14; sample locations for the use of surface water as a drinking water source are shown on Map 2-8 and in Table 2-16.

2.3.3 Groundwater Seep

Chemicals concentrations detected in the groundwater seep at Outfall 22B were compared to the residential tapwater RSLs. As with the soil RSLs, the tapwater RSLs based on a noncancer endpoint were divided by 10 to give values equivalent to a HQ of 0.1. The location of Outfall 22B is shown on Map 2-5, and COPCs are presented in Table 2-15.

2.3.4 Fish and Shellfish Tissue

No appropriate risk-based screening values for fish tissue were available. Although EPA Region 3 has published fish tissue screening levels, the consumption rate of 54 g/day used to derive those values is not considered representative of the range of consumption rates relevant to Portland Harbor. Accordingly, all chemicals detected in fish and shellfish tissue in the BHHRA dataset were considered to be COPCs and evaluated further in the BHHRA. The general locations of fish in a particular composite of smallmouth bass and common carp are shown on Map 2-6. Brown bullhead and black crappie were composited over RM 3-6 and RM 6-9. Shellfish were composited over areas representing their assumed home range, and sample locations on Map 2-7 represent the general spatial distribution of composited samples.

3.0 EXPOSURE ASSESSMENT

Exposure assessment is the determination of the magnitude, frequency, duration, and route of exposure (EPA, 1989). Populations that currently, or may in the future, come into contact with site contaminants are identified along with potential routes of exposure that define the mechanism by which the exposure may occur. Magnitude is determined by estimating the amount, or concentration, of the chemical at the point of contact over an exposure duration, as well as the actual intake, or dose, of the chemical.

According to EPA (1989), an exposure assessment includes three primary tasks:

- Characterization of the exposure setting. This step includes identifying the characteristics of populations that can influence their potential for exposure, including their location and activity patterns, current and future land use considerations, and the possible presence of any sensitive subpopulations.
- Identification of exposure pathways. Exposure pathways are identified for each population by which they may be exposed to chemicals originating from the site.
- Quantification of exposure. The magnitude, frequency, and duration of exposure for each pathway is determined. This step consists of the estimating of exposure point concentrations and calculation of chemical intakes.

3.1 CONCEPTUAL SITE MODEL

The conceptual site model (CSM) describes potential contaminant sources, transport mechanisms, potentially exposed populations, exposures pathways and routes of exposure. As discussed in Sections 4, 5, and 6 of the RI Report, contaminated media within the Study Area are sediment, water, and biota. Current and historical industrial activities and processes within the Study Area have led to chemical releases from either point or nonpoint sources, including discharges to the river from direct releases or via outfalls and groundwater within the Study Area. In addition, releases that occur upstream of the Study Area and atmospheric deposition from global, regional, and local emissions may also represent potential contaminant sources to the Study Area. Chemicals in sediment and water may be accumulated by organisms living in the water column or by benthic organisms in sediments. Fish and shellfish within the Study Area feeding on these organisms can accumulate chemicals in their tissues through dietary and direct exposure to sediment and water. Additional information on potential contaminant sources is provided in Section 4 of the RI Report, and a more detailed CSM is presented in Section 10. A graphical representation of the exposure CSM is presented on Figure 3-1.

3.2 IDENTIFICATION OF POTENTIALLY EXPOSED HUMAN POPULATIONS

Potentially exposed populations were identified based on consideration of current and potential future uses of the Study Area. An analysis of potential exposure pathways for the Study Area was detailed in the Portland Harbor RI/FS Programmatic Work Plan (Integral 2004), including those directed by EPA. Consumption of shellfish by subsistence fishers, in-water exposures by recreational and commercial divers, and potential future domestic water use were subsequently evaluated after direction by EPA (see Attachment F1). The exposure scenarios identified below represent those populations that are anticipated to have the greatest potential for exposure to contaminants within the Study Area for both current and potential future conditions. For this reason, this risk assessment is likely to be protective of other potentially exposed populations that are not evaluated quantitatively in this BHHRA. The receptors evaluated for current and future uses of the Study Area are:

- Dockside workers
- In-water workers
- Transients
- Divers
- Recreational beach users
- Recreational/Subsistence fishers
- Tribal fishers
- Potential future domestic water users

The above populations were identified based on human activities known to occur within the Study Area, with the exception of the use of surface water as a domestic water source. However, public and private use of surface water is a beneficial use of the LWR, and as described in Section 1, this baseline risk assessment evaluates exposures assuming no institutional controls, such as obtaining a permit for use of surface water. Each of these receptors is described in greater detail in the following sections.

3.2.1 Dockside Workers

Portland Harbor supports a large number of water-dependent commercial uses, and many of the facilities adjacent to the LWR rely on ship and barge traffic. Dockside workers were evaluated to be representative of industrial and commercial workers at many of the facilities adjacent to the river. Specific activities are assumed to occur only within natural river beach areas, and include unloading ships or barges, or conducting occasional maintenance activities at specific locations near or at the water's edge. Exposures for dockside workers are evaluated as occurring only within defined areas considered to be industrial sites, rather than on a Study Area or harbor-wide basis. The specific areas evaluated are shown on Map 2-1.

3.2.2 In-Water Workers

In-water workers were evaluated as representative of individuals who conduct activities that typically occur in or over-water, rather than on shore as assumed for dockside workers. Specific activities may include the repair of in-water structures such as docks or pilings, maintenance dredging of private slips or berths, or maintenance and cleaning of equipment. While such activities would not necessarily be restricted to a given area, exposure would most likely be localized to specific facilities, and between the shore and the navigation channel.

3.2.3 Divers

Several different groups of people dive in the Portland Harbor area, including the public for recreation (which may include gathering of biota for consumption), the sheriff's office for investigations and emergency activities, and commercial divers for a variety of purposes including marine construction, underwater inspections, routine operation and maintenance, and activities related to environmental work. The majority of divers are expected to be commercial divers who typically use either wet or dry suits, wet or dry gloves, and a full face mask or a regulator held in the mouth with the diver's teeth. Although dry suits provide greater protection, wetsuits are occasionally used because of the higher cost of dry suits and higher water temperature (Sheldrake et. al, 2009). The Willamette River is 303d listed as a temperature impacted area, with the Lower Willamette reaching average temperatures of over 70 degrees F in the summer months. 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, the use of wet suits by commercial divers may still occur (EPA 2008d). Accordingly, two different diver exposure scenarios are included in this BHHRA, and are differentiated by considering the use of either a wet suit or dry suit. Each scenario assumes that divers are exposed to sediment and surface water through inadvertent ingestion and dermal contact throughout the Study Area.

3.2.4 Transients

Transient encampments are known to exist within the Study Area along the Lower Willamette River. While tents and makeshift dwellings are typically observed above actual beach areas, transients are likely to have direct contact with beach sediment and surface water (including groundwater seeps) during swimming, bathing or other activities, such as washing of clothing or equipment, and may also use surface water as a drinking water source. Although individuals are anticipated to move within or outside the Study Area, some individuals may spend a majority of their time at relatively few areas. Thus, exposure was evaluated as occurring at individual beaches rather than averaged over a larger area. Specific locations where exposure by transients was evaluated in the risk assessment are shown on Map 2-1.

3.2.5 Recreational Beach Users

Adults and children participate in recreational activities at beaches within the Study Area, and the LWR is also used for boating, water skiing, swimming, and other activities. The areas currently used for recreational activities as well as other areas in the Study Area where sporadic beach use may occur were identified as recreational use areas. While certain individuals may frequent a specific area almost exclusively, others users may regularly use various areas throughout the Study Area. Recreational activities are likely to result in exposure to beach sediment and surface water.

3.2.6 Recreational and Subsistence Fishers

A year-round recreational fishery exists within the Study Area. Current information indicates 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, an investigation by the Oregonian newspaper and 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 use fish from the lower Willamette either as a supplemental or primary dietary source (ATSDR 2002). These surveys also indicate that the most commonly consumed species are carp, bullhead catfish, and smallmouth bass, although other species may also be consumed. In conversations that were conducted as part of a project by the Linnton Community Center (Wagner 2004) about consumption of fish or shellfish from the Willamette River, transients reported consuming a large variety of fish, and several said they ate whatever they could catch themselves or obtain from other fishers.

Direct exposures to beach sediments by individuals engaged in recreational or subsistence fishing was evaluated at specific areas designated as transient and recreational use areas, exposures to in-water sediments were evaluated per half mile along each side of the river as well as on a Study Area-wide basis. Fish consumption was evaluated assuming a multi-species diet comprised of equal proportions of the four resident fish species (smallmouth bass, black crappie, brown bullhead, and common carp) for which tissue data are available and generally assuming consumption of fillets with skin. Because only smallmouth bass data are available on a river-mile basis, those data are used as representative of tissue concentrations in the other fish species when evaluating risk associated with fish consumption on a per river mile basis.

3.2.7 Tribal Fishers

The LWR provides a ceremonial and subsistence fishery for Native American tribes. Four Native American tribes (Yakama, Umatilla, Nez Perce, and Warm Springs) 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 consume more fish than the general public. Certain 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.

3.2.8 Potential Future Domestic Water User

According to the City of Portland, the primary domestic water source for the city 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). Although there are currently no known uses of the Lower Willamette River as a source of drinking water, public and private use of the Willamette River as a domestic water source is a designated beneficial use by the State of Oregon. Hence, use of surface water as a source of household water was assessed as a potentially complete pathway. Additionally, although domestic water supply is a designated beneficial use of the Willamette River, OAR 340-041-0340 Table 340A defines the beneficial use only with adequate pretreatment and natural quality that meets drinking water standards. Exposure to surface water could occur via ingestion and dermal contact, as well as volatilization of chemicals to indoor air through household use.

3.3 IDENTIFICATION OF EXPOSURE PATHWAYS

Exposure pathways are defined as the physical ways in which chemicals may enter the human body. 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)
- 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. The potential exposure pathways to human populations at the Study Area include:

- Incidental ingestion of and dermal contact with beach sediment
- Incidental ingestion and dermal contact with in-water sediment
- Incidental ingestion and dermal contact with surface water
- Incidental ingestion and dermal contact with surface water from seeps
- Consumption of fish and shellfish

- Infant consumption of human milk.

A more detailed discussion of potential exposures in the Study Area under current and future conditions and the rationale for including or eliminating pathways from quantitative evaluation are presented in the following sections. The identified receptors, exposure routes, and exposure pathways, and the rationale for selection are also summarized in Table 3-1.

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. Pathways considered potentially complete are quantitatively evaluated in this BHHRA.

Potentially Complete but 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, exposure via the pathway is likely to be negligible relative to the overall risk. Pathways considered potentially complete but 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 for a different receptor: These pathways may be complete for some individuals, but are not evaluated for the identified receptor because the pathways are not considered typical for that receptor. These pathways are evaluated for different receptors where the pathways are considered potentially complete and significant. Overlapping exposures that may occur for the different receptors are discussed further in Section 3.3.8.

The following sections provide a more detailed discussion of the exposure pathways that are quantitatively evaluated in this BHHRA.

3.3.1 Direct Exposure to Beach Sediment

Based on current and future uses within the Study Area, incidental ingestion and dermal contact with beach sediment could occur within natural river beach areas identified as human use areas in the Programmatic Work Plan. These areas were further classified with respect to the type of exposures that could occur, including recreational, fishing, transient, or dockside worker use areas. Human use areas in the Study Area and their associated classifications are shown in Map 2-1. Direct exposure to beach sediments is considered to be a potentially complete pathway for dockside workers, transients, recreational beach users, and fishers.

3.3.2 Direct Exposure to In-Water Sediment

Direct contact with in-water sediment could occur during activities conducted from a boat or other vessel that result in bringing sediment to the surface, during diving, or when fishing as a result of handling anchors, hooks, or crayfish pots. Hence, direct exposure to in-water sediment is considered to be a potentially complete pathway for in-water workers, divers, and fishers. Although recreational beach users may contact in-water sediment while swimming, such exposures are not expected to be significant and were not quantitatively evaluated in the risk assessment. Exposure to in-water sediment was evaluated throughout the Study Area by half-mile river segments for each side of the river rather than at specific areas as was done with exposure to beach sediments.

3.3.3 Direct Exposure to Surface Water

Direct exposure to contaminants in surface water could occur during recreation or occupational activities that occur near or in the water, or from potential future use of the LWR as a domestic water source. Transients may also use surface water as a source of drinking water or for bathing. Accordingly, direct exposure via ingestion and dermal contact with surface water is considered to be a potentially complete pathway for transients, recreational beach users, divers, and potential future domestic water users.

Exposure to contaminants in surface water via dermal absorption and ingestion were considered potentially complete but insignificant pathways for dockside workers, in-water workers, tribal fishers, and fishers. It is unlikely that dockside and in-water workers would have direct contact with surface water on a regular basis, and the potential for significant exposure is considered low while fishing. Additionally, although contaminants may volatilize from water, it is unlikely to result in a significant exposure considering the amount of mixing with ambient air and the relatively low concentrations of VOCs in water. Hence, inhalation of volatiles to ambient air was considered a potentially complete but insignificant exposure pathway for all receptors.

3.3.4 Direct Exposure to Groundwater from Seeps

Direct contact with groundwater is assumed to occur only at seeps where groundwater comes to the surface on a beach above the water line. Direct exposure to groundwater via seeps is considered a potentially complete exposure pathway for transients and recreational beach users. As described in Section 2.1.4, a seep reconnaissance survey identified only Outfall 22B, which is located at approximately RM 7W in an area designated as a potentially used by transients. Therefore, exposure to surface water from the groundwater seep at Outfall 22B was evaluated only for transients.

3.3.5 Consumption of Fish

Many of the contaminants found in Portland Harbor are persistent in the environment and accumulate in the food-chain. Local populations who consume fish caught in Portland Harbor may be exposed to COPCs that bioaccumulate in fish. 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. Consumption of locally-caught fish is evaluated as a potentially complete exposure pathway for dockside workers, in-water workers, recreational beach users, and divers. Consumption of fish by these populations is evaluated under the recreational and subsistence fisher receptor. By definition, ongoing long-term fish consumption by transients would not be expected to occur, and the evaluation of fish consumption for other receptors is considered to be protective of consumption of fish by transients.

3.3.6 Consumption of Shellfish

Certain contaminants can bioaccumulate in shellfish, and populations may be exposed to COPCs through consumption of shellfish that are collected within the Study Area. The actual extent shellfish harvesting and consumption is presently occurring is not known. The Linnton Community Center project (Wagner 2004) reported that some transients reported eating clams and crayfish, although 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. While the degree to which consumption of clams currently occurs in Portland Harbor is unknown, the Linnton Community Center project suggests that it does not occur on an ongoing basis within the Study Area. However, the predominant species found in the LWR during sampling events were Asian clams (*Corbicula*), which are an invasive, non-native species. Oregon law (OAR 635–056–0050) prohibits the possession, transportation, and sale of non-native wildlife, and the actual extent to which freshwater clams or other shellfish are currently harvested from Portland Harbor and consumed is not known. 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 4,300 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, DHS occasionally receives calls from citizens who are interested in harvesting crayfish from local waters and are interested in fish advisory information. According to a member of the Oregon Bass and Panfish club, traps are placed in the Portland Harbor Superfund Site boundaries and crayfish collected for bait and possibly for consumption (ATSDR 2006). Although consumption of shellfish was considered a

potentially complete pathway for dockside workers, in-water workers, recreational beach users, divers, and recreational fishers, it was quantitatively evaluated only for subsistence fishers, as they were considered the most likely population to regularly harvest and consume shellfish.

3.3.7 Infant Consumption of Human Milk

Lipid-soluble chemicals can accumulate in body fat, including lipids found in breast-milk. As a result, breast-feeding represents a potentially complete exposure pathway for nursing infants. Accordingly, infant exposures to PCBs, dioxins/furans, DDx, and PBDEs were evaluated as a potentially complete exposure pathway wherever maternal exposure to those compounds was evaluated.

3.3.8 Potentially Overlapping Exposure Scenarios

An estimate of reasonable maximum exposure should not only address exposure for individual pathways, but also exposures that may occur across multiple exposure routes. Examples of overlapping scenarios include in-water workers who fish recreationally, and may also be recreational beach users. Potentially overlapping scenarios are indicated on Figure 3-1, and risks from potentially overlapping scenarios are discussed in Section 5.

3.4 CALCULATION OF EXPOSURE POINT CONCENTRATIONS

The exposure point concentration (EPC) is defined as the average concentration contacted at the exposure point(s) over the duration of the exposure period (EPA, 1992a). EPA recommends using the average concentration to represent "a reasonable estimate of the concentration likely to be contacted over time" (EPA 1989). Use of the average concentration also coincides with EPA toxicity criteria, which are based on lifetime average exposures. Because of the uncertainty associated with estimating the true average concentration at a site, EPA guidance (EPA 1989, 1992) notes that the 95 percent upper confidence limit (UCL) of the arithmetic mean should be used for this variable. The UCL is defined as a value that, when calculated repeatedly for randomly drawn subsets of data, equals or exceeds the true population mean 95 percent of the time. Use of the UCL can also help account for uncertainties that can result from limited sampling data, and more accurately accounts for the uneven spatial distribution of contaminant concentrations. The process to calculate EPCs for tissue and beach sediment was previously described in the Programmatic Work Plan, and 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), as approved by EPA.

EPCs for RME evaluations represent either the 95 percent UCL, or the maximum detected value when either there was insufficient data to calculate a UCL or the calculated UCL was greater than the maximum reported value. Although inconsistent with EPA guidance (EPA 1992), EPCs for sediment and surface water CT evaluations were calculated as the simple arithmetic mean because such an evaluation is consistent with OAR 340-122-0084(1)(g) and the primary purpose of the CT evaluations is that they provide bounding information to evaluate uncertainties in the RME evaluation in this risk assessment. EPCs for fish/shellfish consumption scenarios are the lesser of the 95 percent UCL or the maximum detected concentration, central tendency evaluations were achieved by using mean or median consumption rates. For analytes with less than 5 detected concentrations, the maximum detected concentration for that exposure area was used as the EPC for the RME evaluation. The uncertainties associated with estimating EPCs from small datasets and with using the maximum detected concentration as the EPC are discussed in Section 6. The 95 percent 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 percent UCLs. If the data did not exhibit a discernible distribution, a non-parametric approach was used to generate a UCL. The 95 percent UCLs were calculated using the method recommended by ProUCL guidance (EPA 2007b).

Prior to calculating EPCs, the data were evaluated to address reporting of multiple results for the same analyte in the same sample and to reduce laboratory duplicates and field splits of samples to derive a single 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). Sample results are reported as not detected when the concentration of the analyte in the sample is less than the detection limit. The actual concentration may be zero, or some value between zero and the detection limit. The following rules were applied to the dataset for tissue, sediment, surface water, and groundwater seep samples:

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

2. A chemical was presumed to be present if it was detected at least once within the Study Area in samples for a given medium. When calculating the 95 percent UCL, non-detects were used in the calculation as recommended by the ProUCL software. ProUCL software output for the 95 percent UCLs calculated in this BHHRA are provided in Attachment F4. When calculating the simple mean, non-detected values were replaced with one half their detection limit in the calculations.

3. Non-detects for which the detection limit was greater than the maximum detected concentration in an exposure area were removed from the dataset prior to calculating EPCs.

Certain toxicity values are based on exposure to chemical mixtures rather than to individual chemicals, as identified in *Human Health Toxicity Values Interim Deliverable* (Kennedy/Jenks Consultants 2004a). Concentrations of the individual isomers or congeners that comprise the mixtures were summed as described in Section 2.2.8 to calculate the EPCs for the mixtures, and the risks from these chemicals were evaluated on the basis of the combined mixture rather than for individual chemicals.

3.4.1 Beach Sediment

EPCs for beach sediment were calculated using data collected during Rounds 1 and 2 from locations designated as human use areas. One composite sample was collected from each beach area, and the results from each composite sample were used as the EPC for the RME and CT evaluations. When evaluating exposure for dockside workers at industrial sites, the same EPC was used to represent adjacent sites in instances where the beach area extended across individual site boundaries. Otherwise, each designated beach area was evaluated as a single exposure area for transients, recreational beach users, and recreational, subsistence and tribal fishers. Beach sediment exposure areas are presented on Map 2-1, EPCs for dockside workers are presented in Table 3-2, EPCs for transient, recreational, and fishing uses are presented in Table 3-3.

3.4.2 In-Water Sediment

Direct contact with in-water sediment is most likely to occur in the near-shore areas outside of the navigation channel. Thus, only surface sediment data collected less than 30.5 cm in depth and outside of the navigation channel were used to exposure to in-water sediments. In–water sediment EPCs are calculated in one-half mile segments along both sides of the river from RM 1.0 to RM 12.2, and for samples within Multnomah Channel. Study Area-wide EPCs were calculated using the sediment data collected between RM 1.9 and 11.8. In-water sediment EPCs for exposures by in-water workers, divers, and recreational/subsistence/tribal fishers are presented in Table 3-4.

3.4.3 Surface Water

Exposure concentrations in surface water were calculated using data collected within the Study Area, as well as the transect data collected from the mouth of Multnomah Channel. Both integrated and non-integrated water column samples were included in the data set, the specific samples used were dependent upon the anticipated exposures

by the different receptors. A summary of the surface water samples used in calculating EPCs is presented in Table 3-5.

Surface water exposures by transients may occur throughout the year, EPCs were calculated using data from all seven seasonal sampling events. The data from each of the five transect locations were combined as described in Section 2.2.6. and EPCs were calculated for those five locations, at Willamette Cove using the discrete surface water samples, and on a Study Area-wide basis using the combined transect data from within the Study Area, excluding the transect location W027, which was collected at the mouth of Multnomah Channel. Surface water EPCs for exposures by transients are presented in Table 3-6.

Exposure to surface water by recreational beach users was assumed to occur primarily during summer months. Therefore, only data from the low-water sampling event conducted in July 2005 were used for calculating the surface water EPCs. These data were collected from recreational beaches in July 2005 included three transect locations and three single-point locations (Cathedral Park, Willamette Cove, and Swan Island Lagoon). Surface water EPCs for exposures by recreational beach users are presented in Table 3-7.

Exposures to surface water by divers were assumed to occur throughout the Study Area and were not considered seasonally dependent. EPCs were calculated in one-half mile intervals along each side of the river, and at each transect location. EPCs in surface water for exposures by divers are presented in Table 3-8.

Use of surface water as a domestic water source was assumed to have the potential to occur at any location through the Study Area on a year-round basis. Accordingly, data from all seven seasonal sampling events were used. EPCs were calculated for all individual transect stations and for single point stations with vertically integrated data. In addition, data from locations where co-located near-bottom and near-surface samples were collected were averaged and used in the domestic water dataset. Study Area-wide EPCs included all vertically integrated samples. EPCs for the use of surface water as a domestic water source are presented in Table 3-9.

3.4.4 Groundwater Seeps

As discussed Section 2.1.4, Outfall 22B, which is located on the west side of the river at RM 7, was the only seep identified where direct contact could occur within the Study Area. Data from two sampling events between 2002 and 2007 at times that did not involve stormwater influence were used to calculate the EPC, and the results are presented in Table 3-10.

3.4.5 Fish and Shellfish Tissue

EPCs for fish and shellfish tissue were calculated using data collected in the Round 1, Round 2, and Round 3 investigations, and the ODHS study. Study Area-

wide EPCs were calculated for recreational and subsistence fishers assuming a multi-species diet comprised of equal proportions of the four resident fish species for which data were collected. This was done by first calculating a harbor-wide concentration for each species, and then assigning a weighted fraction to each concentration corresponding to the portion of the total diet represented by each species to obtain a species-weighted EPC as follows:

$$Conc_{weighted} = \sum \left(C_{fish\,i} \times DF_{fish\,i} \right)$$

where:

$Conc_{weighted}$ = weighted chemical concentration fish tissue (mg/kg-wet weight)
C_{fish} = chemical concentration in fish tissue (mg/kg-wet weight)
DF_{fish} = fraction of total fish consumption represented by each species

Because PBDE data in fish tissue was limited to carp and smallmouth bass collected during Round 3 sampling, it was not possible to calculate a true weighted EPC. Therefore, the concentrations detected in smallmouth bass samples were used as representative of PBDE concentrations in other resident fish species. The EPCs for PBDEs are presented in Attachment F3.

EPCs were also calculated on a river-mile basis using fillet data from smallmouth bass as representative of chemical concentrations in all resident fish that might be collected from the same area. This was done because smallmouth bass are the only fish species for which data were collected and evaluated on a river mile scale. However, because fillet-only data were not collected in Swan Island Lagoon, EPCs in fillet tissue were estimated by calculating the COPC-specific ratios of the mean fillet-to-whole body concentration and applying that ratio to the measured whole body concentrations in Swan Island Lagoon.

Because of the manner in which data were collected and analyzed for other resident fish species, it is not possible to reliably make direct comparisons of chemical concentrations in fillet of smallmouth bass to other resident fish species on these smaller spatial scales. Averaged over a harbor-wide scale, the highest concentrations of persistent chlorinated organic compounds (such as PCBs and dioxins/furans) were detected in common carp, with increasingly lower concentrations detected in brown bullhead, smallmouth bass, and black crappie. PCB concentrations detected in common carp were as much as an order of magnitude greater than detected in smallmouth bass.

Crayfish and clams were collected and composited at each sampling location. EPCs for crayfish were calculated for each individual location as well as for the entire Study Area. EPCs for clams were calculated for both depurated and undepurated samples per river mile on each side of the river, as well as for the entire Study Area. 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 fish tissue are presented in Tables 3-11 through 3-15. The weighted Study Area-wide EPCs for a multi-species diet are presented in Table 3-16. EPCs for shellfish tissue are presented in Tables 3-17 through 3-20.

3.5 ESTIMATION OF CHEMICAL INTAKES

The amount of each chemical incorporated into the body is defined as the dose and is expressed in units of milligrams per kilogram per day (mg/kg-day). The dose is calculated differently when evaluating carcinogenic effects than when evaluating noncarcinogenic effects. Each is described below:

Non-cancer effects: The dose is averaged over the estimated exposure period and is expressed as a chronic daily intake (CDI). Thus, the CDI is used to represent the potential for adverse health effects over the period of exposure.

Carcinogenic effects: The dose is based on the estimated exposure duration, extrapolated over an estimated 70-year lifetime, representing the lifetime average daily intake (LADI). This is consistent with the cancer slope factors, which are based on lifetime exposures, and on the assumptions that the risk of carcinogenic effects is cumulative and continues even after exposure has ceased.

For non-occupational scenarios where exposures to children are considered likely, both adult and child receptors were evaluated. Children often exhibit behavior such as outdoor play activities and greater hand-to-mouth contact that can result in greater exposure than for a typical adult. In addition, children also have a lower overall body weight relative to the predicted intake. Because cancer risks are averaged over a lifetime, they are directly proportional to the exposure duration as well as the dose and the potency of the chemical. Accordingly, cancer risks were also assessed for a combined exposure from childhood through adult years, to account for the increased relative exposure and susceptibility associated with childhood exposures.

Superfund exposure assessments should be conducted such that the intake variables for an exposure pathway should result in an estimate of the reasonable maximum exposure (RME) expected to occur under both current and future land use conditions (EPA, 1989). The RME is defined as the highest exposure that is reasonably expected to occur at a site. The intent is to estimate an exposure that is substantially greater than the average, yet is still within the range of possible exposures. In general, this is accomplished by using a combination of 90th or 95th percentile values for contact rate, exposure frequency and duration, and 50th percentile values for other variables. This BHHRA also evaluated central tendency (CT) exposures, which is intended to represent an average exposure by the affected population. 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 3-21 through 3-28 and discussed in the following sections.

3.5.1 Incidental Ingestion of Sediment

The following equation was used to calculate the intake (expressed as milligrams per kilogram per day [mg/kg-day]) associated with the incidental ingestion of contaminants in soil or sediment:

$$CDI / LADI = \frac{C_s \times IRS \times 10^{-6}\, kg/mg \times EF \times ED}{BW \times AT}$$

Age-weighted exposures for the combined child and adult receptors were calculated consistent with the following equations:

$$CDI / LADI = \frac{C_s \times IFS_{adj} \times EF \times 10^{-6}\, kg/mg}{AT}$$

where:

$$IFS_{adj} = \frac{ED_c \times IRS_c}{BW_c} + \frac{ED_a \times IRS_a}{BW_a}$$

where:

C_s = chemical concentration in soil or sediment (mg/kg)
IFS_{adj} = age-adjusted soil/sediment ingestion factor [(mg-year)/(kg-day)]
IRS_a = adult soil/sediment ingestion rate (mg/day)
IRS_c = child soil/sediment ingestion rate (mg/day)
EF = exposure frequency (days/year)
ED_a = adult exposure duration (years)
ED_c = child exposure duration (years)
BW_a = adult body weight (kg)
BW_c = child body weight (kg)
AT = averaging time (days)

The exposure assumptions for estimating chemical intake from the ingestion of chemicals in sediment are provided in Tables 3-21 and 3-22.

3.5.2 Dermal Contact with Sediment

The following equation was used to calculate exposure resulting from dermal contact with contaminants in soil or sediment:

$$CDI / LADI = \frac{C_S \times ABS \times SA \times AF \times EF \times ED \times 10^{-6}\, kg/mg}{BW \times AT}$$

Combined child and adult age-weighted exposures resulting from dermal contact with contaminants in sediment for the recreational beach user exposure scenarios were calculated consistent with the following equations:

$$CDI / LADI = \frac{C_S \times SFS_{adj} \times ABS \times EF \times 10^{-6}\, kg/mg}{AT}$$

where:

$$SFS_{adj} = \frac{ED_c \times AF_c \times SA_c}{BW_c} + \frac{ED_a \times AF_a \times SA_a}{BW_a}$$

where:

C_s = chemical concentration in soil or sediment (mg/kg)
SFS_{adj} = age-adjusted dermal contact factor [(mg-year)/(kg-day)]
ABS = absorption efficiency
SA_a = adult exposed skin surface area (square centimeters [cm^2])
SA_c = child exposed skin surface area (cm^2)
AF_a = adult soil-to-skin adherence factor (mg/cm^2)
AF_c = child soil-to-skin adherence factor (mg/cm^2)
EF = exposure frequency (days/year)
ED_a = adult exposure duration (years)
ED_c = child exposure duration (years)
BW_a = adult body weight (kg)
BW_c = child body weight (kg)
AT = averaging time (days)

The exposure assumptions for estimating exposure from dermal contact with soil or sediment are provided in Tables 3-21 and 3-22.

Dermal absorption of chemicals from soil or sediment adhered to the skin is dependent on a variety of factors, including the condition of the skin, the nature of adhered soil/sediment, and the chemical concentration. Dermal absorption factors, representing the fraction of a chemical absorbed from soil or sediment adhered to the skin, are presented in Table 3-26. Only those compounds or classes of compounds for which dermal absorption factors are presented were evaluated quantitatively via dermal contact, although assuming less than complete absorption may not fully describe risks associated with dermally active compound such as carcinogenic PAHs. The uncertainties associated with the exposure and risk estimates via dermal exposures with soil and sediments are presented in Section 6.

3.5.3 Ingestion of Surface Water

Exposure resulting from ingestion of surface water was evaluated using the following equation:

$$CDI / LADI = \frac{C_w \times IR_w \times EF \times ED}{BW \times AT}$$

Combined child and adult age-weighted exposures due to ingestion of surface water were calculated consistent with the following equations:

$$CDI / LADI = \frac{C_w \times IFW_{adj} \times EF}{AT}$$

where:

$$IFW_{adj} = \frac{ED_c \times IRW_c}{BW_c} + \frac{ED_a \times IRW_a}{BW_a}$$

where:

C_W = chemical concentration in water (mg/L)
IFW_{adj} = age-adjusted water ingestion factor [(L-year)/(kg-day)]
IRW_a = adult groundwater ingestion rate (L/day)
IRW_c = child groundwater ingestion rate (L/day)
EF = exposure frequency (days/year)
ED_a = adult exposure duration (years)
ED_c = child exposure duration (years)
BW_a = adult body weight (kg)
BW_c = child body weight (kg)
AT = averaging time (days)

The exposure assumptions for estimating chemical intake from the ingestion of groundwater or surface water are provided in Tables 3-23 and 3-25.

3.5.4 Dermal Contact with Surface Water

Dermal absorption of contaminants due to direct contact with surface water was evaluated using the following equation:

$$CDI / LADI = \frac{DA_{event} \times EV \times EF \times ED \times SA}{AT \times BW}$$

The combined child and adult age-weight absorption of contaminants due to direct contact with surface water was evaluated using the following equation:

$$CDI / LADI = \frac{DA_{event} \times EF \times DFW_{adj}}{AT}$$

The dermally-absorbed dose (DA_{event}) is calculated for organic analytes as a function of the length of exposure and the permeability of the skin to the chemical being absorbed. The rate a chemical enters the skin surface can be greater than the rate by which the chemical is leaving the skin and entering the bloodstream. If exposure is long enough, the chemical enters the skin at the same rate that it exits; this is a condition known as steady-state, designated as t^*. When the exposure duration is less the t^*, the DA_{event} is calculated as:

$$DA_{event} = 2 \times FA \times K_p \times C_w \times CF \times \sqrt{\frac{6 \times \tau \times ET_{adj}}{\pi}}$$

When the exposure duration is greater than t^*, DA_{event} is calculated as:

$$DA_{event} = K_p \times C_{water} \times CF \times \left[\left(\frac{ET_{adj}}{1+B} \right) + 2\tau \left(\frac{1+3B+3B^2}{(1+B)^2} \right) \right]$$

The age-adjusted exposure time is calculated as:

$$ET_{adj} = \frac{(ET_c \times ED_c) + [ET_a \times (ED_a - ED_c)]}{ED_r}$$

and the age-adjusted dermal contact factor for water, DFW_{adj} is calculated using the following equation:

$$DFW_{adj} = \frac{EV_c \times ED_c \times SA_c}{BW_c} + \frac{EV_a \times ED_a \times SA_a}{BW_a}$$

Where:

C_w = chemical concentration in water (mg/L)
DA_{event} = dermally absorbed dose (mg/cm^2-event)
DFW_{adj} = age-adjusted dermal contact factor (cm^2-event-day/kg)
K_p = dermal permeability coefficient (cm/hour)
τ = lag time (hours)
EV = events per day
EF = exposure frequency (days/year)
ET = exposure time (hours)
FA = fraction of chemical absorbed
CF = conversion factor (10^{-3} L/cm^3)
ED_a = adult exposure duration (years)
ED_c = child exposure duration (years)
SA_a = adult exposed skin surface area (cm^2)

SA_c = child exposed skin surface area (cm^2)
BW_a = adult body weight (kg)
BW_c = child body weight (kg)
AT = averaging time (days)

Chemical-specific factors are presented in Tables 3-27 and 3-28. These values were obtained from Appendix B of EPA's Supplemental Guidance for Dermal Risk Assessment (2004). The uncertainties associated with calculating DA_{event} for chemicals with factors outside of the Effective Prediction Domain are discussed in Section 6.

3.5.5 Consumption of Fish/Shellfish

The following equation was used to estimate exposure associated with the consumption of fish and shellfish:

$$CDI / LADI = \frac{C_t \times IR \times 10^{-3} kg / g \times EF \times ED}{BW \times AT}$$

Combined child and adult exposure was evaluated consistent with the following equation:

$$CDI / LADI = \frac{C_t \times IR_{t-adj} \times 10^{-3} kg / g \times EF}{AT}$$

where:

$$IR_{t-adj} = \frac{ED_c \times IR_c}{BW_c} + \frac{ED_a \times IR_a}{BW_a}$$

where:

C_t = Contaminant concentration in fish tissue (mg/kg, wet-weight basis)
IR_c = Fish consumption rate - child (g/day, wet-weight basis)
IR_a = Fish consumption rate - adult (g/day, wet-weight basis)
EF = Exposure frequency (days/year)
ED_c = Exposure duration – child (years)
ED_a = Exposure duration – adult (years)
BW_c = Body weight – child (kg)
BW_a = Body weight – adult (kg)
AT = Averaging time (days)

The exposure assumptions used to estimate exposure from fish and shellfish consumption are presented in Table 3-24.

3.5.6 Calculation of Intake due to Infant Consumption of Human Milk

Exposure to breastfeeding infants due to consumption of human milk was evaluated using a methodology developed by ODEQ, OHA, and EPA Region 10, adapted from EPA's Methodology for Assessing Health Risks Associated with Multiple Pathways of Exposure to Combustor Emissions (EPA 1998a) and the Human Health Risk Assessment Protocol for Hazardous Waste Combustion Facilities (EPA 2005a), and is described in detail in Appendix D of the DEQ Human Health Risk Assessment Guidance (DEQ 2010). The evaluation for this pathway focuses on PCBs, dioxins/furans, DDx, and PBDEs because of the propensity of these chemicals to bioaccumulate. Because the concentration of lipophilic chemicals in human milk is most directly correlated with the steady-state body burden, which itself is directly related to the long-term intake of the chemical, the daily maternal absorbed intake is calculated from the average daily dose to the mother (as calculated in the preceding sections) using the following equation:

$$DAI_{maternal} = ADD_{maternal} \times AE$$

where:

$DAI_{maternal}$ = daily absorbed intake of the mother (mg/kg-day)
$ADD_{maternal}$ = age-adjusted soil/sediment ingestion factor (mg/kg-day)
AE = absorption efficiency of the chemical

The steady-state chemical concentration in milk fat is then calculated as:

$$C_{milkfat} = \frac{DAI_{maternal} \times h \times f_f}{ln(2) \times f_{fm}}$$

where:

$C_{milkfat}$ = chemical concentration in milk fat (mg/kg-lipid)
$DAI_{maternal}$ = daily absorbed intake of the mother (mg/kg-day)
h = half-life of chemical (days)
f_f = fraction of absorbed chemical stored in fat
f_{fm} = fraction of mother's weight that is fat

Intake for infants via breastfeeding is then calculated as:

$$Intake = \frac{C_{milkfat} \times f_{mbm} \times CR_{milk} \times ED_{inf}}{BW_{inf} \times AT}$$

where:

f_{mbm}	=	fraction of fat in breast milk
CR_{milk}	=	consumption rate of breast milk (kg/day)
ED_{inf}	=	exposure duration of breastfeeding infant (days)
BW_{inf}	=	average infant body weight (kg)
AT	=	averaging time (days)

Additional information regarding the evaluation of persistent, bioaccumulative COPCs is presented in Section 5.1.3.

3.5.7 Calculation of Intake for Mutagenic COPCs

Early-in-life susceptibility to carcinogens has long been recognized by the scientific community as a public health concern. In its revised Cancer Assessment Guidelines, EPA concluded that existing risk assessment approaches did not adequately address the possibility that exposures to a chemical in early life may can result in higher lifetime cancer risks than a comparable duration adult exposure (EPA 2005b). In order to address this increased risk, the agency recommends use of a potency adjustment to account for early-in-life exposures. When no chemical-specific data are available to assess directly cancer susceptibility from early-life exposure, the following default Age Dependent Adjustment Factors (ADAFs) are recommended to be used when evaluating a carcinogen known to cause cancer through a mutagenic mode of action:

- 10-fold adjustment for exposures during the first 2 years of life;
- 3-fold adjustment for exposures from ages 2 to <16 years of age; and
- No adjustment for exposures after turning 16 years of age.

Of the COPCs evaluated in this HHRA, EPA considers that there is sufficient weight-of-evidence to conclude the carcinogenic PAHs cause cancer through a mutagenic mode of action.

The equations used to calculate intake for mutagenic COPCs are presented in the following sections. The exposure parameters are presented in Tables 3-21 to 3-25.

3.5.7.1 Incidental Ingestion of Sediment

The following equation was used to calculate the intake in mg/kg-day for mutagenic COPCs associated with incidental ingestion of soil or sediment:

$$CDI / LADI = \frac{C_s \times \left(\frac{(ED_{0\text{-}2} \times IRS_c) \times 10}{BW_c} + \frac{(ED_{2\text{-}6} \times IRS_c) \times 3}{BW_c} + \frac{(ED_{6\text{-}16} \times IRS_a) \times 3}{BW_a} + \frac{(ED_{16\text{-}30} \times IRS_a) \times 1}{BW_a} \right) \times EF}{AT}$$

where:

C_s = chemical concentration in soil or sediment (mg/kg)
IRS_a = adult soil/sediment ingestion rate (mg/day)
IRS_c = child soil/sediment ingestion rate (mg/day)
EF = exposure frequency (days/year)
$ED_{0\text{-}2}$ = exposure duration ages 0-2 (years)
$ED_{2\text{-}6}$ = exposure duration ages 2-6 (years)
$ED_{6\text{-}16}$ = exposure duration ages 6-16 (years)
$ED_{16\text{-}30}$ = exposure duration ages 16-30 (years)
BW_a = adult body weight (kg)
BW_c = child body weight (kg)
AT = averaging time (days)

3.5.7.2 Dermal Contact with Sediment

The following equation was used to calculate the intake from dermal contact with contaminants in soil or sediment:

$$CDI/LADI = \frac{C_S \times \left(\frac{ED_{0\text{-}2} \times AF_c \times SA_c \times 10}{BW_c} + \frac{ED_{2\text{-}6} \times AF_c \times SA_c \times 3}{BW_c} + \frac{ED_{6\text{-}16} \times AF_a \times SA_a \times 3}{BW_a} + \frac{(ED_{16\text{-}30} \times AF_a \times SA_a \times 1}{BW_a} \right) \times ABS \times EF \times 10^{-6} kg/mg}{AT}$$

where:

C_s = chemical concentration in soil or sediment (mg/kg)
ABS = absorption efficiency
SA_a = adult exposed skin surface area (square centimeters [cm^2])
SA_c = child exposed skin surface area (cm^2)
AF_a = adult soil-to-skin adherence factor (mg/cm^2)
AF_c = child soil-to-skin adherence factor (mg/cm^2)
EF= = exposure frequency (days/year)
$ED_{0\text{-}2}$ = exposure duration ages 0-2 (years)
$ED_{2\text{-}6}$ = exposure duration ages 2-6 (years)
$ED_{6\text{-}16}$ = exposure duration ages 6-16 (years)
$ED_{16\text{-}30}$ = exposure duration ages 16-30 (years)
BW_a = adult body weight (kg)
BW_c = child body weight (kg)
AT = averaging time (days)

3.5.7.3 Ingestion of Surface Water

The following equation was used to calculate intake of chemicals associated with ingestion of surface water:

$$CDI / LADI = \frac{C_w \times \left(\frac{(ED_{0\text{-}2} \times IRW_c) \times 10}{BW_c} + \frac{(ED_{2\text{-}6} \times IRW_c) \times 3}{BW_c} + \frac{(ED_{6\text{-}16} \times IRW_a) \times 3}{BW_a} + \frac{(ED_{16\text{-}30} \times IRW_a) \times 1}{BW_a} \right) \times EF}{AT}$$

where:

C_w	=	chemical concentration in water (mg/L)
IFW_{adj}	=	age-adjusted water ingestion factor [(L-year)/(kg-day)]
IRW_a	=	adult groundwater ingestion rate (L/day)
IRW_c	=	child groundwater ingestion rate (L/day)
EF	=	exposure frequency (days/year)
$ED_{0\text{-}2}$	=	exposure duration ages 0-2 (years)
$ED_{2\text{-}6}$	=	exposure duration ages 2-6 (years)
$ED_{6\text{-}16}$	=	exposure duration ages 6-16 (years)
$ED_{16\text{-}30}$	=	exposure duration ages 16-30 (years)
BW_a	=	adult body weight (kg)
BW_c	=	child body weight (kg)
AT	=	averaging time (days)

The exposure parameters are presented in Tables 3-21 to 3-25.

3.5.8 Population-Specific Exposure Assumptions

Assumptions about each receptor population evaluated in this BHHRA were used to select exposure parameters used to calculate the pathway-specific chemical intakes. Site-specific values are not available for all populations and pathways. Therefore, default values representative of the general U.S. population (EPA 1991b) or values representing best professional judgment based on known human uses of the Study Area were used. The majority of the exposure parameters used in this BHHRA were previously described in the *Exposure Point Concentration Calculation Approach and Summary of Exposure Factors* (Kennedy/Jenks Consultants 2006), which was approved by EPA. Exposure parameters for divers were provided by EPA in its comments on the Round 2 Report. The exposure parameters are discussed below and presented in Tables 3-21 to 3-25. These values represent potential exposures for application at appropriate areas and/or areas agreed upon with EPA and its partners within the Study Area.

3.5.8.1 Dockside Workers

Exposure frequency for dockside workers was assumed to be 50 days/year for the RME evaluation, and 44 days/year the CT evaluation. The RME value assumes a dockside worker is exposed to beach sediment one day per week for 50 weeks per year (50 weeks/year is based on the average number of days worked by an outdoor

worker as being 225 days/year, according to the U.S. Census Bureau's *1990 Earnings by Occupation and Education Survey,* and assuming a 5-day work week). An exposure duration of 25 years was used, representing an EPA default value for the RME estimate of job tenure. This value is consistent with data from the U.S. Bureau of Labor Statistics showing that the 95th percentile job tenure for men in the manufacturing sector is 25 years. The CT estimate assumed duration of 9 years, representing approximately the 50th percentile of residence time estimates from the U.S. Census Bureau data (EPA, 1997).

A sediment ingestion rate of 200 mg/day was used for the RME evaluation, based on EPA Region 10 supplemental guidance on soil ingestion rates (EPA, 2000a), and is representative of approximately the midpoint between the recommended values of 100 mg/day for outdoor workers and 330 mg/day for construction workers. An ingestion rate of 50 mg/day was used to estimate CT exposure.

Dermal exposure was assessed assuming that the face, forearms and hands are exposed, representing an exposed skin surface area of 3,300 cm^2, which is representative of the median value (50th percentile) for adults. A body weight of 70 kg, representing the 50th percentile of mean body weights of men and women combined (EPA, 1997a) was used for all adult receptors. RME and CT exposure values for dockside workers are presented in Table 3-21.

3.5.8.2 In-Water Workers

According to the Army Corps of Engineers (Siipola 2004), the Port of Portland conducts the most frequent dredging within the Study Area, thus the exposure factors for workers at Terminal 4 are considered protective of in-water workers for potential in-water sediment exposures throughout the Study Area. Exposure factors for in-water workers were developed based on in-depth interviews with several workers at Terminal 4 who either conduct or oversee activities that could result in contact with in-water sediment. For the RME evaluation, in-water sediment exposures were assumed to occur for 10 of 25 years of employment at a given facility, with an exposure frequency of 10 days of sediment contact per year. For the CT evaluation, contact with in-water sediment is assumed for 4 of 9 years employment at a given facility, with an exposure frequency of 10 days of sediment contact per year. Intake rates for in-water sediment are the same as those used for the dockside worker, which are the default ingestion rate of soil for an industrial worker. RME and CT exposure values for the in-water worker are presented in Table 3-22.

3.5.8.3 Divers

Two different scenarios were evaluated, based on whether the divers wear wet or dry suits. Divers wearing wet suits are assumed to be working as commercial divers without a full face mask, and wearing either wet gloves or no gloves. An exposure frequency of 5 days/year for the RME evaluation and 2 days/year for the CT evaluation are based on best professional judgment and discussions between EPA,

LWG, and commercial divers, as well as the experience of EPA divers who work at the Portland Harbor Superfund site. Exposure durations of 25 years and 9 years were used for the RME and CT estimates, respectively, based on the labor statistics for job tenure described in Section 3.5.9.1.

Sediment ingestion rates were assumed to be 50 percent of the ingestion rate for dockside workers, corresponding to values of 50 mg/day and 25 mg/day, respectively for the RME and CT evaluations. Dermal exposure to sediment for divers wearing a wet suit was evaluated assuming the entire skin surface area was exposed. A value of 18,150 cm^2, representing the median skin surface area for men and women was used for both the RME and CT evaluations. Divers wearing a dry suit (with a neck dam) would likely have only their head, neck, and hands exposure, and a RME value of 2,510 cm^2 was used. Sediment dermal adherence factors for of 0.3 mg/cm^2-event and 0.07 mg/cm^2 event was used for the was used for the RME estimate and CT estimate, respectively. A CT evaluation was not done for divers wearing dry suits.

Incidental ingestion of surface water for both diver scenarios was assumed to be 50 mL/hour for both the RME and CT evaluations (EPA 1989). More recent data regarding estimates of the amount of water ingested by commercial divers indicates that on average, occupational divers ingested 6 mL/dive in freshwater and 10 mL/dive in marine water, with the maximum estimated ingestion ranging between 25 and 100/mL/dive (EPA 2011). Exposure via ingestion and dermal contact was assumed to occur for 4 hours/event for the RME estimate and 2 hours/event for the CT estimate.

Tables 3-22 and 3-23 summarize exposure assumptions for the wet suit and dry suit divers for in-water sediment and surface water, respectively.

3.5.8.4 Transients

Little information is available regarding how long individuals may remain at specific locations or within the Study Area itself. Based on professional judgment, an exposure duration of 2 years was assumed for the RME and 1 year for CT evaluations, exposure frequency was assumed to be daily (365 days/year). Incidental ingestion of sediment was evaluated at the same rates used for the dockside workers (200 mg/day). Dermal exposure was assessed assuming that the face, forearms and hands, and lower legs are exposed, representing an exposed skin surface area of 5,700 cm^2, which represents the median value for adults. A soil adherence factor of 0.3 mg/cm^2 was used based on the expectation that beach sediment would have a greater moisture content than dry soil. An ingestion rate of 2 L/day was used for consumption of surface water, which represents the default value for domestic water use. Tables 3-21 and 3-23 summarize RME and CT exposure values for the transient scenario for beach sediment and surface water, and the reference and rationale for each value.

3.5.8.5 Recreational Beach User

In the absence of specific information regarding the frequency of recreational activities in Portland Harbor, potential exposures are based on best professional judgment, assuming that beach use is most frequent in the summer, with less frequent use in the spring/fall, and only intermittent use in the winter. An exposure frequency of 94 days/year (5 days/week during summer, 1 day/week during spring/fall, and 1 day/month during winter) was used for the RME estimate and 38 days/year (2 days/week during summer, 2 days/month during spring/fall) was used for the CT estimate. Exposure duration for recreational activities is based on the assumption that individuals are largely permanent residents of the Portland area. Accordingly, an exposure duration of 30 years, which represents approximately the 95th percentile of the length of continuous residence in a single location in the U.S. population (EPA 1997) was used for the RME estimate. More recent studies described in the 2011 edition of EPA's Exposure Factors Handbook show the 95th percentile value is closer to 33 years, data from the U.S. Census Bureau indicate that 32 years represents the best estimate of residence time at the 90th percentile. However, the value of 30 years is consistent with other Superfund risk assessments nationwide, and represents a reasonably conservative estimate of total residence time in the area. An exposure duration of 9 years was used for the CT estimate.

Sediment ingestion rates of 100 mg/day for adults and 200 mg/day for children were used, approximating the 95th percentile soil ingestion rates. CT estimates assumed sediment ingestion rates of 100 mg/day for children and 50 mg/day for adults. Dermal exposures were evaluated assuming that the face, forearms and hands, and lower legs are exposed. Median values of 5,700 cm^2 and 2,800 cm^2 were used for adults and children, respectively. A soil-skin adherence of 3.3 mg/cm^2-day was used for children to account for the greater moisture content of beach sediment.

Water temperatures in the Lower Willamette River would typically limit swimming to the summer months, thus the RME estimates for swimming were assumed to occur at a rate of 26 days per year for adults and 65 days per year for children. As discussed in Section 3.5.10.3, incidental ingestion of river water was assumed to occur at a rate of 50 mL/hour while swimming. Based on current recommendations, 50 mL/hr represents mean value, assuming 21mL/hr for adults and 49 mL/hr for children, upper-percentile recommended values are 71 mL/hr for adults and 121 mL/hr for children (EPA 2011). Tables 3-21 and 3-23 summarize RME and CT exposure values for beach sediment and surface water, respectively, for adult and child recreational beach users.

3.5.8.6 Recreational and Subsistence Fishers

Because there is limited information regarding the frequency of fishing activities within the Study Area, a range of possible exposures was evaluated for people who engage in recreational or subsistence fishing activities by considering both a high- and a low-frequency rate of fishing. RME estimates for high-frequency fishers assumed a fishing frequency of 156 days/year, approximating a rate of 3 days/week.

Low-frequency fishers were assumed to fish 104 days/year, approximating a rate of 2 days/week. CT estimates assumed a frequency of 52 days/year and 26 days/year for high- and low-frequency fishers, respectively, and are representative of assumed fishing frequencies of 1 day/week and 2 days/month. People engaged in recreational or subsistence fishing were also assumed to be residents of the greater Portland area, therefore exposure durations of 30 years and 9 years, were used for the RME and CT evaluations, respectively, based on the population statistics for residency discussed in Section 3.5.8.5.

Incidental ingestion of beach sediment was evaluated assuming 100 mg/day for the RME estimate and 50 mg/day for the CT estimate, representative of soil ingestion rates in a typical residential setting. Rates of 50 mg/day for the RME estimate and 25 mg/day for the CT estimate were used for incidental ingestion of in-water sediment, representing 50 percent of the rates used for beach sediment. An exposed surface area of 5,700 cm^2, representing the face, hands, forearms and lower legs was used to assess dermal exposure to beach sediments, exposures to in-water sediment was assumed to be limited to the hands and forearms, corresponding to a surface area of 1,980 cm^2. Sediment adherence to skin was evaluated using a weighted adherence factor based on exposure to the hands, forearms, and lower legs (EPA 2004). A factor of 25 percent was used to account for the time spent fishing in a single area within the Study Area. Exposure assumptions for beach and in-water sediment contact for recreational/subsistence fishers are presented in Tables 3-21 and 3-22

Information currently available indicates 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, an investigation by the Oregonian newspaper and 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 either as a supplemental or primary dietary source (ATSDR 2002). These surveys also indicate that the most commonly consumed species are carp, bullhead, catfish, and smallmouth bass, although other species may also be consumed. In conversations that were conducted as part of a project by the Linnton Community Center (Wagner 2004) about consumption of fish or shellfish from the Willamette River, transients reported consuming a large variety of fish, and several said they ate whatever they could catch themselves or obtain from other fishers.

No studies were located that document specific consumption rates of recreational or subsistence anglers in Portland Harbor prior to its listing as a Superfund site, and surveys conducted subsequent to the listing would not be representative of historical, baseline consumption patterns due to subsequent fish advisories and efforts to limit consumption of fish caught from the harbor. Therefore, fish consumption rates from published studies were used to describe the range of reasonably expected exposures relevant to the different populations known to occur in the Portland Harbor area.

Three different rates were evaluated: 17.5 grams per day (approximately 2 eight ounce meals per month), 49 g/ day (approximately 6 ½ eight ounce meals per month), and 142 g/day per day (19 eight ounce meals per month). The term "recreational fishers" is intended to encompass a range of the population while focusing on those who may fish on a more-or-less regular basis, and "subsistence fishers" to represent populations with high fish consumption rates, recognizing that fish are not an exclusive source of protein in their diet. Accordingly, 17.5 g/day is considered representative of a CT value for recreational fishers, and 49 g/day was selected as the RME value representing the higher-end consumption practices of recreational fishers. The consumption rate of 142 g/day represents a RME value for high levels of fish consuming, or subsistence, fishers. No CT value was selected because the evaluations based on 17.5 g/day and 49 g/day inform the risks associated with consumption rates less than 142 g/day. Consumption rates for children aged 6 years and younger were calculated by assuming that their rate of fish consumption is approximately 42 percent of an adult, based on the ratio of child-to-adult consumption rates presented in the CRITFC Fish Consumption Survey (CRITFC 1994). The corresponding rates that were used for children are 7 g/day, 20 g/day, and 60 g/day.

The rates of 17.5 g/day and 142 g/day represent the 90th and 99th percentiles, respectively, of per capita consumption of uncooked freshwater/estuarine finfish and shellfish by individuals (consumers and non-consumers) 18 or older, as reported in the Continuing Survey of Food Intakes by Individuals (CSFII) and described in EPA's Estimated Per Capita Fish Consumption in the United States (EPA 2002b). While the values are presented in terms of "uncooked weight," it should not be construed to imply that the fish are consumed raw, as the consumption rates represent adjusted values to account for the amount of uncooked fish needed to prepare specific meals. No adjustments were made to contaminant concentrations in raw fish tissue because of the uncertainties associated with accounting for specific preparation and cooking practices.

The CSFII surveys recorded food consumption for two non-consecutive days. "Consumers only" were defined as individuals who ate fish at least once during the 2-day reporting period, individuals who reported not consuming any fish during the reporting period were designated as "non-consumers." For comparison, the 90th and 99th percentile consumption rates for consumers-only are 200 g/day and 506 g/day, respectively (EPA 2002b). Because of the short time period over which the survey is conducted, the results characterize the empirical distribution of average daily per capita consumption rather than describe true long-term average daily intakes. Although 17.5 g/day represents a 90th percentile value, it is considered an average consumption rate for sport fishers (EPA 2000d). Similarly, 142 g/day is considered to be representative of average consumption estimates for subsistence fishers when compared to upper percentile values for consumers only. However, the use of values representative of both non-consumers and consumers is appropriate as it accounts for the fact that some portion of the total diet of fish consumed may come from sources other than Portland Harbor. The consumption rate of 49 g/day is from a creel study

conducted in the Columbia Slough (Adolfson 1996), and represents the 95 percent upper confidence limit on the mean, where 50 percent of the mass of the total fish is consumed (Adolfson 1996). Use of this rate accounts for the likelihood that at least some individuals may consume more than just fillets, and that the consumption rates from the Columbia Slough survey are presented as mean values rather than upper percentiles.

The Columbia Slough survey is subject to several deficiencies that limit its usefulness in describing fish consumption patterns in Portland Harbor. Factors that may lead to a positive bias (over predicting consumption) include that the survey was conducted only over one month during the summer (fishing may be less frequent in colder winter months), and that creel surveys tend to encounter anglers who fish more frequently and/or have greater success. Additionally, the consumption rates are reported as the 95 percent UCL on the mean rather than an upper bound percentile as recommended to describe intake in a RME assessment. Factors that may lead to a negative bias include current fish consumption advisories for Columbia Slough, and that individuals who fish more frequently (and/or consume more fish that recommended in the current advisory) may have been reluctant to participate. Other factors that may affect the overall accuracy include the fact that the interviewers often had to estimate the size of fish caught, actual preparation of the fish for consumption may have been someone other than the person interviewed, thus estimates of how much of the fish is actually consumed was not accurate, and not all known fishing locations were surveyed. However, the results of the study do provide some insight into local, non-tribal fishing and consumption patterns.

Consumption of shellfish was evaluated considering only consumption by adults, and assuming that consumption of shellfish is primarily a component of a subsistence diet. Site-specific information regarding consumption of shellfish is not available, thus a range of consumption rates were evaluated. Consumption rates of 3.3 g/day and 18 g/day were selected as representative of CT and RME estimates. These values represent the mean and 95th percentile consumption rates of shellfish from freshwater and estuarine systems for individuals of age 18 and older in the United States (EPA 2002b). Exposure assumptions for recreational/subsistence fish consumption are presented in Table 3-24, and the uncertainties associated with these consumption rates are discussed in Section 6.

3.5.8.7 Tribal Fishers

Specific information regarding population mobility on Native American populations is less readily available than for the general U.S. population. The evaluation of exposures to Native Americans was based on the premise that they spend their entire lives in the area (EPA 2005c), and a typical lifetime was evaluated as 70 years. Fishing frequency was assumed to be 260 days/yr (5 days/week) for the RME estimate and 104 days/year (2 days/week) for the CT estimate.

Incidental ingestion of beach sediment was evaluated assuming 100 mg/day for the RME estimate and 50 mg/day for the CT estimate. Rates of 50 mg/day for the RME estimate and 25 mg/day for the CT estimate were used for incidental ingestion of in-water sediment, representing 50 percent of the rates used for incidental soil ingestion in a typical residential setting. An exposed surface area of 5,700 cm^2, representing the face, hands, forearms and lower legs was used to assess dermal exposure to beach sediments, exposures to in-water sediment was assumed to be limited to the hands and forearms, corresponding to a surface area of 1,980 cm^2. Sediment adherence to skin was evaluated using a weighted adherence factor based on exposure to the hands, forearms, and lower legs (EPA 2004). A factor of 25 percent was used to account for the time spent fishing in a single area within the Study Area. Exposure assumptions for beach and in-water sediment contact for tribal fishers are presented in Tables 3-21 and 3-22.

Fish consumption by tribal members was evaluated assuming a multi-species diet that includes both resident and anadromous fish (salmon, lamprey, and sturgeon). An overall rate of 175 g/day (approximately 23 eight oz meals per month), representing the 95th percentile of consumption rates for consumers and non-consumers in the CRITFC Survey was used for adult tribal fish consumers. A consumption rate of 73 g/day, representing the 95th percentile of consumption for children from the CRITFC Survey was used for child tribal fish consumers. The CRITFC survey reported that none of the respondents fished the Willamette River for resident fish, and approximately 4 percent fished for anadromous fish. Overall fish consumption information from the CRITFC survey was used to determine the consumption rate for each fish species, as shown in the following table:

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 Consumption Rate	175	100

(a) Rates are based on the weighted mean data in Table 18 of CRITFC 1994.

As shown, consumption rates of anadromous species account for approximately 50 percent of total intake. Consumption of salmon, lamprey and sturgeon were evaluated at rates of 67 g/day, 12.3 g/day, and 8.6 g/day, respectively. The remaining portion of the diet was evaluated assuming equal portions of the four resident fish (smallmouth bass, brown bullhead, common carp, and black crappie) for which tissue data were

available. Consumption rates for children were calculated using the same dietary percentages as the adult tribal fish consumers and a total intake of 73 g/day. Exposure assumptions for tribal fish consumption are presented in Table 3-24. Adult salmon, adult lamprey, and sturgeon have life histories such that significant contaminant loading can occur outside of the Study Area, making it problematic to associate tissue concentrations with site contamination. However, including consumption of anadromous fish in conjunction with resident fish provides useful information regarding risks to tribal members who may fish the Lower Willamette River.

3.5.8.8 Domestic/Household Water User

Use of surface water as a household water source was evaluated assuming exposure occurs in a residential setting. Exposure frequency is assumed as 350 days per year (7 days/week for 50 weeks) for both the RME and CT evaluations. As discussed in Section 3.5.9.5, overall exposure duration for residential exposure was assessed as 30 years for the RME estimate and 9 years for the CT estimate. Water ingestion by adults was evaluated at a rate of 2 L/day for the RME estimate, representing the average of the 90th percentiles of two national studies (EPA 1997a). A value of 1.4 L/day was used for the CT estimate, representing the population-weighted means of the same studies. These values are representative of water consumed directly from the tap or used in the preparation of food and beverages for adults. Ingestion rates representing 50th percentile values of 1.4 L/day for RME and 0.9 L/day for CT were used for children aged 6 years and younger.

Dermal exposures during showering or bathing were evaluated assuming a rate of one event per day, with an event duration of 35 minutes (0.58 hr) for the RME and 15 minutes (0.15 hr) for the CT, representing the 95th and 50th percentile values from EPA 1997a. A total skin surface area of 18,000 cm^2, representing estimates of the 50th percentile of mean surface area for adult men and women (EPA 1997a), was used for both the RME and CT estimates. A corresponding mean surface area of 6,600 cm^2 was used for children aged 6 years and younger.

Table 3-25 summarizes the exposure assumptions used to evaluate domestic use of surface water.

3.5.9 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.

3.5.9.1 Arsenic

Although arsenic was analyzed as total arsenic, the toxicity values represent inorganic arsenic. 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 percent to

26.6 percent with an average of 5.3 percent inorganic arsenic in 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 percent is expected to result in a health protective estimate of the potential health effects from arsenic in fish." Therefore, 10 percent of total arsenic in tissue was assumed to be inorganic arsenic when calculating the EPCs in fish and shellfish tissue. Uncertainties associated with the assumption that 10 percent of the total arsenic is in the inorganic form in fish and shellfish are discussed further in Section 6.

3.5.9.2 PCBs

PCBs were analyzed as Aroclors and congeners in tissue. Where PCBs were analyzed as Aroclors, the summed concentration of individual Aroclors was used in calculating the EPCs. Where PCBs were analyzed as congeners, EPCs were calculated for 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 as described in Section 2.2.8.

3.5.9.3 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

The toxicity assessment is composed of two steps: (1) hazard identification and (2) dose-response assessment. Hazard identification is the process of determining whether exposure to a chemical may result in a deleterious health effect in humans. It consists of characterizing the nature of the effect and the strength of the evidence that the chemical will cause the observed effect. Dose-response assessment characterizes the relationship between the dose and the incidence and/or severity of the adverse health effect in the exposed population. For risk assessment purposes, chemicals are generally separated into categories based on their toxicological endpoints. The primary basis of this categorization is whether a chemical exhibits potentially carcinogenic or noncarcinogenic health effects. Because chemicals that are suspected carcinogens may also give rise to noncarcinogenic effects, they must be evaluated separately for both effects.

4.1 TOXICITY VALUES FOR EVALUATING CARCINOGENIC EFFECTS

Cancer slope factors are used to estimate the risk of cancer associated with exposure to a chemical known or suspected to be carcinogenic. The slope factor is derived from either human epidemiological or animal studies, and represents an upper bound, generally approximating a 95 percent confidence limit, on the increased cancer risk from a lifetime exposure by ingestion. Slope factors are generally expressed in units of proportion (of a population) affected per mg of substance/kg body weight-day ([(mg/kg-day)$^{-1}$].

In addition to the numerical estimates of carcinogenic potential, a cancer weight-of-evidence (WOE) descriptor is used to describe a substance's potential to cause cancer in humans and the conditions under which the carcinogenic effects may be expressed. This judgment is independent of consideration of the agent's carcinogenic potency. Under EPA's 1986 guidelines for carcinogen risk assessment, the WOE was described by categories "A through E"—Group A for known human carcinogens through Group E for agents with evidence of noncarcinogenicity. Under EPA's 2005 guidelines for carcinogen risk assessment, a narrative approach rather than the alphanumeric categories is used to characterize carcinogenicity. Five standard weight-of-evidence descriptors are used: *Carcinogenic to Humans, Likely to Be Carcinogenic to Humans, Suggestive Evidence of Carcinogenic Potential, Inadequate Information to Assess Carcinogenic Potential, and Not Likely to Be Carcinogenic to Humans).* Slope factors for assessing dermal exposure were derived as described in Section 4.7, and oral and dermal slope factors are presented in Table 4-1.

4.2 TOXICITY VALUES FOR EVALUATING NONCARCINOGENIC EFFECTS

The reference dose (RfD) provides quantitative information for use in risk assessments for health effects known or assumed to be produced through a nonlinear

(possibly threshold) mode of action. The RfD, expressed in units of mg of substance/kg body weight-day (mg/kg-day) is defined as an estimate (with uncertainty spanning perhaps an order of magnitude) of a daily exposure to the human population (including sensitive subgroups) that is likely to be without an appreciable risk of deleterious effects during a lifetime. The use of RfDs is based on the concept that there is range of exposures that exist up to a finite value, or threshold, that can be tolerated without producing a toxic effect. Reference doses are presented in Table 4-2.

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 cancer slope factor (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.

Trichloroethene cancer potency was evaluated using the geometric mid-point of the slope factor range from EPA 2001b as recommended by EPA Region 10 (EPA 2007b). Recommendations were not provided for evaluating oral exposures for noncancer endpoints for trichloroethene.

4.4 CHEMICALS WITH SURROGATE TOXICITY VALUES

If a toxicity value was not available from the above hierarchy for a specific chemical, 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 were evaluated using surrogate toxicity criteria:

- Butyltin. The toxicity of organotin compounds is somewhat determined by the nature and number of groups bound to tin. In general, toxicity decreases as the number of linear carbons increases and as the number of substitutions decrease. As a health protective approach, RfD for dibutyltin compounds was selected as a surrogate for butyltin.

- Acenaphthylene is classified as category D (not classifiable as to human carcinogenicity). The RfD for acenaphthene, which is the most structurally similar PAH, was selected as a surrogate for acenaphthylene.

- Benzo(e)pyrene. As a health protective approach, the RfD for pyrene was used as a surrogate for benzo(e)pyrene.

- Benzo(g,h,i)perylene is classified as category D (not classifiable as to human carcinogenicity). As with benzo(e)pyrene, the RfD for pyrene was used as a surrogate for benzo(g,h,i)perylene.

- Dibenzothiophene. Fluorene the most structurally similar PAH with available toxicity values. Hence, the RfD for fluorene was used as a surrogate for dibenzothiophene.

- Dibenzofuran. The RfD for flourene, which represents the most structurally similar compound for which an RfD was available was selected as a surrogate for dibenzofuran.

- Di-n-octyl phthalate. The RfD for dibutyl phthalate was selected as a surrogate for di-n-octyl phthalate.

- Perylene. The RfD for pyrene was selected as a surrogate for perylene.

- Phenanthrene. The RfD for pyrene was selected as a surrogate for phenanthrene.

- Retene. The RfD for pyrene 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 RfD for endrin was used 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 RfD for endrin was used as a surrogate for endrin ketone.
- 4-Nitrophenol. The RfD for 4-methylphenol was used as a surrogate for 4-nitrophenol.

4.5 CHEMICALS WITHOUT TOXICITY VALUES

No SF and RfD or other suitable surrogate values were obtained for titanium and delta-hexachlorocyclohexane (delta-HCH). 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). Accordingly, 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 CLASSES

Certain toxicity values are based on exposure to more than one isomer and not to individual chemicals. As a result, the risks were evaluated for the combined exposure rather than on an individual chemical basis. COPCs that were evaluated for toxicity as classes 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. Although individual slope factors are available for DDD, DDE, and DDT based on studies conducted using the 4,4' isomers, the potency of the 2,4' isomers was assumed to be equal to that of the 4,4' isomers, and cancer risks assessed as the sum of the 2,4' and 4,4' isomers. Additionally, the RfD for DDT was used as a surrogate to evaluate the noncancer effects of DDD and DDE.

- Endosulfan: The RfD for endosulfan was derived from studies using technical endosulfan, which includes alpha-endosulfan, beta-endosulfan, and endosulfan sulfate. The individual endosulfan results were summed to give a total endosulfan concentration, and the RfD for technical endosulfan was used to evaluate total endosulfan.

- PCBs: The cancer slope factor for PCBs is based on administered doses of Aroclors (Aroclor 1016, 1242, 1254, or 1260), and was used to assess the cancer risks for total PCBs measured either as congeners or Aroclors. As discussed in Section 2.2.8, total PCB concentrations were calculated as either the sum of Aroclors or individual congeners. Where PCBs were reported as individual congeners, an adjusted PCB concentration was calculated by subtracting the sum of total dioxin-like PCB congener concentrations from the sum of all congeners. Dioxin-like PCB congeners were evaluated separately using the slope factor for 2,3,7,8-tetrachlorodibenzo-p-dioxin (2,3,7,8-TCDD) as described below. This approach may double-count a portion of the toxicity of the dioxin-like PCBs, as discussed in Section 6.3.6. The RfD for Aroclor 1254 was used to evaluate the noncancer endpoint for total PCBs, measured either as total unadjusted congeners or as 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 for dioxin-like PCB congeners (see Table 4-3). Concentrations of individual congeners are multiplied by their respective TEF to provide a 2,3,7,8-TCDD-equivalant concentration (TEQ), the resulting TEQs are then summed into a total 2,3,7,8-TCDD TEQ. Cancer risk were assessed using the slope factor for 2,3,7,8-TCDD was used to evaluate the cancer endpoint of the TEQ for dioxin and furan congeners, as well as for dioxin-like PCB congeners. The ATSDR MRL for 2,3,7,8-TCDD was used in conjunction with the TEQ approach for dioxin and furan congeners, and for dioxin-like PCB congeners.

- Carcinogenic PAHs: Individual carcinogenic PAHs were evaluated for toxicity based on their potency equivalency factor (PEF), which estimates cancer potency 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 ASSESSMENT

Toxicity is a function of contaminant concentration at critical sites-of-action. However, most oral reference doses and slope factors are expressed as an administered dose, whereas exposure estimates for dermal exposures are based on the absorbed dose. Anatomical differences between the gastrointestinal tract and the skin can affect rate as well as the extent of absorption. Thus, the route of exposure may significantly affect the critical dose at the site-of-action. A further complication is that an orally administered dose experiences "hepatic first-pass" metabolism, which may significantly alter the toxicity of the administered chemical. Additionally, some chemicals can cause cancer or other effects through direct action at the point of application. For such locally active compounds, it may be inappropriate to evaluate risks based on oral response data.

As recommended by EPA guidance (EPA 2004), an adjustment to the oral toxicity factor to account for the estimated absorbed dose was applied when the toxicity value derived from the critical study was based on an oral dose and GI absorption of the chemical is less than 50 percent from a medium similar to the one used in the critical study.

Dermal RfDs for assessing dermal exposure were calculated using the following equation:

$$RfD_{dermal} = RfD_o \times ABS_{GI}$$

RfD_{dermal} = dermal reference dose (mg/kg-day)
RfD_o = oral reference dose (mg/kg-day)
ABS_{GI} = fraction of contaminant absorbed in gastrointestinal tract

Cancer slope factors for assessing dermal exposure were calculated as follows:

$$SF_{dermal} = \frac{SF_o}{ABS_{GI}}$$

SF_{dermal} = dermal cancer slope factor $(mg/kg\text{-}day)^{-1}$
SF_o = oral cancer slope factor $(mg/kg\text{-}day)^{-1}$
ABS_{GI} = fraction of contaminant absorbed in gastrointestinal tract

5.0 RISK CHARACTERIZATION

Risk characterization integrates the information from the exposure assessment and toxicity assessment, using a combination of qualitative and quantitative information to provide numerical estimates of potential adverse health effects. Risk characterization is performed separately for carcinogenic and noncarcinogenic effects. Carcinogenic risk is expressed as the probability that an individual will develop cancer over a lifetime as a result of exposure to a potential carcinogen. Noncarcinogenic hazards are evaluated by comparing an estimated exposure level or dose with a reference dose that is without appreciable risk of adverse health effects.

5.1 RISK CHARACTERIZATION METHODOLOGY

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

5.1.1 Noncancer Hazard Estimates

The potential for adverse noncancer health effects is generally addressed by comparing the CDI to the corresponding RfD to yield a hazard quotient (HQ; EPA 1989):

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

The calculation of a HQ assumes that exposures less than the RfD are unlikely to result in adverse health effects, even for sensitive populations. By definition, when the HQ is less than 1, the estimated exposure is less than the RfD and adverse health effects are unlikely. Unlike cancer risks, the HQ does not represent a statistical probability, and the likelihood of adverse effects does not increase in a linear fashion relative to a HQ of 1. Rather, exposures greater than the RfD may result in adverse health effects, but all RfDs do not have equal precision and are not based on the same severity of effects. HQs for individual chemicals were summed to yield a cumulative hazard index (HI). Although a HI provides an overall indication of the potential for noncancer hazards, dose additivity is most appropriately applied to chemicals that induce the same effect via the same mechanism of action. When the HI is greater than 1 due the sum of several HQs of similar value, it is appropriate to segregate the chemical-specific HQs by effect and mechanism of action. In this BHHRA, when the calculated HI was greater than 1, HQs based on the same target organ system were calculated. The target organs or systems on which the RfDs are based are presented in Table 5-1.

5.1.2 Cancer Risk Estimates

The cancer slope factor converts the estimated daily intakes averaged over a lifetime directly to an incremental cancer risk. Cancer risks are calculated by multiplying the estimated LADI of a carcinogen by the SF (EPA 1989):

$$Risk = LADI \times SF$$

The dose-response relationship is generally assumed to be linear through the low-dose portion of the dose-response curve. That is, the risk of developing cancer is assumed to be directly associated with the amount of exposure. However, this linear relationship is valid only when the estimated risk is less than 0.01 (1×10^{-2}). Where contaminant concentrations result in an estimated risk greater than 1×10^{-2}, the following equation should be used (EPA, 1989):

$$Risk = 1\text{-}e^{-LADI \times SF}$$

Because the slope factor typically represents an upper confidence limit, carcinogenic risk estimates generally represent an upper-bound estimate, and EPA is confident that the true risk will not be greater than risk estimates obtained using this model, and they may be less than that predicted. Cancer risk estimates for individual chemicals and different exposure pathways were summed where exposure was assumed to be concurrent to obtain the cumulative excess lifetime cancer risk for each receptor and/or exposure scenario.

5.1.3 Infant Consumption of Human Milk

As discussed in Section 3.3.7, infant exposure to persistent, lipophilic contaminants via breastfeeding was quantitatively evaluated in the BHHRA. Using the methodology presented in Section 3.5.5, DEQ determined that the magnitude of the difference in the hazard estimates between the infant and the mother remain constant regardless of the maternal exposure pathway or dose, and can be expressed as infant risk adjustment factors (IRAFs, DEQ 2010):

$$HQ_{infant} = HQ_{mother} \times IRAF_{nc}$$

where:

HQ_{infant} = hazard quotient for breast-fed infant
HI_{mother} = hazard quotient for the mother
$IRAF_{nc}$ = infant risk adjustment factor for noncancer effects

The chemical-specific IRAFs are presented in the following table:

Chemical	$IRAF_{nc}$	Half Life (days)
PCBs	25	2550
Dioxins/Furans	2	2550
DDx	2	120
PBDEs	38	2550

5.1.4 Risk Characterization for Lead

Health effects associated with exposure to inorganic lead and compounds are well documented and include neurotoxicity, developmental delays, hypertension, impaired hearing acuity, impaired hemoglobin synthesis, and male reproductive impairment. Importantly, many of lead's health effects may occur without other overt signs of toxicity. Lead has particularly significant effects in children, and it appears that some of these effects, particularly changes in the levels of certain blood enzymes and in aspects of children's neurobehavioral development, may occur at blood lead levels so low as to be essentially without a threshold. Because of the difficulty in accounting for pre-existing body burdens of lead and the apparent lack of threshold, EPA determined that it was inappropriate to develop a RfD. The Centers for Disease Control (CDC) has identified a blood lead concentration of 10 micrograms per deciliter (µg/dL) as the level of concern above which significant health effects may occur (CDC 1991), and 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). An acceptable risk is generally defined as a less than 5 percent probability of exceeding a blood lead concentration of 10 µg/dL (EPA 1998).

Using the ALM (EPA 2003c), acceptable lead concentrations in fish tissue that are unlikely to result in fetal blood lead concentrations greater than 10 µg/dL were calculated using the following equation:

$$PbF = \frac{\left(\left[PbB_f / R \times GSD^{1.645}\right] - PbB_o\right) \times AT}{BKSF \times \left(IR_F \times AF_F \times EF_F\right)}$$

Where:

PbB_a	=	Central tendency of adult blood lead level
PbB_o	=	Adult baseline blood lead level
PbB_f	=	Fetal blood lead level
R	=	Fetal/maternal blood lead ratio
GSD	=	Geometric standard deviation PbB
BKSF	=	Biokinetic slope factor
PbF	=	Lead fish tissue concentration
IR_F	=	Consumption rate of fish
AF_F	=	Gastrointestinal absorption of lead from fish
EF_F	=	Exposure frequency for fish consumption
AT	=	Averaging time

The values used in this analysis are presented in Attachment F5. Because the lead models calculate a central tendency or geometric mean blood lead concentration,

median values are typically used as inputs. The mean estimate of national per capita fish consumption of 7.5 g/day (EPA 2000b) was used as the consumption rate for recreational fishers, the median consumption rate of 39.2 g/day from the CRITFC study was used for tribal fishers. Using the equation presented above, the target lead concentrations in fish are 5.2 mg/kg for recreational fishers and 1 mg/kg for tribal fishers.

EPA's Integrated Exposure Uptake Biokinetic (IEUBK) model was used to calculate tissue lead concentrations unlikely to result in blood lead concentrations greater than 10 µg/dL in children. Because site-specific values for concentration of lead in soil, house dust, air and drinking water were not readily available, default values were used for those inputs. The ratio of child-to-adult consumption of 0.42 was applied to the median adult consumption rate of 7.5 g/day to obtain a childhood rate of 3.2 g/day for children of recreational fishers. The corresponding lead concentration in fish is 2.6 mg/kg. Assuming a consumption rate of 16.2 g/day for tribal children, representing the 65th percentile consumption rate from the CRITFC survey, the calculated lead concentration in fish is 0.5 mg/kg. Uncertainties associated with the evaluation of lead are discussed further in Section 6.

5.1.5 Cumulative Risk Estimates for Contaminants Analyzed by More Than One Method

In some instances specific contaminants were analyzed by more than one method, and thus more than one EPC calculated for that contaminant. Cumulative risks are presented using the EPC from only one method to avoid double-counting the risks from a given contaminant. When assessing risks associated with sediment exposures, Aroclor data were used because the data set was larger than for congeners. However, because the congener analysis provided lower detection limits, it was preferentially used when available for assessing risks associated with consumption of fish and shellfish. Where metals were analyzed as both total and dissolved fractions in surface water and groundwater seep samples, the EPCs based on total metals were used in the cumulative risk estimates because unfiltered data is generally more representative of typical human exposure.

5.2 RISK CHARACTERIZATION RESULTS

This section presents a summary of the risk characterization results the scenarios described in Section 3. EPA policy (EPA 1991a) states that CERCLA actions are generally warranted when where the baseline risk assessment indicates that a cumulative site risk to an individual using RME assumptions for either current or future land use is greater than the 1 x 10^{-4} lifetime excess cancer risk end of the cancer risk range of 1 x 10^{-4} to 1 x 10^{-6}, or the HI is greater than 1. Accordingly, risk and hazard estimates are generally presented in terms of whether they are greater than the upper end of the cancer risk range of 1 x 10^{-4} or the HI is greater than 1. 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.

A summary of risks by exposure medium are presented in Tables 5-18, 5-43, 5-57, 5-60, 5-65, 5-77, and 5-86.

5.2.1 Dockside Workers

Risks to dockside workers were estimated separately for each of the eight beaches designated as a potential dockside worker use areas, shown in Map 2-1.

The estimated CT and RME cancer risks are less than 1 x 10^{-4} at all beach areas, and the HI is less than 1 for adults and infants. These results are presented in Tables 5-2, 5-3, 5-16, and 5-17.

5.2.2 In-Water Workers

As discussed in Section 3.2.1.2, in-water workers are described as typically working around in-water structures such as docks, and primarily exposed to in-water sediments. In-water sediment exposure by in-water workers was evaluated in half-mile increments along each side of the river. The estimated CT and RME cancer risks are less than 1 x 10^{-4} at all RM segments, and the RME HIs for adults are less than 1 at any location. The HI for infants at RM 7W is 2 due to dioxins and furans. These results are presented in Tables 5-19, 5-20, 5-32 and 5-33.

5.2.3 Transients

Risks to transients were estimated separately for each beach designated as a potential transient use area, as well as the use of surface water as a source of drinking water and for bathing. Beaches where sediment exposure was evaluated are shown on Map 2-1. Year-round exposure to surface water for four individual transect stations, Willamette Cove, Multnomah Channel, and for the four transects grouped together to represent Study Area-wide exposure are shown on Map 2-3. The CT and RME risk estimates for beach sediment are less than 1 x 10^{-4} for all locations, and the HI is less than 1. The results of the RME and CT evaluations for exposure to beach sediments are presented in Tables 5-4 and 5-5, respectively.

Estimated CT and RME cancer risks associated with surface water exposures are less than 1 x 10^{-4} at all individual and transect locations, and the HI is less than 1. The results of the RME and CT evaluations are presented in Tables 5-44 and 5-45, respectively.

As noted in Section 3.3.4, exposure to surface water by transients was also evaluated at the groundwater seep at Outfall 22B. All risk and hazard estimates are less than 1 x 10^{-4} and 1, respectively, and the results are presented in Tables 5-58 and 5-59.

5.2.4 Divers

Commercial divers were evaluated for exposure to surface water and in-water sediment, and assuming the diver was wearing either a wet or a dry suit. As described in Section 3.4.2, in-water sediment exposure by divers is evaluated in half-mile exposure areas for each side of the river, and on a Study Area-wide basis. Risks associated with exposure to surface water were evaluated for four individual transect stations, and at single-point sampling stations grouped together in one-half mile increments per side of river.

5.2.4.1 Diver in Wet Suit

The estimated CT and RME cancer risk associated with exposure to in-water sediments is less than 1×10^{-4} at all half-mile river segments and for Study Area-wide exposure, and the HI is also less than 1 for adults. The HI for infants is 2 at RM 8.5W for the RME evaluation due to PCBs. The RME and CT estimates for adults are presented in Tables 5-29 and 5-30, respectively. RME and CT risk and hazard estimates for infant exposures are presented in Tables 5-40 and 5-41, respectively.

The estimated CT and RME cancer risk associated with exposure to surface water is less than 1×10^{-4} for all half-mile river segments, and the HI is less than 1. These results are presented in Tables 5-50 and 5-51, respectively, for the RME and CT evaluations. Indirect exposure to contaminants in surface water by infants via breastfeeding was not evaluated.

5.2.4.2 Diver in Dry Suit

The estimated RME cancer risk associated with exposure to in-water sediments is less than 1×10^{-4} at all half-mile river segments and for Study Area-wide exposure, and the HI is also less than 1 for adults and indirect exposures to infants via breastfeeding. The results of the adult RME risk and hazard estimates are presented in Table 5-31, a CT evaluation was not done for a commercial diver in a dry suit. Noncancer hazard for infants is presented in Table 5-42.

The estimated RME cancer risk associated with exposure to surface water is less than 1×10^{-4} for all half-mile river segments, and the HI is less than 1. These results are presented in Tables 5-52. Indirect exposure to contaminants in surface water by infants via breastfeeding was not evaluated.

5.2.5 Recreational Beach Users

Risks associated with exposure to beach sediment were evaluated separately for each beach designated as a potential recreational use area, shown on Map 2-1. Exposure to surface water was evaluated using data collected from three transect locations and three single-point locations (Cathedral Park, Willamette Cove, and Swan Island Lagoon) shown on Map 2-3.

The estimated CT and RME cancer risks associated with exposure to beach sediments are less than 1 x 10^{-4} at all recreational beach areas, and the HI is also less than 1. These results are presented in Tables 5-6 through 5-9. Indirect exposure to contaminants in beach sediment to infants via breastfeeding was not evaluated.

The estimated CT and RME cancer risks associated with exposure to surface water are less than 1 x 10^{-4} at all recreational beach areas, and the HI is also less than 1. These results are presented in Tables 5-46 through 5-49. Indirect exposure to contaminants in surface water to infants via breastfeeding was not evaluated.

5.2.6 Recreational/Subsistence Fishers

Recreational and subsistence fishers were evaluated assuming direct exposure to contaminants in sediment and via consumption of fish and shellfish. As discussed in Section 3.2.1.6, exposures associated with beach sediment were assessed at individual beaches designated as potential transient or recreational use areas, in-water sediment exposures were evaluated on a one-half river mile basis per side of the river and as an averaged, Study Area-wide evaluation. Sediment exposures were further assessed as CT and RME evaluations and assuming either a low- or a high-frequency rate of fishing.

5.2.6.1 Sediment - Direct Contact

The estimated CT and RME cancer risks associated with low-frequency fishing exposures to either beach or in-water sediments are less than 1 x 10^{-4} at all areas evaluated. Noncancer hazards associated with adult exposures to beach or in-water sediment are less than 1 at all locations evaluated, the RME noncancer hazard associated with indirect exposures to infants via breastfeeding is greater than 1 at two locations for in-water sediment: RM 7W (2) due to dioxin/furans and RM 8.5W (2) primarily due to PCBs, with a HQ of 1. These results are presented in Tables 5-14 and 5-15 for beach sediment exposures, and Tables 5-27, 5-28, 5-38, and 5-39 for in-water sediment exposures. Indirect exposure to contaminants in beach sediment to infants via breastfeeding was not evaluated.

The estimated CT and RME cancer risks associated with high-frequency fishing exposures to either beach or in-water sediments are less than 1 x 10^{-4} at all areas evaluated. For beach sediment, noncancer hazards associated with adult exposure are less than 1 at all locations evaluated. RME noncancer hazards associated with adult exposures to in-water sediment are greater than 1 at RM 7W (2) primarily due to dioxin/furans, with a HQ of 1. The RME noncancer hazard associated with indirect exposures to infants via breastfeeding is also greater than 1 at RM 7W (3) due to dioxin/furans and RM 8.5W (2) due to PCBs. These results are presented in Tables 5-12 and 5-13 for beach sediment exposures, and Tables 5-24, 5-25, 5-26, 5-36, and 5-37 for in-water sediment exposures. Indirect exposure to contaminants in beach sediment to infants via breastfeeding was not evaluated.

5.2.6.2 Consumption Resident Fish

Consumption of resident fish species was evaluated on both a per river mile basis to account for localized fishing practices and heterogeneous contaminant distribution in sediments, as well as averaging consumption over the entire Study Area. Consumption of resident fish species on a river mile basis was evaluated only for recreational fishers, whereas consumption averaged over the entire Study Area was evaluated for both recreational and subsistence fishers. With the exception of RM 5, RME risk estimates for fillet-only consumption on a river mile basis are all greater than 1×10^{-4}, the estimated risk at RM 5 is 9×10^{-5}. CT estimates are greater than 1×10^{-4} at RM 7, Swan Island Lagoon, and RM 11. River miles with the highest estimated RME risks are: RM 2 (2×10^{-4}), RM 4 (3×10^{-4}), RM 7 (6×10^{-4}), Swan Island Lagoon (6×10^{-4}), RM 9 (2×10^{-4}), and RM 11 (1×10^{-3}). RME risk estimates for dioxins/furans are greater than 1×10^{-4} at RM 7. RME risk estimates for PCBs are greater than 1×10^{-4} at RM 2, RM 4, RM 7 and Swan Island Lagoon and are 1×10^{-3} at RM 11. These results are presented in Tables 5-73. Study Area-wide RME risks for recreational and subsistence fishers are 4×10^{-3} and 1×10^{-2}, respectively; the Study Area-wide CT estimate for recreational fishers is 1×10^{-3}. These results are presented in Tables 5-74.

RME and CT hazard estimates for fillet-only consumption are greater than 1 at all river miles. Values for river miles with the highest estimated RME hazard are as follows: RM 4 (20), RM 7 (20), Swan Island Lagoon (50), and RM 11 (100). Study Area-wide RME hazards for recreational and subsistence fishers are 300 and 1000, respectively, the CT estimate for recreational fishers is 100. PCBs and dioxin/furans result in the highest hazard estimates at RM 7, PCBs result in the highest hazard estimates at RM 11. These results are presented in Tables 5-66 through 5-72.

RME and CT noncancer hazard associated with indirect exposure to infants via breastfeeding was also assessed. When evaluated on a river mile scale, the RME hazard estimates associated with fillet-only consumption range from 30 to 1,000. CT estimates range from 10 to 500 when assessed on a river mile scale. Study Area-wide, the hazard estimates for recreational fishers are 2,000 and 4,000 for the CT and RME estimates, respectively, the RME estimate for subsistence fishers is 10,000. River miles with the greatest RME hazard estimates are: RM 2 (200), RM 4 (200), RM 7 (200), Swan Island Lagoon (600), and RM 11 (1,000). The majority of the hazard estimates is attributable to PCBs. These results are presented in Tables 5-75 and 5-76.

As detailed in Section 3.4.5, EPCs on a river mile scale use data from smallmouth bass only to represent contaminant concentrations in all resident fish species, while consumption was assumed to consist primarily of just the fillet, rather than other parts of the fish. However, an evaluation of the data collected from Portland Harbor indicates that PCB concentrations in whole body smallmouth bass are typically an order of magnitude greater than those measured in just the fillet. By contrast, in common carp and brown bullhead, the observed ratio of whole body-to-fillet PCB concentrations is less than noted in smallmouth bass, meaning that given the same

overall PCB concentration in whole body fish, the PCB concentration in smallmouth bass fillet tissue will be less than for carp and bullhead.

Differences among these species is reflected in the EPCs; specifically the use of fillet smallmouth bass data on a river mile scale resulted in a greater relative reduction of PCB concentration than would be seen if fillet data from common carp and brown bullhead were included. As such, a diet that consists of some portion of carp and bullhead could result in relatively greater intake of PCBs, and the associated risk and hazard would be correspondingly greater as well. In addition, as discussed in Section 3.5.10.6, the Columbia Slough survey results indicate that at least some of the fishers in the Portland Harbor area consume more than just the fillet. Consumption of other portions of the fish in addition to the fillet can result in greater relative exposure to PCBs and other persistent bioaccumulative chemicals and thus, greater relative risks.

Using smallmouth bass data as an example, the increased risk associated with consumption of the entire fish could be as much as an order of magnitude greater than associated with consumption of fillet only. However, since the relative increased exposure is directly related to the type of fish consumed, as well as any portions of the fish consumed in addition to just the fillet, it is not possible to assess the increased risks associated with consumption of more than just fillet with any degree of accuracy.

5.2.6.3 Consumption of Clams

The estimated RME cancer risks associated consumption of undepurated clams by subsistence fishers are greater than 1×10^{-4} at 10 of the 22 river mile sections evaluated. Values for river miles with the highest estimated risks are as follows: RM 5W (6×10^{-4}), RM 6E (7×10^{-4}), and RM 6W (7×10^{-4}). Other areas where the estimated risk is equal to or greater than 1×10^{-4} are RM 2E, RM 3E, RM 4E, RM 4W, RM 7W, RM 8W, Swan Island Lagoon, RM 9W, and RM 11E. The estimated risk Study Area-wide is 4×10^{-4}. Carcinogenic PAHs and PCBs pose the highest risks on a Study Area-wide basis. Risk estimates for cPAHs are greater than 1×10^{-4} at RM 5W and RM 6W. At RM 7W, dioxins/furans result in the highest risk estimates. PCBs result in the highest risk estimates in Swan Island Lagoon and at RM 11. No estimated CT cancer risks associated with consumption of undepurated clams are greater than 1×10^{-4}. Risks were also evaluated based on consumption of depurated clams at RM 1E, RM 2W, RM 10W, RM 11E, and RM 12E. None of the estimated CT or RME cancer risks are greater than 1×10^{-4}. These results are presented in Table 5-78.

The estimated RME noncancer hazards associated consumption of undepurated clams by subsistence fishers are greater than 1 at 20 of the 22 river mile sections evaluated. Values for river miles with the highest noncancer hazard are as follows: RM 3E (8), RM 6E (40), RM 9W (8), and RM 11E (10). The estimated noncancer hazard Study Area-wide is 9. PCBs result in the highest hazard estimates at RM 2E, RM 3E, RM 6E, RM 8W, Swan Island Lagoon, RM 9W, and RM 11E. The estimated CT

hazards associated with consumption of undepurated clams is greater than 1 at RM 6E, where the HI is 7, and at RM 11E, where the HI is 2; PCBs result in the highest hazard estimate at both river miles. The estimated RME hazard associated with consumption of depurated clams is 2 at four of the five river miles and is 7 at RM 11E. PCBs result in the highest estimated hazard. These results are presented in Tables 5-78 through 5-80.

RME noncancer hazard associated with indirect exposure to infants via breastfeeding was also assessed, and the estimated hazard is greater than 1 at each river mile evaluated. Values for river miles with the highest estimated hazard due to parental consumption of clams are as follows (for infant children of subsistence fishers): RM 2E (100), RM 3E (200), RM 6E (800), RM 8W (100), Swan Island Lagoon (100), RM 9W (100), and RM 11E (300). These results are presented in Table 5-84.

5.2.6.4 Consumption of Crayfish

The estimated RME cancer risks associated consumption of crayfish by subsistence fishers are greater than 1×10^{-4} at two of the 32 individual stations evaluated: 07R006 (3×10^{-4}) located at RM 7W, and CR11E (3×10^{-4}) located at RM 11E. When evaluated Study Area-wide, the estimated risk is 3×10^{-4}. Risk estimates for dioxins/furans are greater than 1×10^{-4} at 07R006 and risk estimates for PCBs are greater than 1×10^{-4} at CR11E. No estimated CT cancer risks associated with consumption of crayfish are greater than 1×10^{-4}. These results are presented in Table 5-81.

The estimated RME noncancer hazards associated consumption of crayfish by subsistence fishers are greater than 1 at seven of the 32 individual stations. Stations with the highest estimated hazard are 03R005 (4) located at the end of the International Slip, 07R006 (6), and CR11E (10). The estimated noncancer hazard Study Area-wide is 10. PCBs result in the highest noncancer hazard at 03R005 and CR11E, dioxins/furans result in the highest noncancer hazard at 07R006. These results are presented in Tables 5-81 through 5-83.

RME noncancer hazard associated with indirect exposure to infants via breastfeeding is greater than 1 at 23 of the 32 stations evaluated. Values at locations with the highest estimated hazard due to parental consumption of clams are as follows (for infant children of subsistence fishers): 03R003 (20) at RM 3E, 03R005 (60) at RM 3E, 07R006 (20) at RM 7W, 08R001 (20) at RM 8W, 09R001 (20) in Swan Island Lagoon, 09R002 (30) at RM 9W, and CR11E (400) at RM 11E. The hazard is 200 when evaluated Study Area-wide. These results are presented in Table 5-85.

5.2.7 Tribal Fishers

Tribal fishers were evaluated assuming direct exposure to contaminants in sediment and via consumption of fish. Exposures associated with beach sediment were assessed at individual beaches, in-water sediment exposures were evaluated on a one-half river

mile basis per side of the river and as an averaged, Study Area-wide evaluation. Fish consumption was evaluated assuming a multi-species diet consisting of anadromous and resident fish species, and fishing was evaluated on a Study Area-wide basis.

5.2.7.1 Sediment – Direct Contact

The estimated CT and RME cancer risks associated with direct contact to beach sediment is less than 1 x 10^{-4} at all beaches evaluated. The estimated RME cancer risk associated with exposure to in-water sediment is greater than 1 x 10^{-4} at two locations: RM 6W (2 x 10^{-4}) and RM 7W (3 x 10^{-4}). Risk estimates for cPAHs are greater than 1 x 10^{-4} at RM 6W, risk estimates for dioxins/furans are greater than 1 x 10^{-4} at RM 7W. These results are presented in Tables 5-10, 5-11, 5-21, and 5-22.

With the exception of in-water sediment exposure at RM 7W, the estimated non-cancer hazard is less than one at all beach and in-water locations evaluated. The estimated RME hazard is 3 at RM 7W, primarily due to dioxins/furans, with a HQ of 2. These results are presented in Tables 5-10, 5-11, and 5-21 through 5-23.

Noncancer CT and RME hazard estimates associated with indirect exposure to infants via breastfeeding was evaluated assuming maternal exposure to in-water sediment. The estimated RME hazard is greater than 1 at 3 locations, RM 7W (5), RM 8.5 (4), and RM 11E (2). These results are presented in Tables 5-34 and 5-35.

5.2.7.2 Fish Consumption

The estimated RME cancer risks for the combined child and adult exposure is 2 x 10^{-2} assuming whole body consumption, and 1 x 10^{-2} assuming consumption of fillets only. Risk estimates for PCBs, dioxins/furans, and arsenic are greater than 1 x 10^{-4}. These results are presented in Table 5-63.

The RME noncancer hazard associated with childhood consumption of whole body fish is 800, and is 600 assuming consumption of fillets only. PCBs, and to a lesser extent dioxins/furans, result in the highest noncancer hazard estimates. These results are presented in Tables 5-61 and 5-62.

The RME noncancer hazard associated with indirect exposure of tribal infants via breastfeeding assuming maternal consumption of whole body fish is 9,000, and is 8,000 assuming maternal fillet-only consumption. PCBs result in the highest hazard estimates. These results are presented Table 5-64.

5.2.8 Domestic Water Use

Use of surface water as a source of household water for drinking and other domestic uses was evaluated using data from five transect and 15 single point sampling locations, as well as averaged over a Study Area-wide basis. The estimated cancer risk for combined child and adult exposures is greater than 1 x 10^{-4} at W031 (9 x 10^{-4}), located at RM 6W. PAHs are the primary contributor to the estimated

cancer risk. However, dermal exposure is the primary pathway contributing to the risk estimate, and as described in EPA 2004, the physical-chemical properties of several PAHs, including benzo(a)anthracene, benzo(a)pyrene, benzo(b)fluoranthene, dibenzo(a,h)anthracene, and indeno(1,2,3-c,d)pyrene), place them outside of the Effective Prediction Domain used to estimate the absorbed dermal dose from water. Although PAHs are direct-acting carcinogens, the risk estimates associated with estimating dermal absorption from water have a greater degree of uncertainty than the other risk estimates presented in this BHHRA. These results are presented in Tables 5-55 and 5-56.

The estimated noncancer hazard based on childhood exposure is equal to or greater than 1 at several sampling locations: W005 (1) at RM 4, W023 (1) at RM 11, W027 (2) near the mouth of Multnomah Channel, and W035 (2) in Swan Island Lagoon. In all instances, MCPP is the primary contributor to the estimated hazard. These results are presented in Tables 5-53 and 5-54.

5.3 CUMULATIVE RISK ESTIMATES

Cumulative risk and hazard estimates were calculated for those populations where concurrent exposure to more than one media was assumed to be plausible. Recreational/subsistence and tribal fishers were further evaluated on the basis of whether they were assumed to fish predominately from the shore or from a boat. Populations for which concurrent exposure to more than one media was considered for are as follows:

- Transients: Beach sediment, surface water
- Divers: In-water sediment, surface water
- Recreational beach users: Beach sediment, surface water
- Recreational fishers (beach): Beach sediment, fish tissue (fillet)
- Recreational fishers (boat): In-water sediment, fish tissue (fillet)
- Subsistence fishers (beach): Beach sediment, fish tissue (fillet), shellfish tissue
- Subsistence fishers (boat): In-water sediment, fish tissue (fillet), shellfish tissue
- Tribal fishers (beach): Beach sediment, fish tissue (fillet and whole body)
- Tribal fishers (boat): In-water sediment, fish tissue (fillet and whole body)

Cumulative risk estimates are generally presented for each one-half river mile per side of the river, and the risk estimates for specific media appropriate to each one-half mile segment were used to calculate the total risk or hazard. For example, cumulative risks for recreational fishers who fish from a boat and consume smallmouth bass would include the risks associated with exposure to in-water sediment at the specific half-mile, shellfish collected within same half-mile and side-of-river specific segment, and smallmouth bass from the larger river mile assessment. The results of the cumulative risk estimates are presented in Tables 5-87 through 5-111. 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 5-112.

5.4 SUMMARY OF RISK CHARACTERIZATION

Cancer risk and noncancer hazard from site-related contamination was characterized based on current and potential future uses at Portland Harbor, and a large number of different exposures scenarios were evaluated. Exposure to bioaccumulative contaminants (PCBs, dioxins/furans, and organochlorine pesticides, primarily DDx compounds) via consumption of resident fish consistently poses the greatest potential for human exposure to in-water contamination. In general, the risks associated with consumption of resident fish are greater by an order of magnitude or more than risks associated with exposure to sediment or surface water. The greatest non-cancer hazard estimates are associated with bioaccumulation through the food chain and exposure to infants via breastfeeding. Because the smallest scale over which fish consumption was evaluated was per river mile, the resolution of cumulative risks on a smaller scale is not informative. The highest relative cumulative risk or hazard estimates are at RM 4, RM 7, Swan Island Lagoon, and RM 11. However, assuming exposure to sediment alone, the risk estimates are greater than 1×10^{-4} at RM 6W and 7W for the tribal fisher; the risk estimates for all other locations and scenarios are less than 1×10^{-4}. Assuming shellfish consumption alone, the highest relative cumulative risk or hazard estimates are at RM 3E, RM 5W, RM 6W, RM 6E, RM 7W and RM 11E.

6.0 UNCERTAINTY ANALYSIS

The presence of uncertainty is inherent in the risk assessment process, from the sampling and analysis of chemicals in environmental media to the assessment of exposure and toxicity, and risk characterization. EPA policy calls for numerical risk estimates to always be accompanied by descriptive information regarding the uncertainties of each step in the risk assessment to ensure an objective and balanced characterization of the true risks and hazards.

The term "uncertainty" is often used in risk assessment to describe what are, in reality, two conceptually different terms: uncertainty and variability. Uncertainty can be described as the lack of a precise knowledge resulting in a fundamental data gap. Variability describes the natural heterogeneity of a population. Uncertainty can sometimes be reduced or eliminated through further measurements or study. By contrast, variability is inherent in what is being observed. Although variability can be better understood, it cannot be reduced through further measurement or study, although it may be more precisely defined. However, the additional cost of further data collection may become disproportional to the reduction in uncertainty.

The risks and hazards presented are consistent with EPA's stated goal of RME representing the high end of the possible risk distribution, which is generally considered to be greater than the 90th percentile. However, these estimates are based on numerous and often conservative assumptions and, in the absence of definitive information, assumptions are used to ensure that actual sites risks are not underestimated. The cumulative effect of these assumptions can result in an analysis with an overall conservativeness greater than the individual components. Accordingly, it is important to note that the risks presented here are based on numerous conservative assumptions in order to be protective of human health and to ensure that the risks presented here are more likely to be overestimated rather than underestimated.

6.1 DATA EVALUATION

As discussed in Section 2, sediment, surface water, groundwater seep, and biota data were collected during the RI. Data of confirmed quality that meet the DQOs for risk assessment were used in this BHHRA to estimate exposures. Although uncertainty is inherent in environmental sampling, the use of the EPA's DQO planning process (EPA 2000e) minimized the uncertainty associated with the data collected during the RI. A discussion of key data evaluation uncertainties is presented in the following sections.

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 fish and shellfish species consumed by humans within the Study Area, as recommended by EPA

guidance (2000a), target resident species were selected to represent the diet of all types likely consumed by humans. Four target species were collected to represent a diet consisting of resident fish: smallmouth bass, black crappie, common carp, and brown bullhead. Crayfish and clam tissue samples were collected to represent a diet containing locally-harvested shellfish. Factors considered in selecting the target species included likely consumption by humans, home range, the potential for bioaccumulation of COPCs, the trophic level of species, and their abundance.

PCBs generally represent the greatest contributors to the estimated risks, and detected concentrations are highest in brown bullhead and common carp. Therefore, the use of target resident species as representative of all biota consumed is unlikely to underestimate potential risks. If non-resident species are consumed, the risks may be less, commensurate with the amount of non-resident species present in the diet.

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

Salmon, lamprey, and sturgeon have traditionally represented a substantial portion of the fish diet of tribal members. These species likely spend a substantial portion of their lives outside of the Study Area, and thus contaminant concentrations in these species may bear little relationship to sediment concentrations in the Study Area.

The Washington Department of Ecology analyzed returning fall Chinook salmon, as fillet tissue with skin, collected from three coastal rivers (the 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 indicates that the concentration of PCBs measured as Aroclors and congeners are noticeably greater in salmon collected from the Clackamas fish hatchery relative to concentrations detected in the Ecology study. The reported concentrations of total DDT and dioxins as TEQs are generally consistent between the Ecology study and results from Portland Harbor. These results are summarized 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. The degree to which contaminant concentrations in anadromous fish are due to exposures that occur within the Study Area is unknown. However, approximately 95 percent of the cumulative tribal fish consumption risk is due to contaminants detected in resident species, even though they only account for 50 percent of the estimated diet. As a result, while sources of bioaccumulative chemicals other than

Portland Harbor may contribute to tissue concentrations in anadromous fish species, the uncertainty associated with the source of chemicals to non-resident fish species should not affect the conclusions of this BHHRA for tribal fish consumption.

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

Different contaminants are preferentially accumulated in different parts of an organism. Organic compounds tend to accumulate to a greater degree in tissues with a higher fat content, while heavy metals accumulate more in muscle tissues. Thus, diets consisting of different parts of the fish would result in varying levels of exposure to the consumer. The COPCs with the greatest contribution to the cumulative risk and hazard are persistent chlorinated organic compounds (PCBs, DDx, and various PCDD/PCDF congeners) that preferentially accumulate in fatty tissue.

Smallmouth bass and common carp fillet-with-skin and the remainder-of-body samples were analyzed separately in Round 3B, and whole body concentrations were calculated from these results on a weighted average basis. This analysis provided the opportunity to compare concentrations of chemicals in fillet tissue with estimated concentrations in whole body tissue for the same fish. This analysis focused on PCBs, since they contribute to the majority of risks from tissue consumption in the BHHRA and preferentially accumulate in fatty tissue. Mercury was also evaluated because it preferentially accumulates in muscle tissue and would provide a range of the differences between concentrations in the two tissue types.

Total PCB concentrations in samples of smallmouth bass fillet ranged from 11 to 22 percent of the whole body concentration (approximately 4 to 10 times higher in whole body tissue); the concentration of total PCBs in fillet of common carp ranged from 50 to 80 percent of the whole body concentration (approximately 1 to 2 times higher in whole body tissue).

Mercury concentrations in smallmouth bass fillet tissue ranged from 100 to 220 percent of the whole body concentration (approximately 1 to 2 times lower in whole body tissue); mercury concentrations in carp fillet tissue ranged from 110 to 140 percent of the whole body concentration (approximately 1 to 1.5 times lower in whole body tissue). The results of this analysis are presented in Table F6-1.

Based on information presented in the Columbia Slough consumption survey (Adolfson 1996), the majority of fishers surveyed consume only the fillet, which may not include skin. According to the CRITFC Survey (CRITFC 1994), tribal fish consumers are also most likely to consume the fillet. However, some individuals or groups consume other portions of the fish. Assuming a diet of whole body or fillet tissue with skin represents a conservative assumption and provides a range of risks associated with different dietary habits. Because it is unlikely that a diet consists entirely of whole body tissue, the evaluation of risks associated with consumption of fish for recreational and subsistence fishers are based on fillet data. As noted in the

Columbia Slough survey, perhaps as many as 25 percent of consumers "use the entire fish or make soup from the fish." To the extent that consumption is more than just the fillet, the intake of persistent chlorinated compounds, and the associated risks and hazards will be greater than estimated assuming fillet-only consumption.

6.1.4 Use of Undepurated Tissue to Represent Clam Consumption

Only a limited number clam tissue samples (five of 22) collected in the Study Area were depurated prior to analysis. Depuration is a common practice in the preparation of clams for human consumption, although they may also be consumed undepurated. With the exception of certain metals, average chemical concentrations detected in clam tissue in the Study Area were higher in undepurated than in depurated samples. However, depurated clam tissue samples were collected from edges of the site at the northern and southern stretches, and the concentrations are shown in Tables 3-24 and 3-25. Using the concentrations from undepurated samples provides a health-protective approach to assessing risk from consumption of clams. A comparison of exposure concentrations calculated using depurated and undepurated clam tissue is presented in Table F6-12.

6.1.5 Use of Different Tissue Sample Preparation to Assess the Same Chemical

Samples of resident fish tissue from Round 1 were analyzed for mercury in fillet tissue without skin, while during Round 3, smallmouth bass and common carp samples were analyzed in fillet tissue with skin. The Round 1 and Round 3 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, concentrations would be expected to be higher in the fillet tissue samples without skin. However, for smallmouth bass, mercury concentrations were generally higher in fillet tissue with skin, while 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 affect the conclusions of this BHHRA.

6.1.6 Exclusion of Non-Detected Chemicals Where Detection Limits Exceeded Analytical Concentration Goals

Although site-specific Analytical Concentration Goals (ACGs) were established for each media, ACGs for some chemicals were not attainable in some instances with present laboratory methods. DLs for chemicals that were analyzed but never detected were compared to the appropriate ACG for each media, and the results of that analysis are presented in Tables 6-5 through 6-7.

Chemicals that were not detected were not quantitatively evaluated in the BHHRA. If chemicals were present at concentrations above the ACGs but below the DLs, those chemicals would contribute to the estimated risk and hazard. 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 affect 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 Section 3.4, if the DL for non-detected result was greater than the maximum detected concentration for an exposure area, that result not included when calculating the EPC. These results are 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, data were qualified using the "N" qualifier, when the identity of the analyte is not definitive, generally a result of the presence of an analytical interference in the sample. Examples include samples analyzed for chlorinated pesticide by EPA Method 8081A, which were most commonly N-qualified as a result of analytical interference due to the presence of PCBs in the samples. These N-qualified data were used in the BHHRA for calculating EPCs in fish and/or clam tissue.

Chemicals identified as contaminants potentially posing unacceptable risks (i.e., resulting in cancer risks greater than 1×10^{-6} or HQs greater than 1) based solely on N-qualified data were evaluated further. These chemicals are:

- Heptachlor epoxide. The identification of heptachlor epoxide as a contaminant potentially posing unacceptable risks was based on one N-qualified result from an undepurated clam sample collected from RM 6 during Round 1. It was also detected in clam samples collected during Round 3, including samples from RM 6. The Round 3 data were not N-qualified and did not result in cancer risks greater than 1×10^{-6}.
- Alpha-hexachlorocyclohexane (alpha-HCH). The identification of alpha-HCH as a contaminant potentially posing unacceptable risks was based on a single N-qualified result in a black crappie whole body sample collected from RM 6 to 9. Alpha-HCH was also detected in smallmouth bass and common carp in the Round 3 samples (which did not include black crappie). The Round 3 data were not N-qualified and did not result in cancer risks greater than 1×10^{-6}.
- Beta-hexachlorocyclohexane (beta-HCH). The identification of beta-HCH as a contaminant potentially posing unacceptable risks was based on a single

N-qualified result in a smallmouth bass fillet sample collected from RM 3 in Round 1. Beta-HCH was not detected in the whole body sample collected from RM 3 in Round 1 or in the fillet samples collected from RM 3 in Round 3. Beta-HCH was detected in other smallmouth bass samples collected during Round 3. The Round 3 data were not N-qualified and did not result in cancer risks greater than 1 x 10^{-6}.

- Heptachlor. The identification of heptachlor as a contaminant potentially posing unacceptable risks was based on a single N-qualified result in a black crappie whole body sample collected from RM 3 to 6.

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

When data are presented as summed values (e.g., total PCB congeners), one-half the detection limit was used as a surrogate concentration when calculating the summed value for those specific analytes reported as non-detect. Use of one-half the detection limit assumes that there is equal probability that the actual concentration in the sample may be greater or less than the surrogate value. 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 surrogate values for non-detect results was minimized.

6.1.10 Contaminants That Were Not Analyzed in Certain Samples

Not all fish tissue samples were analyzed for the same suite of analytes. For example, fillet samples collected in Round 1 were analyzed for PCB as Aroclors, but no analysis was done for dioxins and furans. Fillet samples of smallmouth bass and common carp collected in Round 3B were analyzed for PCB, dioxin, and furan congeners. In samples where congeners were analyzed, the risks from the total dioxin TEQ, which is not otherwise measured, comprise approximately 1 to 70 percent of the cumulative risks. Therefore, the use of black crappie and brown bullhead fillet tissue, which were only analyzed in Round 1, likely underestimate the actual risks particularly in those areas where PCBs and dioxin/furans are the predominant contaminants.

In addition, not all clam samples were analyzed for the same number of contaminants due to limited tissue mass of some composites collected during Round 2. Table 6-8 presents a listing of analyses not completed for specific samples. Additional samples were collected in Round 3B and analyzed for a greater number of specific 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.

6.1.11 Chemicals That Were Not Included as Analytes

As it is not practical to analyze for every chemical, specific chemicals and chemical groups were chosen for analysis based on an investigation of known or probable sources at in the LWR. However, the chemicals expected to have the potential for significant contributions to risk are included in the risk assessment. The list of chemicals for analysis was determined in collaboration with EPA and its partners and presented in the approved sampling and analysis plan. Subsequently, there has been interest in two additional groups of chemicals: 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 tissue or surface water samples. Because of their nature, VOCs are not expected to accumulate in tissue to a sufficient degree to pose significant risk via consumption relative to the other chemicals detected in tissue. Given the magnitude of concentrations and toxicities of other chemicals that were detected in surface water and tissue, VOCs are unlikely to contribute significantly to the overall risks. Therefore, the lack of analysis for VOCs is unlikely to alter the conclusions of the BHHRA.

6.1.12 Chemicals That Were Analyzed But Not Included in BHHRA

Not all detected chemicals were included in the BHHRA. The following analytes were excluded from assessment are either because there are no suspected sources, or the analyte typically only present adverse health risks at high concentrations:

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

6.1.13 Data Not Included in BHHRA due to Collection Date

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

6.1.14 Compositing Methods for Biota and Beach Sediment Sampling

Compositing schemes were developed to be representative of the medium sampled and to be representative of each exposure unit. Fish were composited based on an estimate of the average home range for each species (ODFW 2005). The home ranges for common carp and brown bullhead may be as large or larger than the Study Area, 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. 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. However, the compositing scheme represents only an approximation of the home ranges of the fish collected, and typically consisted of five individual fish. Replicate composite samples were collected, and risks were evaluated using both the composite samples as well as on a Study Area-wide basis. Where contaminants are evaluated on a harbor-wide basis and/or specific species are wide-ranging, this process is not likely to have an appreciable effect on the conclusions of the BHHRA. However, where samples are composited over an area larger than the actual home range of specific fish species, the result may either over- or underestimate risks, depending on the distribution of contaminant concentrations in the area over which samples are composited. For example, the highest DDx concentrations are located on the west side of the river at RM 7.5, while the EPC for smallmouth bass at that river mile combined data collected from both sides of the river.

Beach sediment was composited on a beach by beach basis, resulting in a single sample result for each exposure area. Uncertainty stems from 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 1×10^{-4} to1×10^{-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 percent of

both sample types on a mass basis. However, since smallmouth bass exposure areas were assessed 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 affect the conclusions of this BHHRA.

6.2 EXPOSURE ASSESSMENT

Uncertainties that arise during the exposure assessment can typically have some of the greatest effect on 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 Subsurface Sediment Exposure

A complete exposure pathway requires the presence of a retention or transport medium, an exposure point, and an exposure route. Subsurface sediment was not considered an exposure medium in the BHHRA because it was assumed that potential human contact with river sediment below 30 cm in depth was unlikely, or that if it does occur, the frequency and extent would be minimal. Situations which may result in human exposure to subsurface include: potential scouring, natural hydraulic events that are not well understood, future development of near-shore and upland properties, maintenance of the 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. Due to the low potential of exposure to subsurface sediment, the estimates presented in the BHHRA are considered sufficiently representative of baseline exposures.

6.2.2 Potential Exposure Scenarios

Some of the key uncertainties associated with the exposure scenarios evaluated in the BHHRA are discussed in the following subsections.

6.2.2.1 Shellfish Consumption

A commercial crayfish fishery exists in the LWR, and 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), and no commercial crayfish landings were reported for the Willamette River in Multnomah County from 2005 to 2007. DHS had previously received information from ODFW indicating that an average of 4,300 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.

Evidence of current consumption of freshwater clams from Portland Harbor is limited. According to a project conducted by the Linnton Community Center (Wagner 2004), transients reportedly consume clams from the river on a limited and infrequent basis. 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 and were not documented. Transients reported consuming various fish species, as well as crayfish and clams, and many 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. DEQ and EPA staff have occasionally received calls from individuals who claim to have harvested clams and are inquiring whether consumption is safe, and individuals have been observed harvesting clams from the shore in Portland.

6.2.2.2 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. An evaluation was also performed of helmet diving with use of a neck dam, which allows can allow water to leak 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.

6.2.2.3 Potential Future Domestic Water Use

The evaluation of surface water as a domestic water source is based on the assumption that surface water is drawn from the Study Area. Within the Study Area, the LWR 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). Thus, it is unlikely that individuals at households receiving water from the city would be exposed to contaminants at concentrations greater than the MCL. As presented in Section 5.2.8, cPAHs and MCPP are the only COPCs that posed an estimated cancer risk greater than 1×10^{-4} (cPAHs) or a noncancer hazard greater than 1 (MCPP). The uncertainties associated with assessing dermal exposures to dissolved PAHs are discussed further in Section 6.2.4.2. Although there is no MCL established for MCPP, the associated HQ is greater than 1 at only one of the locations evaluated, W035, located at RM 8.5, where the estimated hazard is 2.

6.2.3 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. Ingestion and dermal absorption of chemicals from surface water were quantitatively evaluated for the populations that are expected to have the most frequent contact with surface water. Surface water exposures were not evaluated were for dockside workers, in-water workers, tribal fishers, and fishers.

The 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 typically expected to be 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 exposure pathways considered insignificant could underestimate risks for the Study Area. Due to the low potential of exposure for these pathways, this uncertainty is not expected to impact the conclusions of this BHHRA.

6.2.4 Exposure Factors

Assumptions about exposure factors typically result in uncertainty in any risk assessment. As discussed previously, the scenarios evaluated are representative of exposures that could occur in the Study Area under either current or future conditions. RME and CT values were used for the exposure scenarios to help assess the overall effect that variability in each of the exposure assumptions has on the risk estimates.

The range of risk estimates between these two exposure scenarios provides a measure of the uncertainty surrounding these estimates.

A range of consumption rates for fish consumption were used to evaluate variability on the risk estimates, thus the resulting risks in this BHHRA represent a range of possible outcomes, including estimates that may be representative of the upper range of plausible exposures.

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

6.2.4.1 Exposure Parameters for Sediment Exposure Scenarios

The parameters used in the BHHRA to evaluate beach and in-water sediment exposure used were intended to provide conservative estimates based on potential uses in 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, and the extent to which the beach may be used and the nature of the contact with sediments is unknown. Future changes in land use may make some beach areas more- or less-accessible to the general public, which increases uncertainty about future exposure. When evaluating in-water sediment, each on-half mile 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.

There are uncertainties associated in the selection of the exposure duration, frequency, and intake parameters used to evaluate both beach and in-water sediment exposures. These scenarios assume long-term repeated use of the same beach or one-half mile river mile segment, which may not accurately reflect actual use practices. The exposure frequencies evaluated range from 94 days/year up to 250 days/year. Default intake parameters for soil exposure were generally used; however, to account for an assumed greater moisture content of beach sediments, the dermal adherence factor used to evaluate child recreational beach exposure was 10-fold greater than the default for soil. Consistent with EPA guidance (2004), only those compounds or classes of compounds for which dermal absorption factors are available were quantitatively evaluated via dermal contact exposure. COPCs for which dermal absorption factors were not available were not quantitatively evaluated, as dermal absorption was essentially assumed to be zero. However, as the majority of COPCs were quantitatively evaluated, this uncertainty does not substantially 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.

6.2.4.2 Exposure Parameters for Surface Water and Groundwater Seep Exposure Scenarios

Although dermal absorption of PAHs from water was quantitatively evaluated in the BHHRA, the dermal permeability coefficient (K_p) falls outside of the effective predictive domain (EPD) for a number of the PAHs, including the following:

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

EPA dermal assessment 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 percent lower confidence level) to 20 cm/hr (95 percent 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 1×10^{-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 1×10^{-4} to 1×10^{-6}.

6.2.4.3 Exposure Parameters for Fish/Shellfish Consumption Scenarios

Site-specific information regarding fish consumption is not available for Portland Harbor. In the absence of specific data, fish consumption data representative from several sources was considered and selected as being representative of the general population of the greater Portland area, as well as that portion of the population that actively fishes the Lower Willamette and utilizes fish from the river as a partial source of food. However, the rates presented in the CSFII study represent per capita consumption rates rather than true long-term averaged consumption rates. Further, the large range between the percentile values is indicative of substantial variability in the underlying data. For example, consumption rates consumers are 200 g/day at the 90th percentile and 506 g/day at the 99th percentile. The consumption rate for consumers and non-consumers is approximately 18 g/day at the 90th percentile and 142 g/day at the 99th percentile. As discussed in Section 3.5.9.6, the RME consumption rate selected for recreational fishers of 49 g/day is based on data from the Columbia Slough study. That study was a creel survey, and the representativeness of the rate is dependent on several factors, 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 consumption rates, uncertainty also exists with respect to the relative percentage of the diet of obtained from the Study Area or within individual exposure areas versus other nearby sources of fish, and the degree to which different methods of preparation and cooking may reduce concentrations of persistent lipophilic contaminants.

Uncertainties associated with tribal consumption rates largely relate to limitations inherent in the CRTFIC consumption survey on which the consumption rates used in the BHHRA are based. These consumption rates may be 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.
- 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.

Conversely, conservative assumptions were used with respect to exposure frequency and duration, as well as the relative contribution of fish from the Lower Willamette to the overall tribal diet. According to the CRITFC survey, none of the respondents fished the Willamette River for resident fish and at most, approximately 4 percent fished the Willamette for anadromous fish. However, future use of the site by tribal members may change if fishery resources improved.

Information regarding consumption of shellfish from the Study Area relies in part from information obtained 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 actually occurs. Because site-specific shellfish consumption rates are not available, nationwide CSFII (USDA 1998) shellfish consumption data were used. As with the rates for fish consumption, these are based on per capita consumption rates from the general population. In the nationwide survey, shrimp accounted for more than 80 percent of the shellfish consumed, crayfish accounted for less than one percent of diet, 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, the mean nationwide shellfish consumption rate from freshwater sources is 0.01 g/day; upper percentiles for freshwater shellfish consumption rates are not available (EPA 2002b).

6.2.4.4 Assumptions about a Multi-Species Diet

The non-tribal multi-species diet assumes equal proportions of all four resident fish species, the tribal multi-species diet assumed equal proportions of the four resident fish species, as well as dietary percentages of salmon, lamprey, and sturgeon derived from the CRITFC survey. Variations of these dietary assumptions would result in different risk estimates. The risks due to consumption of the individual species that make up the multi-species diet were evaluated separately, and the results are presented in Table F6-3. The range of risks from fish consumption scenarios encompasses the potential variations in the composition of a multi-species diet. The range of the magnitude of risks associated with a diet consisting solely of the individual species is generally less than an order of magnitude. The magnitude in the difference of risk estimates based on diet composition shows that an assumption of equal proportions of the four resident fish species could result in over or under-estimation of actual risks from a multi-species diet.

6.2.5 Exposure Point Concentrations

The following uncertainties related to calculation of EPCs for this risk assessment were analyzed further to determine the potential effects on the risk estimates.

6.2.5.1 Using 5-10 Samples to Calculate the 95 percent UCL on the Mean

Data sets with fewer than 10 samples generally provide poor estimates of the mean concentration, defined as a large difference between the sample mean and the 95 percent UCL. In general, the UCL approaches the true mean as more samples are included in the calculation of the EPC. Study Area-wide fish tissue EPCs that were calculated as the 95 percent UCL on the mean using less than 10 samples included EPCs for whole body brown bullhead and common carp fillet. The 95 percent UCLs calculated using less than 10 samples are presented in Appendix F2, and a comparison of maximum reported concentrations and Study Area-wide EPCs for PCBs and dioxins based on fewer than 10 samples is presented Table F6-4.

6.2.5.2 Nondetects Greater than Maximum Detected Concentrations

Consistent with EPA guidance, analytical results reported as non-detect for which the detection limit was greater than the maximum detected concentration in a given exposure area were removed from the dataset prior to calculation of the 95 percent UCL. These sample identifications, detection limits, and associated maximum concentrations are listed by media and exposure area in the tables in Attachment F2. If the actual concentrations were closer to the detection limit for surface water and in-water sediment, the risk estimates would still be less than 1×10^{-6}.

6.2.5.3 Using the Maximum Concentration to Represent Exposure

The maximum concentration was used in instances where there were either less than five detected results or fives samples for a given analyte and exposure area, including EPCs calculated to represent Study Area-wide exposure. 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 percent 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. However, EPA's UCL guidance (EPA 2002) notes that defaulting to the maximum observed concentration may not be protective when sample sizes are very small because the observed maximum may be smaller than the population mean.

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 percent UCL/maximum EPCs to the mean are equal to or less than 2, and the remaining ratios are less than 4. An 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 presented in Tables F6-5 through F6-8 in Attachment F6.

6.2.5.4 Anomalous Antimony and Lead Results in Smallmouth Bass

A smallmouth bass sample collected during the Round 3 sampling effort at RM 10E (LW3-SB010E-C00B) had anomalously high detected concentrations of lead and antimony in the tissue analyzed as whole body-without-fillet. As described in the Round 3B Fish and Invertebrate Tissue and Collocated Sediment Data Report, Addendum 1 (Integral 2008), the sample was reanalyzed, and because of the consistently high detection of these compounds, the results of the lead and antimony re-analyses for this sample were averaged for use in the BHHRA. The resulting concentrations in body-without-fillet tissue from this sample are 1,640 mg/kg lead, and 8.41 mg/kg antimony, which are 160 times higher than the next highest lead and

antimony concentration detected in smallmouth bass within the Study Area. These concentrations are consistent with what would be expected from fish that swallowed fishing gear containing lead and antimony or other similar metal objects, and thus may not be representative of tissue concentrations resulting from exposure to contamination within the Study Area. However, these concentrations were used with the corresponding fillet concentrations to calculate a whole body concentration for use in the BHHRA, which was also anomalously high. The concentrations of lead and antimony for this sample (LW3-SB010E-C00WB) were the maximum concentrations for the RM 10E smallmouth bass exposure area, and due to the low number of smallmouth bass samples within the exposure area, they were used as the EPCs. The maximum concentrations of this sample are a conservative estimate of potential exposures from this river mile stretch. The concentrations from this sample were also used in the calculation of Study Area-wide EPCs for smallmouth bass, creating a high bias in the dataset.

6.2.5.5 Possible Effects of Preparation and Cooking Methods

Cooking and preparation methods of fish tissue can change the concentration of lipophilic contaminants in fish tissues; EPA (1997b) states that "cleaning and cooking techniques may reduce the levels of some chemical pollutants in the fish." PCBs tend to concentrate in fatty tissues. Trimming away fatty tissues, including the skin, may reduce the exposure to PCBs. Removing the skin can reduce PCB concentrations in raw fillet by 50 percent by (EPA 2000c). Cooking can also reduce the concentrations as much as 87 percent, depending on the method (Wilson et al. 1998). However, one study showed a net gain in PCB concentrations after cooking (EPA 2000c). The potential for reduction in PCB concentrations due to cooking is subject to a substantial degree of variability, and some consumption practices make use of whole fish, reductions in PCB concentrations were not considered quantitatively in the risk assessment. EPA's guidance on assessing contaminant concentrations for fish consumption advisories (EPA 2000) includes a summary of contaminant reductions observed in several studies due to skinning, trimming, and cooking. Depending on species and preparation/cooking method, observed PCB concentrations range from 16 to 80 percent of those found in whole body fish. This information is summarized in Table F6-9.

6.2.5.6 Assumptions about Arsenic Speciation

Tissue concentrations of arsenic were reported as total arsenic, while EPA toxicity criteria are based on inorganic arsenic. A study conducted on the middle Willamette River (EVS 2000) measured composites of resident fish (largescale sucker, carp, smallmouth bass, and northern pikeminnow) from a 45-mile section of the river extending from the Willamette (River Mile 26.5) to Wheatland Ferry (River Mile 72). Total arsenic and inorganic arsenic concentrations were determined in composites of whole body, fillet with skin, and composites of that portion of the fish remaining after removing fillets. Percent inorganic arsenic ranged from 2 percent (carp) to 13.3 percent (sucker). The average percent of inorganic arsenic was 4.2 percent for the

carp and 3.8 percent for the smallmouth bass. Consistent with the recommendation in the Columbia River Basin Fish Contaminant Survey (EPA 2002e), the EPC for inorganic arsenic was estimated as 10 percent of the total arsenic detected in tissue.

Inorganic arsenic in clams was found to range as high as 50 percent of total arsenic in data collected in the Lower Duwamish River. However, the Lower Duwamish is an estuarine system, while the Lower Willamette in Portland Harbor is freshwater system. 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 in shellfish. The clam tissue data collected from the Study Area 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:

- All of the arsenic concentrations in clam tissue are within a factor of 2. In addition, the arsenic concentrations in clams are normally distributed.
- 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 percent of the total arsenic rather than the assumption of 10 percent used in the BHHRA, the cumulative risks from consumption of clams increase by a factor of 1.1 to 1.3. Arsenic is not the primary contributor 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 substantially less than that from PCBs. Therefore, the assumptions about inorganic arsenic are not likely to affect the overall conclusions of the BHHRA.

6.2.5.7 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. 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 greater uncertainty in the PCB Aroclor analysis for tissue. However, fillet tissue collected in Round 1 was analyzed for PCB Aroclors only, Round 3 smallmouth bass and common carp samples 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 consumption 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. A comparison of the results for PCBs as Aroclors and PCBs as individual congeners for tissue samples is provided in Table F6-11. This comparison is based on the whole body tissue data from Round 1, which is the only sampling event where Aroclors and congeners were analyzed in the same tissue samples. As shown in Table F6-10, sometimes the congener results are higher and other times the Aroclor results are higher.

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.

6.2.5.8 Bioavailability of Chemicals

The toxicity values used in the risk assessment are often based on laboratory studies in which the chemical is administered in a controlled setting via food or water. 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 exposure to that chemical in that medium. The National Research Council has 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.

6.2.5.9 Exposure Areas for Consumption of Smallmouth Bass

Fish consumption on a river mile basis was assessed using data from smallmouth bass. 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, Round 3, samples were composited on a per river mile basis for each side of river. The Round 1 and Round 3 results were combined, and the EPC thus represents an exposure area of one river mile. A study by ODFW (ODFW 2005) that included tracking the movement of smallmouth bass in the Lower Willamette indicated that their home range is typically between 0.1 and 1.2 km, and they are most frequently found in near-shore areas.

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. 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 was typically used as the RME EPC for the river mile exposure areas. In addition, the area over which fishing occurs should also be considered. Given an exposure duration of 30 to 70 years, it is possible that fish would be collected over an area greater than a single river mile. Therefore, use of an exposure area consisting of a single river mile for evaluating consumption of smallmouth bass is generally health protective and unlikely to underestimate risks.

6.2.5.10 EPCs in Surface Water for Recreational Beach Users

Only data collected from the low water sampling event was used to assess recreational exposures to surface water, in order to represent surface water conditions during the time of year when most frequent recreational use occurs. There is some

uncertainty in the representativeness of this dataset for surface water conditions for recreational users.

Because exposure to surface water by transients can occur throughout the year, 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 affect 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. 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).

EPA applies uncertainty factors, typically a factor 10, when deriving reference doses, to account for limitations in the data. These limitations include variation in susceptibility among the members of the human population, uncertainty in extrapolating animal data to humans, uncertainty in extrapolating from data obtained in a study with less-than-lifetime exposure, uncertainty in extrapolating from a LOAEL rather than from a NOAEL, and uncertainty associated with extrapolation when the database is incomplete. As a result, actual risks within the Study Area are likely to be lower than the estimates calculated in this BHHRA.

In addition, the following specific uncertainties have been identified.

6.3.1 Early Life Exposure to Carcinogens

As discussed in Section 3.5.6, early-in-life susceptibility to carcinogens has long been recognized as a public health concern. EPA's Supplemental Guidance for Assessing Susceptibility from Early-Life Exposure to Carcinogens (EPA 2005b) provides a process to evaluate risks from early-life exposure to carcinogens known to act via a mutagenic mode of action. The only exposure scenarios for which early-life exposures are considered are recreational beach use, fish consumption, and household use of surface water. Of the COPCs identified in the risk assessment, only cPAHs have been identified as mutagenic.

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.

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 affect the conclusions of this BHHRA.

6.3.4 Toxicity Values for Chromium

Chromium was analyzed as total chromium in all media. Although toxicity values exist for both trivalent and hexavalent chromium, hexavalent chromium exhibits greater toxicity that the trivalent form. The reference dose for hexavalent chromium is 0.003 mg/kg-day, versus 1.5 mg/kg-day for trivalent chromium. Hexavalent chromium can be reduced 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). 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.

The highest HQ for chromium from fish consumption was 0.004. Even if a portion of the chromium were present as hexavalent chromium, the HQ would likely still be less than 1. Additionally, EPA currently considers the carcinogenic potential of hexavalent chromium via oral exposure as “cannot be determined.” Toxicity criteria derived by the New Jersey Dept. of Environmental Protection was used as a Tier 3 source for evaluating the cancer risks associated with oral exposures to hexavalent chromium.

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 (dioxin-like PCBs), and coplanar PCBs may have contributed to the carcinogenicity observed in the study. Because the cancer risk from coplanar PCB congeners was evaluated separately, including both the total PCB and coplanar PCB congener risks in the cumulative cancer risk may result in an overestimate of the cancer risks. Although the potential double counting of PCB mass was corrected for by using the PCB adjusted values, there was no correction for the potential double counting of toxicity of dioxin-like PCBs in the PCB TEQ cancer risk estimate.

PCBs are classified as probable human carcinogens based on adequate dose-response data from studies in rats. However, the human carcinogenicity data are inadequate. Several cohort studies have been conducted that analyzed cancer mortality in workers

exposed to PCBs. These 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.

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. In its 2001 evaluation of the 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 sub-linear 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

As discussed in Section 4.7, an adjustment was applied to the oral toxicity factor to account for the estimated absorbed dose when evaluating dermal exposures when the following conditions were 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 percent in a medium similar to the one used in the critical study.

EPA (2004) recommends the adjustment of oral toxicity values to reflect dermal absorption only when GI absorption was less than 50 percent, eliminating 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. Organic chemicals are generally well absorbed across the GI tract, absorption of inorganic chemicals is dependent on a number of factors, but is generally less than for organic chemicals. However, in the absence of a specific value for GI absorption, a default of 100 percent was used. EPA 2004 states that assuming 100 percent absorption may underestimate dermal risk for those chemicals that are poorly absorbed because it overestimates the dose at the site of action. The extent of underestimation is proportional to the actual GI absorption. Inorganic COPCs for which the default value of 100 percent GI absorption was used are 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 some chemicals may have a myriad of health effects as exposures increase. As an example, a majority of the non-cancer affect from the site is 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 reproductive effects. 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 would be a factor of 4 greater than 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, 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 (Tables 5-87 through 5-111). 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 experience 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 affect the conclusions of this BHHRA.

In addition to cumulative exposure scenarios for a given population, an individual may be a member of multiple exposure populations, and thus 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 fish and shellfish consumption is typically at least 10-fold greater than other exposure pathways, if an individual consumes fish, the relative 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

Metals are naturally occurring and the concentrations present in tissue, water, or sediment may not be directly related to contamination. Reported concentrations of arsenic and mercury in samples collected within the Study Area result in estimated risks greater than 1×10^{-6} or an HQ of 1 for one or more of the exposure scenarios evaluated in the BHHRA. Exposure concentrations of arsenic in beach sediment ranged from 0.7 mg/kg to 9.9 mg/kg, within the general range of 7 mg/kg used as a background concentration of arsenic by DEQ (DEQ 2007). At the background concentration of 7 mg/kg, the calculated risk from arsenic would exceed 1×10^{-6} for several of the beach sediment and in-water sediment exposure scenarios evaluated in this BHHRA.

Neither background nor anthropogenic tissue concentrations of COPCs were established for the Study Area. Regional tissue concentrations were measured as part of the Columbia River Basin Fish Contaminant Survey in 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. Concentrations of certain contaminants are higher in tissue collected within the Study Area than observed in the Columbia River study, and the sources of the regional tissue concentrations are unknown. Consistent with EPA policy, risk estimates were presented in this BHHRA without accounting for

contributions from background. However, it is important to recognize that background concentrations may result in unacceptable risk and hazard estimates.

6.4.4 Risks from Exposure to Lead

The maximum EPC calculated for lead are associated with a probability of exceeding protective blood lead levels in the fetus of a pregnant woman who consumes fish from the Study Area. This EPC may be attributable to lead in the gut of the fish rather than tissue concentrations. Protective lead concentrations in tissue 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 as adapted for the Portland Harbor BHHRA focuses on potential affects to the fetus when considering fish consumption by pregnant women. However, the ALM was developed for evaluating exposure to lead in soil and may not be appropriate to use for fish consumption. Furthermore, the ALM is sensitive to the bioavailability of ingested lead. For purposes of calculating a tissue concentration of lead that is expected to be without adverse effects, the default bioavailability of lead in soil was used, and it is not known whether this is an appropriate assumption for lead in tissue.

6.4.5 Future Risks

This BHHRA estimated current and future risks for exposure within the Study Area, based on known and reasonably anticipated future uses of the Study Area. However, the LWR is a dynamic, industrialized waterway, and if the land uses in certain areas of the Study Area were to change in the future in a manner with the uses considered in the BHHRA, risk and hazard estimates presented here may not be representative of conditions in the future.

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 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 in this section. The magnitudes of these ranges are provided in Attachment F6.

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.

7.0 SUMMARY

The overall objective of this BHHRA is intended to provide an analysis of baseline risks and help determine the need for action at the Site, and to provide risk managers with an understanding of the actual and potential risks to human health posed by the site, and any uncertainties associated with the assessment.

The populations evaluated in the BHHRA were identified based on human activities currently known to occur within the Study Area or that could occur in the future, as described in the Programmatic Work Plan or in subsequent direction from EPA. Populations and associated exposure scenarios that were quantitatively evaluated in this BHHRA include:

- Dockside Workers – Direct exposure to beach sediment
- In-water Workers – Direct exposure to in-water sediment
- Recreational Beach Users – Direct exposure to beach sediment and surface water
- Transients – Direct exposure to beach sediment, surface water, and groundwater seep
- Divers – Direct exposure to in-water sediment and surface water
- Recreational and Subsistence Fishers – Direct exposure to beach or in-water sediment, consumption fish and shellfish
- Tribal Fishers – Direct exposure to beach and in-water sediment, consumption of fish
- Potential Future Domestic Water Use – Direct exposure to surface water used as a domestic water source
- Infants - Indirect exposure to bioaccumulative contaminants (PCBs, dioxin/furans, DDx, and PBDEs) in environmental media via indirect exposures due to breastfeeding.

7.1 SUMMARY OF RISKS

A comparison of the estimated risks by exposure media can help focus risk management decisions by identifying the media contributing most to the overall human health risks 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 noncancer HIs, respectively, for each media type, based on RME exposure assumptions for each media evaluated in the BHHRA. The estimated risks associated with consumption of fish and shellfish are orders of magnitude higher than risks from other scenarios, and exceed a cumulative cancer risk of 1×10^{-4} and a HI of 1. Scenarios for which the cumulative estimated cancer risk is greater than 1×10^{-4} or the HI is greater than 1 are consumption of fish and shellfish, and direct contact with in-water sediment by tribal and high frequency fishers.

7.2 CONTAMINANTS POTENTIALLY POSING UNACCEPTABLE RISKS

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.

Contaminants were identified as potentially posing unacceptable risks if the estimated cancer risk is greater than 1×10^{-6} or the HQ is greater than 1 for any of the exposure scenarios evaluated in this BHHRA, regardless of the uncertainties associated with the estimates. Given the uncertainties in the analytical data discussed in Section 6, the preliminary list was further refined to select the final listing of contaminants potentially posing unacceptable risks for this BHHRA. This 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.

α-, β-, and γ-Hexachlorocyclohexane and heptachlor were detected in fish tissue only 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 the subsequent Rounds 2 and 3 sampling events, different analytical methods were used so that the identification of pesticides was not an issue in tissue. EPA guidance (1989) recommends caution in the use of data where there are uncertainties in the identification of contaminants. 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.

Additional considerations in the evaluation of contaminants potentially posing unacceptable risk 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 site use, 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 or other anthropogenic sources of chemicals in the region.
- Magnitude of risk greater than EPA's target range for managing cancer risk of 1×10^{-4} to 1×10^{-6} and noncancer hazard of 1.

7.2.1 Fish Consumption Scenarios

Twenty four contaminants (PCBs, dioxins, four metals, bis(2-ethylhexyl)phthalate (BEHP), PAHs, hexachlorobenzene, and five pesticides) are identified as potentially posing unacceptable risks associated with consumption of fish:

- PCBs (PCBs and PCB TEQs): Based on the magnitude of the estimated risks greater than 1×10^{-4}, the overall spatial scale, and the relative contribution to cumulative risk estimates.

- Dioxins/furans: Based on localized and Study Area-wide exposures, the magnitude of the risk estimates greater than 1×10^{-4}, the overall spatial scale, and the relative contribution to cumulative risk estimates.

- Metals: Antimony, arsenic, and mercury were associated with one or more fish consumption scenarios that resulted in a risk estimate that exceeded a cancer risk of 1×10^{-6} or HQ of 1.
    - The overall estimated risk estimates for arsenic are greater than 1×10^{-4} based on Study Area-wide exposures.
    - The HQ associated with antimony is greater than 1 based on tribal consumption.
    - Lead, based on a measured tissue concentration greater than the protective tissue concentrations derived using blood lead models. However, this is due to only a single result of smallmouth bass whole body sample collected at RM 10 with an anomalously high result, as discussed in Section 6.1.14
    - Mercury, based on an HQ greater than 1 for both localized and Study Area-wide exposures.

- BEHP, based on a cancer risk estimate greater than 1 x 10^{-6} for tribal consumption of whole body tissue.

- PAHs: Benzo(a)anthracene, benzo(a)pyrene, dibenzo(a)anthracene, and total cPAHs, based on cancer risk estimates greater than 1 x 10^{-6}. Cancer risk estimates for total carcinogenic PAH are greater than 1 x 10^{-6} at five river mile segments and Study Area-wide.

- Organochlorine Pesticides: Dieldrin, total chlordane, total DDD, total DDE, and total DDT are identified based on estimated cancer risks greater than 1 x 10^{-6} or an HQ of 1.
    - Dieldrin, based on estimated cancer risks greater than 1 x 10^{-6} for consumption of fish on a localized and Study Area-wide basis.
    - Total chlordane, based on estimated cancer risks greater than 1 x 10^{-6} for consumption of fish on a Study Area-wide basis.
    - DDx compounds, based on estimated cancer risks greater than 1 x 10^{-6} for consumption of fish on a localized and Study Area-wide basis.

- PBDEs: based on an HQ greater than 1 for consumption of fish on a localized basis.

Considering the magnitude and relative contribution to the overall risk estimates, as well as their frequency of detection, PCBs and dioxins/furans are the most significant contributors to risk from fish consumption scenarios. Estimated risks from PCBs and dioxins/furans are greater than 1 x 10^{-4} or an HQ of 1 for both the CT and RME evaluations at both localized and Study Area-wide exposures. Figure 7-3 illustrates the relative contribution of individual contaminants to cumulative risk estimates based on the Study Area-wide multi-species fish consumption by subsistence fishers. PCBs are the most significant contributor to the overall risk estimate, and taken together with dioxins/furans, account for the majority of the estimated risk on a Study Area-wide basis. On a river mile basis, PCBs pose the highest risks at all locations except RM 7, where dioxins/furans pose the highest risks. Figure 7-4 shows the relative contributions to the overall risk estimate based on tribal fish consumption.

PCBs and dioxins/furans have been detected in fish tissue collected outside of the Study Area in both the Willamette and Columbia Rivers. In a risk assessment for the mid-Willamette (EVS 2000), PCB concentrations were found to result in a HQ greater than 1 assuming both a 142 g/day and a 17.5 g/day consumption rate, and an estimated cancer risk greater than 1 x 10^{-4} for the 142 g/day consumption rate. Dioxins and furans were also found to result in an estimated cancer risk greater than 1 x 10^{-4} using a 142 g/day consumption rate (non-cancer endpoints were not evaluated for dioxins and furans). In the Columbia River Basin Fish Contaminant Survey (EPA 2002c), the estimated cancer risks associated with PCBs and dioxins/furans were greater than 1 x 10^{-4} assuming a consumption rate

of 142 g/day, and the estimated risk due to PCBs was greater than 1 x 10^{-4} assuming a consumption rate of 7.5 g/day. While ambient 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 consumption rates. 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.

7.2.2 Shellfish Consumption Scenarios

Seventeen contaminants (PCBs, dioxins, arsenic, PAHs, pentachlorophenol, and five pesticides) were identified as potentially posing unacceptable risks due to consumption of shellfish, based on estimated cancer risks greater than 1 x 10^{-6} or a HQ of 1:

- PCBs (Total PCBs and PCB TEQs): Based on cancer risk estimates greater than 1 x 10^{-4} and/or HQs greater than 1 for shellfish consumption in localized and Study Area-wide exposures, the magnitude and spatial scale of the risk estimates greater than 1 x 10^{-4}, their relative contribution to cumulative risk estimates, and their frequency of detection.

- Dioxins/furans: Based on cancer risk estimates greater than 1 x 10^{-4} and/or HQs greater than 1 for shellfish consumption in localized and Study Area-wide exposures, the magnitude and spatial scale of the risk estimates greater than 1 x 10^{-4}, their relative contribution to cumulative risk estimates, and their frequency of detection.

- Arsenic: Based on cancer risk estimates that greater than 1 x 10^{-6} from clams and crayfish at both consumption rates and on a localized and Study Area-wide scale. No cancer risk estimates exceeded 1 x 10^{-4}. Though arsenic is identified as a contaminant potentially posing unacceptable risk on both a localized and Study Area-wide spatial scale, concentrations in shellfish tissue are due in part to the contribution of background concentrations.

- cPAHs: Based on cancer risk estimates greater than 1 x 10^{-6} from both clams and crayfish at both consumption rates and on a localized and Study Area-wide scale. Cancer risk estimates greater than 1 x 10^{-4} from clams collected at locations RM 5W and RM 6W and assuming a consumption rate of 18 g/day. cPAHs are considered a primary contributor to risk for the shellfish consumption pathway at those locations because of the magnitude of the risk estimates and their relative contribution to the cumulative risk.

- Pentachlorophenol: Pentachlorophenol was detected only in a single crayfish composite sample collected near RM 8. It was not detected in the remaining

40 shellfish samples. This single detection of pentachlorophenol resulted in a cancer risk estimate within the range of 1×10^{-6} to 1×10^{-4}.

- Organochlorine pesticides (Aldrin, dieldrin, DDx): Based on an estimated cancer risk greater than 1×10^{-6} or a HQ of 1.
    - Aldrin, based on an estimated cancer risk greater than 1×10^{-6} for consumption of clams at RM 8W and on a Study Area-wide basis, assuming a consumption rate of 18 g/day.
    - Dieldrin, based on an estimated cancer risk greater than 1×10^{-6} for consumption of clams near RM 8W and Study Area-wide, assuming a consumption rate of 18 g/day.
    - DDx, based on an estimated cancer risk greater than 1×10^{-6} for consumption of clams near RM 6W, RM 7W, RM 8W and Study Area-wide, assuming a consumption rate of 18 g/day.

Considering the magnitude and relative contribution to the total risk estimates and their frequency of detection, PCBs, dioxins/furans, and cPAHs are the most significant contributors to the risk estimates associated with consumption of shellfish. On a Study Area-wide basis, PCBs and dioxins/furans combined contribute approximately 52 percent and 92 percent, respectively, of the cumulative cancer risk from consumption of clams and crayfish, cPAHs contribute approximately 41 percent and 4 percent, respectively, of the cumulative cancer risk from consumption of clams (undepurated samples) and crayfish. PCBs pose the highest risks from consumption of clams at all locations except RM 4W, RM 5W, and RM 6W, where cPAHs pose the highest risks, and RM 4E and RM 7W, where dioxins/furans pose the highest risks.

7.2.3 In-Water Sediment Scenarios

PAHs (primarily benzo[a]pyrene), arsenic, PCBs, and dioxins are identified as contaminants potentially posing unacceptable risk for in-water sediment. PAHs and dioxins are identified for all of the in-water sediment scenarios, arsenic and PCBs were identified for the tribal fisher and high frequency fisher scenarios only. The relative contribution of each contaminant to cumulative cancer risk estimates varied by river mile. On a Study Area-wide basis, estimated risks from cPAHs and dioxins/furans each contributed approximately 50 percent of the cumulative cancer risk estimate. As previously discussed, cumulative cancer risks associated with arsenic are due in part to naturally occurring concentrations in sediment. Cumulative cancer risks from PCBs are greater than 1×10^{-6} at four one-half mile river segments, and from dioxins at two one-half mile segments. Cumulative cancer risks from cPAHs are greater than 1×10^{-6} for at 22 one-half mile river segments. Carcinogenic PAHs contribute significantly to risks associated with in-water sediment exposures at many locations throughout the Study Area. PCBs and dioxins/furans contribute significantly to the risk estimates at RM 2E, RM 3.5E, RM 6.5E, RM 8.5W, RM 9W, RM 11E, and Swan Island Lagoon (PCBs) and RM 7W (dioxins/furans).

7.2.4 Beach Sediment Scenarios

PAHs (primarily benzo[a]pyrene) and arsenic were identified as potentially posing unacceptable risk in beach sediment. Risks greater than 1×10^{-6} associated with exposure to arsenic in beach sediment are due in part to naturally occurring concentrations of arsenic. Risks greater than 1×10^{-6} associated with exposure to benzo(a)pyrene was limited to a few locations, with the maximum cumulative cancer risk at beach location 06B025.

7.2.5 Surface Water Scenarios

PAHs contribute significantly to estimated cancer risks that are greater than 1×10^{-4} assuming use of river water as a domestic water source and greater than 1×10^{-6} for divers at RM 6W. However, as noted in Section 5.2.8, the estimated risks associated with dermal exposure to PAHs in water should be used with caution, as PAHs are not within the Effective Prediction Domain of the model used to estimate the dermally-absorbed dose. Additional risk management considerations during remedy selection should consider the limited spatial scale and degree of uncertainty associated with the diver exposure assumptions. HIs greater than 1 at Multnomah Channel and RM 8.5 were due to MCPP and associated with use of river water as a potential drinking water source.

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OFFICE OF
ENVIRONMENTAL
CLEANUP

April 3, 2013

Mr. Bob Wyatt
Chairman, Lower Willamette Group
c/o Northwest Natural
220 Northwest Second Avenue
Portland, Oregon 97209

Re: Portland Harbor Superfund Site, Administrative Order on Consent for Remedial Investigation and Feasibility Study; Docket No. CERCLA-10-2001-0240
Final Approval of the Final Baseline Human Health Risk Assessment dated March 28, 2013

Dear Mr. Wyatt;

EPA has completed its review of the Final Baseline Human Health Risk Assessment (BHHRA) submitted to EPA on March 28, 2013. The EPA has reviewed the Final BHHRA and has determined that the document adequately incorporates EPA's directed revisions and conditional provisions. Thus, EPA approves the BHHRA as final.

EPA has additionally identified three instances in the final BHHRA where the word "hypothetical" has not been changed to "potential future" that needs to be changed in the electronic version and provided as errata to the hard copy document within 14 days of the date of this letter: the first in a footnote to Table 2-16, the second in a cover sheet to a section of the UCL printouts, and the third being a footnote to Table F2-17.

Finalizing the BHHRA is a significant milestone for the Portland Harbor RI/FS. EPA appreciates the LWG's significant efforts and collaboration in developing this final report.

If you have any questions regarding this letter, please contact Chip Humphrey at (503) 326-2678, or humphrey.chip@epa.gov, Kristine Koch at (206) 553-6705, or

koch.kristine@epa.gov. All legal inquiries should be directed to Lori Cora at (206) 553-1115, or cora.lori@epa.gov.

Sincerely,

Chip Humphrey
Remedial Project Manager

Kristine Koch
Remedial Project Manager

cc: Mr. Jim Anderson
Oregon Department of Environmental Quality

Mr. Rob Neely
National Oceanic and Atmospheric Administration

Mr. Ted Buerger
U.S. Fish and Wildlife Service

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Ms. Rose Longoria
Confederated Tribes and Bands of the Yakama Nation

Mr. Michael Karnosh
Confederated Tribes of the Grand Ronde Community of Oregon

Mr. Tom Downey
Confederated Tribes of the Siletz Indians

Mr. Audie Huber
Confederated Tribes of the Umatilla Indian Reservation

Ms. Erin Madden
Nez Perce Tribe

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