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Columbia River Component Risk Assessment

Volume II: Baseline Human Health Risk Assessment

United States
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Columbia River Component Risk Assessment

Volume II:
Baseline Human Health Risk Assessment

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United States Department of Energy

P.O. Box 550, Richland, Washington 99352

EXECUTIVE SUMMARY

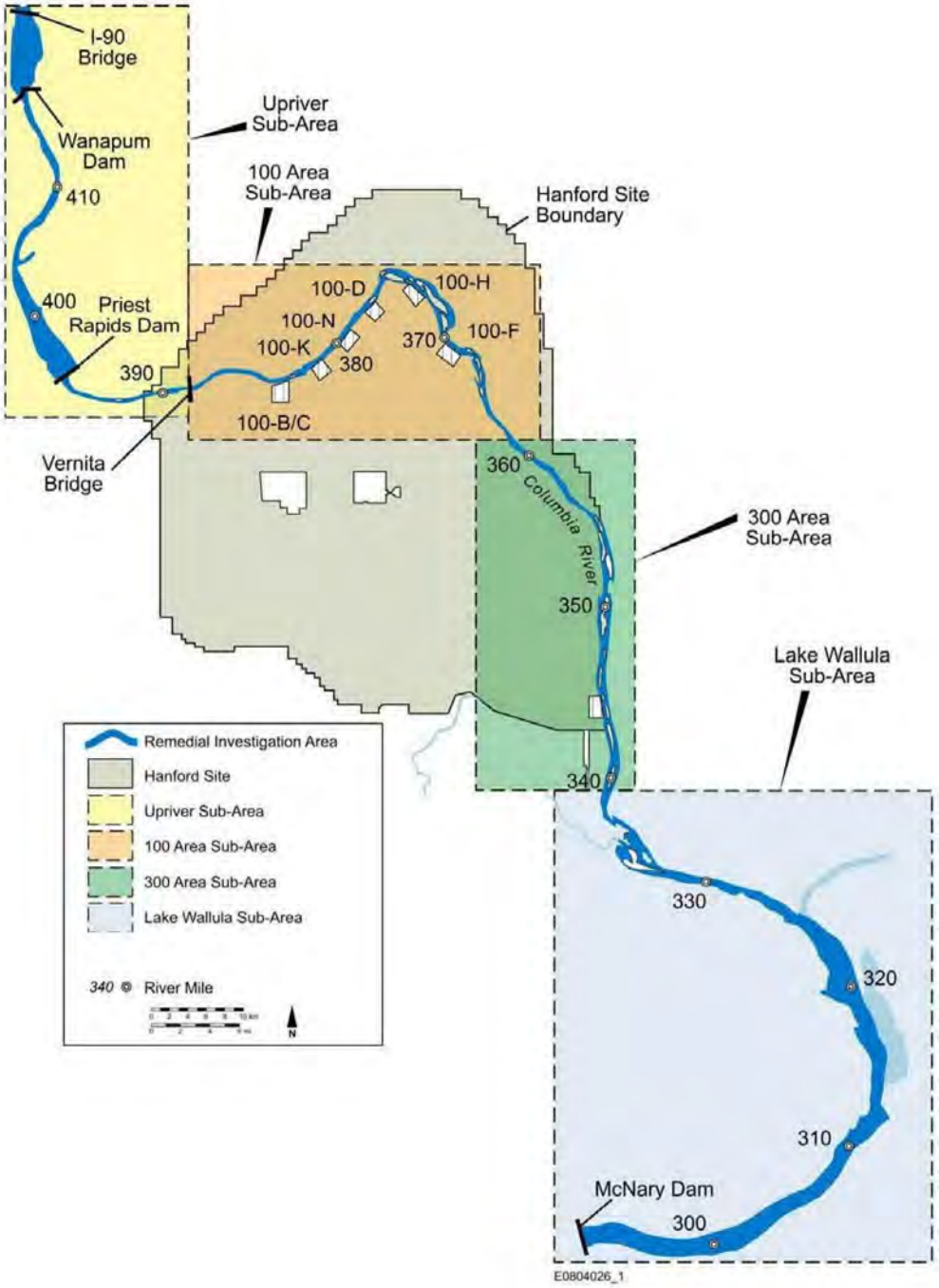
This document presents the methodology and results of a comprehensive human health risk assessment (HHRA) of the surface water, sediment, island soils, and fish of the Columbia River adjacent to and downriver of the Hanford Site in Benton County, Washington. The study was conducted to obtain information about the potential for Hanford Site-related contaminants to affect the health of individuals who use the Columbia River for fishing, recreation, or other purposes. This information will be used, along with the findings from a complementary ecological risk assessment, to support cleanup decisions regarding the Hanford Site that will be protective of human health and the environment.

SITE DESCRIPTION AND HISTORY

The Columbia River stretches 2,000 km (1,243 mi) from the Canadian province of British Columbia through the United States' Washington State, forming much of the border between Washington and Oregon, before emptying into the Pacific Ocean. Measured by the volume of its flow, the Columbia River is the largest river flowing into the Pacific from North America and is the fourth largest river in the United States. In south-central Washington State, the river flows through the U.S. Department of Energy's (DOE's) Hanford Site (Figure ES-1). The area known as the Hanford Reach is an 82-km (51-mi) stretch of the Columbia River that flows unimpeded between Priest Rapids Dam to the head of Lake Wallula upstream of McNary Dam.

The Hanford Site is a 1,517-km^2 (586-mi^2) federal facility located within the semiarid shrub-steppe Pasco Basin of the Columbia Plateau in south-central Washington State. (NOTE: For the purposes of this report, the Hanford Site refers to the boundaries of the Hanford Reservation.) It is situated north and west of the cities of Richland, Kennewick, and Pasco.

Figure ES-1. Columbia River Study Areas.



The Hanford Site became a federal facility in 1943 when the U.S. Government took possession of the land to produce weapons-grade plutonium during World War II. During Hanford Site operations, liquid effluents from plutonium production reactors were discharged directly to the Columbia River, and unplanned overland flows from retention ponds and basins occasionally occurred. In addition, plumes of contaminated groundwater developed in portions of the Hanford Site as a result of the practice of discharging waste waters to the soil column and subsequent migration through the soil. Some of these contaminated groundwater plumes have reached the Columbia River, discharging in seasonal springs along the shoreline and upwelling through the river bottom.

Hanford Site production activities continued until the late 1980s, when the mission focus changed to cleaning up the radioactive and hazardous wastes that had been generated during the previous decades. In 1989, areas of the Hanford Site were placed on the National Priorities List under the authority of the *Comprehensive Environmental Response, Compensation, and Liability Act of 1980* (CERCLA). Placement on the National Priorities List initiated the CERCLA process that would result in the cleanup of contaminated areas.

A primary objective of the Hanford Site cleanup mission is protection of the Columbia River, through remediation of contaminated soil and groundwater that resulted from its production mission. These remedial actions were initiated in 1994 and continue today, with an emphasis on activities in the "River Corridor," a 570-km^2 (220-mi^2) portion of the Hanford Site that includes the former plutonium production reactors in the 100 Area and research and development facilities in the 300 Area.

This HHRA focuses on the Columbia River itself, which contains residue from historical activities at the Hanford Site as well as current upriver and non-Hanford Site sources. The Columbia River is not a part of the Hanford Site, but because it is a potentially affected area, it is being investigated using the same CERCLA process and guidance. The general approach for the entire HHRA was described in DOE/RL-2008-11, *Remedial Investigation Work Plan for*

Hanford Site Releases to the Columbia River (RI Work Plan). This study follows the approach outlined in that work plan.

PURPOSE AND SCOPE

The purpose of this HHRA is to evaluate whether chemical and radiological contaminants in various environmental media in the Hanford Site Study Area are present at concentrations that may pose a potential health risk to individuals (referred to in this report as "human receptors") that visit the shoreline of the Columbia River and its numerous islands. The HHRA identifies the chemical and radiological contaminants present in river media (i.e., sediment, surface water, fish tissue, and island soil); identifies both current and potential future human receptors who may encounter these contaminants through various activities; and characterizes noncancer hazards and cancer risks associated with exposure to contaminants in these media.

Estimation of risk is accomplished through use of standard risk assessment equations that reflect the many different ways that people may be exposed to contaminants in and around the river. These equations take into account both physical characteristics (such as body weight and daily ingestion rates) as well as the different ways in which individuals use the river (for fishing or swimming, for example) to estimate whether individuals may be exposed to contaminants at levels that may have adverse effects on health. The potential for effects are estimated under both central tendency (or "average") and upper-bound (comparable to "worst-case") exposure conditions. The exposure inputs used in the equations and supporting toxicity information incorporate a number of conservative safety factors to account for the uncertainty associated with extrapolating from animal studies to human effects, variability within the human population, as well as other necessary assumptions.

The ultimate objective of the HHRA is to provide a conservative assessment of whether people who use the Hanford Site Study Area portion of the Columbia River for fishing, recreating, or other purposes have the potential to experience adverse health effects under current or reasonably foreseeable river-use scenarios. Risk managers will use the results from this baseline

HHRA in conjunction with other information to determine whether cleanup decisions are required for contamination that exists in or along the Columbia River as a result of historical operations at the Hanford Site.

Integration with Other Hanford Site Risk Assessments and Studies

The DOE, which retains responsibility for the Hanford Site, is currently in the process of conducting remedial investigation (RI) and cleanup activities at the Hanford Site in accordance with the requirements and guidelines of the CERCLA program. This HHRA is being completed in general accordance with the RI Work Plan (DOE/RL-2008-11), which was developed by the Tri-Parties (i.e., U.S. Environmental Protection Agency [EPA], Washington State Department of Ecology [Ecology], and DOE). The results of this risk assessment, in addition to the RI, are important to other Hanford Site cleanup activities in the River Corridor.

Concurrent with the HHRA is the River Corridor Baseline Risk Assessment (RCBRA) (DOE/RL-2007-21, *River Corridor Baseline Risk Assessment, Volume II: Human Health Risk Assessment*) that presents a comprehensive human HHRA for the right[1]-bank source areas along the Hanford Reach. The RCBRA evaluated recreational, industrial, residential, Confederated Tribes of the Umatilla Indian Reservation (CTUIR) and Yakama Nation subsistence living scenarios, and nonresidential Tribal scenarios involving exposure to various Hanford Site media, including soil and groundwater in upland portions of the Hanford Site, and sediments, surface water, and fish along the near-shore areas within the Hanford Reach of the Columbia River. The results of the RCBRA have been reviewed and considered in conjunction with development of the HHRA. Whereas the RCBRA focused on the right bank of the Columbia River, this HHRA evaluated risks from "bank-to-bank" in the Hanford Reach and downstream Lake Wallula, characterizing risk in areas not previously addressed under the RCBRA. The quantitative HHRA was conducted in accordance with EPA Superfund risk assessment guidelines presented in EPA/540/1-89/002, *Risk Assessment Guidance for Superfund, Volume 1,*

[1] Within this report, reference is frequently made to different sides of the river. By convention, all lateral references are made looking downriver. Thus, "right side" of the river or an island refers to the right shoreline, looking downstream; "left side" of the river or an island refers to the left side, looking downstream.

Human Health Evaluation Manual (Part A) (Interim Final), as well as other EPA risk guidance. The following sections summarize the methodology and key outcomes of this HHRA, describing the study components (such as the area of study, selection of contaminants of potential concern, and exposure scenarios), the risk characterization, and the uncertainties associated with estimating hazard and risk.

STUDY COMPONENTS

The components, data, and structure of the HHRA are described below.

Area of Study. For purposes of statistical evaluation and assessment of surface water, sediment, island soils, and fish, the area of investigation within the Columbia River was divided into four distinct but contiguous sub-areas. As described in the RI Work Plan (DOE/RL-2008-11), the boundaries of the sub-areas downriver of Priest Rapids Dam were determined based on spatial distribution of contaminant concentrations observed in surface water and sediment relative to the various sources of contamination from the Hanford Site. The four sub-areas are as follows:

- Upriver Sub-Area (river mile [RM] 420 through RM 388)
- 100 Area Sub-Area (RM 387 through RM 366)
- 300 Area Sub-Area (RM 365 through RM 340)
- Lake Wallula Sub-Area (RM 339 through RM 292).

Figure ES-1 shows these four sub-areas in relation to the Hanford Site. The portion of the study area that is the focus of this HHRA extends from just downstream of Vernita Bridge (RM 388) to McNary Dam, a distance of approximately 154 km (96 mi). This stretch of river is referred to as the "Hanford Site Study Area." Within this area, the lateral area evaluated extends shore to shore (ordinary high water mark to ordinary high water mark).

Analytical Results. The data used for the risk assessment were drawn from a wide variety of sources, reflecting the extensive monitoring and assessment historically associated with the

Columbia River and the Hanford Site. The final data set used for this HHRA is composed of both data collected during the Columbia River RI, which was conducted between 2008 and 2010 specifically to support the risk assessments, and "historical" data, which were collected as part of other studies prior to 2008. Remedial investigation data were described in detail in WCH-398, *Data Summary Report for the Remedial Investigation of Hanford Site Releases to the Columbia River, Hanford Site, Washington*. Historical data were obtained from a variety of sources and were screened to exclude data from outside the geographical or lateral boundaries of the RI study area.

Data from the following date ranges were considered:

Medium	Data Set Range
Surface water	2000 – 2010
Sediment	2000 – 2010
Island soil	2008 – 2010
Fish tissue	2008 – 2010

No island soil data prior to the 2008 to 2010 RI were available, so the RI data form the basis for the soils data set. Although pre-2008 fish tissue data are available, only the RI fish data set was specifically designed to support the HHRA and provided a consistent sampling and analysis approach among species, tissue types, and analytes. Therefore, only RI fish tissue results were included in the HHRA.

Selection of Contaminants of Potential Concern. Contaminants of potential concern (COPCs) are the chemicals and radionuclides that were selected for quantitative assessment in this HHRA. The COPCs are selected from among the analytes detected in each environmental medium, using a method that generally follows the approach described in the RI Work Plan (DOE/RL-2008-11). This approach includes the evaluation of detection frequency, concentration relative to risk-based benchmarks, essential nutrient status, and whether the contaminant is considered to be a known Hanford Site-related contaminant in soil or groundwater. The COPC selection step also includes a process that characterizes the selected COPCs based on a statistical comparison of Hanford Site data to data from reference locations, to identify COPCs that are present in the Hanford Site Study

Area (i.e., 100 Area, 300 Area, and Lake Wallula Sub-Areas) at concentrations inconsistent with or statistically higher than those in reference locations, as described further below.

There are a number of sources unrelated to the Hanford Site that may potentially release contaminants to the Columbia River and therefore contribute to cumulative health risk. These sources include upriver mining; worldwide atmospheric testing; naturally occurring elements; and municipal, urban, and agricultural activities. The contribution of these non-Hanford Site sources has been evaluated in this risk assessment, for purposes of supporting risk management decisions. The end result of this COPC statistical evaluation process was a determination of whether a COPC was either "consistent with Reference" (i.e., a Reference COPC) or "not consistent with Reference" (i.e., a Study Area COPC). A Reference COPC is a constituent present in the Hanford Site Study Area at concentrations similar to or lower than those of Reference areas, whereas a Study Area COPC is a constituent that is present at higher concentrations in the Hanford Site Study Area. However, a Study Area COPC may not necessarily be attributed to a specific Hanford Site release; rather, its designation is due solely to its relative concentration in river media.

In general, many of the COPCs identified in river media (particularly heavy metals and metalloids) are present at concentrations consistent with those of Reference areas. Study Area COPCs were mainly found to be select radionuclides and hexavalent chromium in soil and sediment, and volatile organic compounds in surface water.

Risks related to Study Area and Reference COPCs are distinguished in the risk characterization in order to assist with risk management decisions. However, noncancer hazard and cancer risk were evaluated collectively to provide cumulative risk estimates for each exposure scenario across all COPCs.

Exposure Scenarios. The following exposure scenarios were quantitatively evaluated as part of this HHRA. Each scenario reflects different ways in which individuals who access the Hanford Site Study Area might be exposed to COPCs in fish or other river media. These scenarios are

not inclusive of all uses, but rather focus on those that cover a range of exposure to contaminants. These scenarios are summarized below and are also illustrated in Figure ES-2:

- The **Avid Angler** scenario includes both adults and older children who engage in fishing activities, as well as younger children who consume the catch brought home.

- The **Casual User** scenario is an adult or child who uses the river for seasonal recreational purposes (e.g., swimming, wading).

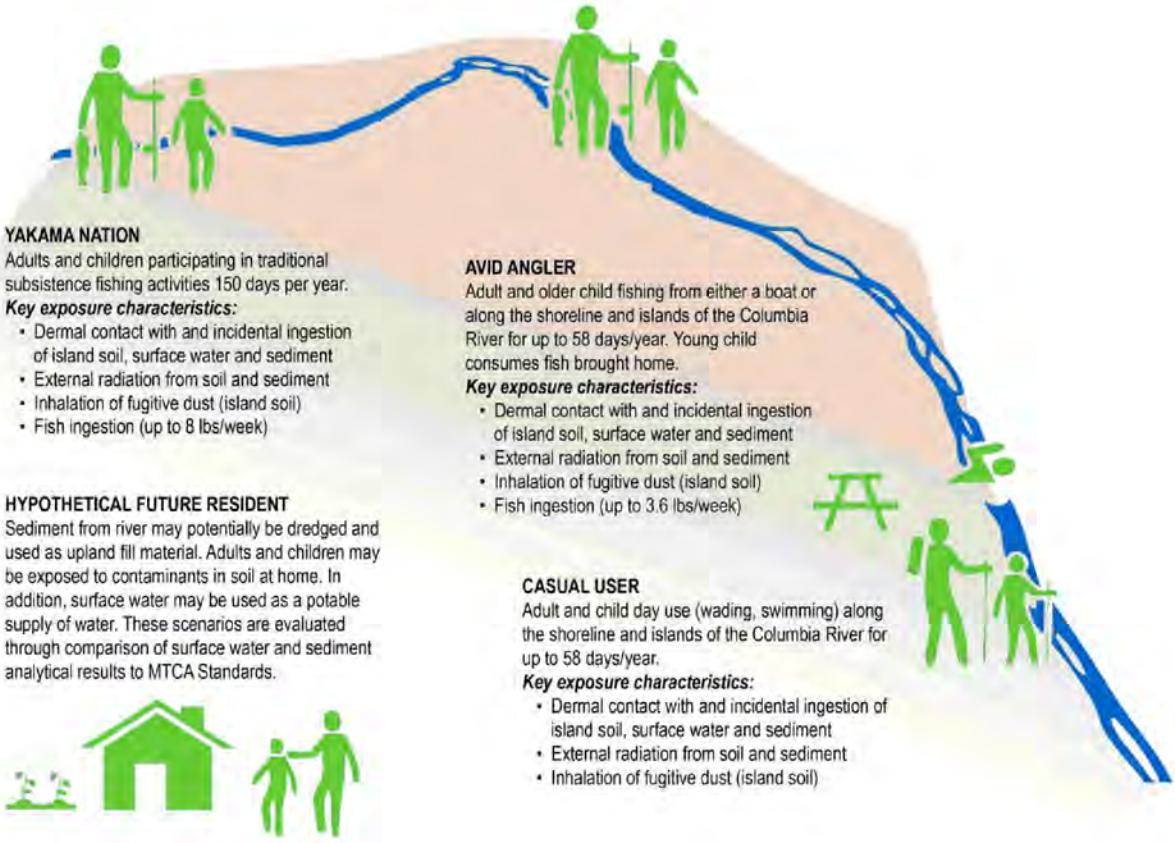
- The **Yakama Nation** scenario includes children and adults of the Yakama Nation who engage in subsistence fishing-related activities in the Columbia River.

- The **Hypothetical Future Residential** scenario involves children and adults who may be routinely exposed to COPCs in dredged sediments that are placed in upland residential areas, as well as to COPCs in surface water that may be used as a potable water supply.

The Casual User and Avid Angler scenarios included evaluation of both average exposures, referred to here as "central tendency" exposures (CTEs), and representative "worst case" exposures, referred to as "reasonable maximum" exposures (RMEs), to provide an estimate of potential health risks under a range of conditions. The Yakama Nation scenario was provided to the DOE by the Yakama Nation[2] and was run in accordance with the RI Work Plan (DOE/RL-2008-11, Rev. 0), using RME exposure point concentrations. The CTUIR scenario, although relevant to the Columbia River Corridor, was not included in this HHRA but instead was evaluated separately in the RCBRA.

[2] *Yakama Nation Exposure Scenario for Hanford Site Risk Assessment, Richland, Washington* (Ridolfi 2007).

Figure ES-2. Summary of Human Receptors Evaluated in the Human Health Risk Assessment.



The Hypothetical Future Residential scenario was evaluated by comparison of sediment and surface water exposure point concentrations to risk-based benchmarks for residential soil and drinking water, respectively[3]. This evaluation, provided in Appendix A, was conducted separately from the quantitative baseline HHRA approach used for the recreational and Tribal scenarios. Note that past dredging projects in the Columbia River conducted by the U.S. Army Corps of Engineers required extensive permitting and evaluations of "beneficial use" of dredged sediments to ensure that the ultimate disposition of dredge spoils would not pose risks

[3] Note that analytical results for untreated surface water samples were used to calculate exposure point concentrations for drinking water for the residential scenario. In actuality, surface water of the Columbia River is processed through a treatment system prior to public distribution as a drinking water supply, which is subject to the federal *Safe Drinking Water Act*.

to future potential receptors/users of such materials. Furthermore, although the Columbia River is currently used as a source of potable water for the City of Richland, filtered and treated water from the river is routinely monitored prior to its distribution and meets federal drinking water standards (maximum contaminant levels), as required by the *Safe Drinking Water Act*.

For each scenario, assumptions are made about a variety of factors: age and body weight of river users, how much they eat or drink, how many years and at what age the use of the river occurs, and similar characteristics. The values used for these characteristics are generally conservative in that they reflect exposure and contaminant levels that are much higher than those that would realistically exist for any individual. This is particularly true in the RME scenarios, in which individuals are assumed to have the highest reasonable exposure characteristics (e.g., for fish consumption, exposure duration) and at the same time are assumed to encounter the highest reasonable contaminant concentrations in fish, surface water, soil. or sediment. Reasonable maximum exposure contaminant concentrations are represented by either the maximum values or by 95th percentile upper confidence limit (UCL) of the mean, which is a statistical value that equals or exceeds the true mean 95% of the time. By evaluating risk under both CTE and RME conditions, the HHRA provides a means to evaluate the uncertainty surrounding risk estimates (EPA/540/1-89/002).

RISK CHARACTERIZATION

Characterization of risk to human health is the estimation of the incidence and severity of adverse effects that may potentially occur in a human population due to exposures to chemicals or radionuclides in fish, water, or other media. Risk is expressed as either a numerical index or as a "probability." Cumulative cancer risk and noncancer hazard estimates, described below, were calculated for each receptor and compared to EPA and Ecology risk management criteria.

- **Cancer Risk:** Cancer risk is calculated for carcinogenic chemicals as well as radionuclides. The potential for carcinogenic health effects is characterized as the incremental lifetime cancer risk (ILCR). The ILCR represents the incremental probability or likelihood of an

individual developing cancer over a lifetime as a result of exposure to a potential carcinogenic COPC. This is considered "incremental" because it is the additional potential risk of developing cancer due to the assumed Study Area exposures, above and beyond the "background" cancer risk (which may be due to genetics, lifestyle choices, sun exposure, etc.). The ILCR is expressed as a single value representing the estimated increase in the chance of getting cancer from Study Area exposures; thus, a one-in-a-million increase in cancer risk is expressed as 1 $x10^{-6}$. The cumulative ILCR for a receptor is compared to EPA's CERCLA target cancer risk range of 10^{-6} to 10^{-4} and the Ecology *Washington Administrative Code* 173-340, "Model Toxics Control Act – Cleanup" (MTCA) risk limit of 1 x 10^{-5}.

- **Noncancer Hazard:** Exposure to contaminants may potentially affect developmental, reproductive, neurobehavioral, and other physiological functions. To account for exposures that a receptor may receive from multiple chemicals and exposure routes, the cumulative noncancer hazard, known as the hazard index (HI), is calculated to estimate potential noncancer effects. Note that the HI conservatively assumes simple additivity across all COPCs, even though the specific toxicological effects of individual COPCs may differ. The cumulative HI for each receptor age group evaluated is then compared with the EPA and MTCA noncancer risk management criterion of 1. If the HI is less than or equal to 1, then it is assumed that the concentrations of chemical COPCs do not pose a risk of harm to human health.

Remedial action is generally not warranted for sites where cumulative cancer risk under an RME condition does not exceed the EPA target risk limits of 10^{-6} to 10^{-4} ILCR or noncancer hazard is below 1 (OSWER Directive 9355.0-30, "Role of the Baseline Risk Assessment in Superfund Remedy Selection Decisions").

Although the HHRA evaluated cumulative risk from all relevant exposure media for each receptor, risks were also discussed separately for abiotic media (surface water, sediment, and soil) and fish tissue to help inform risk management decisions for these media.

Risk Characterization Results

Cumulative risk estimates were calculated for all evaluated receptors, by medium and exposure pathway, and these cumulative noncancer hazard and cancer risk estimates were compared to the relevant EPA and MTCA risk management criteria. Cumulative hazard and risk were first calculated with both Study Area and Reference COPCs combined, and then again with Study Area and Reference risks separated, to help distinguish background effects from potential Study Area risks.

Tables ES-1 and ES-2 present a summary of the range of cumulative noncancer hazard and cancer risk, respectively, for all RME scenarios and across all COPCs. Also presented are cumulative hazard and risk estimates for Study Area COPCs. These tables also provide the EPA and MTCA target noncancer hazard threshold and ILCR risk limits.

Table ES-1. Summary of Noncancer Hazard Indices for Reasonable Maximum Exposure Scenarios.

		Hazard index [a]		
Endpoint	**Exposure Media**	**Casual User**	**Avid Angler**	**Yakama Nation**
Noncancer Hazard	Abiotic - All COPCs [b]	0.2 to 0.7	0.06 to 0.2	1 to 3
	Fish – All COPCs	Not applicable	97 to 146	675 to 1066
	Cumulative hazard index - all COPCs	0.2 to 0.8	97 to 146	676 to 1069
	Cumulative hazard index - Study Area COPCs	0.001 to 0.04	0.6 to 8	6 to 57
EPA and MTCA Target Hazard Index		1	1	1

[a] Ranges for cumulative hazard index reflect risks across the three sub-areas (100 Area, 300 Area, and Lake Wallula).
[b] Includes sediment, island soil, and surface water.

COPC = contaminant of potential concern
EPA = U.S. Environmental Protection Agency
MTCA = Model Toxics Control Act
Shading = exceedance of target hazard index

Table ES-2. Summary of Cumulative Cancer Risks for Reasonable Maximum Exposure Scenarios.

		Incremental lifetime cancer risk [a]		
Endpoint	**Exposure Media**	**Casual User**	**Avid Angler**	**Yakama Nation**
Cancer Risk	Abiotic - All COPCs [b]	7×10^{-6} to 1×10^{-5}	6×10^{-6} to 1×10^{-5}	5×10^{-5} to 1×10^{-4}
	Fish – All COPCs	Not applicable	5×10^{-3} to 6×10^{-3}	2×10^{-2} to 3×10^{-2}
	Cumulative ILCR - all COPCs	7×10^{-6} to 1×10^{-5}	5×10^{-3} to 6×10^{-3}	2×10^{-2} to 3×10^{-2}
	Cumulative ILCR - Study Area COPCs	3×10^{-6} to 4×10^{-6}	3×10^{-6} to 4×10^{-5}	1×10^{-4} to 2×10^{-4}
EPA Target ILCR Range		10^{-6} to 10^{-4}	10^{-6} to 10^{-4}	10^{-6} to 10^{-4}
MTCA Target ILCR		1×10^{-5}	1×10^{-5}	1×10^{-5}

[a] Ranges for ILCR reflect cumulative cancer risk across the three sub-areas (100 Area, 300 Area, and Lake Wallula), for both chemical and radionuclide COPCs.
[b] Includes sediment, island soil, and surface water.

COPC = contaminant of potential concern
EPA = U.S. Environmental Protection Agency
ILCR = incremental lifetime cancer risk
MTCA = Model Toxics Control Act
Shading = exceedance target ILCR

Cumulative noncancer hazards and cancer risk for the Casual User RME scenario did not exceed EPA or MTCA risk management criteria. However, cumulative hazard and risk for the Avid Angler and Yakama Nation scenarios did exceed risk management criteria, primarily due to the fish ingestion pathway.

Results for all of the individual scenarios are discussed in more detail in the following paragraphs.

- **Casual User:** The Casual User scenario evaluated a child and adult who use the Columbia River for recreational purposes such as swimming or wading and therefore may be exposed to COPCs in surface water, sediment, and island soil. For both CTE and RME scenarios and at all exposure points, the following results were obtained:

  – The cumulative noncancer HI did not exceed the EPA and MTCA threshold of 1.

– The cumulative ILCR for both chemical and radionuclide COPCs fell within the EPA cancer risk range of 10^{-6} to 10^{-4} and below or at the MTCA risk limit of 1 x 10^{-5}.

Most of the calculated noncancer hazard and cancer risk was attributable to Reference COPCs (primarily arsenic) in sediment.

- **Avid Angler:** The Avid Angler scenario assumed that a youth and adult use the Columbia River primarily for recreational fishing and wading and that fish that were caught were brought home and consumed by all age groups (child, youth, and adult). Avid Anglers are assumed to be exposed to contaminants through fish consumption, plus incidental contact with surface water, sediment, and island soils while fishing. For both CTE and RME scenarios, and for all exposure pathways and exposure points, results can be summarized as follows:

  – The cumulative noncancer hazard index exceeded the EPA threshold of 1.

  – The cumulative ILCR for both chemical and radiological COPCs exceeded the MTCA cumulative risk limit of 1 x 10-5 as well as the upper end of the EPA cancer risk range of 10^{-4}.

  At all exposure points, noncancer hazard and cancer risk for the Avid Angler were almost exclusively attributed to the fish consumption pathway, which constituted more than 99% of the total risk. Pathways for abiotic media (i.e., contact with sediment, surface water, and soil) contributed overall to a relatively minor amount of cumulative risk, and calculated risks for these abiotic pathways were within or below risk management criteria.

  The fish consumption pathway was evaluated for the Avid Angler scenario using two separate approaches. In the first approach, risk was quantified assuming a receptor consumed a varied diet consisting of all six fish species evaluated (bass, carp, sturgeon, sucker, walleye, and whitefish). In a second approach, risk was quantified for each

individual fish species. Although the concentrations of COPCs and, hence, estimated hazard and risk, varied among the different species, the relative magnitude of risk remained similar among all six fish species. The cumulative HI ranged from 58 in bass to 176 in carp. The cumulative ILCR ranged from 2 x 10^{-3} in bass to 8 x 10^{-3} in carp.

Consumption of any species of fish resulted in excess hazard and cancer risk up to almost two orders of magnitude above the upper end of EPA risk management criteria (i.e., 10^{-4}). The COPCs responsible for most of the calculated risk in fish tissue consisted of polychlorinated biphenyls (PCBs), chlorinated pesticides, cobalt, mercury and other metals, and carbon-14. Carbon-14 was detected in only carp, whitefish, and sucker, whereas other risk drivers were prevalent across all species. Approximately 50% to 80% of the cumulative cancer risk is related to PCBs alone, with the highest PCB-associated ILCR in the 100 Area Sub-Area. Study Area COPCs in fish tissue varied depending on sub-area (when data from all species were combined) and on individual fish species, but included PCBs, carbon-14, mercury, cobalt, beta-hexachlorocyclohexane (beta-HCH), and lithium. In many of the fish species and across exposure points, however, these risk drivers are also classified as Reference COPCs.

- **Yakama Nation:** Similar to the Avid Angler scenario, the Yakama Nation scenario evaluated fishing-related exposures where this receptor may be exposed to COPCs in surface water, sediment, and soil through wading, and who will catch and consume fish (all species) from the Columbia River. Because this scenario reflects subsistence fishing, exposures are assumed to be higher than those of the Avid Angler. Results for this scenario indicate the following:

  - The cumulative noncancer hazard index **exceeded** the EPA threshold of 1.

  - The cumulative ILCR for both chemical and radionuclide COPCs exceeded the MTCA cumulative risk limit of 1 x 10-5 and the upper end of the EPA cancer risk range of 10^{-4}.

Cumulative hazard and risks for the Yakama Nation scenario were also dominated by the fish ingestion pathway, with Reference COPCs (mainly PCBs) accounting for nearly all of the risk in many of the scenarios evaluated. Polychlorinated biphenylss, pesticides, cobalt, mercury, and carbon-14 were the source of most of the risk associated with fish tissue. Cumulative cancer risks from abiotic exposure pathways, exclusive of fish ingestion, were within the EPA cancer risk range, but noncancer risks were above the EPA target hazard index of 1, primarily due to arsenic, iron, and thallium (which are all Reference COPCs) in sediment and arsenic (which is a Study Area COPC) in island soil.

Although the cumulative cancer risk attributed to abiotic media was within the EPA cancer risk range of 10^{-6} to 10^{-4} within the Hanford Site Study Area, it was above the MTCA cumulative risk limit at the 300-B exposure point, mainly due to arsenic, europium-152, and cobalt-60 in island soil on Johnson Island. These radionuclides, as well as europium-154, are also Study Area risk drivers in sediment.

- **Hypothetical Future Residential Scenario (Screening-Level Assessment):** At the request of Ecology, sediments within Lake Wallula shipping channels that may potentially be dredged in the future were evaluated with respect to residential soil screening criteria, assuming that dredged sediments could be placed in upland areas. Additionally, surface water exposure point concentrations were compared to federal drinking water standards and human health risk-based screening levels for surface water. This screening-level assessment was completed as a separate evaluation distinct from the baseline HHRA, and the full results are provided as Appendix A. Unlike the river exposure scenarios evaluated for other receptors (Yakama Nation, Casual User, and Avid Angler), the residential scenario is hypothetical (because it integrated assumptions even less likely to occur beyond those of other scenarios) since dredging and dredge spoil disposal activities would be subject to various U.S. Army Corp of Engineer and State regulations and would require further assessment prior to disposal and/or reuse in upland areas.

The results of the screening-level comparison are, in general, consistent with the findings of the quantitative risk assessment. The COPCs that contributed to the majority of risk, as identified for the other exposure scenarios, were often the COPCs that exceeded residential soil benchmarks. The COPCs in surface water did not exceed federal drinking water standards.

In general, the abiotic media results from the risk characterization indicate that the risks related to exposure to surface water, sediment, and island soil are generally within or below EPA risk management criteria and very small relative to that from the fish ingestion pathway. For abiotic media, Reference COPCs account for the majority of noncancer hazard and, in most cases, chemical cancer risk in all sub-areas. Arsenic, a Reference COPC in sediment at all of the exposure points, accounted for over half of the cumulative risk. Risks from island soil exposures were relatively minor compared to risks from other abiotic media. Arsenic is a Study Area risk driver in island soil.

Of the radionuclides in abiotic media, cobalt-60, europium-154 and europium-152, which are both Study Area COPCs in soil and sediment, and cesium-137, a Reference COPC, account for the majority of radiation cancer risk.

Polychlorinated biphenyls, chlorinated pesticides, cobalt, arsenic, mercury, and carbon-14 are the primary COPCs contributing to cancer risk or noncancer hazards from fish consumption. However, of these, only carbon-14 is a Study Area COPC consistently throughout the Study Area; the other risk drivers are Reference COPCs in the majority of media and exposure points. Carbon-14 is a Study Area COPC throughout all fish species and sub-areas in which it was detected.

Uncertainties in the Risk Assessment

Inherent in all risk assessments are uncertainties associated with key parameters used to estimate risk, including the environmental concentrations, toxicity values, and exposure assumptions used to estimate magnitude of exposure and to quantify health risks. In general, the assumptions used

in the HHRA are intended to be protective of human health. By design, this HHRA has been developed to provide conservative estimates of risk to those who visit or use the Columbia River within the Hanford Site Study Area.

The fish ingestion pathway comprises more than 99% of the cumulative risk for the Avid Angler and Yakama Nation scenarios. Polychlorinated biphenyls, mercury and other metals, and chlorinated pesticides in fish tissue are the primary risk drivers. These types of contaminants are prevalent in fish tissue in many waterbodies due to their widespread historical use, atmospheric deposition and, consequently, high prevalence in abiotic media. Because of this, it is unclear what contribution, if any, Hanford Site releases have had to fish in the Columbia River for these types of constituents. Furthermore, for other risk drivers that were detected infrequently in fish tissue (such as carbon-14), risks were often based on a few individual tissue sample results and do not accurately represent general fish consumption risk across the entire Hanford Site Study Area.

There is also some uncertainty associated with the fish tissue radionuclide results. Carbon-14 was retained as a fish tissue COPC, but other radionuclides that were detected in fish tissue samples were not. It is believed that these sporadic detections represent false-positives. Exclusion of these radionuclides as COPCs may potentially underestimate risk, should these contaminants actually be present. Similarly, hexavalent chromium was sporadically detected in fish tissue. This form of chromium is not expected to be present in biological tissue, due to its biological conversion to its trivalent form once taken up into tissue; this suggests that the hexavalent chromium tissue results may potentially be positively biased. Because hexavalent chromium is rapidly reduced in tissue to trivalent chromium, which is much less toxic than the hexavalent form, the risk from ingestion of fish tissue is expected to be minimal; toxicity from hexavalent chromium is generally associated with direct exposures, such as inhalation of dusts, ingestion of drinking water, and dermal contact (ATSDR, 2000, *Toxicological Profile for Chromium*; Langard and Costa 2007, "Chromium," in *Handbook on the Toxicology of Metals*).

Both CTE and RME risk estimates provide an understanding of the range of potential health risks. However, differences between the exposure parameters used in this HHRA and actual physiological attributes and activity patterns in potentially exposed populations at the Hanford Site introduce some uncertainty in quantifying exposure. Additionally, spatial and temporal variability in COPC concentrations within an environmental medium (particularly in soil and sediment) can be relatively high; therefore, use of environmental data may potentially introduce a low or high bias when estimating exposure.

In light of these uncertainties, it is important to stress that the risks and hazards calculated in this HHRA are *estimated* risks. It must be emphasized that the risks generated in this evaluation are *hypothetical*, not actual, and are by design intended to be conservative (i.e., tend to overestimate actual risks). By using this conservative approach in developing risk estimates, it would be expected that the calculated risk estimates are likely to result in upper-bound estimates of Hanford Site-related risks and hazards. Consequently, these estimates should be used to highlight areas of potential concern and to assist in providing practical risk management information rather than be considered as absolute estimates of health risks.

CONCLUSIONS AND RECOMMENDATIONS

This baseline HHRA provides a comprehensive assessment of potential health risks associated with recreational and Tribal exposures to surface water, sediment, island soils, and fish tissue within the Hanford Site Study Area. Results of the HHRA indicate that:

- Cumulative risks from all COPCs (both Study Area and Reference) estimated for the Casual User scenario, which assumes recreational exposures to surface water, sediment, and island soil, do not exceed either MTCA or EPA risk management criteria (i.e., HI of 1 and ILCR of 10^{-6} to 10^{-4}; 1 x 10^{-5} for MTCA).

- Cumulative risk from all COPCs (both Study Area and Reference) for the Avid Angler and Yakama Nation scenarios exceed EPA and MTCA risk management criteria primarily due

to the fish ingestion pathway. Study Area COPCs in fish tissue that are risk drivers consist of the following:

- Mercury in the 100 and 300 Area Sub-Areas
- Carbon-14 in all three sub-areas in carp, sucker, and whitefish
- Polychlorinated biphenyls, pesticides, cobalt and mercury in various fish species.

However, with the exception of carbon-14, these types of constituents are also prevalent in fish from other portions of the Columbia River and in many areas of the country. Therefore, there is some uncertainty as to whether these Study Area COPCs are related to Hanford Site releases.

- Cumulative risks from Study Area COPCs in abiotic media do not exceed EPA risk management criteria (i.e., HI of 1 and ICLR of 10^{-6} to 10^{-4}), although they do exceed the MTCA cancer risk limit when evaluated under the Yakama Nation scenario. The primary Study Area COPCs that contribute to risk consist of arsenic, europium-152, and cobalt-60 in soil in the 300 Area, and europium-152 and cobalt-60 in sediment throughout the Study Area.

For most exposure points, risk in abiotic media was primarily attributed to Reference COPCs such as arsenic and other metals in sediment, which were distributed heterogeneously throughout all sub-areas. Radionuclide-related cancer risks in abiotic media were attributable to a mix of Study Area and Reference COPCs, with the Lake Wallula Sub-Area containing the highest radiation cancer risk primarily due to the presence of cesium-137 (a Reference COPC) and europium-152 (a Study Area COPC) in sediment.

The River Corridor remedial investigation/feasibility study programs will further evaluate the nature and extent, conceptual site model, and fate and transport of the HHRA COPCs identified here to determine if concentrations (current detected or future predicted) in the river are potentially from current or historical operations associated with the operable unit being evaluated. Based on that assessment, the need for further study or remedial action will be determined.

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ACRONYMS

ABSd	absorption fraction from soil
ABSgi	gastrointestinal absorption fraction
ACOE	U.S. Army Corps of Engineers
ADAF	age-dependent adjustment factor
ADD	average daily dose
ADE	average daily exposure
AF	adherence factor
ATSDR	Agency for Toxic Substance Disease Registry
CD	compact disc
CDDF	chlorinated dibenzo-p-dioxin and dibenzofuran
CERCLA	*Comprehensive Environmental Response, Compensation, and Liability Act of 1980*
CLARC	Cleanup Levels and Risk Calculation (database)
COPC	contaminant of potential concern
CRC	Columbia River Component
CRCRA	Columbia River Component Risk Assessment
CRITFC	Columbia River Inter-Tribal Fish Commission
CSF	cancer slope factor
CSM	conceptual site model
CTE	central tendency exposure
DA_{event}	absorbed dose per event
DCF	dose conversion factor
DDD	dichlorodiphenyldichloroethane
DDE	dichlorodiphenyldichloroethylene
DDT	dichlorodiphenyltrichloroethane
DMA	dimethylarsenic acid
DOE	U.S. Department of Energy
DQO	data quality objective
Ecology	Washington State Department of Ecology
EPA	U.S. Environmental Protection Agency
EPC	exposure point concentration
FA	fraction absorbed
FOD	frequency of detection
FS	feasibility study
HCH	hexachlorocyclohexane
HEAST	Health Effects Assessment Summary Table
HHRA	human health risk assessment
HI	hazard index
HQ	hazard quotient
ICRP	International Commission on Radiological Protection
IDL	instrument detection limit
ILCR	incremental lifetime cancer risk

IRIS	Integrated Risk Information System
Kp	skin permeability coefficient
JMP	JMP® Version 8.0.2
KM	Kaplan-Meier
LADD	lifetime average daily dose
LADE	lifetime average daily exposure
LRL	laboratory reporting limit
MCL	maximum contaminant level
MDA	minimum detectable activity
MDL	method detection limit
MMA	monomethylarsonic acid
MTCA	*Model Toxics Control Act*
NPDES	National Pollutant Discharge Elimination System
NPL	National Priorities List
OCI	other contributing influences
PAH	polycyclic aromatic hydrocarbons
PCB	polychlorinated biphenyl
PNNL	Pacific Northwest National Laboratory
RCBRA	River Corridor Baseline Risk Assessment
RCRA	*Resource Conservation and Recovery Act of 1976*
RfC	reference concentration
RfD	reference dose
RI	remedial investigation
RM	river mile
RME	reasonable maximum exposure
ROD	record of decision
RPD	relative percent difference
SA	surface area
SAI	sampling and analysis instruction
SAP	sampling and analysis plan
SAw	skin surface area
SLERA	screening-level ecological risk assessment
SVOC	semivolatile organic compound
TCDD	tetrachlorodibenzo-p-dioxin
TCDF	tretrachlorodibenzofuran
TCE	trichloroethene
TEDE	total effective dose equivalent
TEF	toxicity equivalency factor
TIAS	total inorganic arsenic
TPH	total petroleum hydrocarbons
UCL	upper confidence limit
UR	unit risk
VOC	volatile organic compound
WAC	*Washington Administrative Code*

1.0 INTRODUCTION

1.1 BACKGROUND

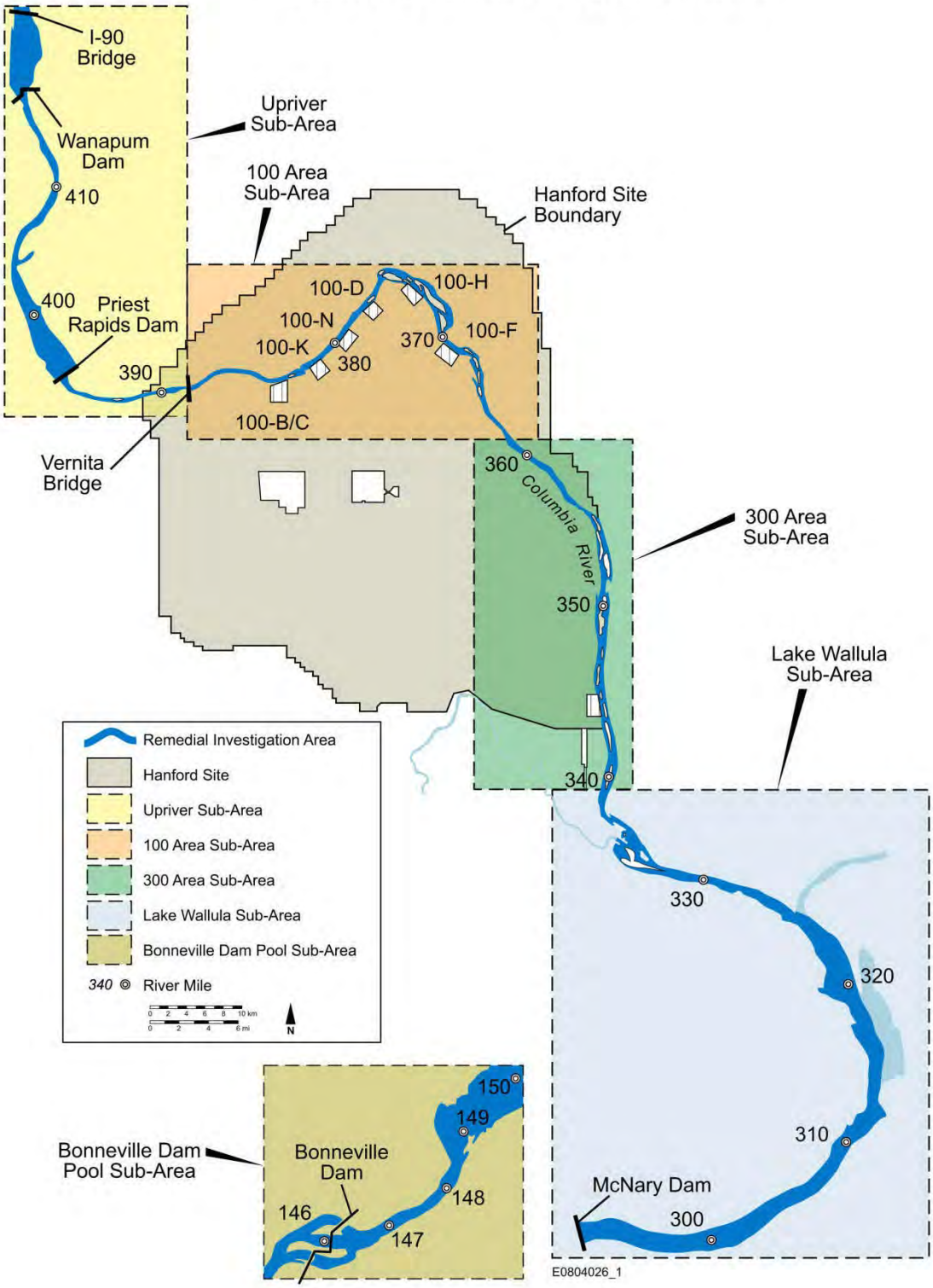
This report presents the results of a baseline human health risk assessment (HHRA) that addresses the potential risk to human health from exposure to surface water, sediment, soil (on in-river islands), and fish tissue potentially impacted by Hanford Site hazardous substance releases to the Columbia River in areas within and downriver of the Hanford Site boundary. Additionally, the HHRA takes into consideration other sources of contamination located upriver of or proximate to the Hanford Site and identifies constituents that are present at concentrations consistent with or inconsistent with those in reference areas that are unlikely to be impacted by past Hanford Site releases. The portion of the river that is the focus of this HHRA extends from just downstream of Vernita Bridge to McNary Dam, a distance of approximately 154 km (96 mi).

The Columbia River stretches 2,000 km (1,243 mi) from the Canadian province of British Columbia through the State of Washington, forming much of the border between Washington and Oregon, before emptying into the Pacific Ocean. Measured by the volume of its flow, the Columbia River is the largest river flowing into the Pacific from North America and is the fourth-largest river in the United States. In south-central Washington State, the river flows through the U.S. Department of Energy's (DOE's) Hanford Site. An area known as the Hanford Reach is a 77-km (48-mi) stretch of the Columbia River that flows unimpeded between Priest Rapids Dam to the head of Lake Wallula upstream of McNary Dam (Figure 1-1). The Hanford Reach is the only free-flowing portion of the river above Bonneville Dam in the United States.

Figure 1-1 includes the Bonneville Dam Sub-Area, although no HHRA analyses were completed for this area. The Bonneville Dam Sub-Area is relevant to the remedial investigation (RI)/feasibility study (FS) project because the Hanford Site operated for a short period of time before McNary Dam was constructed. During that time period, the Bonneville Dam was the first dam downriver of the Hanford Site. This figure includes the Bonneville Dam Sub-Area because of its relevance to the RI/FS project; however, this sub-area was not included in this HHRA because reported radionuclide concentrations in sediment cores collected behind Bonneville Dam were below background concentrations. Bonneville Dam sediment data are presented in WCH-398, *Data Summary Report for the Remedial Investigation of Hanford Site Releases to the Columbia River, Hanford Site, Washington* (Data Summary Report).

The Hanford Site is a 1,517-km^2 (586-mi^2) federal facility located within the semiarid shrub-steppe Pasco Basin of the Columbia Plateau in south-central Washington State and is situated north and west of the cities of Richland, Kennewick, and Pasco. The Hanford Site became a federal facility in 1943 when the U.S. Government took possession of the land to produce weapons-grade plutonium during World War II.

Figure 1-1. Columbia River Study Areas.



During Hanford Site operations, liquid effluents from plutonium production reactors were discharged directly to the Columbia River and unplanned overland flows from retention ponds and basins occasionally occurred. In addition, contaminated groundwater developed in portions of the Hanford Site as a result of the practice of discharging waste waters to the soil column. Some of these contaminated groundwater plumes have reached the Columbia River, discharging in springs along the shoreline and/or upwelling through the river bottom.

Hanford Site production activities continued until the late 1980s, when the mission focus changed to cleaning up the radioactive and hazardous wastes that had been generated during the previous decades. In 1989, portions of the Hanford Site were placed on the National Priorities List (NPL) under the authority of the *Comprehensive Environmental Response, Compensation, and Liability Act of 1980* (CERCLA). Placement on the NPL initiated the CERCLA process that would result in the cleanup of contaminated areas. While the Columbia River is currently not part of the Hanford Site, the river is being investigated under the CERCLA process for consistency with the approach being taken for other Hanford Site operable units.

A primary objective of the Hanford Site cleanup mission is protection of the Columbia River, through remediation of contaminated soil and groundwater that resulted from its production mission. These remedial actions were initiated in 1995 and continue today, with an emphasis on activities in the "River Corridor" because of its proximity to the river and presence of the former production reactors in the 100 Area and research and development facilities in the 300 Area. Current activities in the River Corridor also include performance of a baseline risk assessment of the upland, riparian, and near-shore areas (DOE/RL-2007-21, *River Corridor Baseline Risk Assessment* [RCBRA]).

Within the Columbia River system, surface water, sediment, and fish tissue samples related to potential Hanford Site releases have been collected since the start of Hanford operations. The potential impacts of Hanford Site releases to the Columbia River in areas upstream, within, and downstream of the Hanford Site boundary have been previously investigated as mandated by DOE requirements under the *Atomic Energy Act of 1954*. The current impacts within the Columbia River are now being assessed under CERCLA via the RI activities described in DOE/RL-2008-11, *Remedial Investigation Work Plan for Hanford Site Releases to the Columbia River* (RI Work Plan).

Under the *Resource Conservation and Recovery Act of 1976* (RCRA), *National Environmental Policy Act of 1969*, and CERCLA, and as a requirement of the RI/FS process, DOE is required to assess human and ecological risk via a baseline risk assessment, in order to provide risk managers with an understanding of current and potential future human health and ecological risks posed by a site. This HHRA (Volume II) addresses the human health portion of the Columbia River Component Risk Assessment (CRCRA), complementing the screening-level ecological risk assessment (SLERA; Volume I). Risks for other portions of the Hanford Site within the River Corridor are addressed under the RCBRA (DOE/RL-2007-21). Figures 1-2 and 1-3 depict the areas of the Columbia River that are the focus of this HHRA relative to those areas addressed under the RCBRA.

Figure 1-2. Columbia River Component Risk Assessment Evaluation Area Adjacent to the Hanford Site.

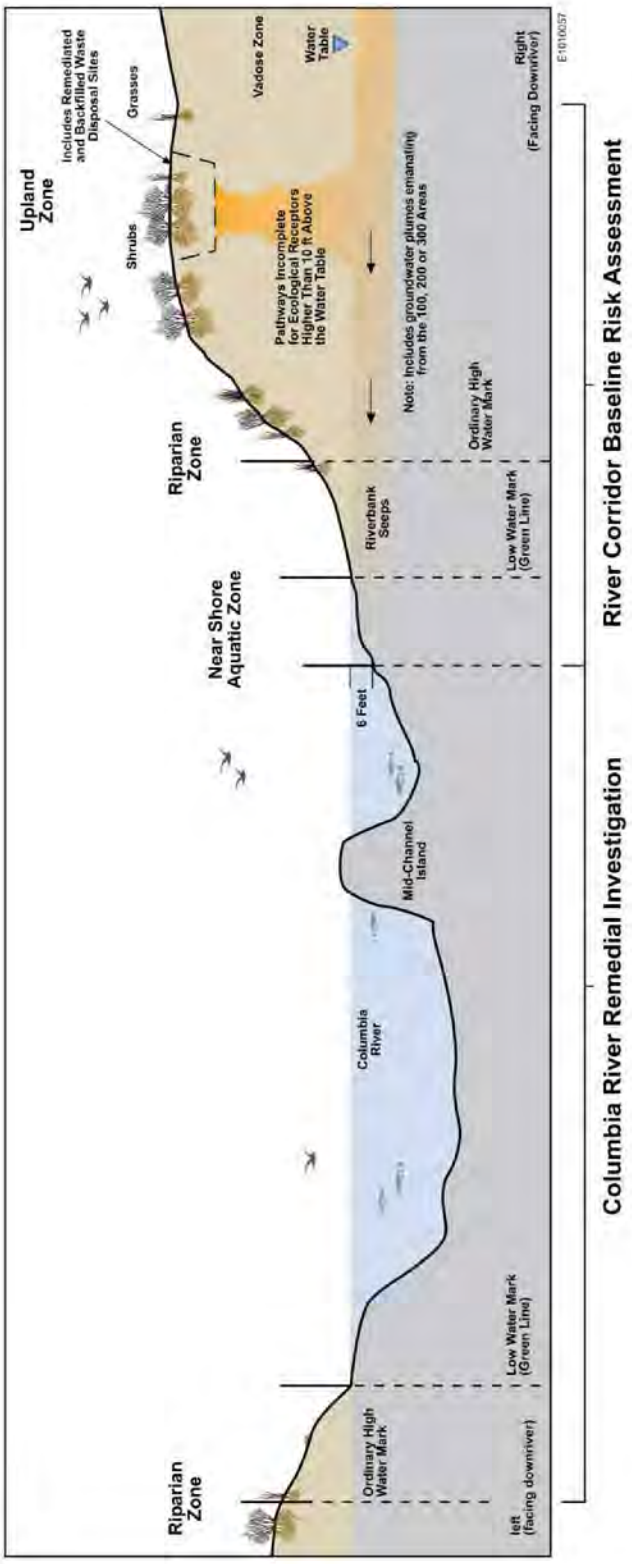
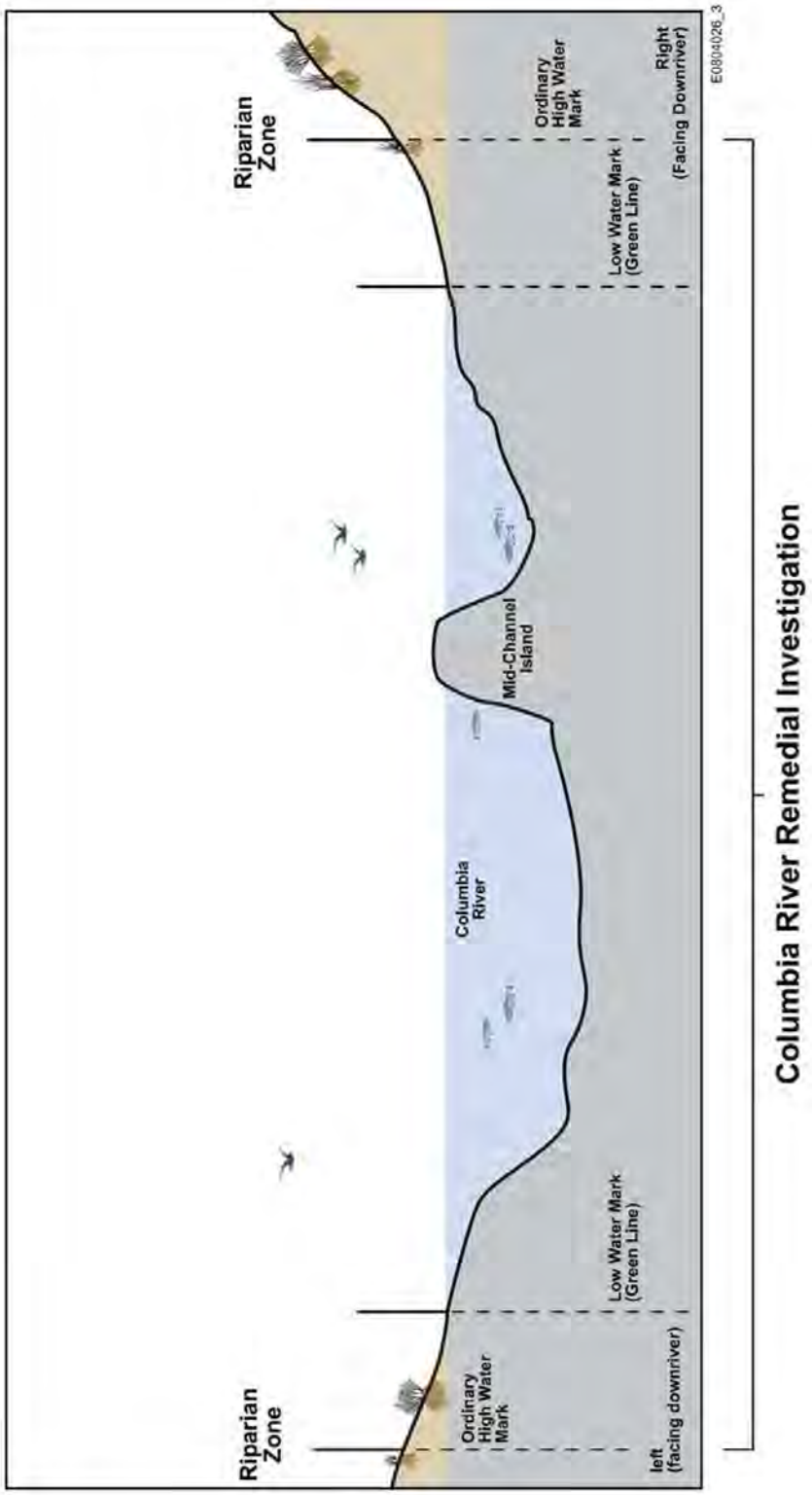


Figure 1-3. Columbia River Component Risk Assessment Evaluation Area Downriver and Upriver of the Hanford Site.



Collectively, the HHRA and SLERA results from the Columbia River Component (CRC) evaluation, along with results from the RCBRA, will be used to support risk management decisions for the River Corridor. Risk managers will use the results from this baseline risk assessment in conjunction with other information from the RI/FS process to support final cleanup decisions, if warranted, that will be protective of human health and the environment. Final risk management decisions applying to all portions of the River Corridor will be identified in proposed plans that will undergo public review and will ultimately be documented in records of decision (RODs).

1.2 PURPOSE

The purpose of this HHRA is to evaluate whether chemical and radiological contaminants in various environmental media in the Columbia River are present at concentrations that may pose a potential health risk to human receptors that frequent the shoreline of the Columbia River and its numerous islands that exist within the river channel. The HHRA identifies the chemical and radiological contaminants present in river media (e.g., sediment, surface water, fish tissue, island soil), identifies both current and potential future human receptors who may encounter these contaminants through various activities, and characterizes noncancer hazards and cancer risks associated with exposure to these media. The results of the HHRA will be used to aid in the decision of whether additional response actions, in terms of either supplemental assessment or remediation, are warranted.

1.3 SCOPE

The HHRA evaluates both current and potential future human exposures to river media in and along the Hanford Reach of the Columbia River. The study area considered in the Columbia River RI consists of the reach of the Columbia River extending from above Wanapum Dam (river mile [RM] 415) to McNary Dam at RM 292 (Figure 1-1). The portion of the river that is the focus of this HHRA extends from just downstream of Vernita Bridge (RM 388) to McNary Dam, a distance of approximately 154 km (96 mi). This stretch of river is herein referred to as the "Hanford Site Study Area" and is the primary focus of the HHRA. The 77-km (48-mi) stretch of river adjacent to the Hanford Site, from RM 388 to Richland at RM 340, is referred to as the Hanford Reach, in accordance with general practice.

For purposes of statistical evaluation and assessment, as well as practicality due to the scale of the study area (spanning 128 RMs), the area of investigation was divided into four distinct but contiguous sub-areas. As described in the RI Work Plan (DOE/RL-2008-11), the boundaries of the sub-areas downriver of Priest Rapids Dam were determined based on spatial distribution of contaminant concentrations observed in surface water and sediment with respect to the various sources of contamination from the Hanford Site. The four sub-areas are as follows:

- Upriver Sub-Area (RM 420 to RM 388)
- 100 Area Sub-Area (RM 387 to RM 366)

- 300 Area Sub-Area (RM 365 to RM 340)
- Lake Wallula Sub-Area (RM 339 to RM 292).

Figure 1-1 shows these four sub-areas in relation to the Hanford Site. Features and characteristics associated with each of these sub-areas are illustrated in Figures 1-4, 1-5, 1-6, and 1-7, respectively. As stated above, Figure 1-1 includes the Bonneville Dam Sub-Area because of its relevance to the RI/FS project; however, this sub-area was not included in this HHRA because reported radionuclide concentrations in sediment cores collected behind Bonneville Dam were below background concentrations (WCH-398).

The Upriver Sub-Area is used as a reference location (i.e., an area unlikely to be impacted by Hanford Site-related releases due to its position upstream). However, contaminants may be present within the Upriver Sub-Area due to other off-site sources (e.g., industrial discharges, naturally occurring geochemical conditions, and agricultural and roadway runoff). Therefore, Upriver is assumed to represent local conditions within the Columbia River, absent the Hanford Site. Other contributing influences also enter the Columbia River within the various sub-areas. Collectively, these are referred to as "other contributing influences" (OCI) areas and are used in the risk assessments as reflections of anthropogenically influenced "reference" concentrations. Upriver and OCI areas are collectively referred to in this report as "reference" areas, or "reference/OCI" areas.

Within the study area for the CRC, the lateral area evaluated adjacent to the Hanford Site differs from the lateral area evaluated upriver and downriver of the Hanford Site. The lateral boundary of the study area adjacent to the Hanford Site on the right shore begins where the RCBRA near-shore investigation stopped. The RCBRA near-shore study area consisted of the right side of the river from the land to a water depth of 2 m (6 ft), as measured at low water. The river's edge at low water is characterized by the presence of the "green line" of algae delineating the permanently inundated portion of the river channel (see Figure 1-2). For the CRC, the lateral boundaries begin on the right shore at the 2-m (6-ft) water depth boundary of the RCBRA near-shore study area and extend to the ordinary high water mark on the left shore as depicted in Figure 1-2.

For areas upriver and downriver of the Hanford Site, the lateral area evaluated extends from right shore to left shore (ordinary high water mark to ordinary high water mark[1]). The lateral boundaries upriver and downriver of the Hanford Site are depicted in Figure 1-3.

[1] From WAC 173-22-030, "the ordinary high water mark on all lakes, streams, and tidal water is that mark that will be found by examining the bed and banks and ascertaining where the presence and action of waters are so common and usual, and so long continued in all ordinary years, as to mark upon the soil a character distinct from that of the abutting upland…"

Figure 1-4. Upriver Sub-Area River Features.

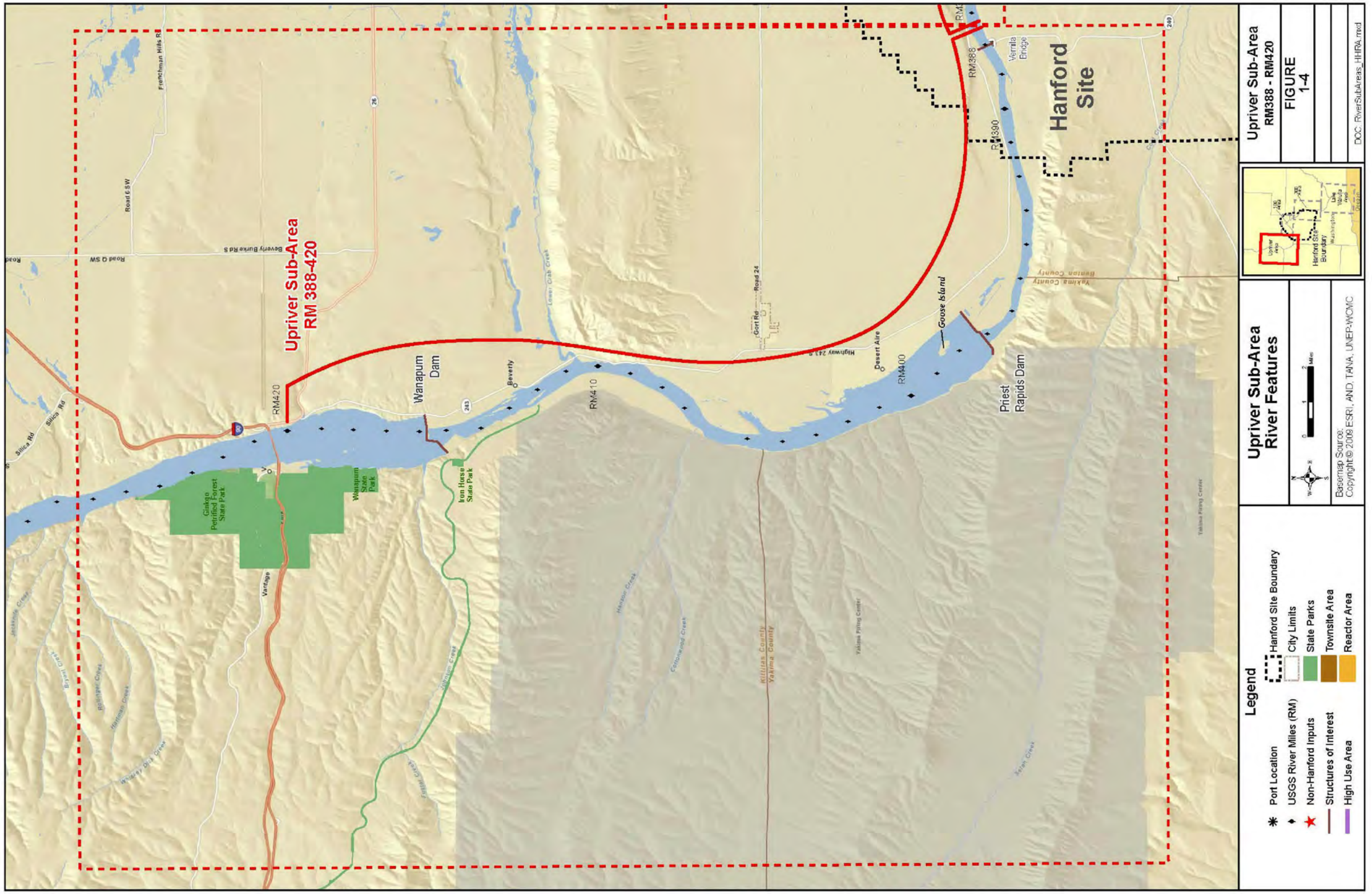


Figure 1-5. 100 Area Sub-Area River Features.

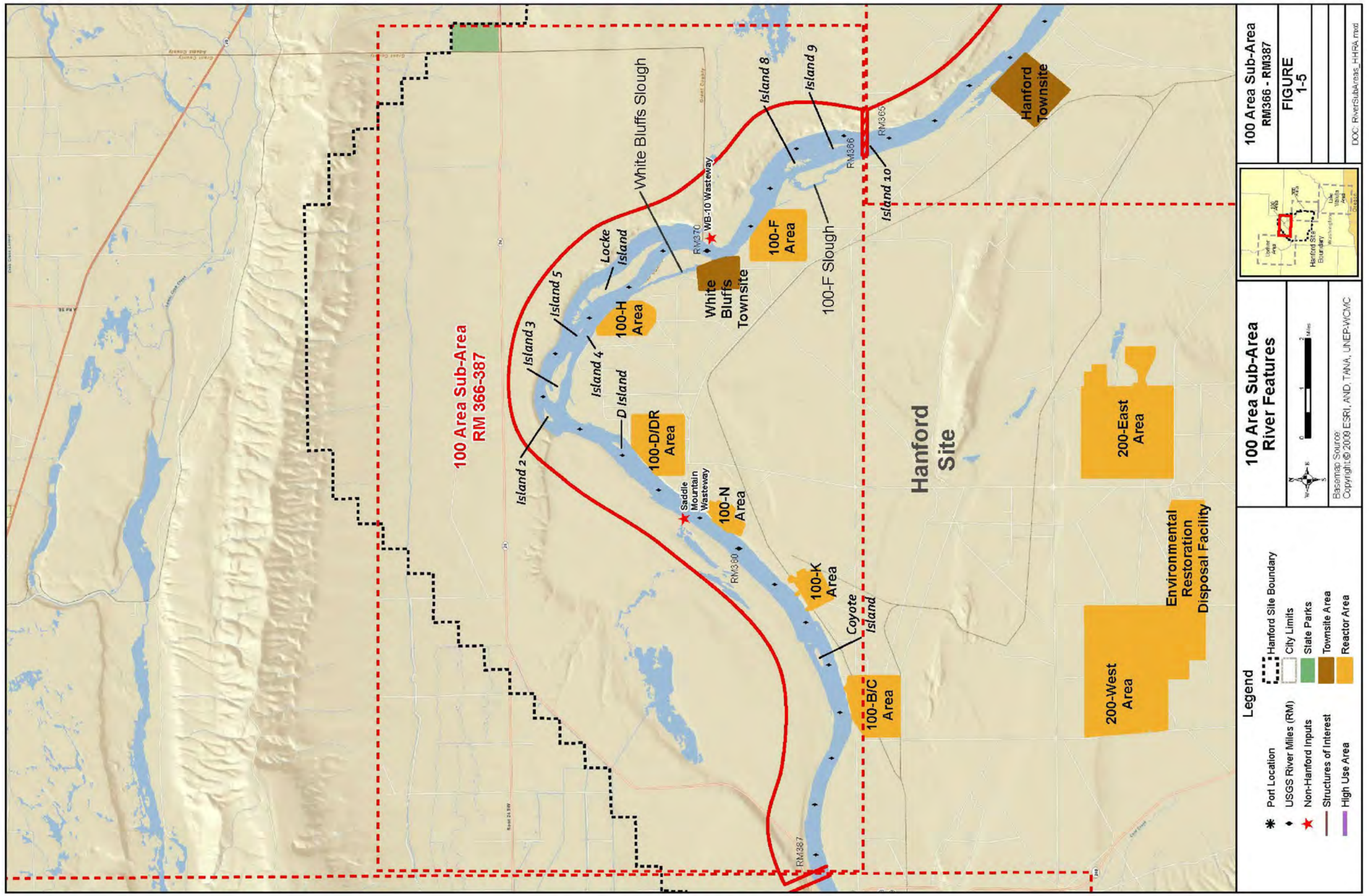


Figure 1-6. 300 Area Sub-Area River Features.

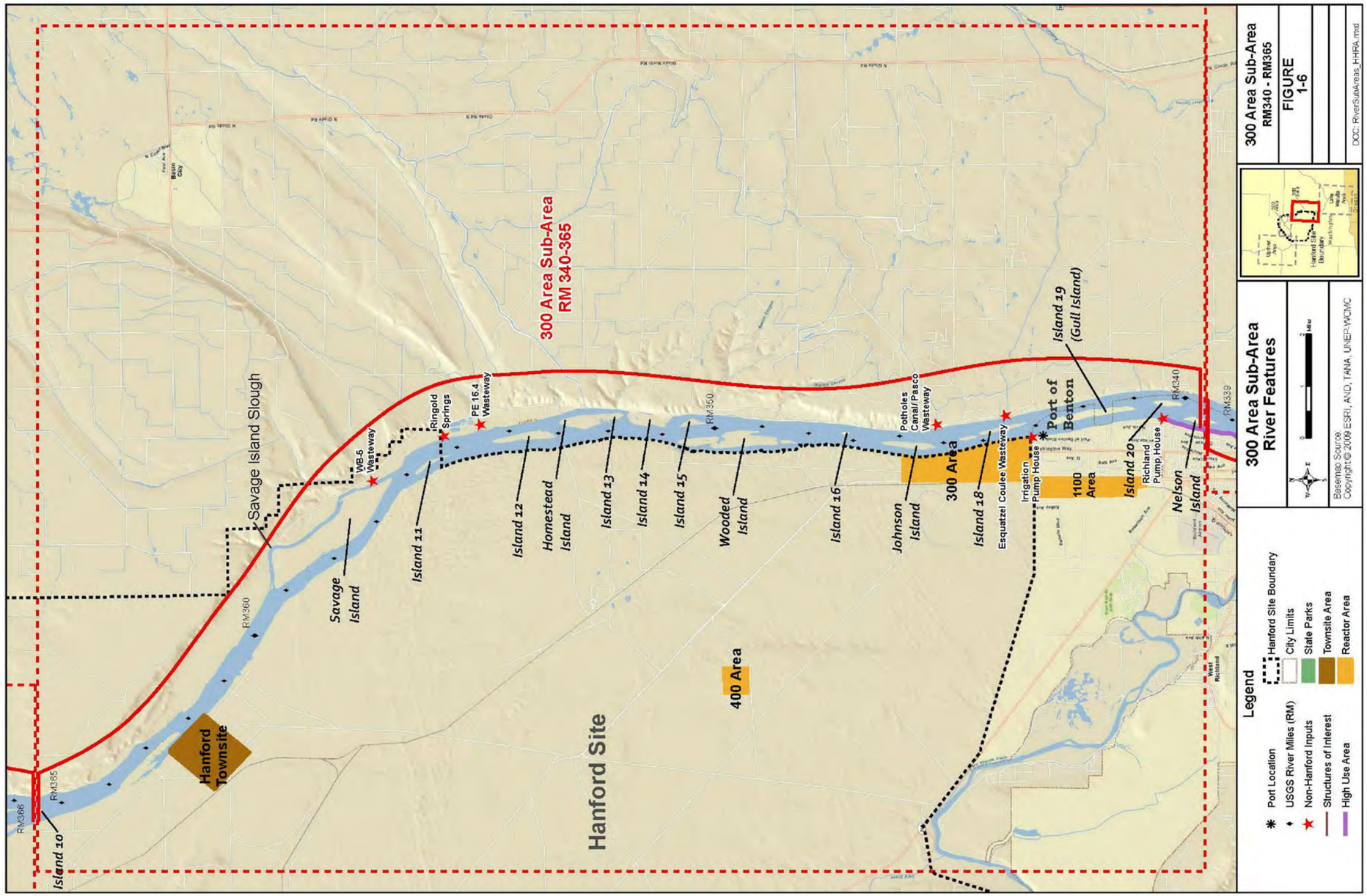
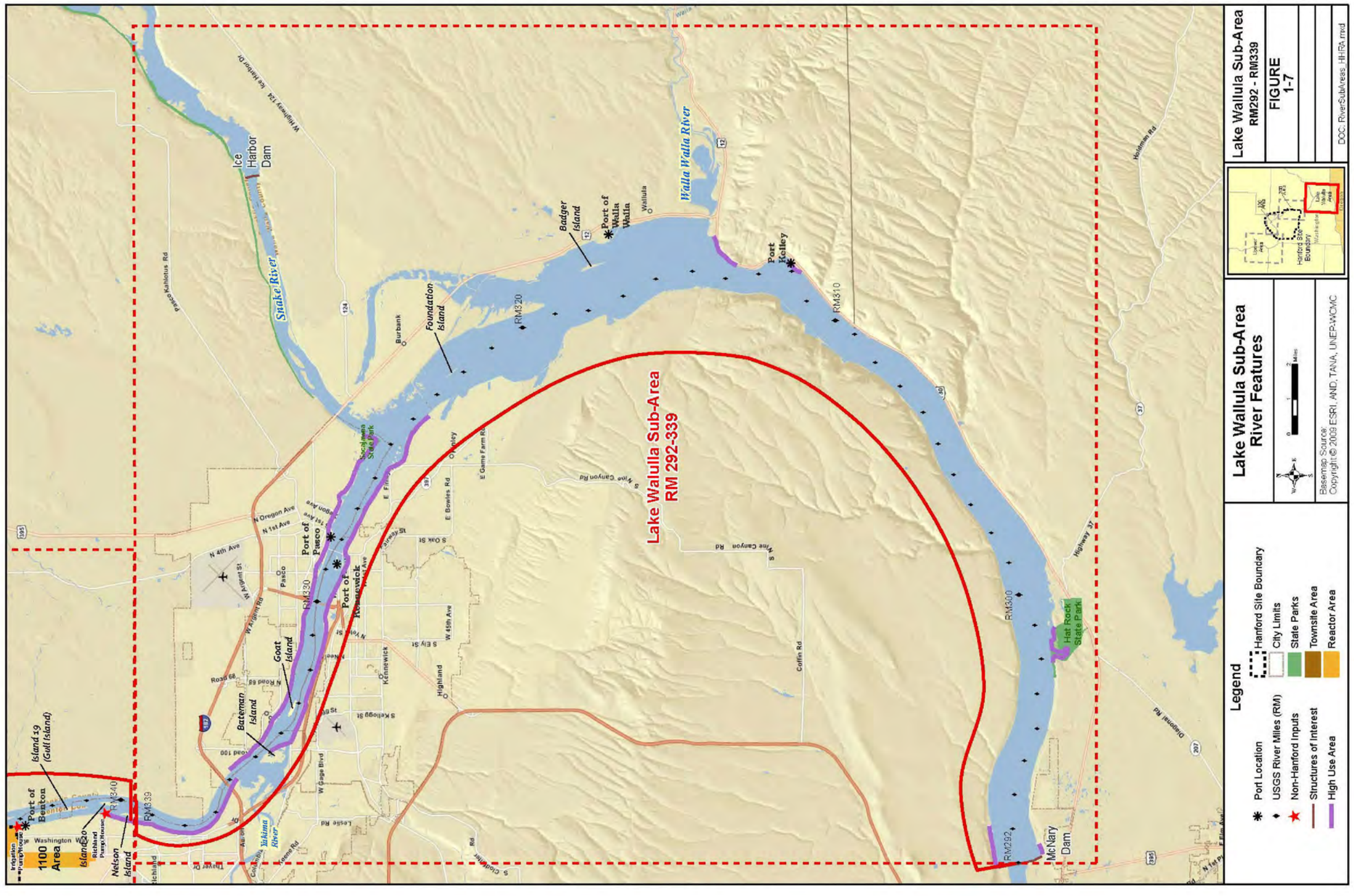


Figure 1-7. Lake Wallula Sub-Area River Features.



Within this report, reference is frequently made to different sides of the river. By convention, all lateral references are made looking downriver. Thus, "right side" of the river or an island refers to the right shoreline, looking downstream; "left side" of the river or an island refers to the left side, looking downstream.

1.4 REQUIREMENTS AND GUIDANCE

This document presents the methodology and results of an HHRA of the surface water, sediment, island soils, and fish tissue of the Columbia River adjacent to and downriver of the Hanford Site in Benton County, Washington. The study was conducted to evaluate the potential for chemical and radiological contaminants to present a health risk to recreational users and Native Americans who visit and live along the Columbia River.

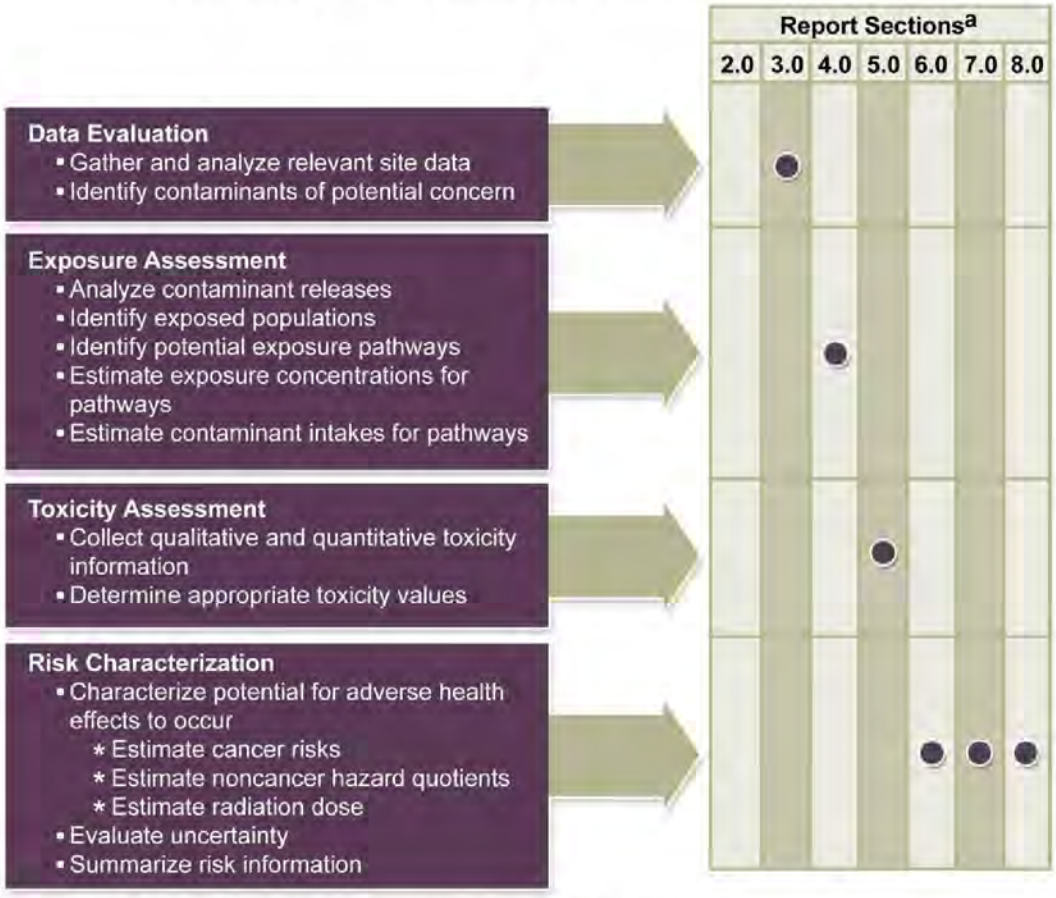
The DOE, which retains responsibility for the Hanford Site, is currently in the process of conducting RIs and cleanup activities at the Hanford Site in accordance with the requirements and guidelines of the CERCLA program. The Columbia River itself, which contains residuals both from historical activities at the Hanford Site as well as current upstream and non-Hanford Site sources (e.g., OCIs), is not formally part of the Hanford Site, but is being investigated under the same CERCLA process.

Additional guidance from the Washington State Department of Ecology (Ecology) was used to conduct a screening-level assessment of contaminants in sediments and surface water using the "Model Toxics Control Act – Cleanup" (MTCA) cleanup levels for soil (unrestricted use) and surface water (potable use) (*Washington Administrative Code* [WAC] 173-340-740). This assessment was described in the RI Work Plan and is provided in Appendix A.

The quantitative HHRA was conducted in accordance with U.S. Environmental Protection Agency (EPA) Superfund risk assessment guidelines presented in EPA/540/1-89/002, *Risk Assessment Guidance for Superfund, Volume 1, Human Health Evaluation Manual (Part A) (Interim Final)*, as well as other EPA risk guidance. The HHRA approach used herein is shown in Figure 1-8 and is consistent with EPA guidance for performance of human health risk assessments at CERCLA or RCRA sites and also reflects recent discussions with representatives from EPA Region 10, Ecology, DOE, and other interested parties. The outcome of these discussions and framework for this baseline risk assessment is reflected in the RI Work Plan (DOE/RL-2008-11).

Although, as discussed, the study area within the Columbia River is not a designated CERCLA site, the CERCLA approach was followed for completion of the HHRA in order to be consistent with the approach undertaken in the RCBRA (DOE/RL-2007-21) and other operable units within the Hanford Site, as well as the process outlined in DOE/RL-2004-49, *Columbia River Component of the River Corridor Baseline Risk Assessment: Basis and Assumptions on Project Scope*. The approach undertaken for this HHRA follows that outlined in the RI Work Plan (DOE/RL-2008-11).

Figure 1-8. Linkage of the Columbia River Component Risk Assessment Volume II Report Sections to the U.S. Environmental Protection Agency Four-Step Human Health Risk Assessment Process.



[a]Sections that present information supporting completion of each step

Subsequent sections of this report describe the area of study, data, methods, screening values, and results of the HHRA.

1.5 INTEGRATION WITH OTHER HANFORD SITE RISK ASSESSMENTS AND STUDIES

As discussed above, these risk assessments (e.g., SLERA and HHRA) are being completed in accordance with the 2008 RI Work Plan (DOE/RL-2008-11). This work plan was developed at the direction of the Tri-Parties (i.e., EPA, Ecology, and DOE). The results of this baseline risk assessment, in addition to the RI, are important to other Hanford Site cleanup activities in areas that border the Columbia River, also known as the "River Corridor."

In the early 1990s, the Tri-Parties decided that enough information was known about contaminated soil and groundwater at the Hanford Site to begin focusing directly on cleanup instead of performing additional studies to help refine the existing information. This decision led to an early start for cleanup of contaminated soil and groundwater in areas of the Hanford Site that border the Columbia River in 1995. As the cleanup progresses, new information on the contamination is gathered. These cleanup activities continue today.

The Tri-Parties have developed a strategy to make final decisions about the actions that are needed to complete cleanup in the River Corridor. Part of the strategy is to split these final cleanup decisions into smaller pieces of work that are more manageable and aligned with Hanford Site operational functions. Final cleanup decisions will be developed for the ROD areas associated with the following:

- 100-B/C Area
- 100-K Area
- 100-N Area
- 100-D and 100-H Areas
- 100-F and IU-2/6.
- 300 Area fuel fabrication and development facilities.

Final remedial decisions for each of these six areas will address the cleanup of contaminated soil and groundwater. The impacts of the Hanford Site releases to the Columbia River are an integral piece of these final remedial decisions. If cleanup actions are needed to address Hanford Site contamination in the river, they may be included with the final remedial decisions for one or more of the six ROD areas. It is also possible that a separate remedial decision could be made that is specific to the Columbia River. The objective for all of these remedial decisions would be to reduce the risk of potential harm to humans and the environment.

Concurrent with the CRCRA is the RCBRA (DOE/RL-2007-21, Rev. 0) that presents a comprehensive HHRA for the right-bank source areas along the Hanford Reach. The RCBRA evaluated recreational, residential, agricultural, and subsistence living scenarios involving exposure to various Hanford Site media, including soil and groundwater in upland portions of the Hanford Site, and near-shore sediments, surface water, and fish along the Hanford Reach of the Columbia River. The results of the RCBRA have been reviewed and considered in conjunction with development of the CRCRA. The intent of this CRCRA is to complete the assessment of the "bank-to-bank" Hanford Reach and downstream areas (i.e., Lake Wallula) of the Columbia River, characterizing risk in areas not previously addressed under the RCBRA.

In addition to the RCBRA, other previous assessments of the Columbia River along, upstream, and below the Hanford Reach have also been reviewed as part of the CRCRA. Those studies specifically cited include but are not limited to the following:

- CH2MHILL, 2007, *Phase I Fish Tissue Sampling Data Evaluation, Upper Columbia River Site, CERCLA RI/FS (Final)*

- CRITFC, 1994, *A Fish Consumption Survey of the Umatilla, Nez Perce, Yakama, and Warm Spring Tribes of the Columbia River Basin*

- EPA 910-R-02-006, *Columbia River Basin Fish Contaminant Survey 1996-1998.*

- Washington Department of Ecology (11-03-067), 2011, *Focus on Fish Testing: Snake River Fish Tested for Chemicals.*

1.6 ORGANIZATION OF REPORT

This report follows a presentation structure designed to both follow EPA risk assessment guidance and to facilitate the understanding of this large and complex site by presenting information in a logical and sequential fashion.

As discussed, the CRCRA is presented in two volumes: Volume I contains the SLERA and Volume II contains the HHRA. The volumes are complementary but are written to stand alone with separate executive summaries, discussions, and conclusions. Both volumes are composed of two parts. Part 1 contains the text, figures, and references, whereas Part 2 contains the tables that support the text in Part 1. The report was structured to facilitate side-by-side review of the information and the supporting data and is consistent with the structure of the RCBRA.

Subsequent portions of this section and following sections provide the following information:

- **Section 1.0 – Introduction.** This section provides the purpose and scope of the HHRA as well as the guidance used and requirements met for this HHRA.

- **Section 2.0 – Site Background Information.** This section provides a summary of former operations, releases, and response actions within the River Corridor portion of the Hanford Site, as well as a description of the environmental and recreational setting within the Hanford Reach and reference/upriver areas.

- **Section 3.0 – Data Evaluation.** This section identifies the analytical data used in the HHRA, summarizes the analytical results, and identifies which contaminants of potential concern (COPCs) will be carried through the quantitative risk analysis.

- **Section 4.0 – Exposure Assessment.** The exposure assessment estimates chemical concentrations in environmental media, identifies who may be exposed (receptor), the applicable exposure media and pathways, and quantifies the exposure to contaminants in the relevant environmental media.

- **Section 5.0 – Toxicity Assessment.** This section provides toxicity data associated with threshold (noncarcinogenic) effects and carcinogenicity that are used in the estimation of cancer risk and noncancer health hazards.

- **Section 6.0 – Risk Characterization**. This section provides estimates of health hazard and cancer risk for each exposure scenario and discusses these results with respect to primary risk drivers and areas potentially requiring further data and/or response actions.

- **Section 7.0 – Uncertainty Analysis.** This section presents key areas of uncertainty associated with various components of the quantitative risk assessment, including data gaps in toxicological or exposure assessment information and the conservative assumptions or scientific judgments used to bridge these data gaps. Uncertainties and assumptions are also are discussed with respect to their impact and biases on the risk assessment results.

- **Section 8.0 – Conclusions and Recommendations.** This section presents a summary of the findings of the baseline risk assessment as well as recommendations for further characterization as needed.

Appendices included with this HHRA are as follows:

- Appendix A – This appendix presents a separate methodology comparing exposure point concentrations (EPCs) with certain cleanup levels in the State of Washington MTCA regulations (WAC 173-340). As per the request of Ecology and as indicated in Section 4.6.7.4 of the RI Work Plan, this methodology consists of a comparison of sediment and surface water exposure point concentrations to medium-specific soil and drinking water benchmarks. Freshwater sediment cleanup levels under WAC 173-340 are typically established on a site-specific basis; however, lacking such values at this time, sediment EPCs are conservatively compared to soil cleanup levels for unrestricted exposure. (On CD only.)

- Appendix B – This appendix consists of the final data set used for this HHRA (termed the "HHRA Data Set") and is composed of island soil, sediment, surface water, and fish tissue data collected during the RI, which were collected between 2008 and 2010 as part of the RI field effort; and "historical" sediment and surface water data, which were collected as part of other studies conducted between 2000 and 2007. (On CD only.)

- Appendix C – This appendix contains figures with the locations of all surface water, sediment, soil, and fish tissue samples used in the HHRA.

- Appendix D – This appendix contains the EPA ProUCL statistical software outputs that were used to calculate EPCs. (On CD only.)

- Appendix E – This appendix provides the output for the statistical comparisons that were completed to identify those contaminants that are present at concentrations consistent with either background or reference areas. (On CD only.)

- Appendices F through L – These appendices include the calculation of exposure doses and risk estimates for each receptor and relevant exposure pathways. Tables within these appendices (arranged by exposure point) show for each receptor the risk and hazard

calculations by exposure route (e.g., dermal contact, ingestion) and by exposure medium (i.e., soil, sediment, surface water, and fish tissue). (On CD only.)

- Appendix M – This appendix contains the comparison of the analytical results (i.e., reporting limits) of nondetect contaminants to human health screening criteria to ensure that data for these undetected contaminants were usable for assessing risk, and that exclusion of these contaminants was unlikely to underestimate risk. This comparison is presented in Appendix M-1 for surface water, M-2 for sediment, M-3 for island soil, and M-4 for fish tissue. (On CD only.)

- Appendix N – This appendix evaluates and discusses whether inclusion of wasteway and irrigation canal data in the reference data set impacts the conclusions of the risk assessment. (On CD only.)

- Appendix O – This appendix contains dose and noncancer hazard/cancer risk calculations for ingestion of select contaminants in fish tissue. The purpose of these calculations is to estimate the potential hazard/risk that is associated with various constituents and evaluate how these calculations may bias the results of the human health risk assessment. This bias is discussed in Section 7.0, the Uncertainty Analysis. (On CD only.)

1.7 REFERENCES

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2.0 SITE BACKGROUND

2.1 OVERVIEW

The Hanford Site is located in the Pasco Basin within the Yakima Fold Belt on the Columbia Plateau in southeastern Washington State and occupies an area of about 1,450 km^2 (560 mi^2). The Hanford Site is considered one of the source areas for chemical and radiological contaminants that enter the Columbia River along a portion of the Hanford Reach.

Hanford Site sources of contamination to the Columbia River include past river effluent pipeline discharges, current contaminated groundwater seepage to the river, and limited overland flow from the operational areas. In addition, the Columbia River receives contributions from other anthropogenic and natural sources unrelated to the Hanford Site. Detailed descriptions of the sources of contaminants to the Columbia River as well as geological, topographical, and other relevant information have been provided in numerous reports, including the following:

- NWPC 2004, *Columbia Gorge Mainstem Subbasin Plan*
- BHI-01648, *Late Pleistocene- and Holocene-Age Columbia River Sediments and Bedforms: Hanford Reach Area*
- WCH-201, *Columbia River Component Data Gap Analysis*
- DOE/RL-2007-21, *River Corridor Baseline Risk Assessment Volume II: Human Health Risk Assessment*
- WCH-398, *Data Summary Report for the Remedial Investigation of Hanford Site Releases to the Columbia River, Hanford Site, Washington* (Data Summary Report).

This section provides a description of the river's environmental setting and an overview of Hanford Site operations and sources of impacts to the Columbia River, as well as a discussion of previous Hanford Site investigation and assessment activities.

2.2 PHYSICAL SETTING

The Columbia River originates in Canada on the west slope of British Columbia's Rocky Mountains and flows 1,954 km (1,214 mi) to the Pacific Ocean along the Washington/Oregon state boundary. Approximately 1,207 km (750 mi) of the river flows through the State of Washington. The Columbia River enters the Hanford Site from the west and flows through the northern portion and along the eastern site boundary. The Hanford Reach is a 77-km (48-mi) stretch of river that flows unimpeded from the base of Priest Rapids Dam downstream to the head of Lake Wallula above McNary Dam. It is the only undammed, free-flowing portion of the Columbia River in the United States above Bonneville Dam.

The Yakima River flows south of the Hanford Site and drains to the Columbia River several miles south of the site boundary. The confluence of the Snake River, the largest tributary to the Columbia River, joins the Columbia River approximately 14 km (9 mi) downstream from the Yakima River confluence. The smaller Walla Walla River drains to the Columbia River downstream of the Snake River confluence. The Yakima and Snake Rivers are the primary contributors of suspended sediment to the Columbia River (FH 1999, *Groundwater/Vadose Zone Integration Project Preliminary System Assessment Capability Concepts for Architecture, Platform, and Data Management, Appendix E: Columbia River Conceptual Model*).

With respect to discharge, the Columbia River and its 30 major tributaries comprise the predominant river system in the Pacific Northwest and the fourth largest in the United States. The Pend Oreille and Spokane Rivers provide the largest annual tributary contributions to flow (over 850 m^3/sec [30,000 ft^3/sec]) on the Columbia River in the upper reach between Canada and Grand Coulee Dam. The tributaries between the Okanogan River and the Snake River contribute approximately 396 m^3/sec (14,000 ft^3/sec), and the Snake River itself contributes approximately 1,529 m^3/sec (54,000 ft^3/sec). Below the Snake River, downstream to Bonneville Dam, the mean annual tributary inflow totals approximately 396 m^3/sec (14,000 ft^3/sec) (CRWMP 2006, *Water Supply Inventory and Long-Term Water Supply and Demand Forecast Report*, Chapter 3, "Columbia River Baseline Assessment").

The flow of water in the Columbia River is regulated by several dams within the United States that were constructed between 1938 and 1967 for several purposes, including flood control, irrigation, and electrical power generation. Of the 11 major dams constructed along the main channel of the Columbia River, only Bonneville and Grand Coulee Dams were in place when the first single-pass reactor (105-B Reactor) came on line in September 1944. Construction began on three additional dams downstream of the Hanford Site after operations began: McNary Dam (the nearest dam downstream of the Hanford Site) in the late 1940s, the Dalles Dam in the early 1950s, and John Day Dam in the late 1950s. The construction of the dams greatly slowed the water travel times and resulted in lower sediment loads being discharged to the Pacific Ocean, as well as created depositional areas behind each of these dams.

Flows through the Hanford Reach fluctuate significantly and are controlled primarily by power demand operations at Priest Rapids Dam (FH 1999), the nearest dam upstream of the Hanford Site. As a result of the fluctuations in discharges at Priest Rapids Dam, the depth of the Columbia River varies significantly over time and may change by up to 1 m (3 ft) within a few hours along the Hanford Reach (FH 1999).

The suspended sediment load of the Columbia River is typically very low, and the bedload consists mainly of fine and medium sand (DOE/RL-2005-09, USACE 1999). The coarser sediments are typically deposited at the head of pools, while the finer sediments are deposited within the impoundments or may be transported past the dams. Because of the relatively high flow rate along the Hanford Reach, the majority of this stretch of river is primarily coarse-grained deposit (e.g., the river bottom is composed of gravel and cobbles with limited amounts of fine-grained material deposited between the coarse-grained material). The sediment thickness on the upstream side of McNary Dam was estimated at up to 9 m (30 ft), with an average annual

depositional rate of 5 to 18 cm/yr (2 to 7 in./yr) in 1976 (BNWL-2305, *Association of Hanford Origin Radionuclides with Columbia River Sediment*). Deposition of sediment also occurs on the shoreline portions of the islands along the Hanford Reach.

Resuspension of fine-grained residual sediment occurs on a daily basis, as well as during flood events. Sediments have been redeposited throughout the Columbia River. While flood events may move sediments to higher levels above normal high water along shorelines during periods of flooding, the majority of the suspended sediment loads will be deposited in the lakes created behind the dams. A significant flood occurred in 1948, which is prior to when most of the dams had been constructed and when increased production began at the Hanford Site. To further evaluate potential impacts of sediment resuspension and redeposition along the Hanford Reach and downriver, a total of 104 sediment core samples were collected from the reactor sites down to the McNary Pool during the RI. In addition, core samples were also collected from the Bonneville Dam pool. These data were used to characterize contaminant distribution.

Groundwater beneath the Hanford Site discharges to the Columbia River. The presence of shoreline seeps and springs depends on the water level in the river. Groundwater flow toward the river is influenced by fluctuations in river stage, with locations near the river being most strongly affected (FH 1999). Changes in river-stage elevation can be correlated to changes in water table elevation up to 360 m (1,180ft) from the river (PNL-8580, *Water Level Measurements for Modeling Hydraulic Properties in the 300-FF-5 and 100 Aggregate Area Operable Units*). In many areas, water flows from the river into the aquifer at high river stages, causing local groundwater levels to rise. During low river stages, riverbank seeps can be observed discharging to the river.

Upwelling data (porewater and sediment) collected as part of the RI indicate that groundwater from the Hanford Site discharges primarily along the right bank of the Columbia River, for most chemical/radiological constituents, consistent with the conceptual site model (CSM) presented in the Data Summary Report (WCH-398). However, hexavalent chromium has been detected in sediment and porewater upwelling samples collected across the river on the far (left) bank, suggesting that groundwater from the Hanford Site may migrate out farther into the river than previously documented, or that hexavalent chromium may be related to another unidentified source. Prior to the upwelling sampling that occurred during this investigation, it was believed that the majority of the groundwater discharged into the river directly adjacent to the shoreline. However, based on the findings of the upwelling study, it is now understood that groundwater upwelling also occurs out into the center of the river channel. The CSM has been updated accordingly. A discussion of the CSM is provided in Section 4.1.

2.3 RIVER ECOLOGY

The Columbia River and associated riparian zones provide habitat for numerous wildlife and plant species, supporting a large and diverse population of plankton, benthic (bottom-dwelling) invertebrates (e.g., insect larvae, clams, crayfish), fish, and wildlife. Large rivers such as the

Columbia River, with its series of large reservoirs, contain significant populations of primary energy producers (e.g., algae and plants) that contribute to the biota's basic energy requirements.

Numerous species of fish, both native and introduced, have been listed in the Hanford Reach of the Columbia River. Of native species, Chinook salmon (*Oncorhynchus tshawytscha*), sockeye salmon (*Oncorhynchus nerka*), coho salmon (*Oncorhynchus kisutch*), and steelhead trout (*Oncorhynchus mykiss*) use the river as a migration route to and from upstream spawning areas and are of the greatest economic importance. Additionally, fall Chinook salmon and steelhead trout spawn in the Hanford Reach. Inundation of other mainstream Columbia River spawning grounds by dams has increased the relative importance of the Hanford Reach to fall Chinook salmon production in the Columbia and Snake Rivers.

Other fish of importance to sport anglers are the native mountain whitefish (*Prosopium williamsoni*) and white sturgeon (*Acipenser transmontanus*). Introduced species like smallmouth bass (*Micropterus dolomieui*), crappie (*Pomoxis nigromaculatus*), catfish (*Ictalurus punctatus*), walleye (*Stizostedion vitreum*), and yellow perch (*Perca flavescens*) are also present. Large populations of rough fish (i.e., freshwater fish considered undesirable as a food or sport fish and often viewed as a competitor of more desirable fish) are also present, including introduced carp (*Cyprinus carpio*) and native species such as redside shiner (*Richardsonius balteatus*), suckers (*Catostomus macrocheilus*), and northern pikeminnow (*Ptychocheilus oregonensis*) (PNNL-6415, *Hanford Site National Policy Act [NEPA] Characterization*).

2.4 RECREATIONAL AND OTHER USES OF THE RIVER

The Columbia River is widely used for recreational purposes such as boating, wading, swimming, fishing, and water-skiing, and a variety of beaches, boat ramps, and wildlife viewing areas are located throughout the study area. The Hanford Reach National Monument consists of an 82-km (51-mi) stretch of the Columbia River and federally owned riparian lands. Below the southern site boundary, recreational use is widespread throughout Lake Wallula, the next 80 km (50 mi) of the McNary Dam impoundment.

Numerous islands are located within the study area. Most of these islands are owned by federal or state agencies and are designated as conservation/recreation areas. Many of the islands (or portions of the islands) are entirely submerged during periods of high water and consequently subject to depositional/erosional forces.

In addition to recreational use, surface water for certain portions of the Columbia River is used for river navigation/transportation; hydropower; and as a domestic, agricultural, and industrial water supply. The City of Richland relies on filtered and treated river water as its source of public drinking water; the Richland Pumphouse, a primary treatment system, is located near RM 340 (City of Richland 2011).

2.5 HANFORD SITE OPERATIONAL HISTORY

In March 1943, construction at the Hanford Site began on three reactors (105-B, 105-D, and 105-F Reactors) and three chemical processing facilities (B, T, and U Plants). The Hanford Site was originally designed, built, and operated as part of the Manhattan Project to produce plutonium for nuclear weapons using production reactors and chemical reprocessing plants. After World War II, six additional reactors were built (105-H, 105-DR, 105-C, 105-KW, 105-KE, and 105-N Reactors) along with two additional chemical separation plants. In the 1950s, energy research and development, isotope use, and other activities were added to the Hanford Site mission. Specific areas of the Hanford Site have been designated for the uses described above. Operational areas generally contain support facilities including maintenance buildings, powerhouses, raw water treatment plants, water storage tanks, electrical maintenance facilities, and subsurface sewage disposal systems.

Reactors and other facilities at the Hanford Site have been grouped into three main operational areas: 100 Area, 200 Area, and 300 Area. Each of these areas is described in the following subsections.

2.5.1 100 Area

The 100 Area is located upstream from the City of Richland along the Columbia River in the northern portion of the Hanford Site and occupies an area of approximately 68 km^2 (26 mi^2) (Figure 1-1). Between 1943 and 1962, nine water-cooled, graphite-moderated plutonium production reactors were built along the shore of the Columbia River. The last single pass water-cooled reactor (100-KE) ceased operations in 1971. The mission of each reactor was to produce weapons-grade plutonium. The main component of each reactor was a large stack of graphite blocks (pile) with process tubes containing the fuel elements and cooling water. The confinement of large numbers of uranium fuel elements within the reactor piles created an intense radiation field and a nuclear chain reaction that converted some uranium atoms to plutonium atoms.

The first eight reactors (105-B, 105-C, 105-D, 105-DR, 105-F, 105-H, 105-KE, and 105-KW) used water from the Columbia River for direct cooling of the reactor pile. The ninth reactor (105-N) recirculated purified water through the reactor core in a closed-loop cooling system. Effluent from the 105-N Reactor was discharged to trenches and cribs near the river. Columbia River water passed through 100 Area reactors, absorbing and removing heat generated by the nuclear process. Cooling water was withdrawn from the Columbia River though the river pump houses located directly on the river and sent to the reservoirs. The reservoirs each stored 25 million gallons of water for primary and secondary (backup) water uses (DOE/RL-97-1047, *History of the Plutonium Production Facilities at the Hanford Site Historic District, 1943-1990*). The water was pumped to a series of support buildings for treatment and filtration prior to use to remove particulate matter, dissolved gases (i.e., carbon dioxide and oxygen), and chemicals. Following injection of water into the reactor at a rate of about 113,562 L/min (30,000 gal/min), processed water was discharged to the retention basins where it cooled to allow for decay of short-lived radionuclides. From the retention basins, the water reentered the Columbia River via

outfall structures and underground pipelines, emerging at the mid-channel of the Columbia River (DOE/RL-97-1047).

Cooling water also contained radioactive materials (fission products) that escaped from the fuel elements or tube walls during the irradiation process (DOE/RL-97-02, *National Register of Historic Places Multiple Property Documentation Form – Historic, Archaeological and Traditional Cultural Properties of the Hanford Site, Washington*). The coolant water was occasionally contaminated while passing through reactors due to failed aluminum jackets. Failure of aluminum jackets allowed cooling water to come in direct contact with irradiated uranium. This resulted in a release of fission products and actinides to the effluent stream. Fission products included isotopes such as cesium, strontium, and iodine. This highly contaminated cooling water was sent to trenches rather than being returned to the Columbia River.

Other past waste disposal practices in the 100 Areas resulted in releases of radionuclides and chemicals to soil and groundwater. Unplanned and planned releases to the soil column in the 100 Areas also created hundreds of waste sites. Unplanned releases were mainly from leaks or overflow of reactor cooling water transfer systems. Planned releases were made at liquid waste sites, solid waste burial grounds, and "remaining sites" (a name used for administrative and remediation purposes).

Liquid waste sites in the 100 Area include retention basins, trenches, cribs, french drains, and effluent pipelines. Contaminated water from process tubes in which fuel cladding failures occurred was generally discharged to cribs distant from the reactors and percolated into the soil (DOE/RL-97-1047). Solid waste containing hazardous and radioactive wastes was managed within burial grounds. Burial grounds contain concrete, construction debris, and other wastes. The "remaining sites" are scattered across the 100 and 600 Areas and include, but are not limited to, septic systems, burn pits, french drains, pre-Hanford Site and Hanford-era waste dumps, small oil spills, nonreactor effluent pipelines, and animal experiment facilities. Additional details on 100 Area waste sites are found in DOE/RL-2004-37, *Risk Assessment Work Plan for the 100 Area and 300 Area Component of the RCBRA*.

2.5.2 200 Areas (Central Plateau)

After cooling in the 100 Areas, the irradiated fuel elements were taken to the 200 Areas for storage, additional cooling, and processing within the chemical separation plants. The 200 Areas (200 East and 200 West Areas) are located in the center of the Hanford Site and are located approximately 8 to 10 km (5 to 6 mi) from the Columbia River, respectively. The 200 Areas occupy approximately 16 km^2 (6 mi^2) and contained the facilities used to separate, isolate, store, and ship the plutonium. To separate the plutonium from the base uranium and activated by-products formed in the irradiation process, the chemical separation plants first dissolved the fuel elements with acids and then chemically separated the plutonium isotopes from the liquefied materials. The plants produced large quantities of high-level radioactive waste that were stored first in single-shell underground tanks and later in double-shell underground tanks. The various

separation processes are described in DOE/RL-98-28, *200 Areas Remedial Investigation/Feasibility Study Implementation Plan Environmental Restoration Program.*

The separation process in the 200 Areas generated large volumes of effluent. Most of the low-level liquid wastes were discharged to the soil column at liquid waste receiving sites (i.e., ponds, trenches, reverse wells, ditches, and cribs). Other wastes such as uranium- and fission product-rich wastes were stored in the underground storage tanks. Unintentional and intentional releases to the ground from chemical separation operations have impacted the soil column and aquifer beneath the Hanford Site.

The discharge of effluent to the soil columns provided the primary driving force for liquid and contaminant migration through the vadose zone to groundwater. Key radionuclides with half-lives longer than 10 years that were discharged to the soil column included cesium-137, barium-137, iodine-129, strontium-90, yttrium-90, technetium-99, uranium, carbon-14, americium-241, plutonium-239/240, and tritium as tritiated water. Major nonradiological chemicals in liquids discharged to the ground include nitrate, sodium, phosphate, sulfate, ammonia, carbon tetrachloride, fluoride, and sodium dichromate. Inorganic chemicals were used and discharged in much greater quantities than organics. The greatest amount of hazardous chemicals in liquids was discharged between 1945 and 1958 (WHC-SD-EN-TI-008, *Geologic Setting of the 200 West Area: An Update*; DOE/RL-98-28).

2.5.3 300 Area

The 300 Area borders the Columbia River on the southeastern edge of the Hanford Site and is located just north of the City of Richland. The 300 Area occupies approximately 1.35 km^2 (0.52 mi^2). In March 1943, construction of a fuel fabrication complex began at the Hanford Site in the 300 Area. As a manufacturer of uranium fuel, the 300 Area housed the first essential step in the plutonium production process. Nuclear fuel was fabricated from uranium shipped in from offsite support facilities. Metallic uranium was extruded into the proper shape and encapsulated in aluminum alloy cladding (during early years) or zirconium alloy cladding (during later years). In addition to housing the Hanford Site fuel fabrication plants, the 300 Area was the center of many research and development projects. Process improvement laboratories were constructed beginning with the Manhattan Project. These facilities included research laboratories, chemical process laboratories, test reactors, and numerous ancillary support structures. The addition of new research and laboratory facilities continued into the 1950s and 1960s to support defense and energy research. New support and laboratory facilities were added in the 1970s for further research on energy, waste management, biological sciences, and environmental sciences. The 300 Area industrial complex is currently undergoing extensive decommissioning and demolition of many of the older facilities that no longer have a defined use. A number of facilities, however, still support the remaining industrial complex and continue to support ongoing Hanford Site missions.

Operations in the 300 Area created both liquid and solid waste sites. Prior to 1973, a series of solid waste burial grounds were used for solid waste and debris (DOE/RL-2004-37). After 1973, the 300 Area burial grounds were no longer used for disposal, and waste was transported to other

Hanford Site burial grounds. Between 1943 and 1994, unlined ponds and process trenches received millions of gallons of contaminated waste water from 300 Area operations. These ponds and trenches are suspected to be the primary source of uranium in the groundwater beneath the 300 Area.

2.5.4 Historical Hanford Site Contaminant Sources and Waste Streams

A comprehensive summary of radionuclides released from the eight single-pass reactors during operation from 1944 to 1971 is provided in DOE/RL-97-1047. A majority of the radionuclides are short-lived and are no longer present. The following radionuclides are known to have been released to the Columbia River: cobalt-60, zinc-65, strontium-90, cesium-137, europium-152, europium-154, thorium-228, radium-226, plutonium-238, plutonium-239, plutonium-240, and americum-241 (BNWL-2305). In addition, nonradioactive chromium is known to have been released to the river through the river effluent pipelines. Groundwater contaminated by past operations continues to flow toward and discharge to the Columbia River. This upwelling groundwater contains chromium, nitrate, strontium-90, tritium, and, in the 300 Area, uranium and volatile organic compounds (VOCs).

Historic spills and overland discharges are also considered releases to the Columbia River. These included an overland discharge of liquid process effluent containing uranium from the 300 Area South Process pond in 1948 (EMO-1026, *Addendum to Data Compilation Task Report for the Source Investigation of the 300-FF-1 Operable Unit Phase I Remedial Investigation*) and a spill from a sodium dichromate storage tank at the 183-C Building in 1965 (DUN-3032, *Chemicals Discharged to the Columbia River from DUN Facilities Fiscal Year 1967*).

2.5.5 River Effluent Pipeline Discharges

From 1943 to the present, the Columbia River has been used as a water supply by the Hanford Site. Most of the nuclear reactors (with the exception of 105-N) in the 100 Area used the single-pass river water for primary reactor cooling purposes. Between 1943 and 1987, pipelines extending from outfall structures at the 100 Area reactors into the Columbia River were used to carry reactor cooling water for discharge to the river. Operation of most river effluent pipelines ended when the associated water-cooled reactors were shut down between the mid-1960s and mid-1980s, with the last shutdown occurring at the 100-N Area in 1987. Today the effluent pipelines remain in place on or beneath the river channel bottom. One of the two river effluent pipelines in the 100-K Area (100-K-96) was active up through April 2011; this pipeline was associated with the dewatering of the 107-KE Retention Basins and had a National Pollutant Discharge Elimination System (NPDES) permit during its operation. The effluent pipelines constitute seven waste sites in the 100-B/C, 100-D, 100-H, 100-F, 100-K, and 100-N Areas and include 15 separate pipelines. Most of the river effluent pipelines are known or suspected to still contain small amounts of residual contamination from past reactor operations.

2.5.6 Contaminated Groundwater Seepage to River

Past waste disposal practices at the Hanford Site have resulted in the presence of several contaminated groundwater plumes. Groundwater beneath the Hanford Site discharges to the Columbia River via springs and subaqueous (within the riverbed) groundwater plume upwellings. Therefore, groundwater provides a means for transporting Hanford Site-related contaminants to the Columbia River.

In general, groundwater discharges are considered to be the current dominant pathway for Hanford Site-related contaminants to enter the Columbia River. At least 115 shoreline springs have been documented along the Hanford Reach, with the predominant areas of discharging springs in the vicinity of the 100-N Area, Hanford townsite, and 300 Area (DOE/RL-2010-11, *Hanford Site Groundwater Monitoring and Performance Report for 2009*). Today, seeps from the 100-N Area have diminished due to declining water-table elevations, a consequence of the end of operations at the 105-N Reactor, which have reduced discharge from the springs. In addition, effluent from the 105-N Reactor was discharged to trenches and cribs near the river. Contaminants from the 100 Area trenches and cribs have impacted groundwater that discharges to the river.

Groundwater contamination exists beneath the Hanford Site and along the Columbia River shoreline and near-shore river where groundwater mixes with the surface soils and Columbia River water (DOE/RL-2004-37). The following are the primary contaminants associated with the 100, 200, and 300 Area groundwater plumes (DOE/RL-2010-11; Data Summary Report [WCH-398]):

- 100 Area plumes: Hexavalent chromium, carbon-14, strontium-90, tritium, trichloroethene (TCE), and nitrate
- 200 Area plumes: Carbon tetrachloride, chromium, technetium-99, tritium, uranium, iodine-129, nitrate, and TCE
- 300 Area plumes: Nitrate, tritium, and uranium.

2.5.7 Limited Overland Flow

While the most significant historic transport mechanism was direct discharge of the single-pass cooling water, historic overland flow was also associated with reactor operations. Historic information, including aerial photographs, clearly shows water seepage from the reactor cribs and trenches flowing across the land surface and discharging directly into the Columbia River. While this transport mechanism is no longer active, it is assumed that overland flow was a significant source of Hanford Site contaminants to the river during operations. Hanford Site contaminants that reached the river during single-pass cooling water operations (1943 to 1972) via this transport mechanism have migrated downriver.

2.6 NON-HANFORD SITE SOURCES OF CONTAMINANTS

In addition to Hanford Site releases, there are numerous other sources of river contaminants, both naturally occurring and anthropogenic. This section summarizes potential current and historical contaminant sources upriver of the Hanford Site and other contributing influences within the RI area (i.e., global, municipal, industrial, agricultural, and commercial sources).

2.6.1 Upriver/Industrial Sources

While the presence of dams upriver from the Hanford Site currently limits the transport of contaminants from upstream sources, the magnitude and duration of historical and current discharges may provide a potential for long-range transport to the Hanford Reach.

Contributions of contaminants to the Columbia River may come from direct sources to the river or indirect sources. Examples of direct and indirect sources include mining operations, smelting, pulp and paper production, runoff from cities and agricultural areas, municipal and industrial wastewater treatment plants, nuclear weapons production and atmospheric testing, and other activities that release materials that ultimately reach the river.

Mining operations at the Teck Cominco Mine in Trail, British Columbia, located 16 km (10 mi) north of the U.S./Canadian border, began in 1890, with smelter operations beginning in 1896 along the headwaters of the Columbia River. These operations began prior to the construction of any dams along the Columbia River. The lead and zinc smelter on the banks of the Columbia River at the Trail facility dumped an estimated 10 million to 20 million tons of slag into the river. The facility released dissolved iron, manganese, zinc, copper, lead, arsenic, cadmium, and mercury via liquid effluent and as solids in the form of slag, a smelting byproduct (WHC-SA-1989-FP, *Sediment Quality and Ecorisk Assessment Factors for a Major River System*). The EPA Region 10 contends that the Trail smelter is the largest source of metals pollution to Lake Roosevelt, a reservoir created when the river was impounded behind Grand Coulee Dam in 1937. In 2006, an EPA study of sediment samples concluded that the portion of the lake from Inchelium, Washington, upstream to the Canadian border already qualified for Superfund listing because of hazards to aquatic life from heavy metals (CH2MHILL 2006, *Phase I Sediment Sampling Data Evaluation, Upper Columbia River Site, CERCLA RI/FS [Draft Final]*). Metal contaminants flow down the river into Lake Roosevelt. The EPA is currently undertaking an RI/FS of the Upper Columbia River/Lake Roosevelt, which encompasses the stretch of the Columbia River from the Canadian border to Grand Coulee Dam and surrounding upland areas. This study, which includes the collection of surface water, sediment, and fish tissue samples, is discussed further in Section 2.6. Contaminants from these historical discharges may also exist downstream in the Hanford Site RI area.

Other smelting operations have taken place in Northport, Washington (EPA 2004a, *EPA to Investigate Upper Columbia River Pollution*). The Celgar pulp mill in Castlegar, British Columbia, was a primary source of historical loading of dioxins and furans to the upper Columbia River (EPA 2004b, *Sediment Sampling Approach and Rationale, Upper Columbia River*). Alcoa's aluminum smelter facility in Wenatchee, Washington, is currently the

only major U.S. industrial NPDES-permitted facility located upstream of the Hanford Site. It may contribute the following contaminants to the Columbia River: fluoride, aluminum, copper, benzo(a)pyrene, cyanide, oil, and grease (WCH-201; WCH-91, *Columbia River Component Data Evaluation Summary Report*). There are also nine municipal treatment plants that discharge effluent to the river upstream of the Hanford Site (WCH-91, WCH-201).

The Bunker Hill Superfund Site, located in the Coeur D'Alene River Basin in northern Idaho, has a long history of mining and metals-processing activities dating to more than 100 years ago. The U.S. Department of the Interior identified more than 1,000 mining or milling-related features in the region surrounding the South Fork of the Coeur D'Alene River, and it is estimated that approximately 62 million tons of mine tailings have been discharged to the Coeur D'Alene River Basin since mining began (EPA 2010). The entire Coeur D'Alene River Basin includes the Upper Basin, the Lower Basin, Lake Coeur D'Alene, and a portion of the Spokane River where the lake drains into Washington State.

The Spokane River, a tributary to the Columbia River, has elevated levels of polychlorinated biphenyls (PCBs) and metals. The Spokane River flows 179 km (111 mi) from Lake Coeur d'Alene, Idaho, to Lake Roosevelt, which was created in the Columbia River by the completion of the Grand Coulee Dam in 1941. Some of the sources of contamination to the Spokane River include the following:

- Mining waste and the associated metals that may have been transported downstream from the Coeur d'Alene River Basin to the Spokane River.

- Midnite Mine, an open-pit uranium mine, operated along the Spokane River in the Selkirk Mountains of eastern Washington from the mid-1950s until 1981 and contributed contaminants upriver of the Hanford Site. Elevated levels of radioactivity (primary uranium) and heavy metals mobilized in acid mine drainage pose a potential threat to human health and the environment (EPA 2010, *Site Description: Midnite Mine, Washington*).

- Kaiser Trentwood, an aluminum plant, discharged PCBs to the Spokane River in excess of 2 kg/day in the early 1990s and as late as 2000 (Serdar et al. 2006, *Spokane River PCBs Total Maximum Daily Load, Water Quality Improvement Report, June*).

- The Spokane Wastewater Treatment Plant discharged 0.25 kg/day of PCBs in 2001 (Serdar et al. 2006).

As indicated above, there are a number of industrial and municipal discharges that have likely impacted surface water and sediments within the RI study area.

2.6.2 Global Sources

Worldwide atmospheric nuclear testing contributed to radionuclide contaminants in surface waters and ultimately to sediments throughout the Pacific Northwest. Fallout from atmospheric testing by the United States, Russia, and China contributed significantly to radionuclide levels in

the environment (WDOH 1994, *Special Report; Radioactivity in Columbia River Sediments and Their Health Effects*). The fallout materials consisted primarily of radionuclides such as cesium-137 (half-life of 30.07 years) and strontium-90 (half-life of 28.78 years) (WDOH 1994). Strontium and cesium are also associated with Hanford Site operations. The Soviet nuclear reactor accident at Chernobyl in 1986 also produced detectable levels of iodine-131 and cesium-137 in precipitation in the Pacific Northwest (WDOH 1994). Various radionuclides have been detected in surface and sediment samples collected from reference/background locations, as discussed further in Section 3.8.

2.6.3 Naturally Occurring Sources

Naturally occurring chemical and radiological contaminants associated with the chemical composition of bedrock and soil features in the Columbia River basin are present in sediment and surface water (EPA 910-R-02-006). For example, aluminum, arsenic, barium, cadmium, manganese, and elemental uranium have been detected in sediment and surface water samples collected from background (upriver) locations. In addition, radionuclides such as uranium-234 and uranium-238 have also been detected in background samples.

While these elements and radionuclides are naturally occurring in the environment as a result of local geochemistry, their presence in sediment and surface water upstream of the Hanford Site may not necessarily be representative of only naturally occurring conditions, and may in part be related to upstream or other non-Hanford Site sources (e.g., industrial, agricultural, or mining), as previously described. The nature and distribution of contaminants related to these sources in the Upriver/Reference Area sampling locations are inconsistent with the Hanford Site-wide CSM and are therefore not believed to be Hanford Site related (e.g., as a result of atmospheric transport and deposition).

2.6.4 Municipal/Urban Sources

Municipal and urban activities contribute as point and nonpoint sources of contamination to the river. Other NPDES-permitted discharges to the Columbia River include stormwater, minor industrial process wastewater, contact and noncontact cooling waters, treated waters, and construction sites. Effluents from municipal sewage treatment plants also contribute to waste loading within the Columbia River system. A total of 41 municipal sewage treatment plants were identified in 2005 that discharge effluent to the Columbia River (WCH-201).

Urban contributions including unpermitted residential, municipal, and commercial stormwater runoff; use of fertilizers and pesticides; and septic sewage systems are some of the potential sources of contamination from communities along the banks of the Columbia River. Stormwater runoff can contain a number of contaminants such as pesticide and weed control products, contamination from leaking transformers, hydraulic and lubricating fluids, petroleum products, metals, polycyclic aromatic hydrocarbons (PAH), and deicing salts. Runoff containing naturally occurring contaminants such as uranium also contributes to river contamination (WCH-201, Becker 1990).

2.6.5 Agricultural Sources

By the 1920s, major irrigation projects along the Columbia River and tributaries operated with the benefit of federal programs. In 1948, the Columbia Basin Project began transporting Columbia River water by canal to the more than 600,000 acres of farms in central Washington (CCRH 2007, *Promoting the Study of the Columbia River Basin History*).

Agricultural activities are a potentially significant source of contaminants to the river. Water from the irrigation returns in the Hanford Reach has been periodically sampled; identified contaminants include nitrogen, phosphate, copper, uranium, pesticides, and suspended solids (Ecology 1981, *Irrigation Return Flow Quality South Columbia Basin Irrigation District May - August 1980*; Data Summary Report [WCH-398]).

Sampling of irrigation return water from Franklin County and associated irrigation-related seeps entering the Columbia River, opposite the Hanford Site, have measured total uranium values of 8.6 pCi/L (PNL-7500 1990, *1988 Hanford Riverbank Springs Characterization Report*). Uranium is commonly present in phosphate-based fertilizers and is a natural constituent that weathers from some types of rocks in the region. In recent years, total uranium concentrations in the Hanford Reach have been elevated along the Franklin County shoreline. Previous studies have indicated these elevated concentrations are likely the result of groundwater seepage and surface water from irrigation returns that contain naturally occurring and anthropogenic uranium (PNL-7500).

2.6.6 Commercial and Recreational Sources

Recreational and commercial activities on the Columbia River also contribute contamination to surface water and sediments via marinas, boats, or other recreational watercraft. Discharge of bilge and ballast water, engine oil, spills, and materials associated with boat and shipyard maintenance are potential sources of contamination. These sources may contain old paint scrapings (e.g., lead), anti-foulants (e.g., copper), solvents, oil and grease, fuels, PCBs, and cleaning agents. Pilings, docks, and bulkheads associated with marine structures treated with creosote, chromated copper arsenate, or copper zinc arsenate are other potential sources of contamination.

2.7 RIVER INVESTIGATION AND ASSESSMENT ACTIVITIES

Previous investigations and assessments of the Columbia River include those conducted for the RCBRA, CRCRA, and other environmental assessment and monitoring programs operated by DOE, EPA, and other entities. Studies most relevant to the Hanford Site Study Area are briefly summarized in the following subsections.

2.7.1 RCBRA Investigations

The groundwork for the RCBRA (i.e., 100/300 Area studies) was initiated in the spring of 2003. Work conducted to support the risk assessment effort included defining the basis and assumptions of work scope (DOE/RL-2003-61, *100 Area and 300 Area Component of the River Corridor Baseline Risk Assessment: Basis and Assumptions on Project Scope*); development of a work plan (DOE/RL-2004-37); public and stakeholder participation; identification of issues through a series of agency and stakeholder interviews; identification of data quality objectives (DQOs) (BHI-01757, *DQO Summary Report for the 100 Area and 300 Area Component of the RCBRA*); development and implementation of a sampling and analysis plan (SAP) (DOE/RL-2005-42, *100 Area and 300 Area Component of the RCBRA Sampling and Analysis Plan*); and completion of the River Corridor baseline human health and ecological risk assessment.

The purpose of the RCBRA is two-fold: (1) to evaluate human health and ecological risks resulting from conditions subsequent to the implementation of the remedial actions in the 100 Area and 300 Area of the Hanford Site and (2) to use results to support risk management decision making and to support development of RODs for River Corridor areas.

The RCBRA focused on the potential risk from post-remediation conditions in operational areas, historical townsites, riparian areas adjacent to operational areas, and related groundwater plumes emerging in the near-shore river environment (DOE/RL-2005-37, *Status of Hanford Site Risk Assessment Integration, FY 2005*; DOE/RL-2004-37). Known emergent groundwater contaminant plume areas were evaluated as part of the RCBRA investigation.

After completion of the initial RCBRA sampling effort, an additional study was identified to complete data gaps of various locations, media types, and potential contaminants. The primary purpose of the subsequent assessment was to evaluate risks from current concentrations of chemicals and radionuclides in the riparian and near-shore aquatic zones between operational areas in the 100 Area and 300 Area. This included evaluating areas from emerging 200 Area groundwater plumes (under current conditions), slough and backwater areas, and habitats found predominantly in areas between reactor and operational areas (DOE/RL-2005-42).

Between October 2005 and December 2006, field sampling and surveys of soil, sediment, surface water, porewater, groundwater (well water), and fish tissue (sculpin and sucker) were conducted. Results of this assessment have been included in Rev. 0 of the RCBRA (DOE/RL-2007-21). Fish tissue results from the RCBRA investigation are briefly discussed in Section 3.6.4.5.

2.7.2 Columbia River Component Investigation

The October 2007 *Columbia River Component Data Gap Analysis* (WCH-201) was conducted to review the adequacy of the existing surface water and sediment data set from the Columbia River, with specific reference to the use of the data in future site characterization and baseline risk assessments. The goal was to determine if there were sufficient data to characterize

the current effects of Hanford Site operations on the Columbia River. The Data Gap Analysis identified site analytes and potential data gaps as well as the study area boundaries for the Columbia River RI. The field investigation program for the RI was developed collaboratively by DOE, EPA Region 10, Ecology, and other stakeholders, and was based on the outcome of the DQO process (WCH-265, *DQO Summary Report for the Remedial Investigation of Hanford Site Releases to the Columbia River*) to address the data needs identified in the Data Gap Analysis report (WCH-201). The rationale for the sampling approach and strategy are detailed in the RI Work Plan (DOE/RL-2008-11).

The RI Work Plan was implemented from the fall of 2008 to the summer of 2010, entailing the collection of more than 2,000 environmental samples consisting of surface water, porewater, sediment, island soil, and fish tissue. The RI field activities associated with the collection of these samples are documented in the following reports:

- WCH-352, *Field Summary Report for Remedial Investigation of Hanford Site Releases to the Columbia River, Hanford Site, Washington: Collection of Surface Water, River Sediments, and Island Soils*

- WCH-380, *Field Summary Report for Remedial Investigation of Hanford Site Releases to the Columbia River, Hanford Site, Washington: Collection of Surface Water, Pore Water, and Sediment Samples for Characterization of Groundwater Upwelling*

- WCH-387, *Field Summary Report for Remedial Investigation of Hanford Site Releases to the Columbia River, Hanford Site, Washington: Collection of Fish Tissue Samples.*

Collection activities and evaluation of the resultant data are described in detail in the Data Summary Report (WCH-398). The data assessment process and resulting data qualification actions are described in WCH-381, *Data Quality Assessment Report for the Remedial Investigation of Hanford Site Releases to the Columbia River, Hanford Site, Washington.*

The data from the RI sampling efforts, in addition to historical data collected under the RCBRA and other sampling programs, have been used to evaluate the nature and extent of past releases of Hanford Site contaminants to the Columbia River and to support this baseline HHRA as well as the SLERA. Data specifically relevant to the CRCRA are further discussed in Section 3.0.

2.7.3 Upper Columbia River/Lake Roosevelt River Media Sampling

In 2005, EPA led a Phase I sampling program in which sediment, porewater, and fish tissue samples were collected in the Upper Columbia River. This program was conducted as part of the Upper Columbia River RI/FS and was designed to update the preliminary CSM for sediment, gather data in support of the human and ecological risk assessments, and support issuance of an updated fish advisory for Lake Roosevelt (CH2MHILL 2006; CH2MHILL 2007, *Phase I Fish Tissue Sampling Data Evaluation, Upper Columbia River Site, CERCLA RI/FS (Final).* Sediment samples were analyzed for various parameters including metals, semivolatile organic compounds (SVOCs), pesticides/PCB Aroclors, and dioxins and furans. Additionally, fish tissue

samples from walleye, rainbow trout, whitefish, largescale sucker, and burbot were collected and analyzed for a variety of chemical parameters. These analyte lists were based on historical information about upstream facilities, industry-related chemical literature, and information about releases, as well as results from previous investigations and the preliminary CSM.

Using the Phase I data, EPA prepared a screening assessment in response to public concern regarding the safety of recreating on beaches along the Upper Columbia River. The results were reported in the *Screening-Level Risk Assessment for Recreational Use of Beaches, Upper Columbia River, Remedial Investigation and Feasibility Study*, Draft, dated August 2006 (EPA 2006). Results of this assessment indicated that arsenic and lead in beach sediment were present at levels exceeding human health risk-based screening criteria, and that a baseline HHRA was warranted.

The constituents of interest in sediment were identified during the data evaluation that is documented in CH2MHILL (2006). These constituents of interest included the following:

- Metals: Antimony, arsenic, cadmium, chromium, copper, iron, lead, manganese, mercury, nickel, uranium, and zinc
- Chlorinated pesticides: 2,4-dichlorodiphenyldichloroethylene (2,4-DDE), 2,4-dichlorodiphenyltrichloroethane (2,4-DDT), 4,4-dichlorodiphenyldichloroethane (4,4-DDD), 4,4-DDE, 4,4-DDT, and aldrin
- PCBs: Aroclor-1016 and Aroclor-1260
- Dioxins and furans: 2,3,7,8-tetrachlorodibenzo-p-dioxin (2,3,7,8 TCDD) toxicity equivalent and 14 congeners
- PAH: Benzo(a)anthracene, benzo(a)pyrene, benzo(b)fluoranthene, benzo(k)fluoranthene, chrysene, dibenz(a,h)anthracene, and indeno(1,2,3-cd)pyrene.

The following analytes were selected as preliminary constituents of interest in fish during the Phase I Fish Tissue Sampling Data Evaluation (CH2MHILL 2007):

- Metals: Aluminum, arsenic, cadmium, chromium, copper, iron, lead, nickel, selenium, uranium, zinc, and mercury
- PCBs: Total PCBs
- Dioxins and furans: 2,3,7,8-tetrachlorodibenzofuran (2,3,7,8 TCDF).

Barium was included as a preliminary constituent of interest to help to illustrate potential differences in species-specific exposure and accumulation.

Syracuse Resource Corporation, on behalf of the EPA, prepared the *Human Health Risk Assessment Work Plan for the Upper Columbia River Site Remedial Investigation and Feasibility Study,* dated March 2009 (EPA 2009). Syracuse Resource Corporation found several environmental media for which the currently available Lake Roosevelt data may not be adequate to support reliable risk calculations in a baseline HHRA. The preliminary risk estimates evaluated in the work plan are intended to be used to guide future data collection efforts and prioritize data needs for the baseline HHRA. As identified in the *Upper Columbia River Work Plan for the Remedial Investigation and Feasibility Study* dated December 2008 prepared by Teck Cominco American Incorporated and modified by EPA (EPA 2008), additional studies and supplemental data collection will be required to meet the data needs for the HHRA. Subject to EPA approval, the following field and laboratory studies were planned for 2009 and 2010, including collection of additional surface water, sediment and fish tissue samples, as well as conductance of recreational and Tribal resource use surveys and an evaluation of upstream sources of contamination.

A 2009 beach sediment study was also designed for Teck Cominco American Incorporated to ensure that the nature and extent of contamination in exposed beach surface and subsurface sediments is sufficiently well characterized to allow a reliable evaluation of potential risks to humans who may be exposed via direct contact (ingestion and dermal). The study, the sampling for which was initiated in 2009, is intended to expand and augment information provided by prior investigations, which include EPA's Phase I investigation and other historical studies of exposed sediments along the Upper Columbia River.

2.7.4 Columbia River Basin Fish Tissue Sampling

In 1994, the EPA and Columbia River Inter-Tribal Fish Commission's (CRITFC's) member tribes initiated a survey of contaminants in fish tissue in the Columbia River Basin. The contaminant survey was designed by a multi-agency group including the CRITFC, Ecology, the Washington State Department of Health, the Oregon Department of Environmental Quality and Health, the Confederated Tribes of Warm Springs, the Yakama Nation, the Confederated Tribes of the Umatilla Indian Reservation, the Nez Perce Tribe, the U.S. Geological Survey, and the U.S. Fish and Wildlife Service. Sample collection took place between 1996 and 1998 with the help of CRITFC's member tribes and staff from federal and state agencies. The results of this study were published by EPA in 2002, in EPA 910-R-02-006, *Columbia River Basin Fish Contaminant Survey 1996-1998.*

A total of 281 samples of fish and fish eggs were collected from the Columbia River Basin from 5 anadromous species (Pacific lamprey, smelt, coho salmon, fall and spring chinook salmon, and steelhead) and 6 resident species (largescale sucker, bridgelip sucker, mountain whitefish, rainbow trout, white sturgeon, and walleye). The following four types of samples were collected: whole-body with scales, fillet with skin and scales, fillet without skin (white sturgeon only), and eggs. All the samples were composites of individual fish, except white sturgeon. The number of fish in a composite varied with species, location, and tissue type. Eleven samples of eggs were collected from steelhead and salmon.

The fish tissues were analyzed for pesticides, metals, PCB Aroclors, PCB homologues, dioxin and furan congeners, and other organic chemicals. The results of the study showed that all species of fish had some levels of chemicals in their tissues and in the eggs of salmon and steelhead. Of the 132 chemicals analyzed in this study, DDE, PCBs, zinc, and aluminum had the highest detected concentrations in most of the fish tissues sampled throughout the basin.

The distribution in contaminant type and concentration across sample stations was variable, although fish collected from the Hanford Reach of the Columbia River and the Yakima River tended to have higher concentrations of organic chemicals than other study sites.

Using EPA's risk assessment models, fish tissue analytical results were used to estimate noncancer hazard indices and cancer risks from fish ingestion. For adults, hazard indices and cancer risks were lowest for the general public at the average ingestion rate and highest for the CRITFC member tribes at a higher ingestion rate.

Chemicals that contributed the most to the hazard indices and cancer risk are the persistent bioaccumulative chemicals (PCBs, DDE, chlorinated dioxins, and furans) as well as inorganic constituents including arsenic and mercury. Many of the chemical residues in fish identified in the EPA study were not unlike levels found in fish from other studies in comparable aquatic environments in North America. The concern raised in the Columbia River Basin also gives rise to a much broader issue for water bodies throughout the United States. The results of the EPA study, therefore, have implications not only for tribal members but also for the general public.

Results from this study are briefly summarized and compared to fish tissue data from the RI in Section 3.6.4.5.

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3.0 DATA EVALUATION

The objective of the data evaluation is to present the relevant sampling data used in the baseline HHRA, discuss the use of such data, discuss the nature and extent of constituents, and select the COPCs for each medium (e.g., island soil, sediment, surface water, and fish tissue). The COPCs are the chemicals and radioisotopes that are carried through the quantitative risk assessment.

In addition to samples collected from the Hanford Site Study Area (i.e., 100 Area, 300 Area, and Lake Wallula Sub-Areas), samples have been collected from the Upriver Sub-Area, major tributaries to the Columbia River (i.e., Snake, Yakima, and Walla Walla Rivers), and agricultural drainage wasteways/irrigation return channels to document the concentration of constituents that may be present in these media, but whose presence is unrelated to Hanford Site releases. This data evaluation therefore also includes a discussion of the island soil, sediment, surface water, and fish tissue data that have been collected within the study area with respect to concentrations of contaminants observed in reference areas.

3.1 THE COLUMBIA RIVER COMPONENT DATABASE AND HUMAN HEALTH RISK ASSESSMENT DATA SET

A significant amount of historical environmental data exist for the Columbia River, dating back to the 1940s and reflecting a number of individual state and federal monitoring programs and studies. These data have been extensively examined and a subset, considered as usable for characterization purposes, has been compiled into an electronic database referred to as the "CRC database" (WCH-64, *Existing Source Information Summary Report Compilation/ Evaluation Effort: December 2004 to September 2005, Columbia River Component of the River Corridor Baseline Risk Assessment*; WCH-91, *Columbia River Component Data Evaluation Summary Report*).

The data used for the HHRA, identified herein as the "HHRA data set," were drawn from a wide variety of sources, reflecting the high level of monitoring and assessment historically associated with the Columbia River and the Hanford Site. The final data set used for this HHRA, presented in electronic format in Appendix B, is composed of island soil, sediment, surface water, and fish tissue data collected during the remedial investigation ("RI" data), which were collected between 2008 and 2010 as part of the RI field effort, and select "historical" sediment and surface water data, which were collected as part of other studies conducted between 2000 and 2007 (see Section 3.3). The data from these two time periods are described separately in Section 3.2 (2008 to 2010 data) and Section 3.3 (pre-2008 data). Because both recent and historical data are combined, the HHRA data set is expanded from that described in the Data Summary Report (WCH-398), which focused exclusively on the more recent RI data. The number of samples included in the HHRA data set (from 2000 to 2010) by area and medium is summarized in the table below.

Medium	Upriver/ OCI Samples [a]	100 Area Sub-Area Samples [a]		300 Area Sub-Area Samples [a]		Lake Wallula Sub-Area Samples [a]		Total (Not Including OCI)
		2000 - 2007	2008 - 2010	2000 - 2007	2008 - 2010	2000 - 2007	2008 - 2010	
Surface water	53	27	45	46	19	6	20	163
Sediment	126	0	133	2	160	21	156	472
Soil	10	0	29	0	48	0 [b]	0 [b]	77
Fish tissue [c]	30	0	35	0	35	0	31	101

[a] Includes duplicate samples.
[b] No islands were sampled in this sub-area; thus, no island soil data are available.
[c] With the exception of sturgeon, fish samples were obtained from a composite of multiple individual fish. Fish samples were further divided by tissue type: fillet, liver/kidney, viscera (sturgeon only) and carcass. Viscera data were not used in this human health risk assessment and these samples are therefore not reflected in the number of samples listed in this table.

OCI = other contributing influences, i.e., reference areas

3.2 2008 TO 2010 REMEDIAL INVESTIGATION DATA

As part of the Columbia River RI, a large and comprehensive field program was conducted during the years 2008 to 2010 to document conditions in the surface water, sediment, soil, and fish tissue of the Columbia River adjacent to, upstream, and downriver from the Hanford Site. This effort was conducted in accordance with DOE/RL-2008-11, *Remedial Investigation Work Plan for Hanford Site Releases to the Columbia River* (RI Work Plan), which produced a comprehensive data set that reflects the use of the common collection techniques, analytical methods, laboratories, staff, and other parameters that could otherwise introduce variability between unrelated sampling programs. In consequence, the RI program produced a high-quality and consistent data set that provides an accurate depiction of current conditions in the Columbia River, as discussed in WCH-381, *Data Quality Assessment Report for the Remedial Investigation of Hanford Site Releases to the Columbia River, Hanford Site, Washington, June 2010.* The RI data set thus forms the primary basis of the risk assessment data set used to support both the HHRA and the SLERA. However, as described in Section 3.1, the HHRA data set also includes select historical data collected between 2000 and 2007.

The RI data, described in detail in the Data Summary Report (WCH-398), were obtained from samples from the Columbia River, as well as from islands and along left-bank (facing downriver) shorelines and right-bank areas outside of the RCBRA study area, as previously discussed in Section 1.0. The RI data collection effort was composed of three separate components:

- 2008 to 2009 Surface water, sediment, and soil sample collection
- 2010 Groundwater upwelling investigation (co-located porewater, surface water, and sediment sample collection)
- 2009 to 2010 Fish tissue sample collection.

The scope of the 2008 to 2010 RI sampling program was based on the outcome of the DQO process (WCH-265, *DQO Summary Report for the Remedial Investigation of Hanford Site Releases to the Columbia River*) to address the data needs. The rationale for the sampling

approach and strategy are detailed in the RI Work Plan (DOE/RL-2008-11). Appendix A to the RI Work Plan is the SAP that describes the sampling activities. Requirements for sampling methods, sample handling and custody, and analytical methods are detailed in WCH-286, *Sampling and Analysis Instructions for the Remedial Investigation of Hanford Site Releases to the Columbia River* (SAI). The RI Work Plan, SAP, and SAI directed the sample collection methods and locations.

The 2008 to 2010 RI field activities associated with the collection of sediment, river water, and island soil are documented in WCH-352, *Field Summary Report for Remedial Investigation of Hanford Site Releases to the Columbia River, Hanford Site, Washington (Collection of Surface Water, River Sediments, and Island Soils)*. WCH-352 provides a description of the sampling locations, identification of samples collected, and a description of modifications and additions made to the SAP.

In addition, a groundwater plume upwelling survey was completed to delineate areas of groundwater plume upwelling into the Columbia River for subsequent sampling. During Phase III of that study, co-located porewater, surface water, and sediment samples were collected from 49 stations identified previously in the RI as being areas of groundwater upwelling. The groundwater upwelling field activities and data collection are documented in WCH-380, *Field Summary Report for Remedial Investigation of Hanford Site Releases to the Columbia River, Hanford Site, Washington: Collection of Surface Water, Pore Water, and Sediment Samples for Characterization of Groundwater Upwelling*.

The RI field activities associated with the collection of fish tissue samples are documented in WCH-387, *Field Summary Report for Remedial Investigation of Hanford Site Releases to the Columbia River, Hanford Site, Washington: Collection of Fish Tissue Samples*.

While detailed information about the RI sampling program methodology and results is contained in the documents listed above, a brief summary of the data collected is provided below, by medium. Section 3.6 describes the nature and extent of contaminants detected in the various media.

3.2.1 Remedial Investigation Island Soil Sample Collection

Island soil samples represent sediment that has historically been transported during high river conditions from other portions of the river onto islands. Island soil samples were collected from the Upriver Sub-Area, 100 Area Sub-Area, and 300 Area Sub-Area, specifically Island 3, Locke Island, White Bluffs, Homestead Island, Wooded Island, Johnson Island, Island 19 (Gull Island), and an unnamed island in Wanapum Pool (Upriver reference). These samples were collected from areas above the normal zone of inundation and were composed of soil to a maximum depth of 9 cm (0.3 ft) below the soil surface. No island soil samples were collected from islands within the Lake Wallula Sub-Area.

The sampling approach for island soils was similar to a stratified-random sampling design. For island soils, the target "population" is the river-transported sediments from the Hanford

Study Area. To ensure that samples were representative of this population, a single-cell sample grid was established prior to sample collection. Samples were collected at random locations within each grid cell. This random sampling enhances the representativeness of these samples for the population.

A total of 87 island soil samples were collected as part of the RI and used in the HHRA.

3.2.2 Remedial Investigation Sediment Sample Collection

Sediment sampling consisted of the collection of sediment from varying water depths in the river channel, shoreline sediment, and sediment cores. Sediment was also collected as part of the groundwater upwelling investigation.

A stratified random approach was used for the design of the sediment sampling program. Because most of the river bottom consists of coarse to medium gravel, a fine-grained sediment survey was conducted prior to the selection of sample locations to identify depositional areas where fine-grained material is present in quantities sufficient for sampling. The survey was conducted by sonar, which was initially verified by petite ponar sediment collection to verify the accuracy of the technique. In the subsequent RI, all sediment samples were collected within these pre-identified areas of fine-grained sediment deposition, which comprise the population "strata" for statistical purposes.

To ensure that samples are representative of the sediment at each location, sampling locations must be positioned within a sampling grid designated as part of the sample design. Because of the nonhomogeneous distribution of fine-grained sediment within the river, however, each area of fine-grained deposits was considered to be a single cell from a sample design grid, and the exact location of sample collection was selected at random from within the cell.

A total of 598 sediment samples were collected as part of the RI and used in the HHRA.

"Shallow" sediment samples were collected from shallow water, less than 1.8 m (6 ft) in depth. The samples consisted of the upper 10 cm (4 in.) of sediment near island and river shorelines, and from the shallow areas of irrigation returns, tributary deltas (Yakima, Snake, and Walla Walla), and other depositional areas between the Hanford Site reactors and McNary Dam.

"Deep" sediment samples were collected in deep water, in areas where water depth was greater than 1.8 m (6 ft). These samples consisted of the upper 10 cm (4 in.) of sediment from deep water areas of the Columbia River, as well as depositional areas upriver of the Yakima River confluence and downriver of the Walla Walla River confluence.

Shoreline sediment samples were collected from downriver islands and along the left (non-Hanford Site) bank within the Hanford Site Study Area. These samples were collected from the lower riparian zone, defined as the area devoid of terrestrial vegetation and inundated on a daily basis by water-level fluctuations.

Shallow sediment core samples were collected using a vibracore drilling tool in selected sediment deposits that were generally thinner than 3 m (10 ft) thick in total. Sampled sediment deposits potentially date back to reactor operations and were located at selected reactor water intake structures at the 100-B/C, 100-K, 105-N, and 100-D Reactors; the head end of the Lake Wallula pool (near the 300 Area); and the Yakima and Snake River deltas.

Deep cores were completed at water depths of up to 27 m (90 ft) with anticipated thick sediment sequences greater than 3 m (10 ft) thick. Deep sediment cores were collected from areas in Lake Wallula (Port Kelley, Hat Rock, and just upriver of McNary Dam) where sediment deposits may date back to the era of reactor operations. Core samples were collected at depths up to 3.4 m (11.3 ft) below the sediment-water interface.

Lastly, sediment was collected as part of the groundwater upwelling investigation. These samples are not considered statistically random samples; they are "judgmental samples" meant to focus on specific areas of concern, as described in the RI Work Plan (DOE/RL-2008-11). Sediment was collected from the top 10 cm (4 in.) of sediment in areas previously determined to be the zone where Hanford Site groundwater plumes discharge to the surface water of the Columbia River. These samples, which were collected within the discharge plume, are designated as "groundwater upwelling" in the database and are co-located with porewater and surface water samples.

3.2.3 Remedial Investigation Surface Water Sample Collection

Surface water samples were collected from the 100 Area Sub-Area (reactor areas); 300 Area Sub-Area; recreational locations (parks and boat launches); Lake Wallula; McNary Dam; irrigation returns; and tributary deltas at the Yakima, Snake, and Walla Walla Rivers. Upriver Reference samples were collected from above Priest Rapids Dam. The target population was considered to be upriver surface water above Priest Rapids Dam, and sample locations were identified at random within this area. Within the river, surface water samples were collected at approximately two-thirds of the depth of the water column, and within the irrigation wasteways samples were collected approximately 15 cm (6 in.) from the surface. Within each of these areas, samples were collected in a stratified random method, so as to be representative of the area of interest.

Two surface water sampling events (fall and spring) were conducted. The fall sampling event occurred between October 16 and November 13, 2008, and the spring sampling event occurred between June 1 and 9, 2009.

In addition to these surface water samples, several judgmental or focused samples were collected. These were deep samples taken near sediment and in areas of groundwater upwelling. Three deep surface water samples were collected during RI field sampling activities directly above the riverbed within Lake Wallula, downriver of the Walla Walla River confluence, and behind McNary Dam. Surface water samples were also collected as part of the groundwater upwelling investigation. For this evaluation, surface water was collected from within a foot of

the sediment surface in areas of documented plume release. Sediment and porewater samples were collected at the same locations.

A total of 216 RI surface water samples were collected for the RI and used in the HHRA. The vast majority of these samples were collected in a random manner; a small fraction are focused samples. Thus, the final data set is a mixture of sample types. For purposes of characterizing human health risk, only unfiltered (i.e., total) metals data, rather than filtered, were considered relevant to and used in the HHRA. This is consistent with EPA guidance (EPA/540/1-89/002, *Risk Assessment Guidance for Superfund, Volume I: Human Health Evaluation Manual [Part A], Interim Final*) for potable water supplies. The Columbia River is a potable source of water for a number of communities. The use of only total metals data (e.g., unfiltered), however, is a conservative assumption; the water from the river is treated and filtered prior to its distribution as a municipal potable supply (e.g., Richland).

3.2.4 Remedial Investigation Fish Tissue Sample Collection

In accordance with the SAP (DOE/RL-2005-42), specimens of the following six fish species were collected as part of the fish sampling program:

- Common carp (*Cyprinus carpio*)
- Mountain whitefish (*Prosopium williamsoni*)
- Walleye (*Stizostedion vitreum*)
- Smallmouth bass (*Micropterus dolomieui*)
- Bridgelip sucker (*Catostomus columbianus*)
- White sturgeon (*Acipenser transmonatnus*).

These six fish species are year-round resident fish that reflect a range of trophic levels and have a higher rate of harvest and consumption among the local population. As described in the RI Work Plan (DOE/RL-2008-11), salmon were not sampled as part of this study because they spend a majority of their life cycle in the ocean as opposed to the Hanford Site Study Area and therefore are not representative of local river conditions.

The numbers of fish tissue samples collected during the 2009 to 2010 sampling event and used in the HHRA data set are shown in the table below.

		Carp	Whitefish	Walleye	Bass	Sucker	Sturgeon
Upriver Sub-Area	Individual fish	21	27	25	25	25	5
	Composites	5	5	5	5	5	0
	Number of samples	5	5	5	5	5	5
100 Area Sub-Area	Individual fish	25	25	26	25	25	9
	Composites	5	5	6	5	5	0
	Number of samples	5	5	6	5	5	9
300 Area Sub-Area	Individual fish	25	27	25	25	25	10
	Composites	5	5	5	5	5	0
	Number of samples	5	5	5	5	5	10

		Carp	Whitefish	Walleye	Bass	Sucker	Sturgeon
Lake Wallula Sub-Area	Individual fish	25	26	27	25	25	6
	Composites	5	5	5	5	5	0
	Number of samples	5	5	5	5	5	6
Total number of samples per species	**131**	20	20	21	20	20	30

For all species except sturgeon, fish tissue samples were composite samples composed of tissue from approximately five fish. Generally, five samples of each fish species were collected from each sub-area, and each sample included separate fillet, carcass (which included the head and skeleton of the fish), and combined liver and kidney tissue for analysis. For carp, sufficient tissue mass was available to obtain separate liver and kidney samples. Fillet samples for all of these species except sturgeon were prepared with the skin on, since skin for these types of fish is often left on during preparation and consumed.

Sturgeon samples were not composited, and thus samples represent tissue from individual fish. Sturgeon fillet samples were collected with the skin off, and separate liver and kidney samples were prepared. Twenty-five sturgeon were collected from the 100 Area, 300 Area, and Lake Wallula Sub-Areas, while five reference sturgeon were collected from upriver of Wanapum Dam.

For both the Study Areas and the Reference Area, fish samples were obtained from where they were available, rather than at specific sampling points. The constraints of fish sampling make it impractical to conduct sampling in a statistically random fashion. The degree to which fish collections are representative of the population of fish is unknown. Thus, the fish sample analytical results are considered to be suitable for statistical comparisons.

Analytical results from the RI fish tissue study were used to support the HHRA.

3.2.5 Remedial Investigation Analytical Methods and Reported Results

As detailed in the Data Summary Report (WCH-398), samples from all media were analyzed for a wide variety of constituents. Analyses varied somewhat by medium and sampling objective, but typical analyses for most constituents included metals, hexavalent chromium, total petroleum hydrocarbons (TPH), PCBs (as Aroclors and individual PCB congeners), pesticides, radionuclides, VOCs, and SVOCs. Surface water samples included both dissolved and total metals analysis. All fish tissue samples were analyzed for PCB congeners, metals, pesticides, and radionuclides. Fillet and carcass samples were analyzed for total inorganic arsenic (TIAS) in addition to total arsenic. Sturgeon samples were also analyzed for methyl mercury and hexavalent chromium. Specific analytical details for all medium types are provided in the Data Summary Report (WCH-398). Table 3-1 presents a general summary of parameters analyzed for each medium.

Fish tissue results are reported in wet weight; all other solid media results are reported in dry weight. Sediment results were received from the laboratory in wet weight and converted to dry weight using percent moisture data, as described in the Data Summary Report (WCH-398). The RI effort produced a large, consistent, and high-quality data set focused specifically on the needs of the risk assessment. For this reason, these RI data formed the bulk of the data used to evaluate river conditions in the HHRA. However, historical data were also reviewed for usability in the risk assessment; the data incorporated from these sources are described in the following section.

3.3 PRE-2008 HISTORICAL DATA

Pre-2008 historical data were compiled into a single database as part of the effort for the 2007 CRC data gap analysis (WCH-201, *Columbia River Component Data Gap Analysis*). The combined database created for the data gap analysis consisted of data from the following sources:

- The original CRC database
- U.S. Army Corps of Engineers (ACOE) data
- 2003 to 2006 data used to support the RCBRA
- Mid-Columbia River sediment data provided by the EPA Region 10 Watershed Restoration Unit on June 8, 2007
- Annual reports from Pacific Northwest National Laboratory (PNNL) through 2009.

As described in WCH-201, the original CRC database was a compilation of data obtained from the detailed data collection effort conducted as part of WCH-64 and WCH-91. As part of those efforts, data were obtained, reviewed, and selected by a team composed of researchers from universities, PNNL, WCH, and a Native American consulting firm through a process that involved extensive review and input by DOE, Trustees, and regulators. The extensive details of the data collection and evaluation method are provided in those documents, particularly WCH-64, and specific decisions about what data to include or exclude were made by those researchers. Data quality was categorized into tiers, and only Tier I data, the highest quality category, were retained for use in the SLERA and this HHRA.

Prior to use, the historical data set for each medium was reviewed on a sample-by-sample basis to identify samples appropriate for use in the risk assessments. For all media, samples were omitted if they were collected from outside the boundaries of the study area (Section 1.2) or included in the RCBRA evaluation of the near-shore area of the river. Other factors used to select historical samples are described by medium, below. Historical data exist for sediment, surface water, and fish tissue, but not soils, in the area of study.

3.3.1 Historical Sediment Data

Sediment sample results were reviewed for the period 1990 through 2007 to determine comparability and consistency with the RI data set. Specific characteristics that were reviewed included data reporting practices (e.g., consistent units, nomenclature issues, duplicate reports), categories of constituents analyzed and detected, the relative number of samples, and the frequency of detection (FOD) and concentration of constituents relative to the RI data set. This evaluation was conducted separately for the 1990 through 2007 data set as well as for the more recent subset of 2000 through 2007 analytical results. The goal of the separate evaluations was to determine if sediment conditions, as reflected by the historical data, had remained consistent over the last 20 years or had changed enough to warrant the use of more recent data.

In general, the analysis showed that the sediment data from 2000 and later were more comparable in concentration and detected constituents to the RI data than the older data from 1990 to 2000. This reflects the river as a dynamic system, where daily flow changes and periodic flooding continually transport material and, in general, realign the sediment characteristics with the changing array of Hanford and non-Hanford Site discharges that influence sediment chemical composition. Certain constituents, such as various heavy metals, appeared to have declined in concentration, presumably due to a reduction in upstream mining activities. In addition, the process of radioactive decay will naturally reduce the concentrations of short-lived radionuclides over time. The results of the data review show that river conditions for the last 10 years have been relatively consistent, and suggest that conditions reflected by the older (pre-2000) sediment data no longer exist in the river. For this reason, only historical sediment data from 2000 forward were retained for use in the HHRA, since these data were shown to be more representative of current conditions in the river than data from 10 or more years ago.

In summary, the historical sediment data used for the HHRA consisted of selected data from year 2000 through 2007.

3.3.2 Surface Water Historical Data Review

Surface water analytical results were reviewed in a manner similar to that of sediment with regard to sample location and data characteristics. As with sediment, surface water records were reviewed on a sample-specific basis, and some samples were removed due to locations or sample content. Samples removed from the data set consisted of samples not collected from the Columbia River or nearby tributaries; samples collected from Hanford Site Study Area springs, seeps, sloughs, or other source areas; and samples from the right bank of the Columbia River in the Hanford Site Study Area (which was addressed as part of the RCBRA).

For this review, surface data collected prior to the year 2000 were not included in either the review or the resulting risk assessment data set, for the following reasons:

- As an inherently transient medium, surface water most accurately reflects recent conditions and current influences on water quality. Thus, current river conditions, and resulting risks, are more accurately estimated by the use of recent surface water data.

- The results of the sediment analysis suggest that conditions in the river have remained relatively consistent over the last 10 years. Thus, surface water from this time period is expected to be similarly consistent, and so these historical data were included to provide a robust data set that captures a variety of seasonal and flow conditions.

Thus, historical surface water samples from 2000 through 2007 were included in the HHRA data set.

3.3.3 Fish Tissue Historical Data Review

Fish tissue has been a part of monitoring at the Hanford Site for many years, resulting in a wide variety of species and fish tissue in the database of historical samples. As mobile and relatively long-lived components of the river biota, fish may reflect the conditions in both surface water and sediment during the years they live in the river and may be appropriate monitors of bioaccumulative constituents over time. Thus, while the analytical results from a large number of fish samples collected as part of the RI comprise the bulk of the data set, selected historical fish tissue samples were evaluated for inclusion in the HHRA as well.

Within the historical fish tissue data set, there is considerable inconsistency in species evaluated, tissue type (whole body, fillet, skin on, skin off, etc.), and analytes. Additionally, multiple collection and analysis approaches, as well as variability in bioaccumulation among species and age of specimen, have introduced significant variability in analytical results. Because of these inconsistencies, it was determined that these older data were not suitable for combining with the RI fish tissue data set and not usable in assessing current or future human health risks from fish ingestion in the HHRA.

The RI fish sampling program was specifically designed to support the HHRA and provided a consistent sampling and analysis approach among species, tissue types, and analytes. Therefore, fish tissue data from only 2009 to 2010 were used in this HHRA.

3.4 SUMMARY OF RISK ASSESSMENT DATA

As described above, the data used for this HHRA are composed largely of the data produced by the 2008 through 2010 RI sampling, supplemented as appropriate by historical sediment and surface water data collected from 2000 and later. To summarize, the data spans for each medium are shown in the following table.

Medium	Data Set Date Range
Surface water	2000 - 2010
Sediment	2000 - 2010
Island soil	2008 - 2010
Fish tissue	2009 - 2010

The location of all surface water, sediment, island soil, and fish tissue samples used in the HHRA is provided in Appendix C.

The appropriateness of using statistical methods on the HHRA data set was considered. Inferential statistical methods are typically specified for use with random samples. The sample design of the data set for each media (soils, sediment, surface water, and fish tissue) was reviewed to identify any influence the sample design and data set characteristics, described in Section 3.2, may have on the statistical outcome. Sample design is reviewed by medium below.

Soils were randomly collected from a single-cell grid in general areas of interest; they were collected in a manner similar to a stratified random sampling design. The strata are composed of the separate islands, within which the sample locations were randomly identified. Not all islands were sampled; however, the data are suitable for use in statistical analyses.

The sediment samples collected during the RI were from locations randomly selected within general areas of interest. Sediment samples were collected according to a stratified random design, with the depositional areas targeted for sampling representing the individual "strata" for analysis. This data set also includes historical samples; these are judgmental samples, also called 'focused' samples in the RI Work Plan (DOE/RL-2008-11).

Surface water samples in the Upriver Sub-Area were collected according to a random design, where sample locations were selected at random from within the area upriver of Priest Rapids Dam. Within the downriver sub-areas, samples were collected at random from within general areas of interest, which were typically areas where data were lacking. The surface water data set also includes focused samples. These are historical data, samples collected near groundwater upwelling, and samples collected proximate to the sediment. All these sample locations are described in detail in the RI Work Plan (DOE/RL-2008-11).

Due to the sampling practice of obtaining fish from where they were available rather than at specific sampling points (see Section 3.2.4), and the use of the same approach in both Study and Reference areas, these samples are considered to be suitable for statistical comparisons.

The influence of the focused samples on the entire data set was considered. Concentrations in focused samples are assumed to be higher than elsewhere, since these areas consisted of reactor outfalls, groundwater plume discharge areas, and other locations of known or suspected contaminant presence. Under these circumstances, the effect of including focused samples in the otherwise random data set would be to over-estimate the magnitude of the means and variances in the Study Area data. When comparing Study Area data to Reference data, this potential bias would increase the chance that the null hypothesis (that samples in the two groups come from the

same underlying population) would be rejected. Thus, concentrations between Study Area and Reference/OCI are more likely to be designated as different, when in fact they may be the same. This introduces bias of a conservative nature. Additionally, targeting areas of known contamination will bias high estimates of exposure, resulting in overestimation of risk. Thus, the conservative bias introduced through the sampling program is considered to be acceptable for purposes of characterizing exposure and risk.

3.5 METHODOLOGY FOR DATA USE AND EVALUATION

The general treatment of the HHRA data is summarized in the following subsections.

3.5.1 Interpreting Analytical Results

Analytical results used in this HHRA are based on the results reported by the analytical laboratory. There are two types of results that are used in the risk assessment: detect and nondetect (i.e., censored) results.

3.5.1.1 Chemical Analysis. For nonradionuclides, each result involves a laboratory reporting limit (LRL) (this may also be referred to in laboratory reports as an estimated quantitation limit). The LRL is the lowest concentration that can be reliably reported within the specified limits of precision and accuracy during routine laboratory operating conditions and is unique to each sample and compound (SW-846, *Test Methods for Evaluating Solid Wastes, Physical/Chemical Methods*).

Note that the LRL is a value different from the instrument detection limit (IDL) and method detection limit (MDL). The IDL is a concentration equivalent to an instrument signal due to the analyte of interest that is equal to a multiplier of the standard deviation of a series of replicate measurements of a reagent blank's signal, measured at the same response (SW-846). In effect, the IDL determines the baseline background "noise" of an analytical instrument for the specific analyte of interest. The IDL determinations are typically made using reagent water and do not incorporate any potential effects or the components on the analytical instrument (i.e., matrix effects). The IDL is then typically used to estimate a likely MDL.

The MDL is the minimum concentration of a constituent that can be measured and reported with 99% confidence that the analyte concentration is greater than zero. The MDL is determined from analysis of a given matrix type containing the analyte at a level derived from the IDL (SW-846). It is standard laboratory practice to perform MDL studies with spiked reagent water or simple solid matrix materials (e.g., silica sand). However, MDLs are unique to a method and not a particular sample.

The laboratories providing data for this report use limits of detection to define the detect/nondetect decision point during analyses, in accordance with the National Environmental Laboratories Accreditation Conference 2003 Standard (EPA/600/R-04/003).

Limits of detection are derived from MDLs adjusted for potential real-life sample effects on the analytical process. Analytical results reported as detected below the LRL reflect the presence of the analyte but with less precision and/or accuracy than results reported at or above the LRL. These results are flagged to identify the as lower precision/accuracy values.

The LRLs are sample-specific and are highly matrix-dependent; as a result, the LRL of a given sample may be 5 to 10 times higher than the MDL. For many analytes, the base LRL analyte value is selected as the lowest nonzero standard in the calibration curve. For reporting of actual sample results, base LRLs are adjusted if necessary to account for sample-specific parameters (e.g., initial aliquot quantity, conversion to dry-weight reporting, additional instrument dilutions). For nonuniform matrices, such as sediment in particular, the LRLs within a sample group may vary substantially; it is not uncommon to have individual sample results within a sample delivery group with a 10-fold difference in LRLs.

For chemical data used in this risk assessment, positive chemical results are those results reported at or above the limit of detection, and nondetect results (U-qualified) are reported at the LRL. The LRL is used in generating statistics for nondetect results.

3.5.1.2 Radionuclide Analysis. Radionuclides are reported relative to a minimum detectable activity (MDA) rather than an LRL. The MDAs are established based on analytical detector baseline instrument activity (background). The MDA establishes a statistical confidence that radionuclide activity is present in the sample (i.e., detected versus not detected). Radionuclide analytical results can be positive, negative, or zero. Results above the MDA are treated as detected, results below the MDA as nondetected (i.e., censored). Positive results below the MDA and negative results were used without modification in a manner similar to that of detected results in generating the various statistics employed in the HHRA.

3.5.2 Units

Nonradionuclide chemical results are presented in units of micrograms per liter (µg/L) for aqueous media and milligrams per kilogram (mg/kg) for solid media. For the solid media, island soil and sediment results are expressed on a dry-weight basis, whereas fish tissue results are expressed on a wet-weight basis.

Radionuclide results are presented in units of activity (pCi) per volume in liters (L) for aqueous media (porewater and surface water) or units of activity (pCi) per mass in gram (g) for solid media (soil, sediment, and fish tissue). Although radionuclide results are expressed on an activity (mass or volume) basis, the results are colloquially referred to as "concentrations" in this report.

3.5.3 Data Qualifiers

Sample results include various levels of data validation. With the exception of samples qualified as rejected ("R"-flagged), all U- (nondetect) and J- (estimated) qualified data were considered to

be usable for purposes of risk assessment. Data that had been qualified as "rejected" during the data quality assessment process were omitted from the data sets. The data assessment process and resulting data qualification actions for the RI data set are described in WCH-381. Sample results qualified in any other way (e.g., estimated values qualified with a "J") were used as reported in this statistical analysis.

3.5.4 Duplicate Samples

Duplicate samples (or split samples) are two samples taken from the same medium and sample locations and are processed and analyzed identically. Duplicates are collected as a means of evaluating sample reproducibility. Relative percent difference (RPD) between the concentration in the primary and duplicate samples is used to evaluate reproducibility. As discussed in the Data Quality Assessment Report (WCH-381), the majority of the RPDs calculated (86%) were within the evaluation criteria.

Where both a primary and duplicate sample was collected, results from only the primary sample were used. There is the potential for a concentration to be higher in the duplicate sample than in the primary sample. However, because the number of duplicate samples was relatively small compared to the total number of samples within the entire HHRA data set, and RPDs were within acceptable limits within the overall data set, the exclusion of duplicate results is unlikely to bias sample results.

3.5.5 Duplicate Analyses

Where a constituent was analyzed in a particular sample via more than one analytical method (e.g., naphthalene may be analyzed by VOC and/or SVOC analyses), the maximum detected result or minimum LRL among the various results was used as the representative concentration for that constituent (i.e., the value used for a sample in calculating statistics, such as the arithmetic mean).

For radionuclides analyzed by both gamma spectroscopy and plate methods (e.g., uranium-235, uranium-238, thorium-228, and thorium-232), the values reported for the plate analysis were used in the lieu of the gamma values, because the plate methods had overall lower MDAs and, consequently, fewer censored results.

3.5.6 Censored Data (Chemical Constituents)

Censored data (i.e., results reported as nondetected at or above the LRL) were evaluated as part of the HHRA.

Use of censored results in generating summary statistics is often problematic as the constituent may be present at levels just below the LRL or may not be present at all. Sample-specific reporting limits potentially may be elevated relative to typical LRLs among other sample results and, in some cases, may even be reported as higher than detected results. Consequently, reporting limits of censored results were evaluated with respect to maximum

detected concentrations within a medium as well as the DQOs set forth in the RI Work Plan (DOE/RL-2008-11). Nondetect results were determined to be adequate for use in the HHRA. Where a large number of elevated LRLs were present within a medium, a discussion of this condition was addressed in the uncertainty analysis (Section 7.0).

Use of censored data in derivation of summary statistics and EPCs (i.e., arithmetic mean, 95% upper confidence limit [UCL] of the mean) is treated by a process described in Section 3.5.8 and uses EPA-published methods that are included in the latest version of the EPA ProUCL software program.

3.5.7 Polychlorinated Biphenyl Congener and Dioxin/Dibenzofuran Data

In the 2008 to 2009 RI sampling event, PCBs were analyzed for in various media via two different methodologies: one provides results for individual Aroclor mixtures (e.g., Aroclor-1260), while the other provides results for PCB congeners. Congener analysis is a more sensitive analytical method than Aroclor analysis that provides more accurate quantification of PCB concentrations, and has lower detection limits. Although Aroclors were infrequently detected among samples, PCB congeners were detected in all samples analyzed for this parameter. Rather than evaluate each of the 209 PCB individual congeners, results from PCB congeners were combined to calculate total PCB concentrations for use in the HHRA.

Polychlorinated biphenyls may be categorized as either "dioxin-like" or "nondioxin-like" in their toxicity. Congener results were used to calculate a total "dioxin-like" PCB concentration and a total "nondioxin" PCB concentration for each sample. Dioxin-like PCBs are those congeners that exhibit a toxicological mode of action common to chlorinated dibenzo-p-dioxins and dibenzofurans. Nondioxin-like PCB congeners were assumed to have similar toxicity and mode of action to PCB Aroclors, as further discussed in Section 5.0.

When calculating a total "dioxin-like" PCB concentration, it is assumed that each congener has a toxicity equivalent to some fraction of 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD), the most toxic TCDD. Individual congener concentrations (per sample) are first multiplied by a toxicity equivalency factor (TEF), if available, to calculate a weighted congener concentration. The dioxin TEFs for PCB congeners used in this HHRA are the values published by the World Health Organization (Van den Berg et al. 2006, "The 2005 World Health Organization Reevaluation of Human and Mammalian Toxic Equivalency Factors for Dioxins and Dioxin-Like Compounds"). These values, summarized in Table 3-2, are recommended for use in risk assessments by both EPA/100/R-10/005, *Recommended Toxicity Equivalence Factors (TEFs) for Human Health Risk Assessments of 2,3,7,8-Tetrachlorodibenzo-*p*-dioxin and Dioxin-Like Compounds*, and by *Evaluating the Toxicity and Assessing the Carcinogenic Risk of Environmental Mixtures Using Toxicity Equivalency Factors* (Ecology 2008). The individual TEF-weighted congener concentrations are then summed together to calculate a weighted "total TCDD equivalent" concentration. The PCB congeners without assigned TEFs are categorized as "nondioxin-like" and are not weighted with a TEF; rather, the individual congener results are summed to calculate a "Total Nondioxin PCB" value.

Some of the historical sediment samples were analyzed for individual chlorinated dibenzo-p-dioxins and dibenzofurans (CDDFs). As with PCBs, CDDF results for individual dioxins/furans were TEF weighted and summed to generate a total TCDD equivalent concentration in each sample. The TEFs are summarized in Table 3-2. In order to differentiate between dioxin-like PCBs and CDDFs in this HHRA, the TCDD equivalent concentration of PCBs is called "Total Dioxin-Like PCBs."

Individual congener or CDDF results reported as not detected (i.e., censored results) were included in the derivation of TCDD equivalent concentrations using Kaplan-Meier (KM) statistics rather than using simple substitution approaches. The KM method is not based on any assumptions about data distribution and is useful in addressing variable reporting limits (Helsel 2010, "Summing Nondetects: Incorporating Low-Level Contaminants in Risk Assessment"). See Section 3.5.8 for further discussion of this statistical method.

3.5.8 Summary Statistics

Data from each medium sampled (island soil, sediment, surface water, and fish tissue) were compiled and statistically summarized across the Hanford Site Study Area (i.e., 100 Area, 300 Area, and Lake Wallula Sub-Areas). Sets of summary statistics were calculated for all constituents detected at least once per medium and are presented in Tables 3-3 through 3-6 for sediment, surface water, island soil, and fish tissue (all species and tissue types), respectively. Tables 3-7 through 3-12 present summary statistics for individual fish species. The basic summary statistics that were calculated for each detected constituent included measures of detection frequency (number of detected samples, number of samples analyzed, FOD that equates to number detected/number analyzed), the arithmetic mean, the range of detected values, and location of the maximum detected concentration. Statistical analyses were completed using JMP® Version 8.0.2 (JMP), a commercially available statistical package by SAS Institute, Inc. The data obtained from the sampling events described in Sections 3.2 and 3.3 were downloaded from the HHRA database into either Microsoft® Access® or Excel® for initial processing and quality assurance checks, and then further analyses were completed in JMP.

The statistical methods used to calculate measures of central tendency were dependent on the number of samples collected and the FOD of each constituent. For constituents with a FOD of 100% in a given medium and river sub-area (and species, for fish tissue), standard statistical methods were used to calculate the mean.

Parametric statistical tests (e.g., two-sample *t*-test) assume, as a theoretical basis, that the two populations be normally distributed and have equal variances. Nonparametric tests do not require assumptions about the nature of the underlying distribution (e.g., normality). Parametric tests can therefore be more powerful, especially with environmental data that typically do not

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follow a normal distribution. For this evaluation, a mix of both parametric (where supported) and nonparametric tests were used, based on the underlying distribution of the data.

For constituents with the FOD between 30% and 100%, summary statistics were calculated using the KM estimation method. The KM method is used frequently in survival analysis for "right-censored" data, but can be successfully applied to "left-censored" data typical of environmental data sets (Helsel 2005, *Nondetects and Data Analysis: Statistics for Censored Environmental Data*; Helsel 2010). Left-censored data sets result from the inability to accurately report results below the practical quantitation limit. The KM method has a number of advantages over other methods (such as substitution methods, regression on order statistics, or maximum likelihood estimation) for calculating summary statistics for data sets with left-censored values. The KM method is nonparametric and therefore does not rely on the data set conforming to a specific underlying distribution, which is often the case for environmental data such as those evaluated herein. Additionally, the KM method is capable of being computed for data sets with multiple LRLs.

JMP® Version 8.0.2 implements the KM method through the use of a survival statistics platform. The statistical output includes estimates of the mean, standard error, 25th percentile, the median (50th percentile), and the 75th percentile. Some or all of these statistics may not be calculated for analytes where the FOD is very low (<5%) or when the number of samples is very small (<5).

Methods for calculating summary statistics when greater than 70% to 80% of the data are nondetects are generally considered unreliable or biased (Antweiler and Taylor 2008, "Evaluation of Statistical Treatments of Left-Censored Environmental Data Using Coincident Uncensored Data Sets: I. Summary Statistics"; Helsel 2005). In cases where the FOD of a constituent was less than 25%, an evaluation of the constituent was made based on a comparison of observed LRLs or MDAs and maximum detected concentrations. KM statistics were calculated and are presented in these cases. However, for low-frequency (i.e., 25% or fewer detects) data sets, KM statistics were not used for comparative statistical analyses (see Section 3.8.1.4).

Statistical summaries sometimes required computation of 95% UCL of the mean concentration (95% UCL). The 95% UCL calculations for the appropriate data sets for each medium of concern were generated using the EPA Technical Support Center for Monitoring and Site Characterization's ProUCL program, Version 4.00.05 (EPA/600/R-07/038, *ProUCL Version 4.00.05 User Guide*). This program computes an appropriate UCL of the unknown population mean using a distinct probability distribution (e.g., normal, lognormal, gamma) and/or an appropriate nonparametric method (EPA/600/R-07/038). Since this program calculates multiple parametric/nonparametric UCL values, the program-recommended UCL was used, unless more than one UCL was recommended. In such instances, the maximum UCL was selected. This approach is consistent with that described in the EPA Office of Emergency and Remedial Response document OSWER 9285.6-10, *Calculating Upper Confidence Limits for Exposure Point Concentrations at Hazardous Waste Sites*. ProUCL outputs are provided in Appendix D.

3.6 SUMMARY OF ANALYTICAL RESULTS

This section contains a description by environmental medium of the analytical results for the Hanford Site Study Area (i.e., 100 Area, 300 Area, and Lake Wallula Sub-Areas). Upriver and reference data are used in the HHRA to determine reference conditions; these data are addressed later in Section 3.8.

3.6.1 Sediment

Sediment analytical results for samples collected from 2000 to 2010 (all depths) are summarized in Table 3-3. This table shows constituents that were detected in at least one sample in sediment. Up to approximately 480 samples were collected per class of constituents (i.e., VOCs, SVOCs, PCBs, pesticides, metals/inorganics, and radionuclides), with most samples analyzed for heavy metals and radionuclides.

Of the VOCs, only acetone, toluene, and methylene chloride were detected in sediment, with methylene chloride being most prevalent (FOD of 25%). Both acetone and toluene were detected at much lower frequencies. The highest concentrations of acetone and methylene chloride were observed in the 100 Area Sub-Area at different river miles; the highest concentration of toluene, the other VOC detected, was observed in a sample collected from the Lake Wallula Sub-Area. All detected concentrations of VOCs were less than 1 mg/kg. Note that these constituents are typical laboratory contaminants. However, since no contamination was noted for the laboratory blank samples that corresponded to these samples, VOC sediment results were accepted as valid and usable for purposes of this HHRA.

Semivolatile organic compounds, such as phthalates and PAHs, were detected at a relatively low frequency (<5% of samples) and concentration (up to 2 mg/kg). Total petroleum hydrocarbon fractions (diesel range organics and high boiling motor oil) were detected in up to 40% of samples analyzed and at concentrations up to approximately 700 mg/kg. In general, SVOC concentrations were highest in the Lake Wallula Sub-Area; their presence is likely attributed, at least in part, to the high volume of recreational and commercial boating that occurs in this stretch of river, as well as roadway runoff and other anthropogenic sources of these types of contaminants.

Chlorinated pesticides (such as aldrin, heptachlor, and DDT) were detected at a relatively low (typically less than 10%) FOD at concentrations generally less than 0.1 mg/kg. Total PCBs (via congener analysis) were detected in 100% of sediment samples analyzed for PCBs. The highest reported dioxin-like PCB and nondioxin-like PCB concentrations occurred in samples from the Lake Wallula Sub-Area, with nondioxin-like PCBs approximately five orders of magnitude greater than the dioxin-like PCBs. Total nondioxin-like PCB concentrations in all samples across all sub-areas were less than 0.01 mg/kg. A review of the congener distribution, presented in the Data Summary Report (WCH-398), shows a similar fingerprint among samples in the 100 Area and 300 Area Sub-Areas, suggesting a similar source of PCBs to and/or similar degradation pattern within the river. Lake Wallula Sub-Area samples showed a slightly different pattern of congener distribution (WCH-398), which may be due in part to preferential sediment deposition

and input from major tributaries within this sub-area. TCDD (as Total TCDD equivalents) was detected in 63% of sediment samples at concentrations in the parts-per-trillion range. As with PCBs and pesticides, the highest TCDD concentrations were observed in the Lake Wallula Sub-Area. These data suggest a non-Hanford Site source of PCBs, dioxins, and pesticides to the river.

Metals were detected in most sediment samples. With the exception of antimony, bismuth, hexavalent chromium, selenium, silver, thallium, tin, and elemental uranium, metals were typically detected at a FOD at or greater than 50%. In general, the detected concentrations spanned approximately one to two orders of magnitude, and there does not appear to be an overall consistent pattern in contaminant distribution or concentration, although the highest concentrations were often observed in Lake Wallula sediment. Higher levels of known Hanford Site-related contaminants such as chromium, elemental strontium, and elemental uranium were observed in 100 Area and 300 Area Sub-Areas.

With the exception of cobalt-60, uranium isotopes, and cesium-137, most radionuclides were detected infrequently across the sub-areas. In shallow sediment, the levels of radionuclides such as plutonium, tritium, and uranium-238 are elevated in certain sub-areas within the Hanford Site Study Area and typically occur in the 100 Area and 300 Area Sub-Areas. Sediment core samples collected at depths greater than 0.3 m (1 ft) below the sediment-water interface from behind McNary Dam and at the head of Lake Wallula showed elevated concentrations of other radionuclides, such as cobalt-60 and europium-154, suggesting historical burial by sediment deposition and accumulation.

3.6.2 Surface Water

Table 3-4 presents a statistical summary of surface water data. As discussed, metals statistics presented in this table are based on unfiltered metals data. Similar to sediment results, VOCs and SVOCs were detected infrequently and at relatively low concentrations in surface water, whereas PCBs and metals/inorganics were detected at a relatively high frequency.

Chlorinated VOCs (e.g., TCE, 1,2-dichloroethane) were highest in the 100 Area and 300 Area Sub-Areas. Note that of the recent surface water data (2008 to 2010), no VOCs were detected in any samples.

Detected SVOCs consisted of several PAHs, bis-2-ethylhexylpthalate, di-n-butylphthalate, and TPH ranges (diesel and motor oil). The highest concentrations of TPH were observed in Lake Wallula, similar to sediment, whereas PAH concentrations were higher in the 100 Area and 300 Area Sub-Areas. It is likely that boat traffic in Lake Wallula is the likely source of TPH, as the highest concentrations were observed at samples collected by a marina.

No PCB Aroclors were detected in any surface water sample. Polychlorinated biphenyl congeners were detected in all four samples analyzed, however, at varying concentrations. Similar to the pattern observed in sediment samples, the types and concentrations of PCB

congeners detected are generally similar across the sub-areas (WCH-398). Highest total PCB concentrations were observed in the Lake Wallula Sub-Area.

Metals and other inorganic constituents such as nitrate and sulfate were detected in most surface water samples, at concentrations generally spanning one to two orders of magnitude. The distribution of detections and concentrations was variable among samples, although chromium, nitrate, and sulfate concentrations are somewhat elevated in the 100 Area and 300 Area Sub-Areas relative to levels observed downriver in Lake Wallula. The types of inorganics and the levels observed in the 100 Area and 300 Area Sub-Areas may potentially be related to the discharge of the nitrate and chromium groundwater plumes in this area of the river. The elevated nitrate concentration at sample location HL357 (B1L848) at RM 344 is likely associated with discharges from the fish aquaria in the 331 Life Sciences Laboratory in the 300 Area. However, there are numerous other (non-Hanford Site) sources of these constituents to the river, such as upstream industrial sources and adjacent agricultural sources, as well as naturally occurring geochemistry.

With the exception of strontium-90, tritium, and uranium isotopes, radionuclides were detected infrequently, in only 1% to 4% of all surface water samples. Highest levels of cobalt-60 and plutonium isotopes were observed in individual samples from the Lake Wallula Sub-Area, whereas maximum levels of other radionuclides occurred in either the 100 Area or 300 Area Sub-Areas. The positive results for the single Lake Wallula sample (LW-2SW) in which plutonium-238 and plutonium-239/plutonium-240 were detected may potentially reflect plutonium adsorbed to particles of suspended sediment in the surface water sample. Since this sample was collected approximately 0.3 m (1 ft) off the river bottom, the very act of sampling may have resuspended particles from the underlying fine-grained sediment. It is noted that americium-241 was not detected above the MDA in this sample and that the ratio of plutonium-238 to plutonium-239/plutonium-240 in this sample is not consistent with that produced at the Hanford Site, suggesting that plutonium may be related to atmospheric testing rather than to reactor production. This is further supported by the fact that plutonium-239/plutonium-240 was also detected in a sample (WBW-1SW-F; fall 2008) from the WB-5 irrigation return/wasteway (a Reference/OCI location), although at a lower activity level (0.234 pCi/L). Tritium levels in the 300 Area Sub-Area downstream of the Hanford townsite are elevated in an area of a known plume discharge.

3.6.3 Island Soil

Island soil data are summarized in Table 3-5. Fewer analytes were detected in soil than in either surface water or sediment, although the pattern and prevalence of constituents detected in this medium was similar to that observed in the other media.

No VOCs were detected in island soil samples. With the exception of TPH, SVOCs were detected in only 4% of samples analyzed and at concentrations less than 1 mg/kg. Total petroleum hydrocarbon was detected in relatively higher numbers of soil samples and at higher concentrations; however, neither TPH nor SVOCs showed a discernible spatial pattern.

Chlorinated pesticides were detected infrequently in soil and at concentrations in the parts-per-billion range. DDE was the most prevalent pesticide. As with SVOCs, no spatial pattern was observed.

Polychlorinated biphenyls were detected in each soil sample analyzed for congeners. Total PCB concentrations were highest in 300 Area Sub-Area samples, although the range in concentrations was relatively small, less than an order of magnitude. A review of congener patterns among soil samples shows similarities in congener distribution among all samples (WCH-398).

Most metals analyzed were detected in all samples. The pattern of prevalence was similar to that noted for sediment. In general, the range of observed concentrations for most metals was small, spanning less than an order of magnitude. Maximum concentrations were distributed across the multiple islands, although most often occurred on Johnson Island and Locke Island. According to the Data Summary Report (WCH-398), many of the detected soil metals concentrations were within the range of background soil concentrations published by Ecology (Ecology 1994, *Natural Background Soil Metals Concentrations in Washington State*). Note that the arsenic concentrations observed in island soil samples are higher than those of Reference samples, but within the range of concentrations observed for sediment. Furthermore, the levels of arsenic in soil observed in island soils are consistent with background arsenic levels published by Ecology (1994), suggesting that arsenic in this medium may potentially be naturally occurring or at least present at levels consistent with local conditions.

Nine radionuclides were detected in soil. Of these, cesium-137 and uranium isotopes were detected most frequently (90% to 100% of samples), whereas the others were detected in less than 10% of samples. Highest levels of most radioisotopes were found on samples collected from Johnson Island. However, the levels of radioisotopes detected in island soil samples were generally similar to or lower than those found in adjacent sediment samples.

Johnson Island soils are more similar to sediments than to upriver soil samples. The islands within the Reach and 300 Sub-Area are typical braided stream sequences (e.g., sediment redeposits during high water events). Because of very different river morphology above Wanapum Dam, the Upriver island soils are more similar to upland soils than to reworked sediments. The upriver island soil data (10 samples total) were available for this evaluation, and so the relatively few upriver sample results from an island with a dissimilar soil morphology and geological origin may not fully characterize reference conditions in island soil within the 100 Area and 300 Area Sub-areas.

3.6.4 Fish Tissue

Fillet, carcass, liver/kidney, and viscera samples were collected from six fish species under the RI sampling program. Samples were analyzed for pesticides, PCBs, metals, and radionuclides, which are potentially bioaccumulative constituents. A comprehensive presentation of these data by tissue type and species is presented in the Data Summary Report (WCH-398) and is briefly summarized in this section. Fillet, carcass, and liver/kidney are considered to be the consumable portions of fish, whereas viscera is not. Therefore, only fillet, carcass, and liver/kidney fish

tissue data were used in this HHRA. However, fillet is preferentially consumed, whereas carcass (e.g., pin bones) and organ meat are assumed to comprise only a small fraction of the total amount of fish consumed by humans. Viscera data were not considered to be relevant to the HHRA, which evaluates the dietary ingestion of finfish.

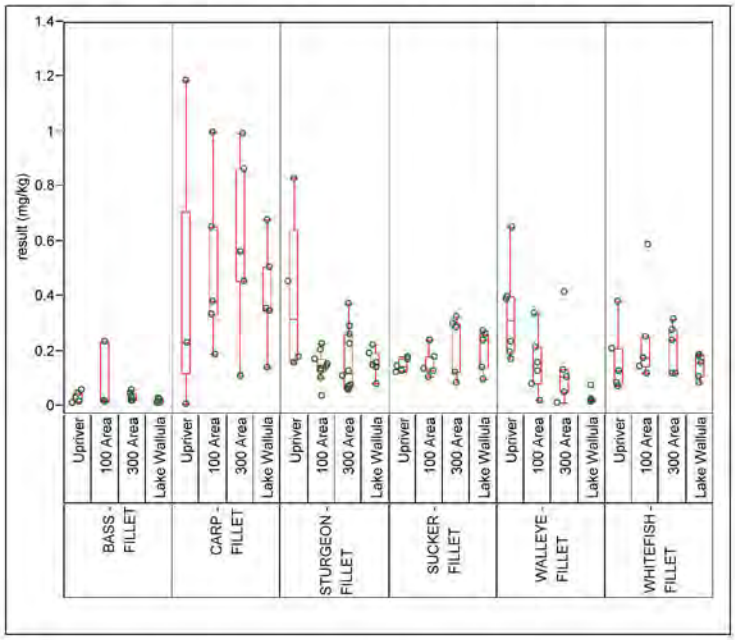
Table 3-6 of this report presents a statistical summary of fish data combined into one data set (i.e., all species and tissue types, except for viscera) for the 100 Area, 300 Area, and Lake Wallula Sub-Areas. Fillet data were used to select COPCs for fish as a medium of concern, as later discussed in Section 3.7. The statistical summaries of the individual fish species data for the combined tissues, fillet, and carcass are presented in Tables 3-7 through 3-12, for bass, carp, sturgeon, sucker, walleye, and whitefish, respectively. Additionally, data for select constituents detected in individual fish species are provided in Figures 3-1 through 3-25. These box plot figures are reproduced from Appendix A of the Data Summary Report (WCH-398) and depict the detected concentration (indicated by an open circle, "o") or, if nondetect, the LRL (indicated by an "x") of a constituent for each sample within each sub-area. For those constituents for which the FOD and/or sample size was adequate, box and whisker plots were generated to display measures of central tendency and the spread of the data. The boxes represent the 25th to 75th interquartile range, with the median represented as the horizontal line within the box. Points falling outside of the "whiskers" represent potential outliers in the data set.[1]

Analytical results for classes of constituents analyzed are discussed in the following sections. Note that all fish tissue data are presented on a wet-weight basis in units of mg/kg for chemical constituents and pCi/g for radionuclides.

3.6.4.1 Chlorinated Pesticides. Select chlorinated pesticides, including DDD, DDE, DDT, and several hexachlorocyclohexane (HCH) isomers, were detected relatively frequently in fish tissue samples with respect to other pesticides. Figures 3-1 through 3-3 present the concentrations of DDE, which was detected most frequently among the chlorinated pesticides, in fish fillet, carcass, and liver/kidney for the individual species. In general, concentrations of pesticides in fish tissue were 1 mg/kg (wet weight) or lower. Liver and kidney samples had the highest levels of pesticides, and maximum concentrations were often observed in the sturgeon, whitefish, and walleye. However, there was no pattern observed in the data set to suggest that levels were elevated in a particular species or sub-area on a consistent basis.

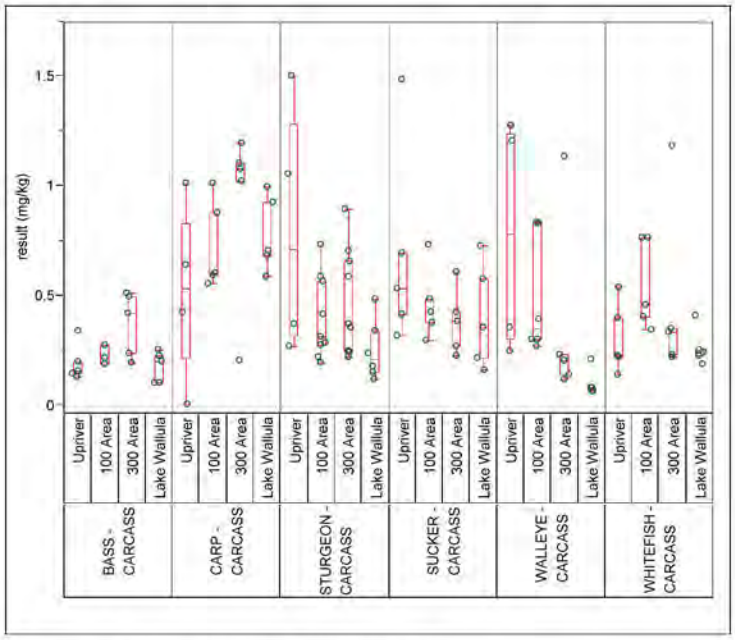
[1] The whiskers extend to the outermost data point falling within 1.5 times the interquartile range (the range between the 25th and 75th quartiles).

Figure 3-1. Box Plot of DDE in Fish Fillet.



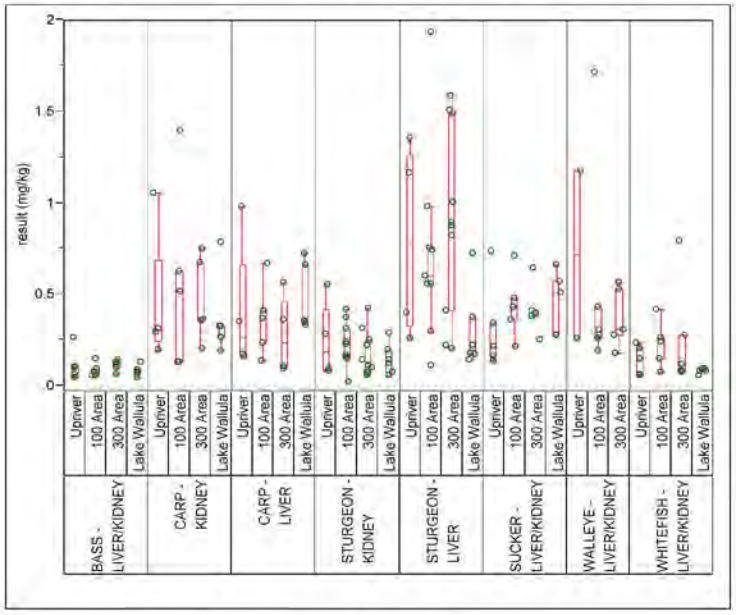
○= detected above laboratory reporting limit

Figure 3-2. Box Plot of DDE in Fish Carcass.



○= detected above laboratory reporting limit

Figure 3-3. Box Plot of DDE in Fish Liver/Kidney.

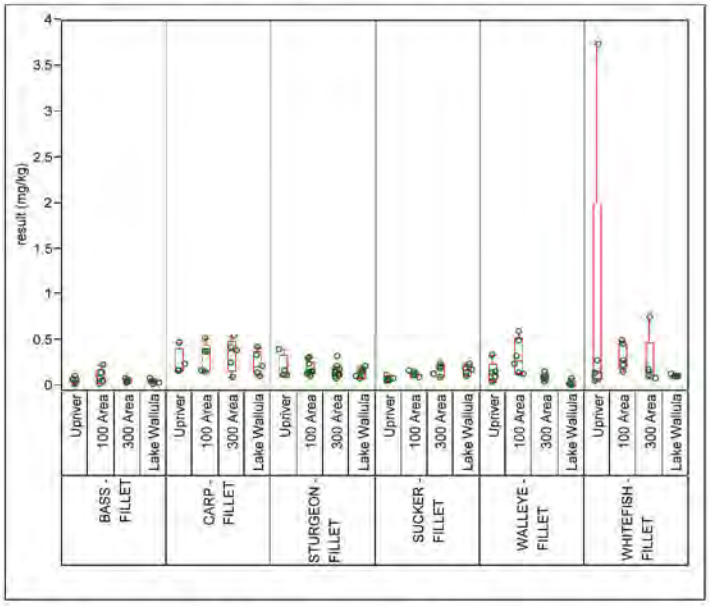


O = detected above laboratory reporting limit

3.6.4.2 Polychlorinated Biphenyls. Polychlorinated biphenyl congeners were detected in each of the fish tissue samples analyzed. Concentrations of PCBs were variable, ranging over approximately two orders of magnitude. Figures 3-4 through 3-6 present the levels of total PCBs in fish fillet, carcass, and liver/kidney, respectively, for the individual species. Polychlorinated biphenyl levels were generally higher in carcass and liver/kidney and lowest in fillet. A review of individual congener data in fish tissue shows that show a similar pattern of congener distribution exists across species and sub-area and reflects patterns observed in sediment (WCH-398).

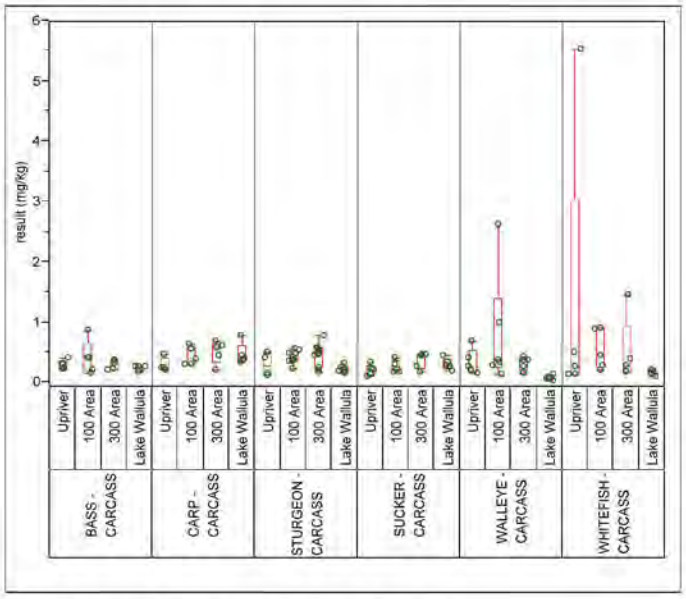
3.6.4.3 Metals. Numerous heavy metals were detected in fish tissue samples. The range of detected concentrations often spanned two orders of magnitude across species and tissue types, reflecting the variability inherent in this data set. Tables 3-7 through 3-12 present statistical summaries of the various metals detected in fish tissue samples by species. Individual fish species data for a subset of detected metals considered among the more toxic of the analytes are described in the following subsections.

Figure 3-4. Box Plot of Total PCB Congeners in Fish Fillet.



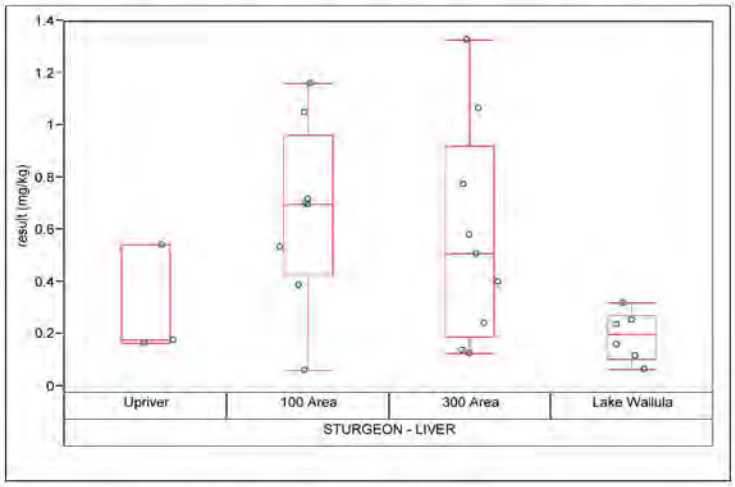
x = not detected above laboratory reporting limit
O = detected above laboratory reporting limit

Figure 3-5. Box Plot of Total PCB Congeners in Fish Carcass.



x = not detected above laboratory reporting limit
O = detected above laboratory reporting limit

Figure 3-6. Box Plot of Total PCB Congeners in Fish Liver/Kidney.

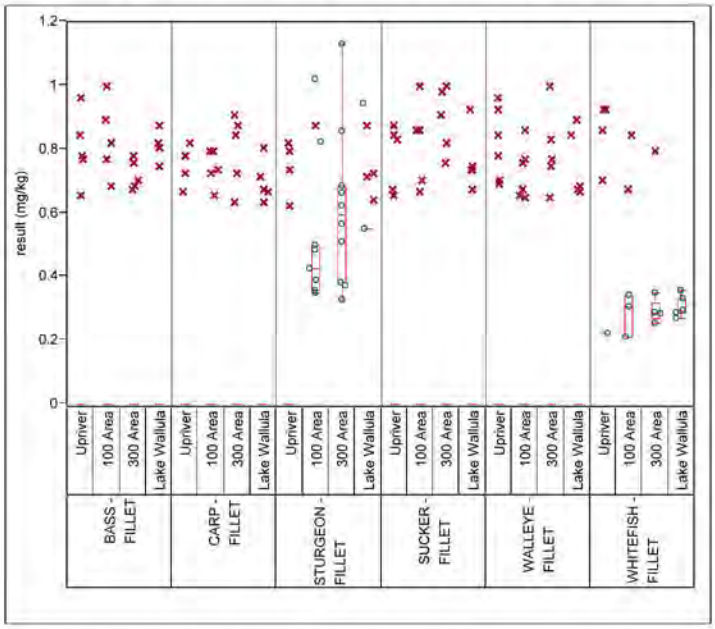


x = not detected above laboratory reporting limit
O = detected above laboratory reporting limit

3.6.4.3.1 Arsenic. Arsenic exists in fish tissue in two forms: inorganic and organic (such as arsenobetaine). The common organic forms of arsenic in tissue are generally not considered toxic, unlike inorganic forms (e.g., As3+, As5+) of arsenic (ATSDR 2007, *Toxicological Profile for Arsenic*). Therefore, fish tissue samples were analyzed for both total arsenic and for TIAS; inorganic arsenic results are used in estimating health risks from fish ingestion exposure, as further described in Section 4.2.2.4.3.

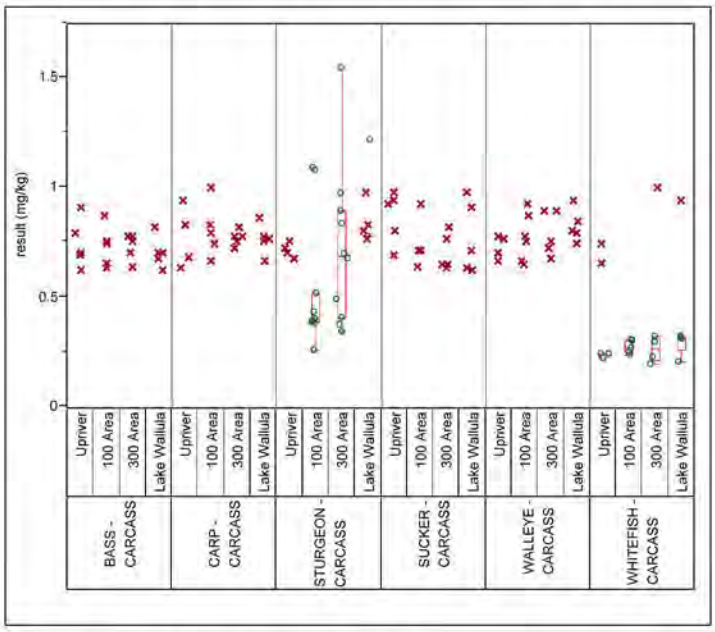
Figures 3-7 through 3-9 present the levels of total arsenic in fish fillet, carcass, and liver/kidney, respectively, for the individual species. For fillet samples, the results indicate that total arsenic was detected in only sturgeon and whitefish, although it should be noted that the reporting limits of results for other fish species samples in many instances exceeded the detected concentrations in these two species. Total arsenic concentrations in carcass and liver/kidney samples were generally consistent with those observed in fillet.

Figure 3-7. Box Plot of Total Arsenic in Fish Fillet.



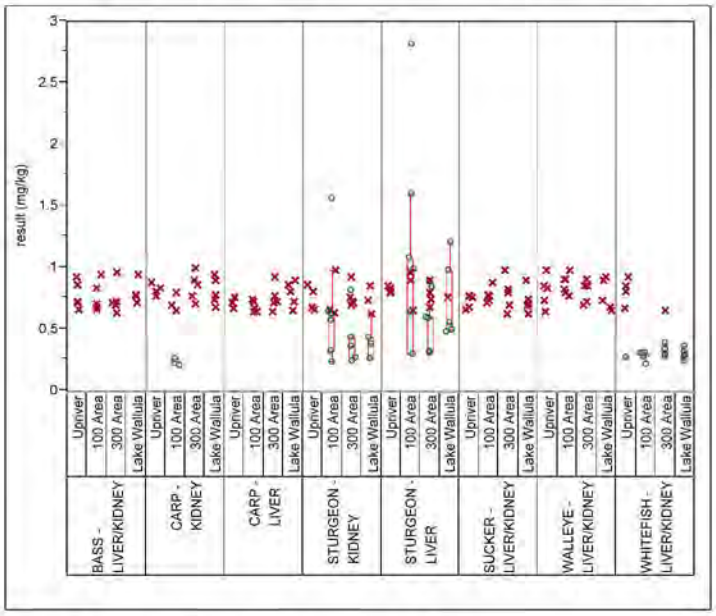
x = not detected above laboratory reporting limit
O = detected above laboratory reporting limit

Figure 3-8. Box Plot of Total Arsenic in Fish Carcass.



x = not detected above laboratory reporting limit
O = detected above laboratory reporting limit

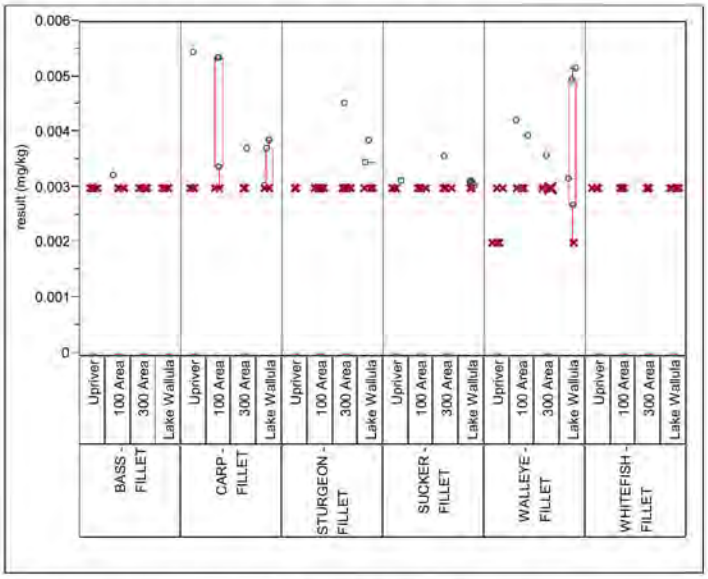
Figure 3-9. Box Plot of Total Arsenic in Liver/Kidney.



x = not detected above laboratory reporting limit
O = detected above laboratory reporting limit

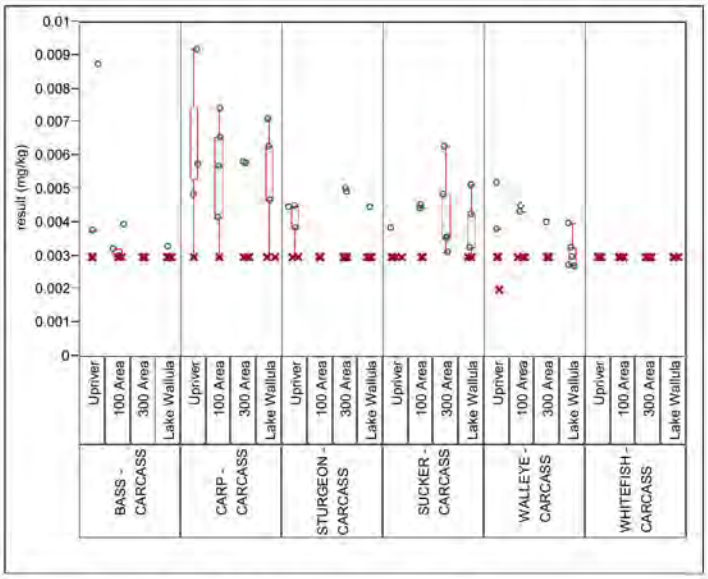
Figures 3-10 and 3-11 depict TIAS concentrations detected in fish tissue samples of fillet and carcass, respectively. Results show that inorganic arsenic was detected most often in carp, sucker, and walleye, less often in sturgeon and bass, and not detected in whitefish. Concentrations of TIAS were variable, highest in carp tissue samples and lowest in bass and sucker. In general, the range of detected concentrations in all fish was narrow, spanning a factor of approximately two.

Figure 3-10. Box Plot of Total Inorganic Arsenic in Fish Fillet.



x = not detected above laboratory reporting limit
O = detected above laboratory reporting limit

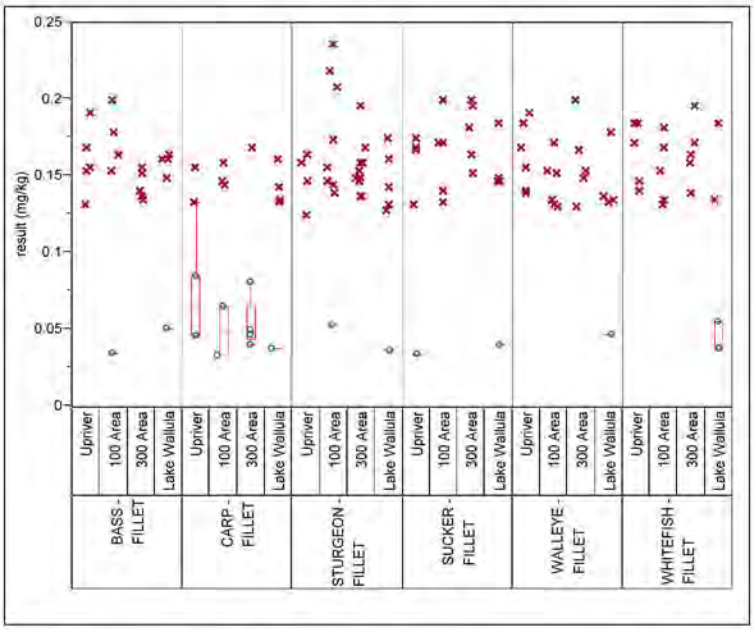
Figure 3-11. Box Plot of Total Inorganic Arsenic in Fish Carcass.



x = not detected above laboratory reporting limit
O = detected above laboratory reporting limit

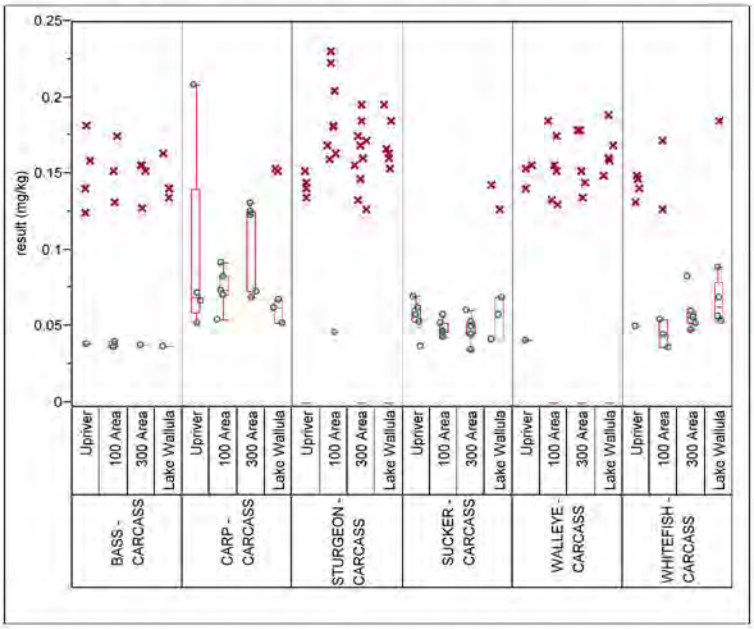
3.6.4.3.2 Cadmium. Figures 3-12 through 3-14 depict cadmium concentrations detected in fish tissue samples of fillet, carcass, and liver/kidney, respectively. Cadmium was detected at a low frequency, or not at all, in fish fillet and carcass samples, with the exception of carp. Upriver carp fillet samples also had the overall highest mean concentration among species and sub-areas, although the means across species and sub-areas were generally within a factor of two. Cadmium levels in fillet/carcass samples in any of the Hanford Site Study Area (100 Area, 300 Area, and Lake Wallula Sub-Areas) were similar to (or lower than) those in Upriver locations. Cadmium was detected in 100% of liver and kidney samples and at much higher concentrations than those observed in other tissue types. Note that many of the nondetect results for cadmium in fillet and carcass tissues have reporting limits much higher than the few detected results (as discussed in Section 3.5.1, LRLs are sample-specific). However, these reporting limits were used in the HHRA. Uncertainties associated with inclusion of elevated LRLs are discussed in Section 7.0.

Figure 3-12. Box Plot of Cadmium in Fish Fillet.



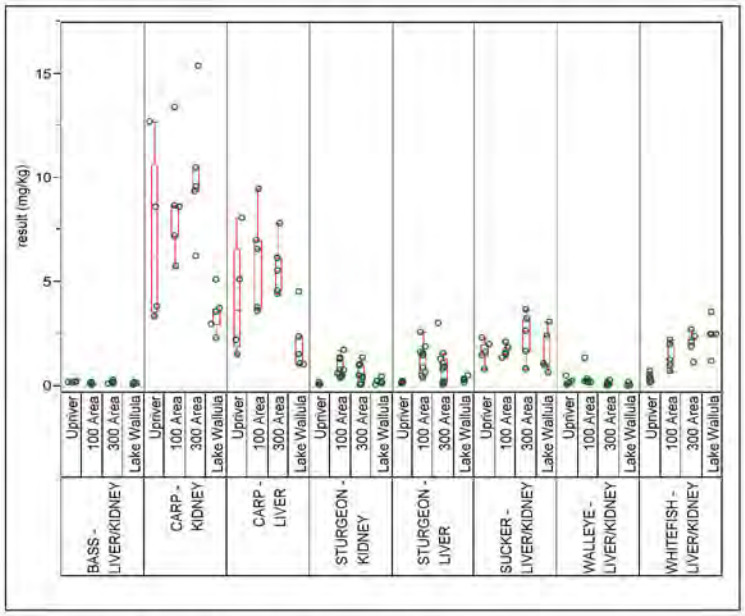
x = not detected above laboratory reporting limit
O = detected above laboratory reporting limit

Figure 3-13. Box Plot of Cadmium in Fish Carcass.



x = not detected above laboratory reporting limit
O = detected above laboratory reporting limit

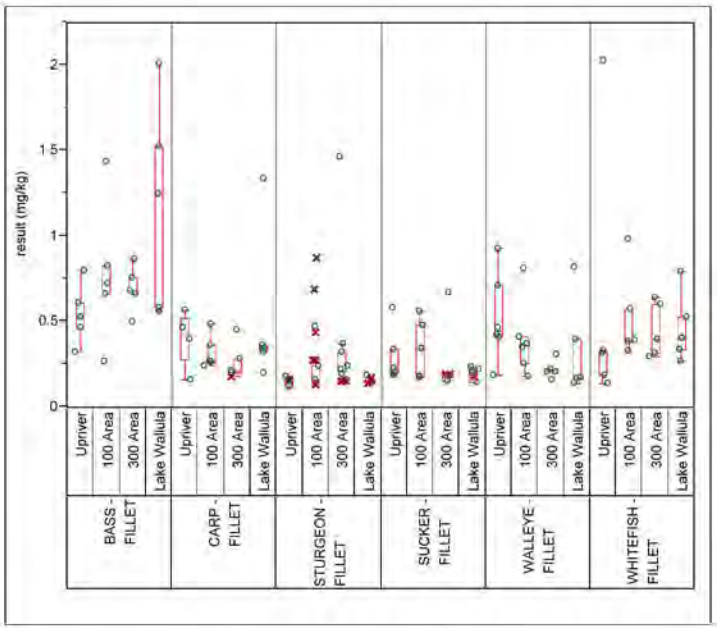
Figure 3-14. Box Plot of Cadmium in Fish Liver/Kidney.



O = detected above laboratory reporting limit

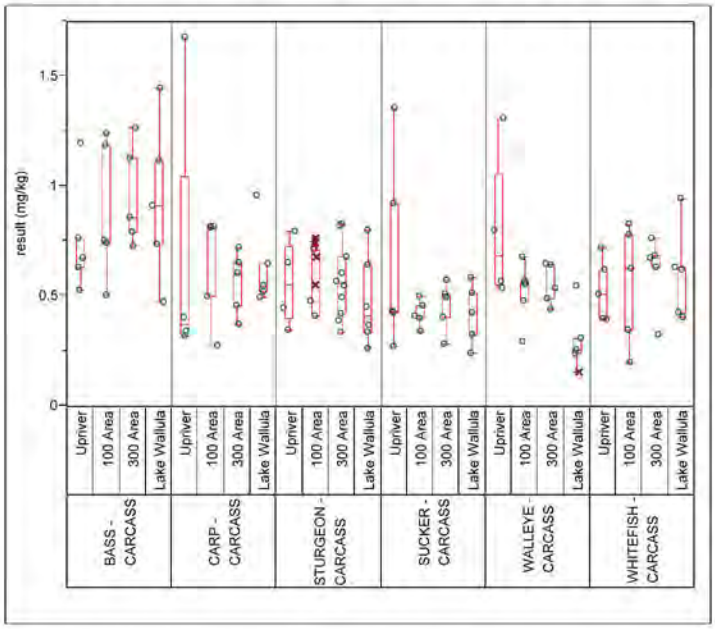
3.6.4.3.3 Chromium. Figures 3-15 through 3-17 depict concentrations of total chromium in fillet, carcass, and liver/kidney samples, respectively, from the six fish species. Chromium was detected in most of the tissue samples analyzed, at variable concentrations. Chromium concentrations are generally higher in fillet and carcass samples, relative to those of liver and kidney. The spread of detected concentrations was greatest in the bass and whitefish samples, in which concentrations were also the highest. Hanford Site Study Area concentrations of chromium are generally consistent with those of Upriver samples, with the exception of bass, which had higher concentrations in the Lake Wallula Sub-Area.

Figure 3-15. Box Plot of Chromium in Fish Fillet.



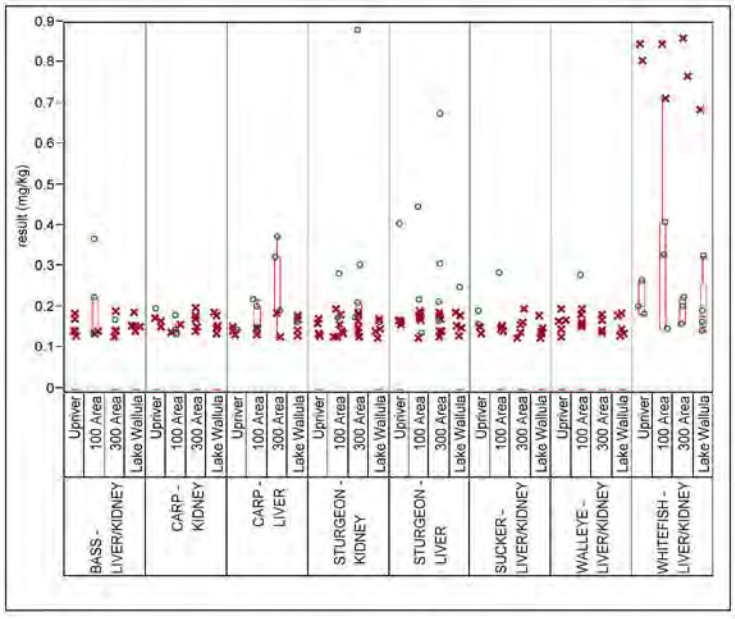
x = not detected above laboratory reporting limit
O = detected above laboratory reporting limit

Figure 3-16. Box Plot of Chromium in Fish Carcass.



x = not detected above laboratory reporting limit
O = detected above laboratory reporting limit

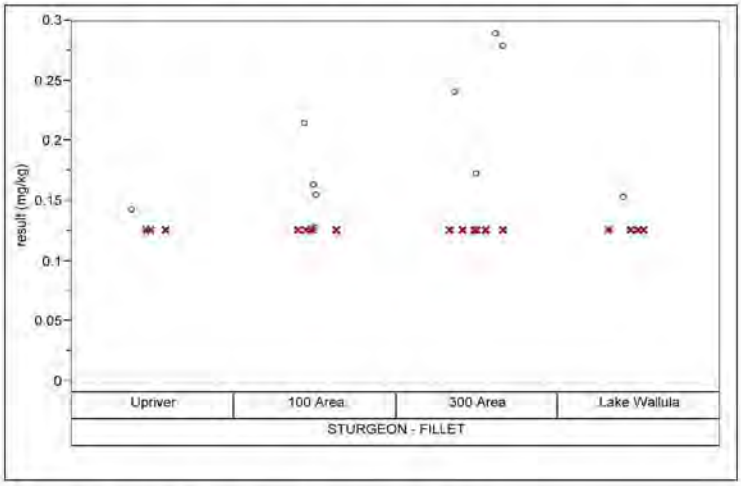
Figure 3-17. Box Plot of Chromium in Fish Liver/Kidney.



x = not detected above laboratory reporting limit
O = detected above laboratory reporting limit

Hexavalent chromium concentrations in fish fillet samples are depicted in Figure 3-18. Only sturgeon fillet and carcass samples were analyzed for this parameter, and hexavalent chromium was detected only in fillet samples. Hexavalent chromium in fillets was detected from all sub-areas, including Upriver, with the highest concentrations detected in samples collected from the 300 Area Sub-Area.

Figure 3-18. Box Plot of Hexavalent Chromium in Fish Fillet.



x = not detected above laboratory reporting limit
O = detected above laboratory reporting limit

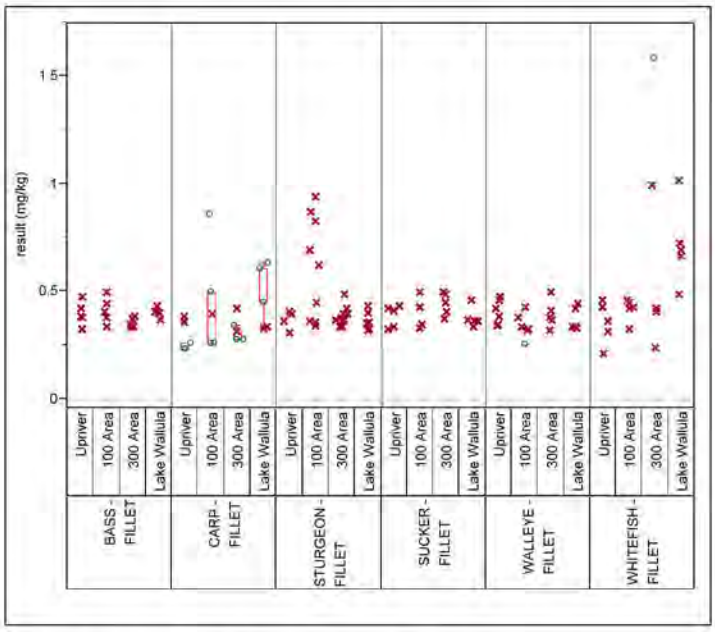
The hexavalent chromium results are consistent with total chromium results for sturgeon fillet samples in that concentrations were slightly higher in the 100 Area and 300 Area Sub-Areas than in either the Upriver or Lake Wallula Sub-Areas. However, fewer tissue samples were collected in the Upriver and Lake Wallula Sub-Areas, and so these higher concentrations in the 100 and 300 Area Sub-Areas may simply reflect the variability within a larger sample size. The presence of detected concentrations of hexavalent chromium in Upriver fish tissue samples could suggest that there may be sources of chromium and hexavalent chromium to the river that are unrelated to the Hanford Site.

However, it is also possible that these results are positively biased, and the detected concentrations may represent false-positives. Although hexavalent chromium is the most biologically active species of chromium (Langard and Costa 2007, "Chromium," in *Handbook on the Toxicology of Metals*), this form is not anticipated to substantially accumulate in biological tissue, since upon uptake it will be reduced to the trivalent form through oxidation with organic matter (EPA 2003; ATSDR 2000, *Toxicological Profile for Chromium*; Langard and Costa 2007). Furthermore, it is unlikely that trivalent chromium is converted to hexavalent chromium in biological systems (EPA 2003). Sample extraction methods could potentially oxidize trivalent chromium back to hexavalent chromium, thus confounding accurate quantitation of this constituent (Applied Speciation 2011, "Hexavalent Chromium

Speciation Analysis"). Because of this, it is believed that the hexavalent chromium detects are actually false-positives. As such, the analytical results for hexavalent chromium in fish tissue are deemed not representative and are not carried through the quantitative risk assessment.

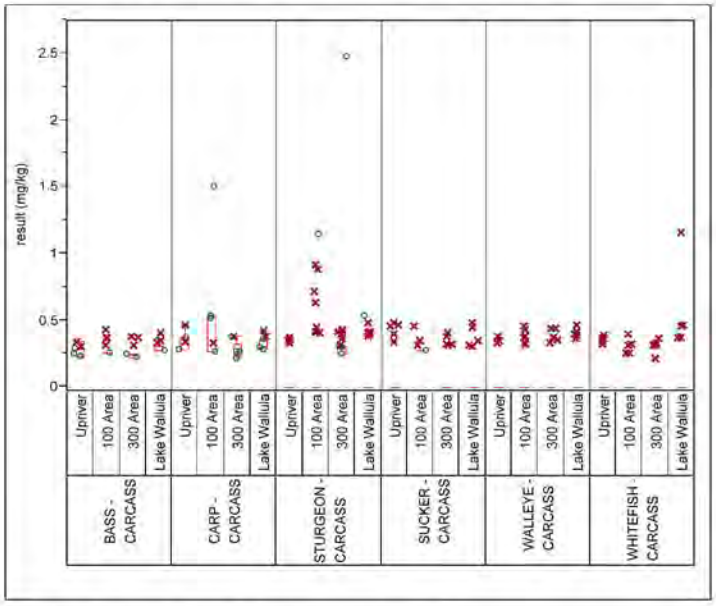
3.6.4.3.4 Lead. Lead concentrations in fish tissue are presented in Figures 3-19 through 3-21 for fillet, carcass, and liver/kidney tissue samples, respectively. Overall, lead was infrequently detected in fish tissue, and the lead levels were variable among species, tissue type, and sub-area. Lead was detected in only walleye, carp, and whitefish fillet samples, with the highest concentration observed in whitefish. In carcass, the maximum lead concentration was observed in sturgeon and carp, at concentrations up to 2.5 mg/kg. On average, bass had the lowest mean carcass concentrations of lead.

Figure 3-19. Box Plot of Lead in Fish Fillet.



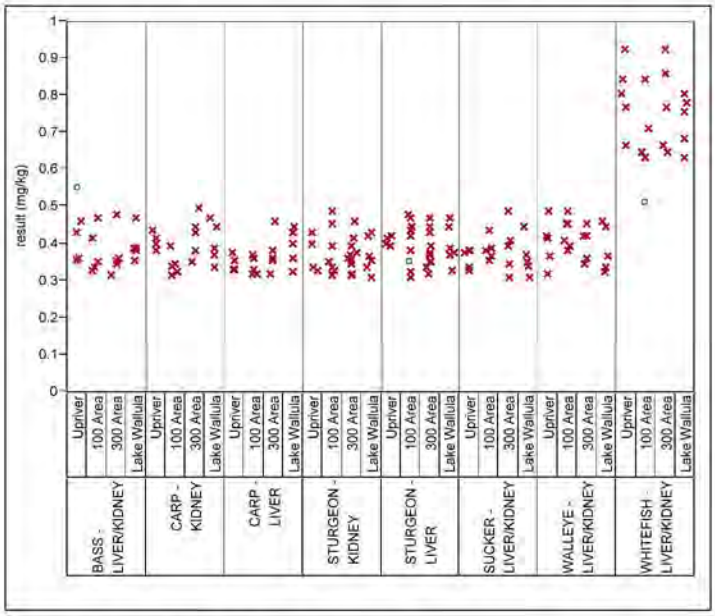
x = not detected above laboratory reporting limit
O = detected above laboratory reporting limit

Figure 3-20. Box Plot of Lead in Fish Carcass.



x = not detected above laboratory reporting limit
O = detected above laboratory reporting limit

Figure 3-21. Box Plot of Lead in Fish Liver/Kidney.



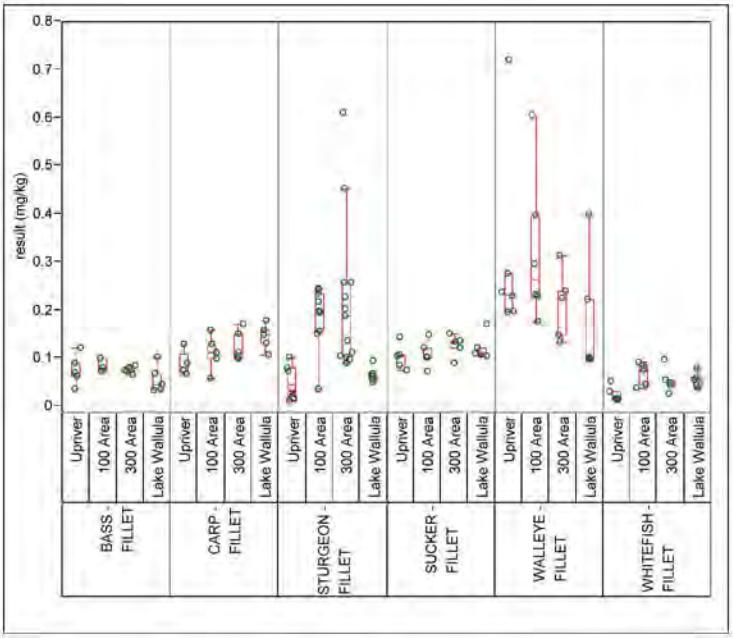
x = not detected above laboratory reporting limit
O = detected above laboratory reporting limit

Liver and kidney sample results (Figure 3-21) indicate concentrations generally similar to those of fillet, although at a lower FOD. Lead was detected more frequently and at higher levels in carcass samples, with concentrations ranging up to 2.5 mg/kg. Because lead preferentially accumulates in bone, this difference in lead levels among the three tissue types is expected.

3.6.4.3.5 Mercury. Total mercury concentrations in fish tissue are presented in Figures 3-22 through 3-24 for fillet, carcass, and liver/kidney samples, respectively. Total mercury was detected in nearly 100% of all samples. Highest concentrations were detected in walleye fillet samples from Upriver and the 100 Area Sub-Area and in sturgeon liver and kidney samples collected from the 100 and 300 Area Sub-Areas. The lowest overall concentrations were observed in whitefish and bass samples, although the range of detected concentrations within these species overlapped with those observed in other species.

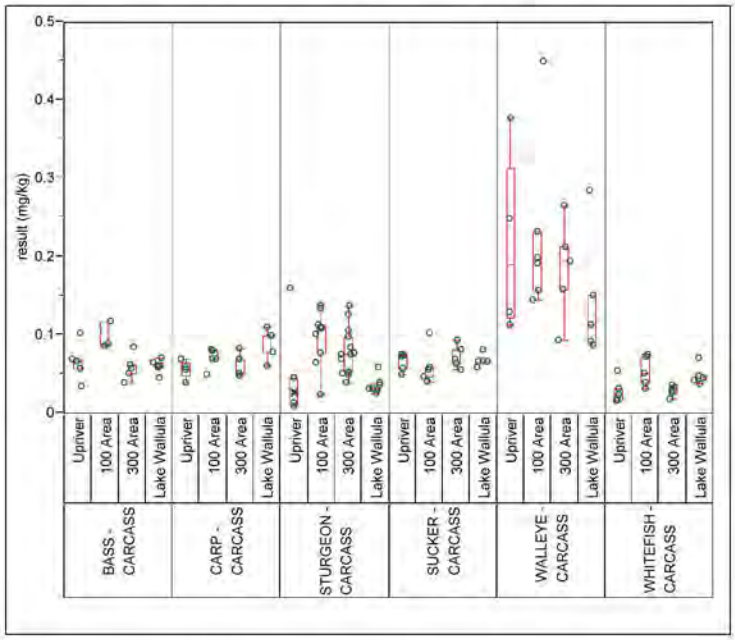
In addition to total mercury, sturgeon fillet samples were analyzed for methylmercury (Figure 3-25). Results from this analysis show that methylmercury comprises most of the total mercury load. See Section 4.2.2.4.4 for further discussion on this analyte.

Figure 3-22. Box Plot of Mercury in Fish Fillet.



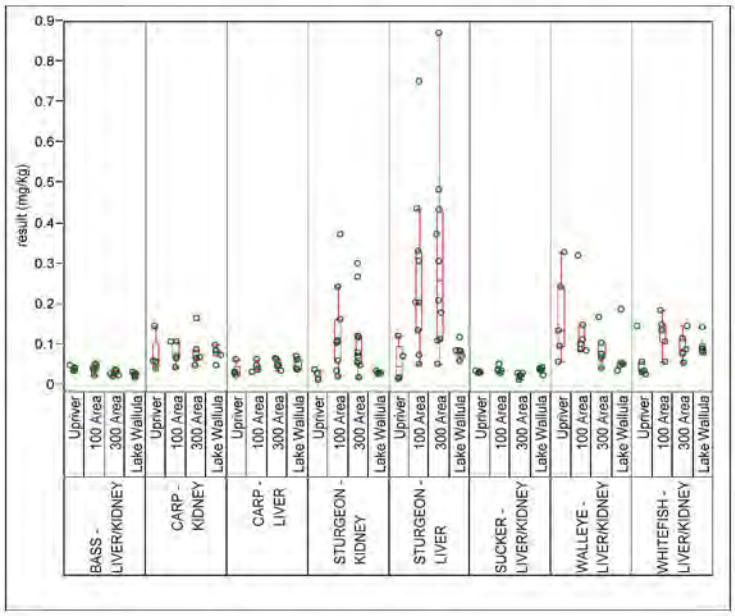
x = not detected above laboratory reporting limit
O = detected above laboratory reporting limit

Figure 3-23. Box Plot of Mercury in Fish Carcass.



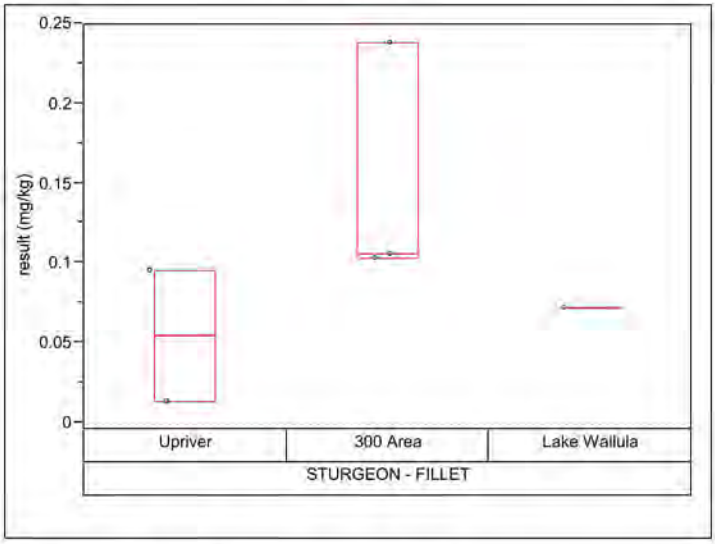
x = not detected above laboratory reporting limit
O = detected above laboratory reporting limit

Figure 3-24. Box Plot of Mercury in Fish Liver/Kidney.



O = detected above laboratory reporting limit

Figure 3-25. Box Plot of Methylmercury in Fish Fillet.



O= detected above laboratory reporting limit
Note: Methylmercury was analyzed in only six sturgeon fillet samples, from Lake Wallula, 300 Area and Upriver. 100 Area Sub-Area sturgeon samples were not analyzed for methyl mercury.

3.6.4.4 Radionuclides. Only six radionuclides (carbon-14, cesium-137, plutonium-239, plutonium-240, strontium-90, technetium-99, and tritium) were detected in fish tissue samples collected within the Study Area. The FOD for any of these elements was 2% and lower. Figures 3-26 through 3-43 present analytical results for the radionuclides detected at least once in fish tissue samples collected from within the 100 Area, 300 Area and Lake Wallula Sub-Areas. As with other classes of contaminants, there was no strong pattern in radionuclide presence or concentration in tissue type, species, or sub-area. In general, there were only a few isolated detections observed in fish tissue samples, and many of the detected concentrations were similar to the sample MDAs, suggesting a high bias for false-positive results. In most instances, these sporadic occurrences of radionuclides do not coincide with areas of known radionuclide contaminant plume discharge areas. While most of the detected concentrations were observed in the Hanford Site Study Area (100 Area, 300 Area, and Lake Wallula Sub-Areas), some radionuclides, such as plutonium-239/plutonium-240 (whitefish fillet) and uranium isotopes (carp liver), were also detected in Upriver fish tissue samples.

Because of the very low FOD of radionuclides in fish tissue, radionuclide results were further evaluated to determine whether the results were likely valid and should be included in the quantitative HHRA. A review of the laboratory data packages has been conducted by WCH and DOE, as well as the analytical laboratory (Eberline Analytical Services) that performed the work. Factors considered in this evaluation included the FOD, the magnitude of the detected activity relative to the sample MDA, the tissue type in which the radionuclide was detected, the

consistency between the result and results for abiotic media, the location the samples were collected, and any other information that may be useful in making this determination.

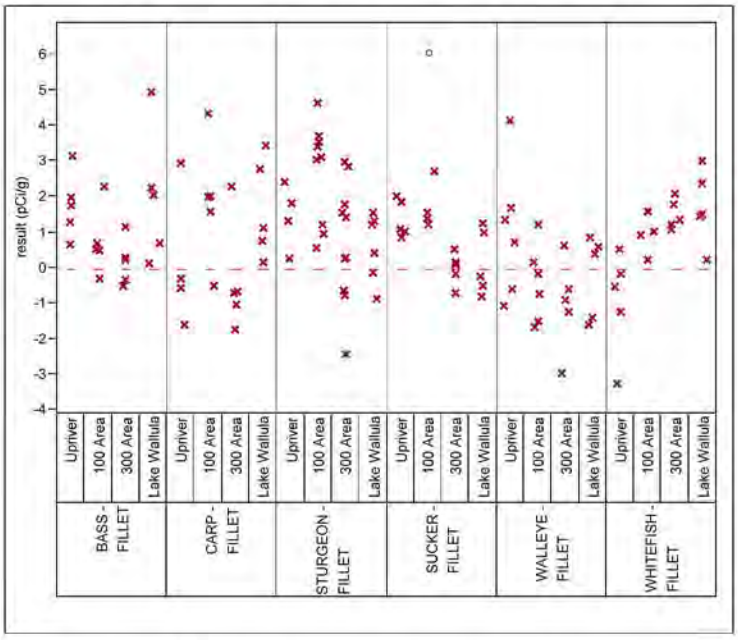
There is always the potential for a truly random event to cause an unexpected detect. Most commonly, a small highly contaminated particle from an internal or external source is included in the analytical preparation. Random electrical "noise" can also be interpreted as a detection. It is also possible that a contaminated laboratory planchet may have been used, which would not be identified as part of quality assurance/quality control reviews or tests. Unexpected detects are rarely confirmed on reanalysis, and a specific cause is rarely identified.

For reasons discussed in the following subsections, data for all detected radionuclides in fish tissue, with the exception of carbon-14, were deemed to be likely false-positive results and were not used for quantitative evaluation in this HHRA. However, uncertainties associated with the exclusion of the fish tissue radionuclide data, relative to cumulative risk, are discussed in Section 7.0.

3.6.4.4.1 Carbon-14. Figures 3-26 through 3-28 present box plots of carbon-14 found in fish fillet, carcass, and liver/kidney samples, respectively. As indicated in these plots, this radionuclide was detected very infrequently and inconsistently among species. One sucker sample (100SA-SUCKER 4) collected from the 100 Area Sub-Area had detected carbon-14 concentrations in both fillet (6 pCi/g) and carcass (8 pCi/g) samples. This sample was collected from RM382 by the 100-K Reactor area. The highest detected carbon-14 concentration (141 pCi/g), however, was observed in a whitefish carcass sample from Lake Wallula (LWSA-WF5). Other detected concentrations were either within or slightly greater than the range of MDAs reported for all samples.

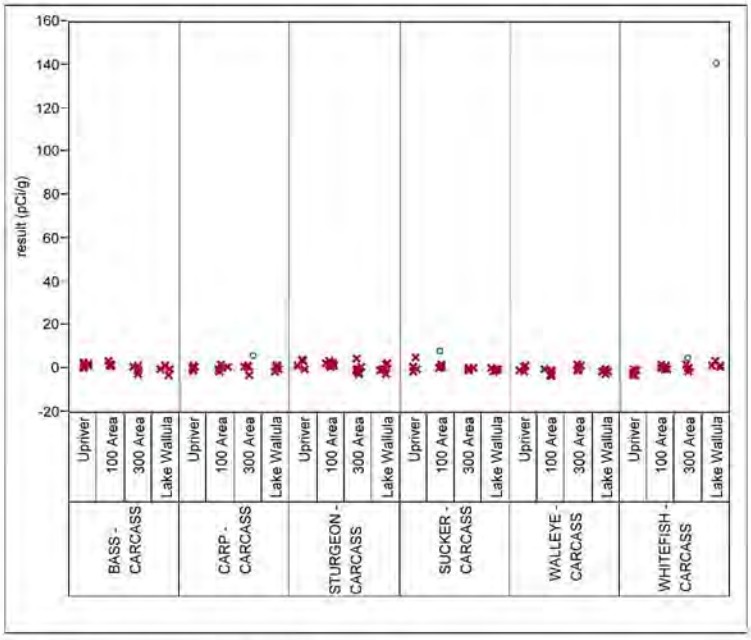
It is unlikely that three false-positives would occur in samples from the same fish composite, as was found for the 100 Area sucker sample. However, the tissue samples were actually run in two different preparation batches and were the only detects in either batch, which further argues that the carbon-14 results are not false-positives. The fish composite sample consisted of fish caught near RM382 along the Hanford shoreline and adjacent to the 100-K Reactor area, which has a known carbon-14 plume. Because there is some consistency of carbon-14 detects in the 100SA-Sucker sample, carbon-14 was therefore carried through the quantitative HHRA.

Figure 3-26. Box Plot of Carbon-14 in Fish Fillet.



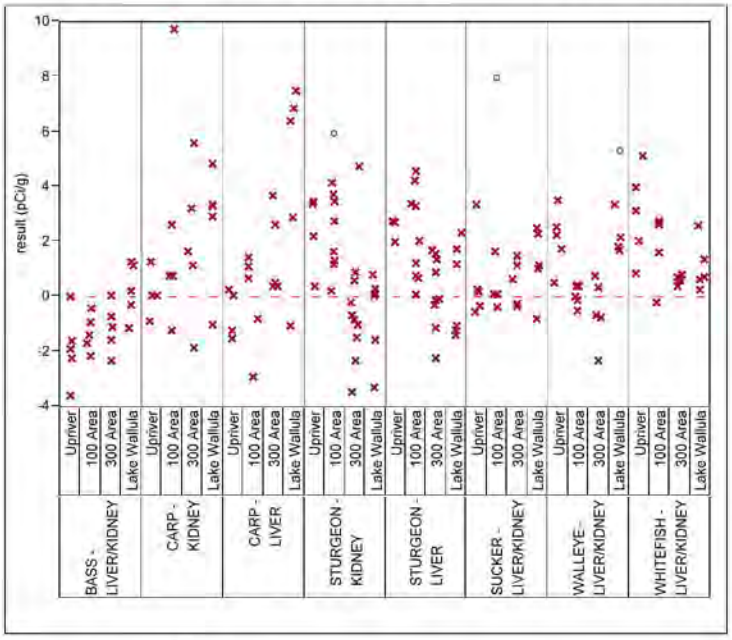
x = not detected above MDA
O = detected above MDA

Figure 3-27. Box Plot of Carbon-14 in Fish Carcass.



x = not detected above MDA
O = detected above MDA

Figure 3-28. Box Plot of Carbon-14 in Fish Liver/Kidney.

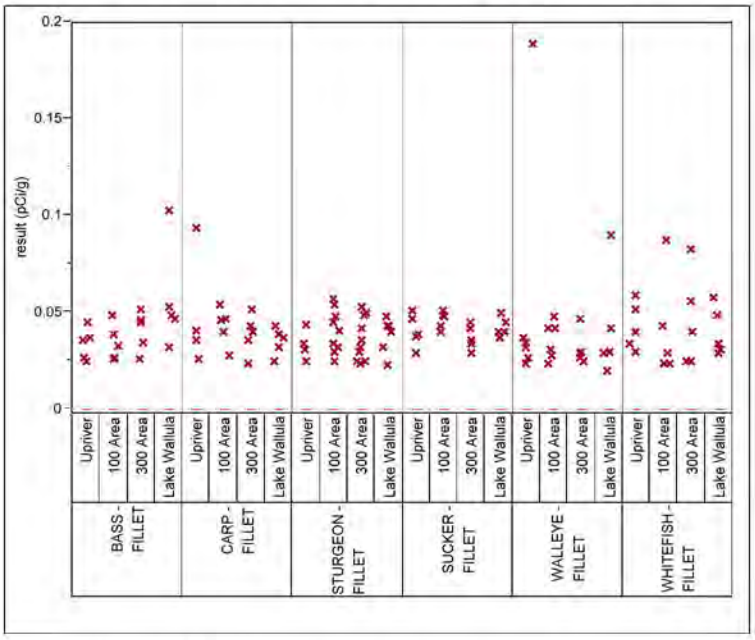


x = not detected above MDA
O = detected above MDA

3.6.4.4.2 Cesium-137. Figures 3-29 through 3-31 show results for cesium-137 in fish tissue. This radionuclide was detected in only one sample (300SA-WF4, whitefish liver/kidney) at an activity (0.358 pCi/g) within the range of reported MDAs. This sample was collected in the 300 Area Sub-Area at RM 350.

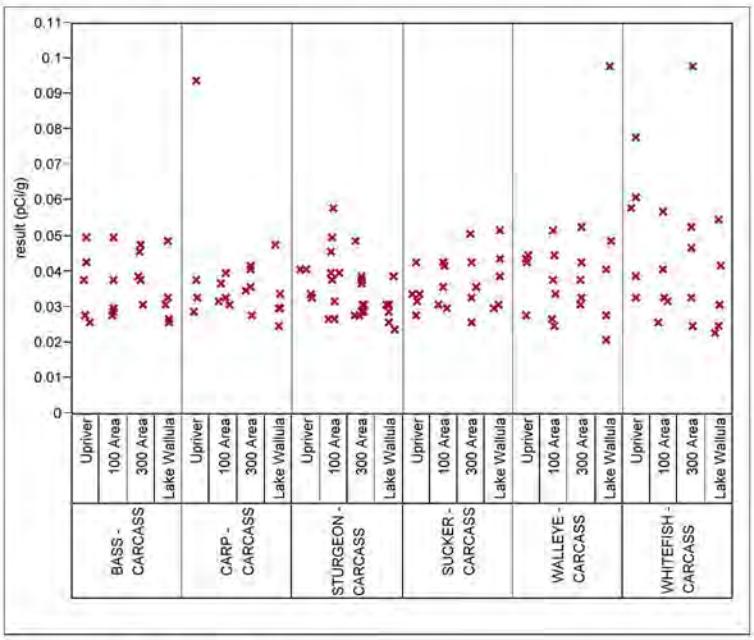
Because cesium-137 was detected in only one single sample at an activity within the range of MDAs reported for all nondetect results, it is assumed that this result is likely a false-positive. Therefore, this radionuclide was not retained for further quantitative evaluation in the HHRA.

Figure 3-29. Box Plot of Cesium-137 in Fish Fillet.



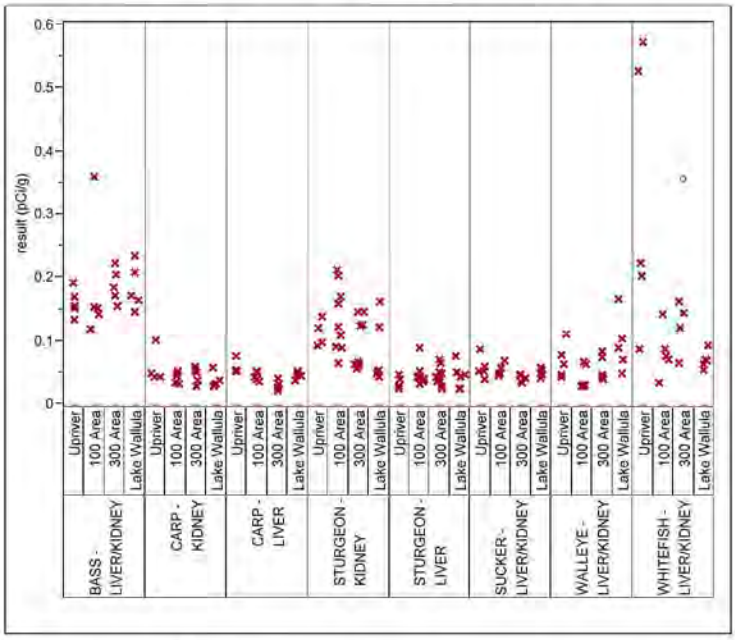
x = not detected above MDA

Figure 3-30. Box Plot of Cesium-137 in Fish Carcass.



x = not detected above MDA

Figure 3-31. Box Plot of Cesium-137 in Fish Liver/Kidney.

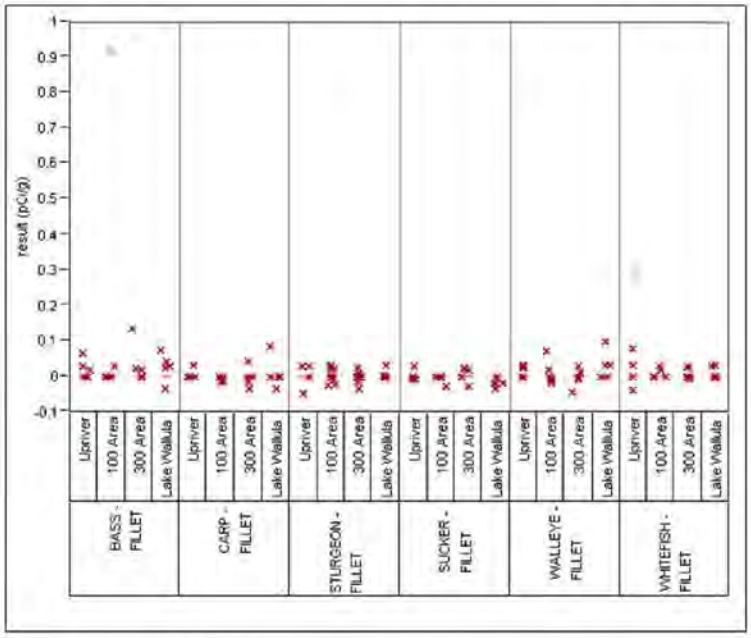


x = not detected above MDA
O = detected above MDA

3.6.4.4.3 Plutonium-239/Plutonium-240. Figures 3-32, 3-33, and 3-34 present box plots of plutonium-239/plutonium-240 results for fish tissue.

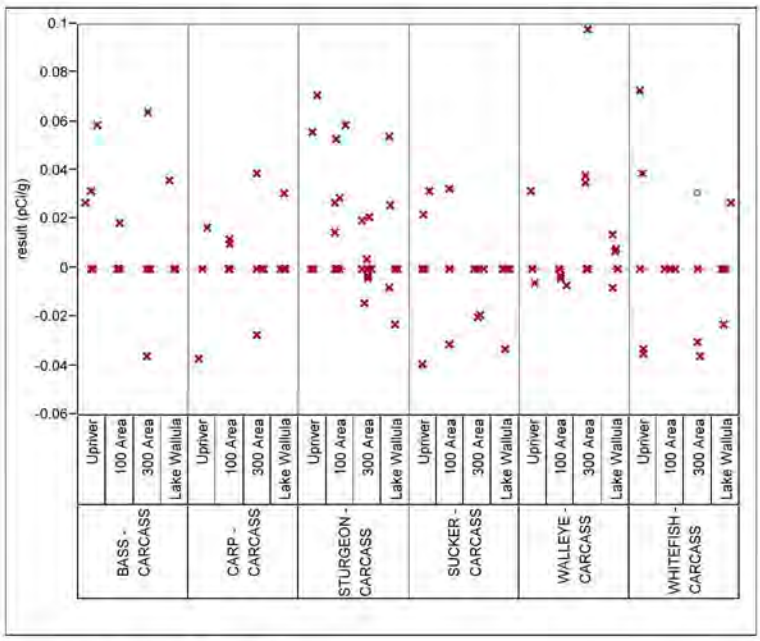
Plutonium-239/plutonium-240 was detected in only one fillet sample within the Study Area, in a sample composited from five bass caught in the area of Coyote Island across from the 100-K Reactor area. This reported bass fillet composite sample result of 0.916 pCi/g is greater than five times the MDA (0.14 pCi/g), which suggests a low potential for a counting-based false-positive. A low potential for a counting-based false-positive is also supported by the analytical error values associated with the results (0.28 pCi/g; errors are a fraction of the result). A review of other analytical information in the data package (sample volumes, counting times, associated analytical batch quality control samples) shows no deviations from routine analytical processing. This sample was processed as part of a large analytical batch, and no other anomalous results were noted in other reported results. Analytical batch blanks were within control limits, and the lack of significant detectable activity in the other samples in the batch indicates no systemic contamination control issues at the laboratory.

Figure 3-32. Box Plot of Plutonium-239/Plutonium-240 in Fish Fillet.



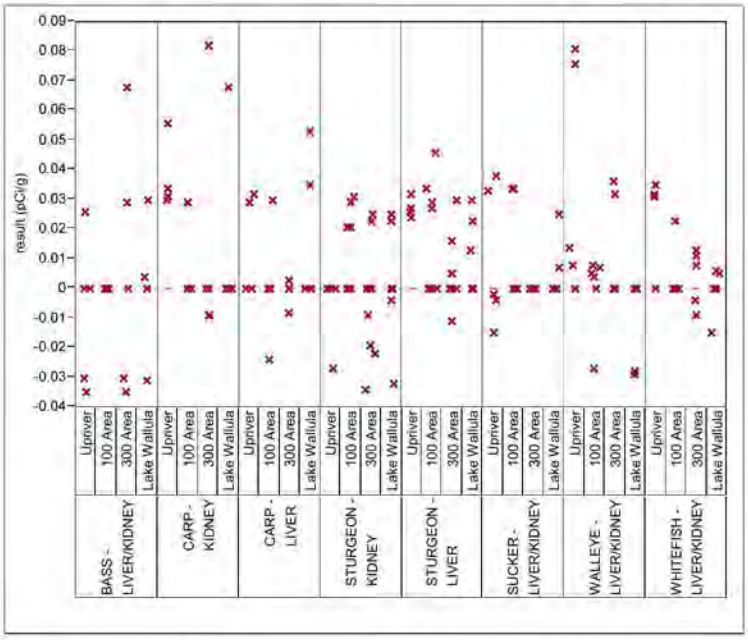
x = not detected above MDA
O= detected above MDA

Figure 3-33. Box Plot of Plutonium-239/Plutonium-240 in Fish Carcass.



x = not detected above MDA
O= detected above MDA

Figure 3-34. Box Plot of Plutonium-239/ Plutonium-240 in Fish Liver/Kidney.



x = not detected above MDA

Regarding a potential source for the plutonium, the data do not point to a specific source. From the data, the plutonium-239/plutonium-240 to plutonium-238 ratio is greater than 4.4/1 (based on the MDA of plutonium-238). This would imply a low-exposure (weapons-grade production) source (long exposure in reactors, as is typical for power production, yields much lower plutonium-239/plutonium-240 to plutonium-238 ratios, and plutonium-238 can predominate) (BNWL-478, *High Exposure Plutonium Studies Analyses of Shippingport Fuel*). The lack of a detectable result (i.e., the result was reported as a nondetect at the MDA) for americium-241 also suggests an anomalous plutonium result. Americium-241 is normally detected when plutonium is detected, particularly for longer exposure fuels (i.e., 105-N Reactor, or much more so in power reactors) (PNL-6866, *Technical Basis for Internal Dosimetry at Hanford*). The lack of an americium-241 detection could result from three potential sources: (1) separation by biological or environmental chemical processes of the elements, (2) extremely low exposures (very little of this type material was generated at the Hanford Site), or (3) the plutonium result is an analytical anomaly. The result appears to be anomalous based on the reported activity of americium-241 relative to observed plutonium-239/plutonium-240 activities in fish samples from other studies.

Note that plutonium-239/plutonium-240 values of 0.289 and 0.031 pCi/g (wet weight) were also reported for a composite fillet sample (J18J07, SDG K1618) from five whitefish caught upriver of Priest Rapids Dam (i.e., in the Upriver Sub-Area) in the vicinity of Beverly. The presence of plutonium-239/plutonium-240 in Upriver samples further suggests that these results may be false-positives. In the Upriver whitefish samples, the detected plutonium-239/plutonium-240 activities were very close to the reported MDA and within a range that has high potential to be considered as counting-based false-positives. The reported concentration for the composite

whitefish fillet sample was 0.289 pCi/g, compared with an MDA of 0.276 pCi/g. The reported concentration for the composite whitefish carcass sample was 0.031 pCi/g, just slightly above the sample MDA of 0.03 pCi/g. The fillet results are further suspect due to the magnitude of activity detected relative to those observed in other studies (BNWL-1867, *Ecological Behavior of Plutonium and Americium in a Freshwater Ecosystem. Phase I. Limnological Characterization and Isotopic Distribution*; Emery et al. 1978, "The Ecological Export of Plutonium from a Reprocessing Waste Pond"; Emery et al. 1981, "Potential Radionuclide Dose from Eating Fish Exposed to Actinide Contamination"), and the presence of this radionuclide in fillet but not bone or carcass, where it is expected to preferentially accumulate (ATSDR 2010, *Toxicological Profile for Plutonium*).

The detected result is also inconsistent with results observed in other studies, and suggests that the result is suspect. In the mid-1970s, PNNL performed research (BNWL-1867; Emery et al. 1978, 1981) at the 216-U-10 pond evaluating the ecological behavior of transuranic elements in a freshwater ecosystem. This pond supported a population of goldfish. In subsequent years, studies were performed to establish the food chain transfer of plutonium in algae, sunfish, and bass. Sediment concentrations of plutonium-239/plutonium-240 were on the order of 1 to >1,000 pCi/g dry weight. Because the pond was very shallow, the exposure of goldfish to plutonium was very high. Of all the biological components of the pond, algal floc had the highest concentrations of plutonium and this formed the basis of the food web for the goldfish that lived and reproduced there in huge numbers.

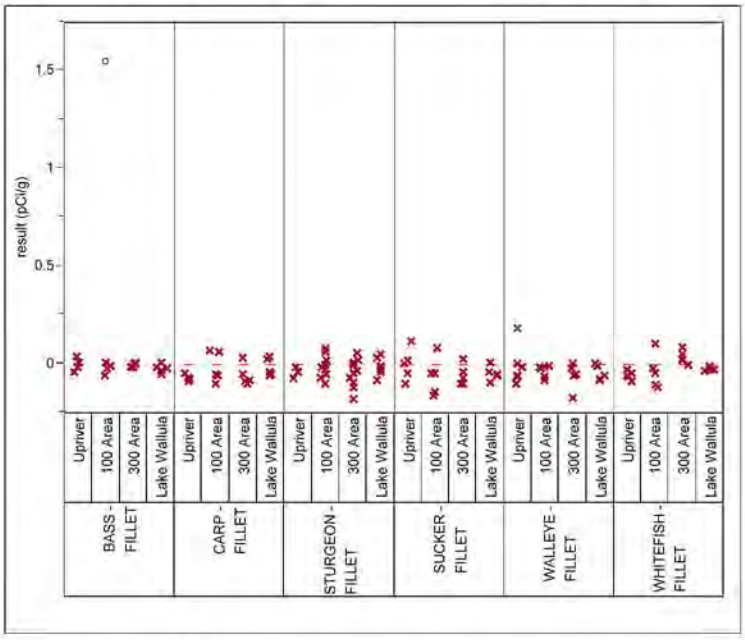
Average muscle concentrations in the goldfish from the pond were 0.89 pCi/g, and most of the activity was found in the gut. Comparable levels of americium-241 were also found. In the studies with bass and sunfish, maximum measured fillet concentrations were 0.013 pCi/g for plutonium-239/40. When compared to available fish data at that time, the fish from this pond had the highest reported plutonium-239/plutonium-240 concentrations in the world (Emery et al. 1981).

The finding of a single detect of plutonium-239/plutonium-240 in a composite sample of bass fillet does not appear realistic in the context of these previous studies. The 216-U-10 pond fish lived in an environment where plutonium-239/plutonium-240 activities were at least 100 times greater than those in the Columbia River, and yet plutonium-239/plutonium-240 in fish tissue never reached levels similar to that found in the reported samples from the Columbia River. Plutonium is not known to preferentially accumulate in muscle tissue (ATSDR 2010).

The fact that plutonium-239/plutonium-240 was not detected in either carcass or liver/kidney samples in this single fish sample further supports that the fillet result is a false-positive. The lack of correlation among muscle, carcass, liver, and kidney results, in conjunction with other information discussed above, form the basis for elimination of these data from further quantitative evaluation in the HHRA.

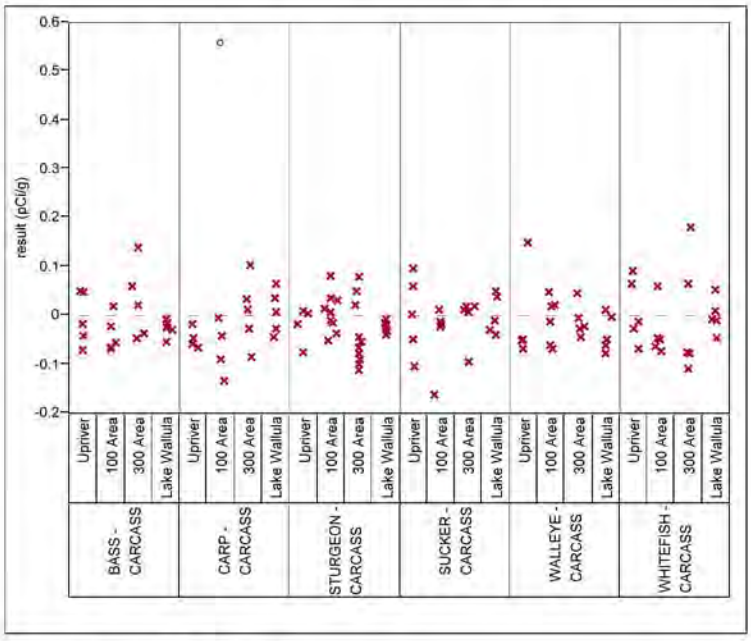
3.6.4.4.4 Strontium-90. Figures 3-35, 3-36, and 3-37 present box plots of the analytical results for strontium-90 in fish tissue samples. Strontium-90 was detected in only one bass fillet sample (100SA-BASS2) from the 100 Area Sub-Area.

Figure 3-35. Box Plot of Strontium-90 in Fish Fillet.



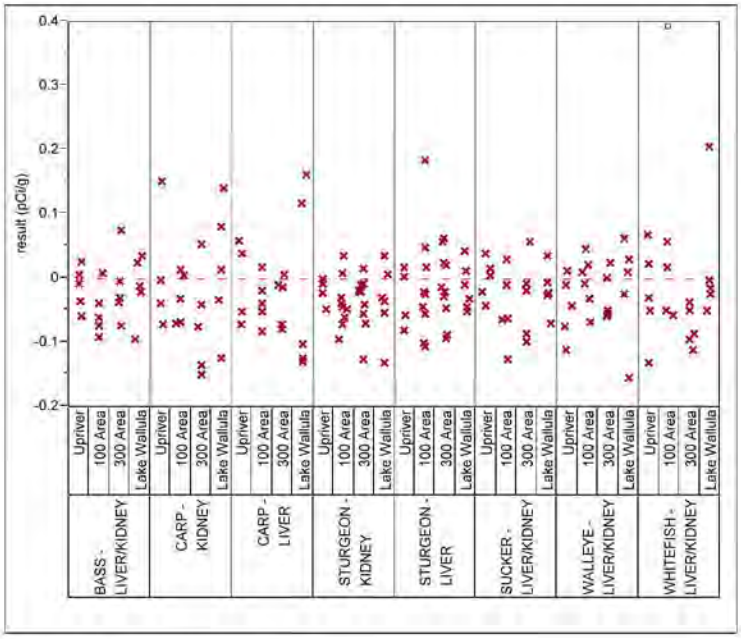
x = not detected above MDA
O = detected above MDA

Figure 3-36. Box Plot of Strontium-90 in Fish Carcass.



x = not detected above MDA
O = detected above MDA

Figure 3-37. Box Plot of Strontium-90 in Fish Liver/Kidney.



x = not detected above MDA
O = detected above MDA

Similar to plutonium, the single detected strontium-90 result found in each tissue type appears anomalous and suspect. In fish tissue, strontium-90 was detected in only one fillet sample at an activity of 1.55 pCi/g and only three times in total out of all fish tissue samples analyzed (FOD <1%).

The activity reported in the bass fillet sample is at least 2.5 times higher than any activity reported in carcass, liver/kidney, or even viscera samples. Furthermore, there was no consistency in strontium-90 detection in species or tissue type. Three of the detects were found in the 100 Area Sub-Area, over a stretch of approximately 23 km (14 mi). In addition to the bass fillet sample, strontium-90 was reported in a carp carcass sample (J196C2) and in whitefish liver/kidney (J18K12) at activities of 0.558 pCi/g and 0.392 pCi/g, respectively. The only other fish tissue sample in which strontium-90 was detected was in a sturgeon viscera sample collected from Lake Wallula (J195W0; 0.456 pCi/g). However, viscera data were not evaluated in this HHRA.

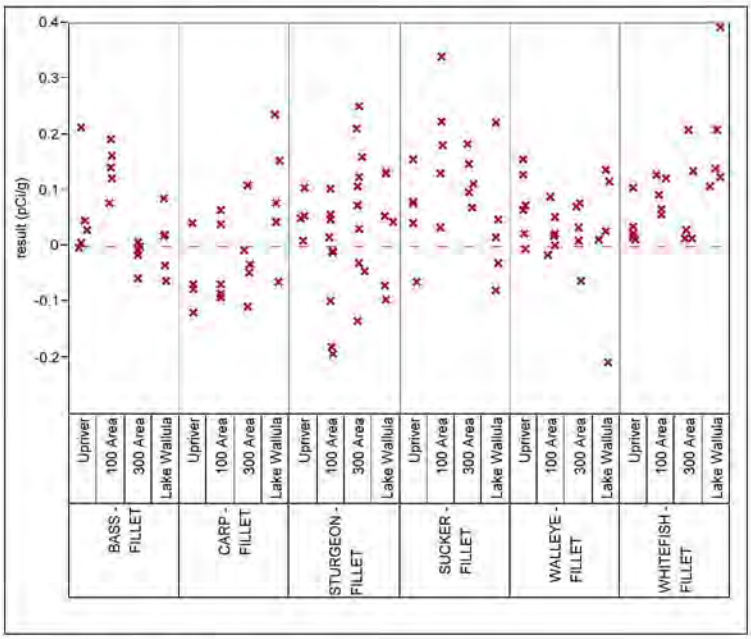
Strontium is known to preferentially accumulate in bone tissue (ATSDR 2004, *Toxicological Profile for Strontium*). However, three of the detected results occurred in nonbone tissue, and the highest detect occurred in bass fillet. Note that the maximum strontium-90 detection occurred in the same tissue sample in which the maximum plutonium-239/plutonium-240 detection was found, further suggesting that sample contamination and/or laboratory error may have biased the results.

Strontium (both elemental and radioactive forms) is abundant in surface water at the Hanford Site. The highest historical groundwater concentrations of strontium-90 have been noted in the 100-N Reactor area, and a known strontium-90 plume discharges to the river adjacent to the 100-N Reactor area near RM 379. Strontium-90 was detected at a lower frequency (26%) in sediment within the Hanford Site Study Area. One would anticipate that were this radionuclide readily accumulating in fish tissue, it would be prevalent at a higher FOD or on a more consistent basis within a species than what was observed in fish tissue samples.

Based on this evaluation, strontium-90 results in fish tissue samples were not retained for further quantitative evaluation in the HHRA.

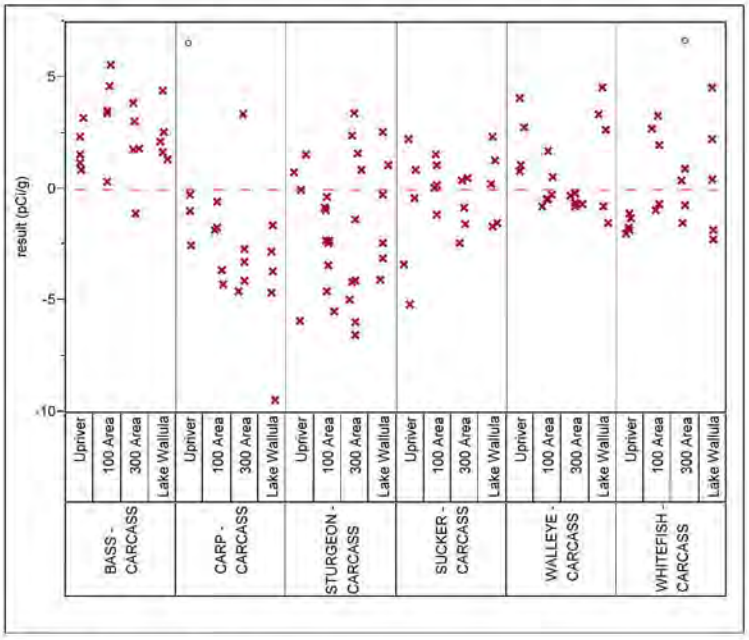
3.6.4.4.5 Technetium-99. Fish tissue results for technetium-99 are presented in Figures 3-38 through 3-40. Technetium-99 was detected in only 1 of 347 fish tissue samples collected from the Study Area. This radionuclide was detected in a bass sample collected from the 300 Area Sub-Area (300SA-BASS5) at an activity of 0.327 pCi/g. Technetium-99 was also detected above the MDA in a sucker sample obtained from the Upriver Sub-Area at a slightly higher activity (0.489 pCi/g, URSA-SUCKER 5). Both detected results are within a factor of two times the MDA in corresponding liver/kidney samples.

Figure 3-38. Box Plot of Technetium-99 in Fish Fillet.



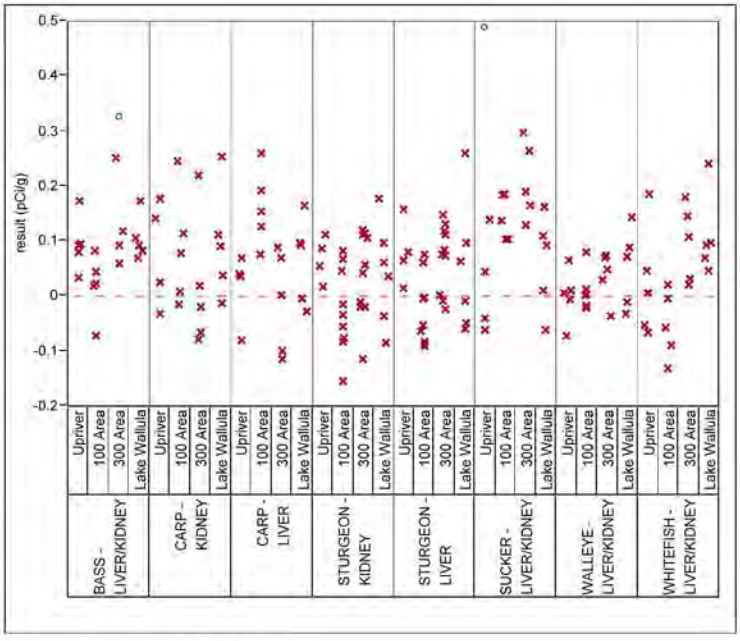
x = not detected above MDA

Figure 3-39. Box Plot of Technetium-99 in Fish Carcass.



x = not detected above MDA
O = detected above MDA

Figure 3-40. Box Plot of Technetium-99 in Fish Liver/Kidney.

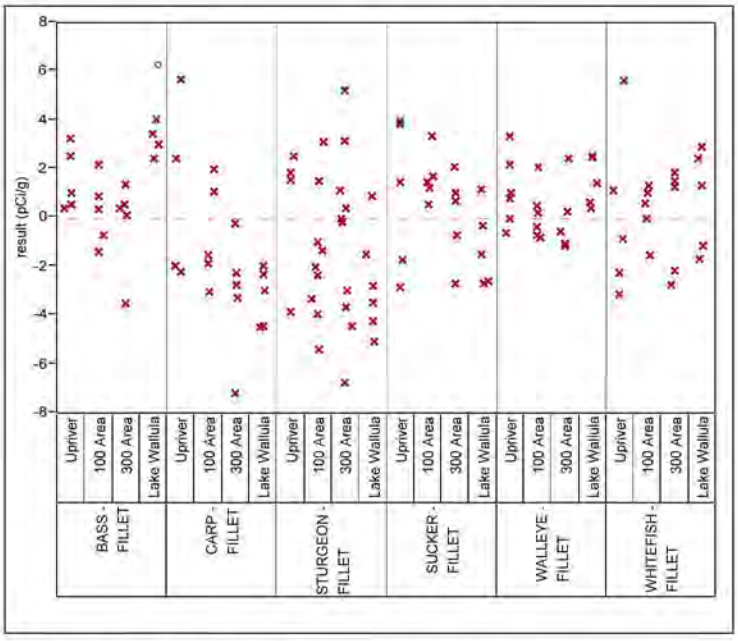


x = not detected above MDA
O = detected above MDA

Because technetium-99 was detected in only one bass liver/kidney sample from the Study Area, at an activity similar to the MDA of other samples and lower than the activity reported for the Upriver sucker sample, it is suspected that this result may be a false-positive. Therefore, this radionuclide was eliminated from further quantitative evaluation in the HHRA.

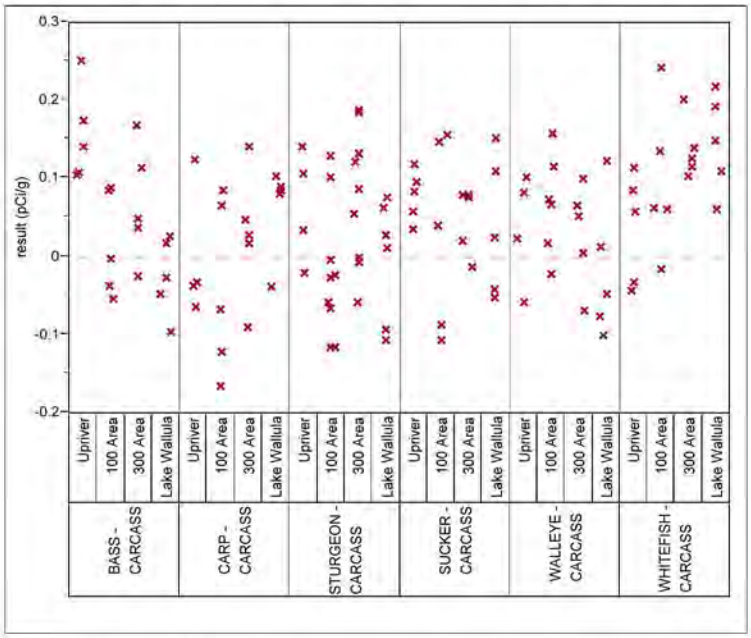
3.6.4.4.6 Tritium. Tritium results for the six fish species are presented in Figure 3-41 for fillet, Figure 3-42 for carcass, and Figure 3-43 for liver and kidney samples. In fillet, tritium was detected in only one bass sample collected from Lake Wallula (LWSA-Bass4). The reported activity of 6.25 pCi/g slightly exceeded the MDA of 5.49 pCi/g from the corresponding sample. In other tissues, tritium was detected in three 300 Area Sub-Area samples (whitefish carcass, 300SA-WF4, and sturgeon liver/kidney, STURGEON-17 and STURGEON-18) and one Upriver carp carcass sample (URSA-CARP4).

Figure 3-41. Box Plot of Tritium in Fish Fillet.



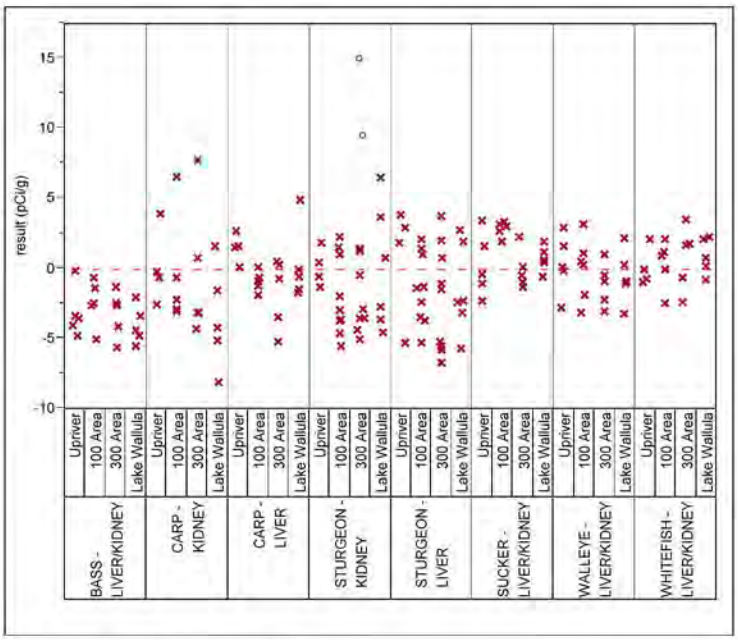
x = not detected above MDA
O = detected above MDA

Figure 3-42. Box Plot of Tritium in Fish Carcass.



x = not detected above MDA

Figure 3-43. Box Plot of Tritium in Fish Liver/Kidney.



x = not detected above MDA
O = detected above MDA

Activities reported for fillet and carcass results were similar, ranging from 6.25 to 6.54 pCi/g. Activities reported in liver/kidney samples were higher, ranging from 9.54 to 15 pCi/g; both of these results were qualified as estimated values (J-qualifiers).

With respect to other media, analytical results for tritium in Study Area samples are somewhat consistent. Tritium levels in surface water in the 300 Area Sub-Area downstream of the Hanford townsite are elevated in an area of a known plume discharge. The Study Area fish carcass and liver/kidney samples in which tritium were detected were also obtained from the 300 Area Sub-Area, at RM 350 for whitefish and at RM 347 for sturgeon samples. However, there is no known corresponding tritium source for the bass fillet sample, which was composited from five fish caught near RM 338 along Leslie Groves Park in Richland, nor for the Upriver sample carp sample result, which was similar in magnitude to the 300 Area whitefish carcass tritium result. Because of this, tritium fish analytical results are not carried through the quantitative risk assessment; instead, exclusion of these data and implications for addressing health risk are discussed in the Uncertainty Analysis (Section 7.0).

3.6.4.5 Comparison of RI Fish Tissue Results to Results from Other Fish Studies. As discussed in Section 2.6, other fish studies have been conducted in the Columbia River. The fish tissue results from the CRC RI were compared to previously collected data from the following studies:

- EPA 910-R-02-006, *Columbia River Basin Fish Contaminant Survey 1996-1998*
- DOE/RL-2005-42, *100 Area and 300 Area Component of the RCBRA Sampling and Analysis Plan*
- CH2MHILL 2007, *Phase I Fish Tissue Sampling Data Evaluation, Upper Columbia River Site, CERCLA RI/FS (Final).*

Note that fish analytical results from these other studies were not used in this HHRA to quantify health risk from fish consumption, as discussed in Section 2.6. The CRC RI fish study was designed with the specific goal of providing a data set appropriate for evaluation of human fish consumption. Rather, the comparison discussed in this section is provided to the reader for additional information about the nature and distribution of fish body burdens within the Columbia River. This discussion is intended to provide an overall context of other fish tissue analytical results from the river.

These previous river investigation studies were summarized in Section 2.6. Although fish species, sample types (e.g., fillet, liver, carcass), analytical methods, and sample preparation methods were not identical among the studies, preventing a direct comparison of results to those of the CRC RI, general observations are summarized below for select constituents, including DDE, PCBs, arsenic, cadmium, lead, and mercury. In general, results from all of the studies are similar with regard to the types of constituents detected, their prevalence, and relative magnitude

of concentration. Elevated concentrations of similar constituents are seen from the CRC RI and these other studies.

3.6.4.5.1 EPA/CRITFC Survey. In 1994, the EPA and CRITFC's member tribes initiated a survey of contaminants in fish tissue in the Columbia River Basin. Sample collection took place between 1996 and 1998 and the results of this study were published by EPA in 2002, in EPA 910-R-02-006.

A total of 281 samples of fish and fish eggs were collected for the EPA/CRITFC study from 5 anadromous species (Pacific lamprey, smelt, coho salmon, fall and spring chinook salmon, and steelhead) and 6 resident species (largescale sucker, bridgelip sucker, mountain whitefish, rainbow trout, white sturgeon, and walleye). The following four types of samples were collected: whole-body with scales, fillet with skin and scales, fillet without skin (white sturgeon only), and eggs. All the samples were composites of individual fish, except white sturgeon. The number of fish in a composite varied with species, location, and tissue type. Eleven samples of eggs were collected from steelhead and salmon.

While analytical methods and sample preparation were not identical between the CRC RI Data Summary Report (WCH-398 [2008 to 2010]) and EPA 910-R-02-006, and therefore cannot be directly compared, the following general observations are provided:

- **Species collected.** Both studies sampled resident species, including sucker, whitefish, sturgeon, and walleye. The EPA study also included anadromous fish species (as discussed above), whereas the RI sampling did not because their life cycle includes primarily nonriver habitat (i.e., ocean).

- **Sample preparation.** Both studies composited fish tissue prior to analyses. However, sample preparation techniques were generally dissimilar. This is a major obstacle to a direct comparison of results from the two studies. The EPA study sampled primarily whole fish and fish eggs, while the CRC RI study collected individual samples from fillet, carcass, and organs. The fillet samples that were collected by EPA do, however, appear to have been prepared in a manner similar to that of the RI study, and thus some of these results may be directly compared.

- **Sample analysis.** Fish tissue samples from the CRC RI were not analyzed for either dioxins or furans as they were in the EPA study. Additionally, EPA analyzed PCBs by Aroclors and a subset of the 209 PCB congeners, whereas the CRC RI study analyzed all 209 PCB congeners. Because of differences in the analytical methodologies used in the two studies to evaluate PCBs, direct comparison between total PCB concentrations in fish tissue may not be appropriate. However, both studies did report that the highest concentrations of total PCBs in fish tissue were found in whitefish.

The comparison of minimum and maximum detected concentrations in fish fillet (with skin on, with the exception of sturgeon) by species for both studies is presented below for select

constituents, where a direct comparison may be possible: DDE, PCBs, arsenic, cadmium, lead, and mercury.

Species	Study	DDE		PCBs [a]		Arsenic		Cadmium		Lead		Mercury	
		Min	Max	Min	Max	Min	Max	Min	Max	Min	Max	Min	Max
Sturgeon [b]	1996-1998	0.1	1.4	0.12		0.15	0.64	<0.004	0.006	<0.01	0.029	0.038	0.43
	2009-2010	0.041	0.833	0.088	0.4	0.32	1.1	0.036	0.053	ND		0.0134	0.612
Whitefish	1996-1998	0.008	0.91	0.19		0.051	0.14	<0.004	0.014	<0.01	0.026	<0.049	0.14
	2009-2010	0.0736	0.592	0.0658	3.74	0.21	0.36	0.038	0.055	1.59	1.59	0.015	0.099
Walleye	1996-1998	0.044	0.052	0.03		0.29	0.4	ND		<0.01	<0.01	0.16	0.2
	2009-2010	0.0135	0.655	0.0123	0.598	ND		0.047	0.047	0.26	0.26	0.098	0.721
Smallmouth bass	1996-1998	0.48	1.2	--		0.11	0.17	ND		0.01	0.055	0.38	0.47
	2009-2010	0.0118	0.239	0.0226	0.233	ND		0.035	0.051	ND		0.035	0.122

NOTE: Concentrations in mg/kg, wet weight.

[a] 1996-1998 data for average concentration of total aroclors (1242, 1254, 1260). 2009-2010 data for total PCB congeners. 1996-1998 data from EPA 910-R-02-006, *Columbia River Basin Fish Contaminant Survey 1996-1998.*

[b] Data presented for sturgeon without skin; fillet only.

-- = indicates no data available ND = constituent was not detected

> = constituent not detected above given reporting limits PCB = polychlorinated biphenyl

DDE = dichlorodiphenyldichloroethylene

The DDE concentrations from the EPA study ranged from a minimum of 0.008 mg/kg in whitefish samples to a maximum of 1.4 mg/kg in sturgeon. The range from the CRC RI data was similar, although slightly lower for smallmouth bass. Overall, the 1996 to 1998 levels were slightly higher than those of the CRC RI.

For PCBs, the average of the total Aroclors (Aroclor-1242, Aroclor -1254, and Aroclor -1260) from 1996 to 1998 (no ranges provided) was compared to the average total PCB results from the CRC RI samples (the total PCB concentration is equivalent to the sum of the detected congeners). For all species, mean concentrations reported in the CRC RI were generally within an order of magnitude of the mean concentrations reported by EPA.

Arsenic concentrations from the EPA study ranged from a minimum of 0.051 mg/kg in whitefish to a maximum of 0.64 mg/kg in sturgeon. Results from the RI were similar, ranging from nondetect in walleye and smallmouth bass to a maximum concentration of 1.1 mg/kg in sturgeon. The CRC RI arsenic levels for sturgeon and whitefish were slightly higher than those reported in the EPA study, whereas the walleye and smallmouth bass results were lower.

Cadmium concentrations from the EPA study ranged up to 0.014 mg/kg, with highest concentrations reported in whitefish. Fish samples from the RI were generally up to eight times higher, ranging from 0.035 mg/kg in smallmouth bass to 0.055 mg/kg in whitefish. Maximum concentrations for all four species were higher in the RI samples than the EPA samples.

Lead concentrations from the EPA study ranged from nondetect in sturgeon, whitefish, and walleye to 0.055 mg/kg in smallmouth bass. Concentrations reported in the RI ranged from nondetect in sturgeon and smallmouth bass to 0.26 and 1.59 mg/kg in walleye and whitefish, respectively. Within a particular species, CRC RI concentrations were higher for whitefish and walleye and lower for sturgeon and smallmouth bass.

Mercury concentrations from the EPA study ranged from nondetect in whitefish to 0.47 mg/kg in smallmouth bass. During the CRC RI, concentrations ranged from 0.013 mg/kg in sturgeon to 0.72 mg/kg in walleye. Within a particular species, concentrations detected in RI samples were up to approximately four times higher than those found in the EPA study for sturgeon and walleye, but were lower for whitefish and smallmouth bass.

Overall, for a given contaminant, with the exception of DDE, the highest concentrations were detected in the samples collected during the RI. Of the constituents evaluated, results of only PCBs and cadmium were higher during the CRC RI sampling across all species evaluated. DDE, arsenic, lead, and mercury maximum concentrations within an individual species were higher during the EPA study for some constituents and higher during the CRC RI for other constituents, reflecting the overall variability observed in fish tissue data in both studies.

3.6.4.5.2 RCBRA. Sculpin and juvenile sucker tissue samples (liver, kidney, and whole organism samples) were collected under the SAP (DOE/RL-2005-42). Data for the RCBRA study were collected between 2005 and 2007, with supplemental fish tissue sampling in 2008. For the CRC RI, bridgelip sucker tissue samples (carcass, liver/kidney, and fillet) were collected in 2009 and 2010; however, sculpin were not collected as part of the CRC RI.

The minimum and maximum detected concentrations in sculpin and juvenile sucker liver and kidney tissue samples from the RCBRA are presented below for select constituents. These results are compared to the bridgelip sucker results (i.e., liver/kidney tissue composite) from the CRC RI.

Although a direct comparison is not possible because the preparation of tissue samples differs, these results are included for general observations in the concentrations of arsenic, cadmium, lead, and mercury in the liver and kidney fish tissue samples. Because the RCBRA study did not include fillet samples, no comparison of fillet results is presented. The sculpin data are included in the table for informational purposes only. A comparison is not made because sculpin were not collected for the CRC RI.

Species	Study	Tissue Type	Arsenic		Cadmium		Lead		Mercury	
			Min	Max	Min	Max	Min	Max	Min	Max
Bridgelip sucker	CRC RI	Liver/ kidney	ND	ND	0.709	3.7	ND	ND	0.014	0.054
Juvenile sucker	RCBRA	Kidney	3.2	4.1	0.18	2.8	ND	ND	ND	ND
Juvenile sucker	RCBRA	Liver	ND	ND	0.14	0.36	ND	ND	ND	ND
Sculpin	RCBRA	Kidney	0.77	1.7	0.42	2.7	1.3	3.1	0.05	0.12
Sculpin	RCBRA	Liver	0.72	1.6	0.33	4	0.3	0.75	0.02	0.19

NOTE: Concentrations in mg/kg, wet weight.
CRC = Columbia River Component
ND = constituent was not detected
RCBRA = River Corridor Baseline Risk Assessment
RI = remedial investigation
Source of RCBRA data: Tables 6-28, 6-29, 6-34, and 6-35 of DOE/RL-2007-21, Rev. 0.

Arsenic was detected in only two of nine sucker kidney tissue samples from the RCBRA, but was not detected in any of the sucker liver/kidney composite samples from the CRC RI. The reporting limits for arsenic for sucker liver/kidney samples from the CRC RI were generally between 0.5 mg/kg and 1 mg/kg (see Figure 3-9).

Cadmium was detected in eight of nine kidney sucker tissue samples and in four of seven sucker liver tissue samples from the RCBRA at slightly lower concentrations than the CRC RI. Cadmium concentrations in liver/kidney sucker samples from the CRC RI were generally within the range of concentrations observed in the RCBRA study sculpin and sucker samples.

Neither lead nor mercury was detected in any of the kidney or liver samples from the RCBRA. Lead was not detected in the liver/kidney samples from sucker collected for the CRC RI. Mercury was detected in the CRC RI samples at concentrations lower than those reported for the sculpin samples.

The PCB Aroclors and DDE were analyzed for only in whole organism juvenile sucker and sculpin samples from the RCBRA, which are not directly comparable to the CRC RI tissue samples. Relevant points of comparison are as follows:

- Of eight samples of suckers analyzed for PCB Aroclors, only Aroclor-1254 was detected in one sample at a concentration of 0.0057 mg/kg. In the sculpin whole organism tissue. samples, Aroclor-1254 was detected in three samples at concentrations ranging from 0.024 mg/kg to 0.025 mg/kg.

- Aroclor-1260 was detected in one sculpin sample at 0.015 mg/kg.

- Total PCB concentrations (based on congener analysis) in sucker samples from the CRC RI study were generally higher, with most of the observed concentrations within a range of approximately 0.25 to 0.5 mg/kg (e.g., see Figure 3-4).

- DDE was detected in seven of the eight sucker whole organism samples from the RCBRA at concentrations ranging from 0.013 mg/kg to 0.071 mg/kg. In sculpin, DDE was detected in 34 of 35 samples at concentrations ranging from 0.0055 mg/kg to 0.29 mg/kg. In the CRC RI sucker tissue samples, DDE concentrations ranged from 0.0849 mg/kg in fillet samples to a maximum of 1.49 mg/kg in the carcass.

As reported in the RCBRA HHRA (DOE/RL-2007-21), tissue concentrations of these contaminants in sculpin captured in the near-shore environment as part of the RCBRA investigation are either comparable to or below concentrations in various game fish reported in EPA/910/R-02/006 and EPA 2007, "Recommendations for Human Health Risk-Based Chemical Screening and Related Issues at EPA Region 10 CERCLA and RCRA Sites." However, PCB and DDE concentrations found in the CRC RI study appear generally higher than those reported in these studies. Again, this comparison should be interpreted with caution, since the analytical and sampling methods may vary greatly among all of these fish tissue studies.

3.6.4.5.3 Lake Roosevelt/Upper Columbia River. The September/October 2005 Phase I fish tissue sampling program in Lake Roosevelt conducted as part of the Upper Columbia River RI/FS was designed to gather data to support (1) human and ecological risk assessments and (2) analyses to consider issuance of an updated fish advisory for Lake Roosevelt (CH2MHILL 2007). As part of this program, 198 fish composite samples were collected from walleye (whole body, offal [e.g., internal organs], and fillet samples), rainbow trout (whole body, offal, and fillet samples), whitefish (whole body samples only), largescale sucker (whole body samples only), and burbot (whole body samples only) at six locations between the U.S./Canadian border and Grand Coulee Dam.

While the Lake Roosevelt results cannot be directly compared with the fillet, carcass, kidney, and liver results from the CRC RI due to the way the fish tissue samples were segregated prior to analysis, they are presented below for informational purposes. Because this study was conducted in an area upriver of the Hanford Site Study Area, the results from this study may be used as an additional line of evidence in elucidating "background" contaminant body burden of various contaminants that have accumulated in fish tissue.

The following analyses were conducted during the Lake Roosevelt sampling event: target analyte list metals, PCB Aroclors, dioxins and furans, PCB congeners, inorganic arsenic, percent lipids, and percent moisture.

General conclusions of Phase I fish tissue sampling include the following (CH2MHILL 2007):

- Tissue concentrations of cadmium, copper, lead, nickel, and uranium were greatest in the largescale sucker, with concentrations tending to be higher in the most upstream portions of the site (near the U.S./Canadian border). Zinc was also elevated in largescale suckers and mountain whitefish, particularly in the most upstream area (i.e., Reach 1).

- Tissue concentrations of arsenic were three to five times higher in burbot compared to other species. Total arsenic tissue concentrations in burbot increased downstream (i.e., higher in the lake-like portion of the site).

- Mercury was detected in tissues of all species evaluated, with the highest concentrations in walleye, burbot, and largescale suckers. The elevated concentrations in walleye and burbot are consistent with their feeding habits (i.e., both are higher trophic-level consumers that feed on other fish). There is a significant downstream increase in mercury tissue concentrations.

- Total PCB tissue concentrations (as Aroclor) were similar for walleye, wild and hatchery rainbow trout, whitefish, and burbot. Concentrations in largescale suckers were about 2.5 times higher than other species.

Because no whole fish or offal samples were analyzed as part of the CRC RI, these Phase I results cannot be directly compared with the fillet, carcass, kidney, and liver results from the CRC RI. Analytical results from fillet samples are the only sample types that can be directly

compared between the Upper Columbia River RI/FS and this CRC RI; however, the only fish species common to both the Upper Columbia River and CRC RI studies with fillet analysis is walleye. A comparison of arsenic, cadmium, lead, and mercury results for walleye fillet samples is provided below. Phase I fish tissue data are from Tables 3-9 and 3-32 of CH2MHILL (2007). Pesticides were not analyzed for in the Phase I samples.

Species	Study	Tissue Type	Arsenic		Cadmium		Lead		Mercury	
			Min	Max	Min	Max	Min	Max	Min	Max
Walleye	CRC RI	Fillet	ND	ND	0.047	0.047	0.259	0.259	0.098	0.721
Walleye	Phase I	Fillet	0.06	0.18	0.005	0.0054	0.005	0.062	0.181	0.417

NOTE: Concentrations in mg/kg, wet weight.
CRC = Columbia River Component
ND = constituent was not detected
RI = remedial investigation

Arsenic was detected in walleye fillet samples from the Phase I study at a mean concentration of 0.11 mg/kg. Arsenic was not detected in the 22 walleye fillet samples collected for the CRC RI; however, the reporting limits from the CRC RI walleye fillet samples were higher than the detected concentrations from the Phase I study.

Cadmium was detected in only one walleye fillet sample from the CRC RI at a concentration almost an order of magnitude higher than the Phase I maximum detected concentration. Similarly, the one detection of lead of 0.259 mg/kg from CRC RI walleye fillet sample was approximately 50 times higher than the maximum concentration reported in the Phase I study.

Mercury concentrations from the Phase I study had a mean concentration of 0.267 mg/kg. Mercury was detected at similar concentrations in the walleye fillet samples from the CRC RI, with a mean concentration of 0.263.

3.6.4.5.4 Summary. In summary, although the results of these three studies are not directly comparable to those of the CRC RI due to differences in sampling methodologies and target analytes, the following general observations are provided. Heavy metals such as cadmium and mercury are routinely detected in fish samples at levels generally similar to those observed in the CRC RI, with some exceptions. PCBs and chlorinated pesticides such as DDE are also prevalent in fish tissue. It is interesting to note that there is not a large difference in the concentrations of these types of contaminants in fish tissue despite, in some instances, over a decade between sampling events, attesting to the environmental persistence of organochlorine compounds.

3.7 SELECTION OF CONTAMINANTS OF POTENTIAL CONCERN

Contaminants of potential concern are selected from among the analytes detected in each environmental medium and constitute those constituents for which risk is quantitatively evaluated. Selection of the appropriate COPCs is useful in streamlining the risk assessment process to focus on potentially significant risk drivers and for making remedial action decisions. Contaminant of potential concern selection should occur through a process that is deliberate,

systematic, and based on established selection criteria (EPA/540/1-89/002). This section describes the approach developed to identify and focus the COPCs identified for the risk assessment evaluation.

The COPC selection process is consistent with EPA guidance pertaining to selection of COPCs for risk assessment (EPA/540/1-89/002) and the approach specified in the RI Work Plan. This process generally follows the approach discussed by the Tri-Parties during meetings held in January through April 2008 for the RCBRA (DOE/RL-2007-21, Rev. 0). However, because the exposure media for the Columbia River consist largely of sediments and surface water rather than upland soils, the approach has been modified to reflect the characteristics of the data set for those media. The selection process presented herein reflects that presented in the RI Work Plan (DOE/RL-2008-11) as well as subsequent discussions with the Tri-Parties.

The COPC refinement process includes a number of complementary steps and criteria, including consideration of a pre-selected list of contaminants that will be excluded or included, spatial distribution, and an evaluation of potential toxicity through a comparison of concentrations to risk-based screening criteria. In addition, the approach for COPC refinement outlines a process for distinguishing COPCs based on a statistical comparison of Hanford Site data to data collected from background or reference, to identify which COPCs are potentially related to or relatively elevated at the Hanford Site (this process is further described in Section 3.8).

The quantitative methods used as part of the statistical analysis provide valuable information for the included analytes and also provide a sound technical basis for eliminating less relevant analytes from the quantitative risk assessment. Figure 3-44 provides an overview of the COPC selection process. Each step of this process is discussed in the following subsections.

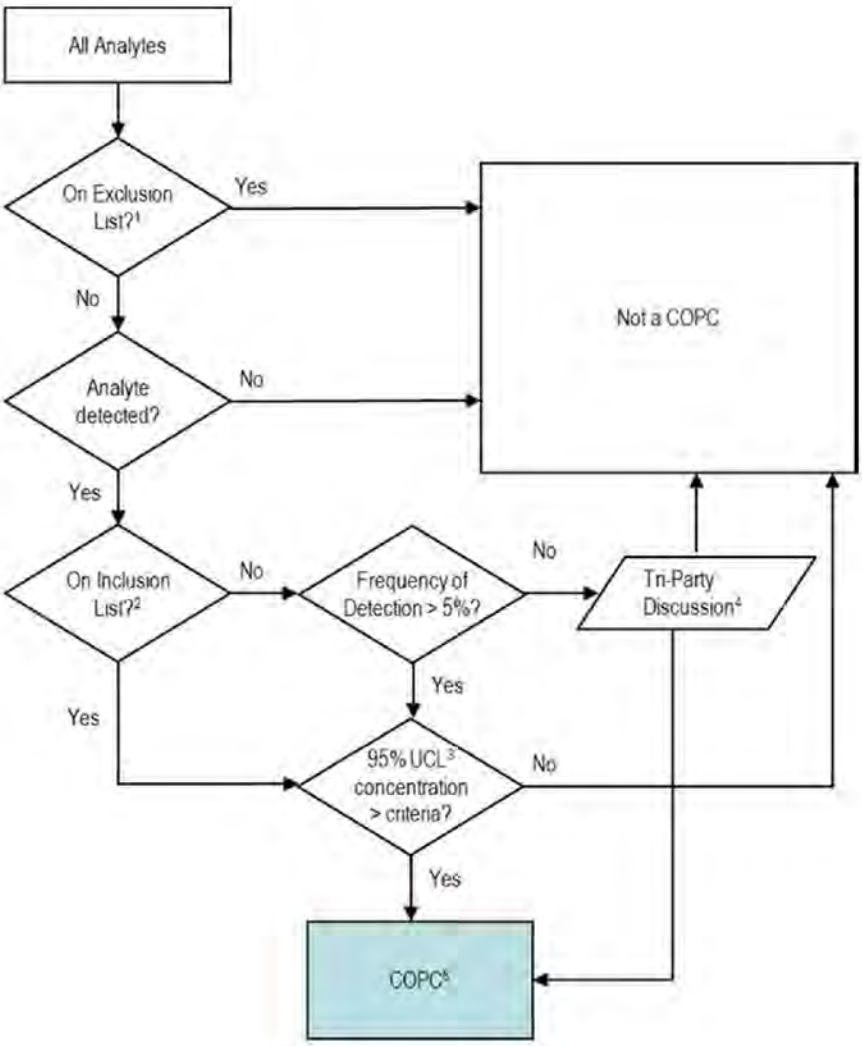
3.7.1 Consideration of Inclusion and Exclusion List Constituents

The COPC refinement process includes consideration of a pre-selected list of contaminants that are to be automatically excluded ("exclusion list" contaminants) from the risk assessment or included for further evaluation ("inclusion list" contaminants). The Inclusion and Exclusion Lists recognize and take advantage of the knowledge gained through decades of Hanford Site characterization and cleanup work that has preceded this assessment.

The use of automatic inclusion and exclusion lists has a number of advantages and disadvantages. The use of inclusion lists ensures that key Hanford Site contaminants are more likely to be retained as COPCs and evaluated in the risk assessment. As indicated in the previous section, inclusion list constituents were not ruled out as COPCs based on low FOD but instead were carried through to the next step (i.e., comparison to benchmarks).

Exclusion lists, on the other hand, save time and money by eliminating from the assessment constituents acknowledged to present negligible risk or that are known to be unrelated to Hanford Site releases (e.g., essential nutrients). However, exclusion of these contaminants may potentially underestimate risks if such constituents are present in elevated concentrations. Each of these lists is discussed in the following subsections.

Figure 3-44. Contaminant of Potential Concern Refinement Process Flow Diagram for All Media.



[1] Consisting of four groups of analytes
1) Half-life less than 3 years
2) Essential nutrient / element
3) Physical property of sample medium
4) Consensus of Tri-Party managers

[2] By consensus of Tri-Party managers, these analytes are retained for further evaluation.

[3] If a 95% UCL can not be calculated due to a low number of detected results, then the maximum concentration is compared.

[4] Consideration of factors such as spatial distribution, relative magnitude of concentration and frequency of detection are used to determine inclusion or exclusion as a COPC.

[5] Once identified, COPCs are segregated into Study Area and Reference COPCs, this process is outlined in Figure 3-45.

3.7.1.1 Exclusion List Constituents. Table 3-13 provides a summary of the exclusion list constituents that have been excluded as COPCs per the RCBRA (DOE/RL-2007-21, Rev. 0). These analytes have been excluded from consideration as COPCs in the river corridor by agreement among the Tri-Parties and based on relevant Hanford Site data. Primary reasons for excluding these contaminants include the following:

- Short (less than 3 years) half-life for radionuclides
- Essential nutrient status
- Water quality parameters such as alkalinity
- Physical measurements such as grain size or temperature
- Radionuclides known to be ubiquitous due to background sources (e.g., potassium-40).

Separate exclusion lists have been developed for upland waste sites and groundwater contaminant plumes. Constituents listed in Table 3-13 were not evaluated further in this HHRA.

3.7.1.2 Inclusion List Constituents. Inclusion list analytes were preferentially retained for further evaluation for all media and subareas in which they were detected at least once, based on evaluation of the commonly reported analytes in waste site cleanup reports or based on the most prevalent contaminants in the groundwater plumes. The analytes included as COPCs per the RCBRA (DOE/RL-2007-21, Rev. 0) are summarized in Table 3-14 and are based on soil and groundwater analytical results from the Hanford Site. This list includes analytes known or expected to be associated with former operations and activities at the Hanford Site, and which may or may not be detected in river media. Constituents that were detected at least once in a medium (i.e., regardless of FOD) and are listed on the "inclusion list" in the RCBRA were further screened relative to risk-based benchmarks (see Section 3.7.3) to determine whether these constituents should be included as COPCs.

An exception to this process is that related to inclusion list radionuclides in fish tissue. As discussed in Section 3.3.6.4.4, the sporadic detections of five radionuclides (cesium-137, plutonium-239/plutonium-240, strontium-90, technetium-99, and tritium) are suggestive of the presence of false-positives. Because these results were considered unrepresentative of true-positive results, these five radionuclides were excluded as COPCs from the HHRA.

3.7.2 Consideration of Detection Status

Detected analytes are the focus of COPC refinement. Constituents detected very infrequently or never detected are assumed to pose a relatively low health risk relative to more frequently detected constituents. Constituents that were never detected in a medium were not considered as COPCs and not carried through the quantitative risk assessment. Constituents detected in fewer than 5% of samples, where 20 or more samples were analyzed, were evaluated further to determine whether they should be retained as COPCs, with the exception of inclusion list analytes (see Section 3.7.1). This additional evaluation, which is generally consistent with EPA risk assessment guidance (EPA/540/1-89/002), considered spatial distribution and magnitude of concentration relative to human health screening benchmarks or other criteria. The evaluation

of nondetect or low frequency constituents is further discussed in Section 7.0, the uncertainty analysis.

3.7.3 Comparison of Upper-Bound Concentrations to Screening Criteria

The next step of the COPC selection process is a comparison of upper-bound concentrations of an analyte to human health risk-based screening criteria. This comparison was conducted as a means of focusing the COPC list on constituents that are the most toxicologically relevant to human health. All inclusion list compounds were carried through this screening process. All non-inclusion list analytes present at a FOD greater than 5% were also carried through this step, as indicated in Figure 3-44.

In accordance with the RI Work Plan, the 95% UCL of the mean concentration (or maximum, where a UCL could not be calculated due to low number of samples or low FOD) of each relevant analyte was compared to a variety of medium-specific human health risk-based screening criteria. This concentration screening approach is consistent with EPA risk assessment guidance (EPA/540/1-89/002). Constituents with a 95% UCL (or maximum) concentration exceeding the screening criteria were identified as COPCs that were then carried through the quantitative HHRA. Conversely, constituents with 95% UCL or maximum concentrations below these conservative benchmarks are assumed to pose relatively negligible risk and not evaluated further in the HHRA.

The human health benchmarks that were used to select COPCs for surface water, island soil, sediment, and fish tissue include both risk-based concentrations, which reflect potential health effects, as well as other regulatory standards and criteria, as available. The selection of the appropriate criteria relies on the EPA Region 10 Memorandum dated April 17, 2007 (EPA 2007), which provided recommendations for human health screening at EPA Region 10 CERCLA and RCRA sites. As per this memorandum, risk-based screening values for noncarcinogenic effects were adjusted downward by a factor of 10 to reflect a hazard quotient of 0.1; cancer-based values were based on 1×10^{-6} cancer risk and were not adjusted. A summary of the benchmarks to be considered in this evaluation is presented below by medium. Summaries of the benchmarks considered in this process are provided in Table 3-15 (sediment and soil), Table 3-16 (surface water), and Table 3-17 (fish tissue).

For each constituent, risk-based or regulatory criteria from a variety of sources, including both EPA and Ecology, were reviewed, and then the lowest potentially relevant value from among these individual sources was chosen as the final human health screening value. This value was then compared to the 95% UCL (where a 95% UCL could be calculated) concentration for each analyte detected at an FOD >5% (when at least 20 samples were analyzed for that constituent) in sediment, island soil, or surface water data collected from the study area. Constituents with 95% UCL concentrations exceeding the human health-based benchmarks were retained as COPCs to be carried through the quantitative risk assessment. For cases in which a benchmark was not available for a particular constituent, the benchmark for another constituent that was structurally similar to the chemical of interest was used, as appropriate (EPA/540/1-89/002). If a reasonable surrogate was not available, then the constituent was excluded as a COPC but was

addressed qualitatively in the uncertainty analysis (Section 7.0). Screening values were, however, available for most detected contaminants.

The following sections discuss the benchmarks for each environmental medium.

3.7.3.1 Sediment and Soil. Directly applicable human health criteria for sediment or island soils under a recreational or subsistence fishing exposure scenario have not been identified.[2] Therefore, to be protective, available human health-based benchmarks for residential exposures to soils were used as screening criteria for both sediment and island soil. The use of residential soil benchmarks for the evaluation of nonresidential soil/sediment exposures is likely very conservative, because the frequency of access to and contact with island soils and sediments is expected to be much lower than those for soils in a residential setting. Furthermore, some of the exposure pathways considered in soil benchmarks (e.g., produce ingestion for radionuclides) are not relevant for recreational exposure scenarios. Nevertheless, these benchmarks were used as conservative screening criteria for selection of sediment and island soil COPCs in order to refine the COPC selection process. The soil benchmarks were drawn from the following sources:

- OWSER 9355.4-24, *Supplemental Guidance for Developing Soil Screening Levels for Superfund Sites*.
- EPA/540/R95/128, *Soil Screening Guidance: Technical Background Document.*
- ORNL, 2012, "Regional Screening Levels for Chemical Contaminants at Superfund Sites," screening level/preliminary remediation goal website. Regional screening levels for residential soil were used. Regional screening levels were available for both chemical and radiological constituents.
- Ecology, 2012, Ecology Cleanup Levels and Risk Calculation (CLARC) Searchable Database, Method B Unrestricted Land Use Values for Soil. The lower of noncancer- and cancer-based values was applied.
- EPA/540-R-00-006, *Soil Screening Guidance for Radionuclides, Technical Background Document,* "Table D.1, Generic (No Accounting for Decay) Soil Screening Levels for Radionuclides," was used in this comparison. The minimum value between direct ingestion of soil and external radiation exposure was applied.

Table 3-15 summarizes these benchmarks for island soil and sediment.

3.7.3.2 Surface Water. Surface water in the stretch of the Columbia River comprising the study area is used for both recreational purposes (i.e., boating, fishing, swimming) and, after filtering and treatment, as a drinking water source for various municipalities (e.g., Richland).

[2] The freshwater sediment benchmarks identified to date are either focused exclusively on protection of ecological biota or are stated to be protective of both human health and ecological receptors and, thus, are not directly relevant.

For surface water, the following criteria were considered as relevant criteria:

- ORNL, 2012, "Regional Screening Levels for Chemical Contaminants at Superfund Sites" screening level/preliminary remediation goal website. Values for residential tap water were used. Regional screening levels were available for both chemical and radiological constituents.

- EPA 2009, *National Recommended Water Quality Criteria*. Values for "consumption of water and organisms" were used. These criteria are available only for chemical constituents.

- EPA 822-S-12-001, *2012 Edition of the Drinking Water Standards and Health Advisories*. Values for maximum contaminant levels (MCLs) were used if available; if not available, then Health Advisories were used (EPA 822-S-12-001). Radiological drinking water MCLs for radioisotopes were obtained from EPA/540-R-00-006.

- Ecology CLARC Searchable Database, Method B Surface Water Standards and Method B Groundwater Standards (Ecology 2012). The lower of noncancer- and cancer-based values was applied. CLARC values are available for only chemical constituents.

Surface water benchmarks are summarized in Table 3-16.

3.7.3.3 Fish Tissue. Fish screening criteria consisted of EPA regional screening levels for fish (ORNL 2012). These levels were adjusted to account for the enhanced fish consumption rate of the Avid Angler scenario (see Section 4.0). Fish screening levels are summarized in Table 3-17.

The fish data set was evaluated for COPC selection in two different ways, although the same overall approach was used to select fish tissue COPCs in either case. The initial analysis supported the assessment of all fish species (combined). For this approach, data from all fish species were combined from across all three sub-areas and evaluated for COPC selection for fish as a single exposure medium. The second analysis supported the assessment of ingestion risks of individual fish species. In this analysis, each of the six fish species was evaluated separately and COPCs selected for each fish species.

Fillet, carcass, and liver/kidney analytical results were used to select fish tissue COPCs. However, the majority of the fish consumption diet is assumed to come from fillet, and the risk-based screening levels are based on consumption rates for a diet assumed to consist primarily of fillet. Carcass and liver/kidney only comprise a small fraction of the total fish diet. For a typical angler, who mainly eats fillet, it was assumed that a small fraction of carcass (5%) could be inadvertently consumed along with fillet, from pin bones, etc. Native American groups may use bones for soup as well as consume liver and kidney, although fillet comprises most of their fish diet (Harris and Harper, 1997 and 2004). Therefore, carcass and liver/kidney comparison concentrations (i.e., 95% UCL or maximum) were multiplied by a factor of 5% to account for the small percentage of the total diet that these tissues comprise. This adjusted concentration was then compared to the screening criterion to select COPCs.

The selection of fish tissue COPCs based on all species is shown in Tables 3-21 through 3-23. For the six individual fish species (bass, carp, sturgeon, sucker, walleye, and whitefish), the COPC selection process is shown in Tables 3-24 through 3-35, respectively.

3.7.4 Summary of Contaminants of Potential Concern

Tables 3-18 through 3-20 summarize the selection process for each COPC based on this screening comparison for sediment, island soil, and surface water, respectively. Fish COPC selection for all species combined is presented in Tables 3-21 through 3-23, and for individual species in Tables 3-24 through 3-35. These tables show for each detected constituent basic summary statistics, 95% UCL and maximum concentrations, the final human health screening values, and the rationale for COPC inclusion or exclusion.

Note that for the purposes of identifying COPCs, the summary statistics and 95% UCLs (or maximum concentrations) used in the screening comparison for each medium were based on combined data from all three sub-areas. Sediment statistics included both shallow and deep (core) sediments. The combination of such data allowed for a robust data set per medium and more accurate determination of the 95% UCL (by reducing intra-media variability).

Table 3-36 provides a summary of COPCs for soil, sediment, surface water, and fish tissue (all species combined). Table 3-37 summarizes the COPCs selected for the individual fish species. As indicated in that table, the following observations can be made:

- VOCs and SVOCs were identified as COPCs for only one medium: surface water.
- Island soil and sediment had comparable lists of COPCs, due to use of the same conservative screening benchmarks and similar nature/distribution of detected analytes.
- Metals and radionuclides were most consistently selected as COPCs across all four media.
- Fish tissue had the highest number of COPCs compared to other media, with metals and PCBs/pesticides comprising the COPCs.

These COPCs are the constituents that were carried through the quantitative risk assessment.

3.8 EVALUATION OF REFERENCE CONCENTRATIONS

There are a number of sources unrelated to the Hanford Site releases that may potentially release contaminants to the Columbia River and contribute to cumulative health risk. Therefore, it is important to understand the contribution of these sources when evaluating risk. To accomplish this, data collected from the Hanford Site Study Area were compared to data collected from Reference areas (e.g., Upriver or tributaries or wasteways) that drain into the Columbia River, and were considered along with information about Hanford Site releases as well as local and regional sources of contaminants. The end result of this process was a determination of whether

a COPC was either "consistent with Reference" (i.e., a Reference COPC) or "not consistent with Reference" (i.e., a Study Area COPC) conditions.

The approach for COPC refinement includes a process for identifying those COPCs that are present at concentrations consistent with those in Reference areas using statistical comparisons. This process is consistent with guidance pertaining to selection of COPCs for risk assessment (EPA/540/1-89/002, Part A, Chapter 5, "Data Evaluation"). Additionally, analytical results were evaluated with respect to results reported in other published studies and databases, and considered along with process knowledge, history, and fate and transport information specific to the Hanford Site.

Note that this HHRA did not eliminate any COPCs from further evaluation based on these comparisons with reference concentrations (RfCs). Rather, these comparisons were used to classify COPC as either "not consistent with Reference" (Study Area COPC) or "consistent with Reference" (Reference COPC), such that the FS ultimately performed, if necessary, for the Hanford Site Study Area can focus on Site areas/constituents that may pose excess risks above and beyond baseline conditions.

It is also important to note that the objective of the Study Area-Reference comparison is to evaluate whether Study Area concentrations of COPCs are higher than, lower than, or consistent with those of Reference/OCI areas. This evaluation does *not* attribute the presence of a COPC directly to a Hanford Site source, although potential sources of contaminants are discussed where relevant. Thus, the presence of a constituent designated as a "Study Area" COPC is not necessarily related to Hanford Site releases; conversely, a constituent designated as a "Reference" COPC is not necessarily related solely to other anthropogenic or natural sources of contaminants.

3.8.1 Reference Comparison Approach

3.8.1.1 Reference Data Set. Contaminants unrelated to Hanford Site releases have been introduced into the Hanford Site Study Area by various sources, such as mining industries located upriver from the Hanford Site, irrigation returns, and locations where other rivers enter the Columbia River (collectively termed "other contributing influences" or "OCIs"). The potential current and historical contaminant sources upriver of the Hanford Site and in OCIs within the study area are described in detail in the RI Work Plan (DOE/RL-2008-11) and WCH-91, and include the following:

- **Upriver sources** – Mining operations, smelting, pulp and paper production, runoff from cities and agricultural areas, municipal and industrial wastewater treatment plants, atmospheric testing, and other activities that have released materials that reach the river.

- **Global contributing sources** – Worldwide atmospheric nuclear testing contributed to radionuclide contaminants in surface waters and ultimately to sediments throughout the Pacific Northwest. Associated contaminants consist primarily of radionuclides such as cesium-137 and strontium-90, along with shorter lived radionuclides such as cerium-141, zirconium-95/niobium-95, and ruthenium-103/106.

- **Naturally occurring sources** – The following naturally occurring inorganic elements and radionuclides have been detected at background sediment locations: antimony, arsenic, barium, cadmium, manganese, nickel, potassium, uranium, zinc, uranium-234, uranium-238, and potassium-40. Aluminum, arsenic, barium, cadmium, manganese, elemental uranium, tritium, uranium-234, and uranium-238 have been detected in surface water samples from Reference locations.

- **Municipal/urban sources** – NPDES-permitted discharges to the Columbia River include stormwater, minor industrial process wastewater, contact and noncontact cooling waters, treated waters, and construction sites. Urban contributions including unpermitted residential and commercial stormwater runoff, residential use of fertilizers and pesticides, and septic sewage systems are some of the potential sources of contamination from communities along the banks of the Columbia River.

- **Agricultural sources** – Water from the irrigation returns in the Hanford Site Study Area has been sampled, and contaminants include nitrogen, phosphate, copper, uranium, and suspended solids. Uranium is commonly present in phosphate-based fertilizers and is a natural constituent that weathers from some types of rocks in the region. Historical pesticide applications on agricultural land may have resulted in releases of arsenic, lead, and chlorinated pesticides to the Columbia River and its tributaries.

- **Commercial/recreational vessels** – Recreation and commercial activities on the Columbia River contribute contamination to surface water and sediments via marinas, boats, or other recreational watercraft, and discharge of bilge and ballast water, engine oil, spills, and materials associated with boat and shipyard maintenance.

- **Anadromous fish returns** – Fish throughout the world have body burdens of PCBs, pesticides, mercury, and other constituents known to biomagnify. Because the Columbia River provides spawning habitat to a variety of anadromous fish species, the return and death of these fish may potentially act as a source of such contaminants to the Columbia River (Rice and Moles 2006, *Assessing the Potential for Remote Delivery of Persistent Organic Pollutants to the Kenai River in Alaska*; Krummel et al. 2005, "Concentrations and Fluxes of Salmon-Derived Polychlorinated Biphenyls (PCBs) in Lake Sediments").

Sediment, surface water, island soil, and fish tissue data from the Hanford Site Study Area were compared to separate Reference/OCI data that consisted of samples collected from a subset of Reference/OCI areas. The Reference/OCI data set was different for each sub-area: wasteways

and tributaries that empty directly into or upstream of the Hanford Site Study Area were included as part of the Reference/OCI data set, as well as all samples from the Upriver Sub-Area. Table 3-38 summarizes the locations at which Reference/OCI samples were collected and used.

As described above, the Reference data set for all sub-areas includes a small number of samples from wasteways and irrigation returns, which convey runoff from agricultural fields located near the Columbia River. Because water and sediment of wasteways and irrigation returns may contain higher concentrations of some constituents than other Reference areas, the potential exists that inclusion of these analytical results may bias the Study Area-Reference comparison such that Reference area concentrations may be inflated and COPCs may be erroneously identified as Reference COPCs, when in fact they may be at higher concentrations in the Study Area.

To evaluate the potential effects of including wasteway and irrigation return data in the Reference data set, a Wasteway Supplemental Analysis (Appendix N) was conducted to determine whether including wasteway and irrigation return (WW/IR) data in the Reference data set had any effect on the findings or outcome of the Study Area-Reference comparison.

The results of this evaluation indicate that inclusion of the WW/IR data does not impact designation of a COPC as a Reference COPC. In nearly all instances, the detected concentrations of COPCs in the WW/IR data set were within or below those of other reference locations, including upriver areas and major tributaries. Where WW/IR concentrations were higher than those of Upriver and major tributary (UR/MT) locations, exclusion of WW/IR results from the comparative analysis did not result in a change from a COPC's status as a Reference COPC. Therefore, inclusion of the WW/IR does not change the conclusions of the HHRA. Full details of this evaluation can be found in Appendix N.

Reference/OCI samples were collected for four media: sediment, surface water, soil, and fish tissue. The sediment Reference/OCI samples were all "shallow" samples, collected from depths of 0 to 30 cm (0 to 12 in.) below the sediment/water interface. Soil and fish tissue Reference/ OCI samples were collected only in the Upriver Sub-Area (upstream of RM 388, Vernita Bridge).

Samples collected in the Upriver Sub-Area are shown in Table 3-39 for sediment, surface water, soil, and fish tissue. Sediment and surface water Reference/OCI samples for the 100 Area Sub-Area are shown in Table 3-40. Table 3-41 presents the Reference/OCI surface water and sediment samples in the 300 Area Sub-Area, and Table 3-42 contains this information for the Lake Wallula Sub-Area.

Summary statistics for the Reference/OCI data set are presented in Tables 3-43 through 3-60. These tables contain the number of samples, FOD, minimum, maximum, mean, and sampling location of maximum. All summary statistics are presented for only the constituents identified as COPCs in Study Area media. Sediment, soil, and surface water statistics are shown in Tables 3-43 through 3-45, respectively. The fish tissue data are presented for all species, combined in Tables 3-46 through 48 (fillet, carcass, and liver/kidney, respectively) and for the

individual fish species in Tables 3-49 through 3-60. These data are presented in this manner, since health risk from fish is addressed using two different approaches (across all species and by individual species). More discussion related to these approaches is provided in subsequent sections in this chapter.

3.8.1.2 Study Area Data Set. The "Study Area" analytical data used in the comparative analysis is the same HHRA data set described previously in Sections 3.2 and 3.3 and includes analytical results from the 100 Area, 300 Area, and Lake Wallula Sub-Areas (i.e., the Hanford Site Study Area). The data from each sub-area were compared to data from the appropriate Reference/OCI data set specific to each sub-area.

In the following sections, the methodology used to compare analytical results between the Hanford Site Study Area and the Reference/OCI area is described and then the results of those comparisons are presented.

3.8.1.3 Methodology to Select Study Area and Reference COPCs. Figure 3-45 depicts the process used to determine whether a COPC was classified as either a Study Area COPC or a Reference COPC. In general, the following questions were asked to make this determination:

1. Are there enough data to conduct a statistical analysis that will indicate whether COPC concentrations in the Study Area are elevated with respect to those in the corresponding Reference Area?

2. If there are adequate data, is there a statistically significant difference between concentrations observed in the Study Area relative to Reference/OCI areas?

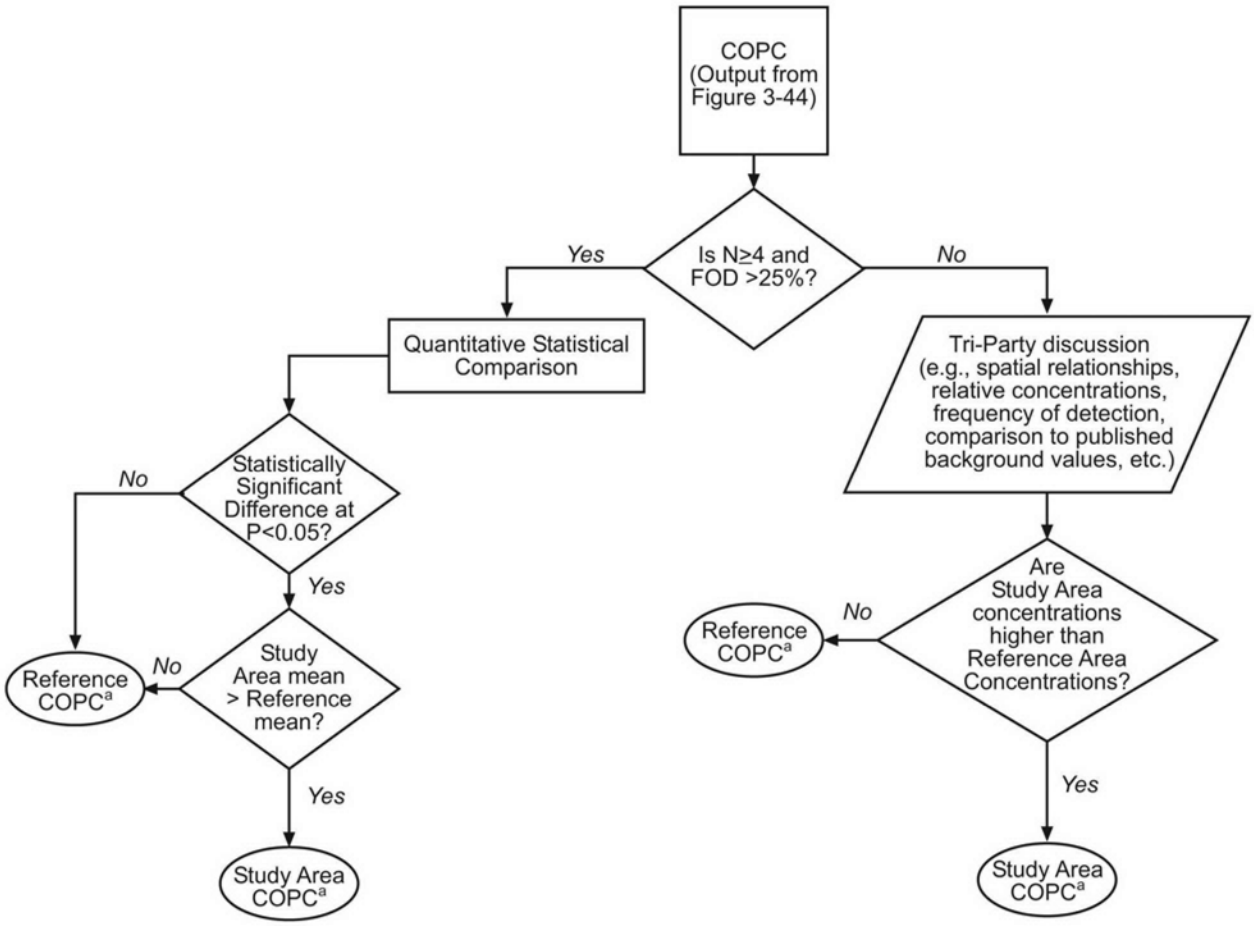
3. If there are few data or positive (i.e., detected) results, are Study Area concentrations generally consistent with or higher than those in Reference areas and/or levels cited in published studies?

4. Is there other information that would indicate or suggest that the COPC is related to reference conditions or to the Hanford Site?

3.8.1.4 Statistical Comparison. For each COPC in each medium (island soil, sediment, surface water, and fish tissue), two-sample statistical tests were used to compare concentrations of the constituents between Study Area and Reference/OCI locations, where adequate data were available (i.e., FOD >25%, n>4 samples). The specific test used for comparisons, described below and shown as a flow diagram in Figure 3-45, was dependent on characteristics of the Study Area and Reference/OCI data sets.

- If there were no detections of a constituent in either the Study Area and/or the OCI data set, no statistical comparison of means was made. Other information was used to make the determination.

Figure 3-45. Study Area to Reference Comparison Flow Diagram.



- If there were no censored results for a given COPC in both the Study Area and Reference data sets and the sample size (n) was greater than four, the Shapiro-Wilk test was used to test whether the distribution of the data sets approximated a normal distribution. If both data sets were normally distributed, a Student's t-test was used to compare the data sets. If either the Study Area or Reference/OCI data set was not normally distributed, a nonparametric Wilcoxon Rank-Sum test was used to compare the two data sets.

- If either the Study Area or Reference/OCI data set contained at least one censored value for a given constituent, the data sets were compared using a Generalized Wilcoxon Test (Kalbfleisch and Prentice 1980, *The Statistical Analysis of Failure Time Data*). This is a nonparametric test that tests the null hypothesis that the Study Area and Reference/OCI concentrations are the same and is a recommended approach over substitution methods for censored values (Helsel 2005). As with a standard Wilcoxon Rank-Sum test, the comparison is made between the sum of the ranks of the data in each data set. The Generalized Wilcoxon Test assigns an estimated rank to those data below the detection limit. This statistical test does not rely on a specific data distribution (e.g., is nonparametric) and addresses the fact that concentrations below a specific value (the reporting limit) are not known. This test is implemented in JMP's survival statistics platform, which was used to generate KM summary statistics (see Section 3.5.8). Standard comparative statistics were not calculated for data sets with more than 75% nondetect values and less than four detected sample results, as noted above. A qualitative analysis, as described below, was performed.

Statistical analyses were completed using JMP® Version 8.0.2. The input data set used in this analysis is the same as that described in Section 3.4.

The results of the statistical comparisons were used as an initial assessment of whether certain COPCs are present at concentrations elevated in the Hanford Site Study Area with respect to Reference/OCI areas. The null hypothesis being tested as part of the statistical comparisons is that analyte concentrations are the same between the Reference/OCI and Study Area locations. The alternative hypothesis is that these concentrations are different. An alpha (α), or Type I error rate, of 0.05 was used to determine if Study Area and Reference/OCI concentrations were significantly different. Setting α to 0.05 means that there is 95% confidence that the Study Area concentrations are different from Reference/OCI concentrations. Two-tailed statistical tests, described above, resulting in a p-value of less than 0.05 indicated Study Area concentrations are significantly greater or significantly less than Reference/OCI concentrations. Using an α of 0.05 provides a trade-off between Type 1 error ("false positive") and Type 2 error ("false negative").

Appendix E provides the output for these statistical comparisons. Tables in this appendix include results of the Shapiro-Wilk distribution test, Generalized Wilcoxon Test, Wilcoxon Rank-Sum, and Student's t-test.

The statistical p-values represent the answer to the null hypothesis (H_o): "Do the samples from the Reference/OCI areas and the Study Area sub-areas come from the same underlying population?" If the p-value is statistically significant, (i.e., less than 0.05), then the null hypothesis is rejected (i.e., false) and it is concluded that the samples come from different underlying populations. In those cases, the "different underlying populations" may have higher concentrations in the Reference/OCI data set or the Study Area data set. If the p-value is greater than 0.05, then the null hypothesis is not rejected and it is concluded that the samples come from the same underlying population. In such cases, the Study Area data are considered "consistent with Reference" because there is no statistical difference between the Reference and Study Area data sets.

Therefore, a COPC is considered "consistent with Reference" for two different statistical conditions: (1) if the p-value is less than 0.05 *and* the Reference/OCI mean is higher than the Study Area mean and (2) if the p-value is greater than 0.05. This second condition means there is no statistical difference between the Study Area and Reference/OCI mean concentrations and that the COPC is then considered to be consistent with Reference.

Thus, constituents detected at concentrations in the Study Area that are statistically significantly lower than concentrations in Reference/OCI areas, or are not significantly different from Reference/OCI concentrations are considered Reference/OCI-related and are categorized as Reference COPCs. If the p-value is less than 0.05 and the Study Area mean is greater than the Reference mean, however, then the COPC is considered "not consistent with Reference" and is categorized as a Study Area COPC.

3.8.1.5 Qualitative Evaluation. Across all media, there were constituents that had an insufficient number of detects for the statistical results to be considered valid (i.e., if greater than 75% of results were nondetect) or had a very low sample size, less than or equal to five. For this sub-set of constituents, a qualitative process was developed to determine if the Study Area data were consistent with Reference/OCI concentrations. For constituents with an FOD of less than 25% and/or a sample size less than or equal to four detected results, the qualitative analysis focused on detected results. A flow diagram of the decision-making process employed for this qualitative evaluation is presented in Figure 3-45.

The criteria used in determining consistency with Reference concentrations are as follows:

- COPC concentrations in Study Area samples are generally lower than or similar to those in Reference Area samples. As a rule of thumb, where maximum and mean concentrations were similar (i.e., the maximum was within a factor of two and the means were similar) in both data sets, a COPC was considered to be consistent with Reference. In some instances where Reference data were all nondetect, Study Area concentrations were evaluated with respect to the LRLs/MDA of Reference samples. If the maximum detected concentration in the Study Area was less than the LRL/MDA, then other information, such as typical background levels, was used in making the determination of Study Area or Reference COPC.

- COPC concentrations in soil, sediment, surface water, and fish tissue in the Study Area are similar to typical "background" values presented in published literature or databases. Sources of background concentrations included the Agency for Toxic Substances and Disease Registry Toxicological Profiles (www.atsdr.cdc.gov), U.S. Geological Survey values for western U.S. soils (Shacklette and Boerngen 1984, *Element Concentrations in Soils and Other Surficial Materials of the Conterminous United States*), Ecology (1994), and Ecology's Environmental Information Management database (fish tissue concentrations. Available online at http://www.ecy.wa.gov/eim/). Overall, this reference was used to evaluate a very small number of compounds, since adequate site-specific Reference/OCI information was available.

- A COPC is not associated with Hanford Site releases, based on historical information on releases, Hanford Site soil and groundwater data, and/or fate and transport characteristics of the COPC.

If a constituent was determined to be consistent with Reference, then that constituent was identified as a "Reference COPC." Otherwise, the COPC was identified as a Study Area COPC. Note that assignment of a constituent to this category indicates only that the constituent is present at levels higher than those of Reference/OCI areas and does not indicate that the constituent's presence is necessarily attributable to Hanford Site releases. Results of these comparisons are discussed by sub-area in Section 3.8.2.

3.8.1.6 Fish Tissue Evaluation. The Reference comparison for the fish tissue components (fillet, liver/kidney and carcass) poses a challenge because multiple species were analyzed separately, and sample sizes for each sub-area were small relative to those of abiotic media. Additionally, because fish consumption risks were evaluated for individual fish species as well as all species combined (see Section 4.0 for discussion), two separate Reference comparisons were required for fish tissue:

1. All species combined by tissue type (i.e., fillet, carcass, liver/kidney), by sub-area within the Hanford Site Study Area

2. Each individual species by tissue type (fillet and carcass) within the Hanford Site Study Area.

Furthermore, fish are mobile and may swim among the various sub-areas as well as in tributaries to the Columbia River and may therefore accumulate contaminants from locations outside of the Study Area. Likewise, fish within the Hanford Reach may swim to upriver or downriver locations transporting contamination accumulated in the Hanford Reach. Therefore, the Reference evaluation took into consideration factors other than tissue concentration alone.

The criteria described above in previous sections were used to identify Study Area and Reference COPCs in fish tissue. Where an adequate number of detected results for both Study Area and Reference populations existed, comparative statistical tests were conducted to determine whether the two populations were statistically different. Constituents determined to have a statistically higher mean concentration in Study Area data sets were categorized as Study Area COPCs; conversely, constituents with a statistically lower mean, or no significant difference between means, were categorized as Reference COPCs.

For the qualitative evaluation, straight-forward comparisons of maximum and mean fish tissue concentrations were conducted. However, the analytical results for the Study Area and Reference data sets did not always support this type of evaluation due to discrepancies in reporting limits, FOD, and other factors. Therefore, the Reference status of sediment and surface water COPCs was also used to assist in determining whether a constituent was a Study Area or Reference COPC. In some instances, a COPC present in fish was not identified as a COPC in abiotic media (e.g., lead). A statistical comparison for abiotic media was then conducted to

determine whether the constituent was consistent with Reference conditions in surface water and sediment, as an additional line of evidence. Results of all statistical analyses are presented in Appendix E. Constituents determined to be consistent with Reference in surface water or sediment were identified as Reference COPCs in fish tissue.

Lastly, published information on regional fish tissue concentrations was used as another line of evidence in determining whether a COPC was related to the Study Area or consistent with Reference conditions. For published background fish tissue concentrations, contaminant levels in fish tissue were obtained from Ecology's Environmental Information Management database (http://www.ecy.wa.gov/eim/). Database queries were performed on a contaminant-by-contaminant basis. Results meeting the following criteria were included in the data set used for the analysis:

- Data for only freshwater locations were used; saltwater/estuarine locations were excluded
- Only "fillet" or "muscle" tissue type data were included
- Only detected results were used in the evaluation.

The resulting data set includes freshwater bodies from all counties in Washington State but does not include data from the Hanford Site Study Area. Fish species include common variants of bass, burbot, carp, catfish, crappie, perch, pikeminnow, rockfish, salmon, sculpin, sturgeon, sucker, sunfish, tonguefish, trout, walleye, whitefish, and other species. Summary statistics (minimum, maximum, mean, median, 75th percentile, and number of results) were then obtained on these data using Microsoft Excel (2003); these statistics are summarized for select metals in the following table.

Constituent	Number of Detected Results	Minimum Concentration	Maximum Concentration	Median Concentration	75th Percentile	Mean Concentration
Arsenic (total inorganic)	9	0.002	0.01	0.004	0.01	0.004
Cobalt	151	0.001	0.71	0.02	0.04	0.04
Lead	309	0.001	4.60	0.03	0.08	0.12
Mercury, total	1,944	0.0001	1.92	0.11	0.22	0.17
Vanadium	111	0.01	4.16	0.04	0.15	0.12
Zinc	400	2.1	71	6.81	12	9.27

NOTE: Concentrations are reported in milligrams per kilogram wet weight.

Note that these alternative background statistics were not used solely as a determinant of whether a contaminant was identified as either a Study Area or Reference COPC. These values were used only as an additional line of evidence to support the Reference analysis decision, where site-specific background data were deemed inadequate to make such a conclusion.

Finally, the fish tissue evaluation sometimes yielded conflicting results for a given COPC among the multiple tissue types (e.g., the COPC may be classified as a "Study Area COPC" in fillet, but a "Reference COPC" in carcass and liver/kidney). Because of this, the determination was made to base the final fish tissue classification decision on the results for fillet, because this tissue type

comprises the vast majority of the fish ingested (>90%). The only instances where this approach was not used was for constituents that were not identified or detected as COPCs in fillet but were found only in other tissue types. Summary tables are provided that indicate the outcome of the reference comparison for each tissue type, and the final decision made for fish, all tissue types, considered as one complete medium. Uncertainties associated with this approach, and implications for the conclusions of the risk assessment, are further discussed in Section 7.0.

Results of the Study Area to Reference comparisons are presented in the following section.

3.8.2 Results for Study Area and Reference Contaminant of Potential Concern Comparison

Study Area to Reference comparisons were made for all COPCs in all media according to sub-area. In addition, a separate evaluation of individual fish species across the Hanford Site Study Area was made, as described in the previous section. Results for each sub-area and medium are presented in tables containing the number of detected results, total N (number of samples analyzed), FOD, and maximum detected concentration for Study Area and Reference/OCI locations. Where comparative statistical tests were conducted, the p-value from the statistical test is presented to indicate if there is a statistically significant difference between the means of the Study Area and Reference data sets. If this p-value is significant (less than 0.05), then the location with the higher concentration is listed. The second to last column of the tables indicates whether the Study Area COPC is deemed as consistent or inconsistent with Reference data, and the final column contains the rationale for that answer. Overall, many of the COPCs in all media appear to be consistent with Reference conditions.

Subsections 3.8.2.1 through 3.8.2.3 discuss results of the Reference Comparison for the 100 Area, 300 Area, and Lake Wallula Sub-Areas, respectively. Subsection 3.8.3.4 presents results of the Reference comparison for individual fish species for the Hanford Site Study Area.

3.8.2.1 100 Area Sub-Area. The results of the Study Area to Reference comparisons for the 100 Area Sub-Area are presented in Tables 3-61 through 3-63, for surface water, sediment, soil and fish tissue (fillet; all species combined), respectively. Results for abiotic media are as follows:

- In surface water (Table 3-61), all COPCs with the exception of fluoride were identified as Reference COPCs. Fluoride was identified as a Study Area COPC, because the mean Study Area concentration was statistically higher than that of the Reference/OCI Area; however, note that the maximum detected concentration of fluoride in the Study Area was actually lower than that detected in Upriver areas.

- Sediment (Table 3-62) has five COPCs not consistent with Reference data and therefore were identified as Study Area COPCs: hexavalent chromium, uranium (elemental), cobalt-60, europium-152, and technetium-99. The judgment for all these COPCs is based on a qualitative evaluation, and each of these constituents was detected at relatively low frequency (1% to 27%). Technetium-99, in particular, is an inclusion list constituent and was detected

in only 1 of 95 sediment samples in the Study Area. Therefore, the inclusion of these constituents as Study Area COPCs is based primarily on results for only a few sample locations; levels at most of the Study Area sample locations are similar to those of Reference areas.

- Island soil (Table 3-63) has three COPCs identified as Study Area COPCs: arsenic, chromium, and carbon-14. Arsenic and chromium are present at concentrations statistically greater than those of Reference areas. The classification of carbon-14 (an inclusion list constituent) as a Study Area COPC is based on a qualitative evaluation because all Reference data are nondetects. However, this radionuclide was detected in only 1 of 29 island soil samples.

- Fish tissue results for all species are presented in Tables 3-64 through 3-66 for fillet, carcass, and liver/kidney, respectively. Table 3-67 provides a summary of the reference comparison outcome for each tissue type. Fish tissue has a number of chlorinated pesticides and PCBs that were retained as Reference COPCs in fillet, although in some instances had higher Study Area concentrations in carcass and/or liver and kidney samples. Note that none of these constituents was retained as a COPC in abiotic media, with the exception of PCBs in surface water. These types of constituents are ubiquitous in animal tissue world-wide due to their strong tendency for environmental persistence and biomagnification through the food web. PCBs in abiotic media were determined to be consistent with Reference conditions. The fact that these contaminants were either not detected in surface water or sediment (such as the case for many of the pesticides) or were detected at concentrations consistent with those of Reference areas (PCBs) throughout the Hanford Site Study Area suggests that these COPCs in fish tissue are consistent with Reference conditions. Reference area sediment and surface water concentrations were higher than those in the 100 Area Sub-Area. Therefore, these two classes of COPCs (pesticides and PCBs) were designated as Reference COPCs in fish tissue for the 100 Area Sub-Area.

  With the exception of mercury and uranium, nonradionuclide detected metal COPCs in the 100 Area were present at concentrations lower than those of Reference areas, consistent with fillet concentrations from other waterways located within Washington, or were present in surface water/sediment at concentrations consistent with Reference conditions. Mercury was found consistent with Reference conditions in abiotic media. Uranium (elemental) was identified as a Study Area COPC in sediment.

  The only radionuclide retained as a COPC in fish tissue, carbon-14, was identified as a Study Area COPC. As previously discussed, carbon-14 was only sporadically detected in fish tissue. Note that carbon-14 was identified as consistent with Reference in sediment.

Table 3-68 summarizes the Study Area to Reference comparisons across the four media for the 100 Area Sub-Area. There is a general consistency across media for a COPC to be classified as either a Reference or Study Area COPC. Within sediment, surface water, and fish tissue, most metals were categorized as Reference COPCs. As discussed, PCBs and pesticides in fish tissue were identified as Reference COPCs. Pesticides were not a COPC in any of the abiotic media;

nondioxin PCBs were identified as a Reference COPC in surface water and were not a COPC in either sediment or soil.

3.8.2.2 300 Area Sub-Area. The 300 Area Sub-Area results for the statistical and qualitative comparison between Study Area and Reference locations are presented in Tables 3-69 through 3-74 for surface water, sediment, soil, and fish tissues. Results of these comparisons indicate the following:

- In surface water (Table 3-69), there are five identified Study Area COPCs: three are VOCs (1,1,2-trichloroethane, 1,2-dichloroethane, and chloroform) and two are metals (thallium and fluoride). Each of the VOCs was detected at a low frequency (6% to 8% of samples). However, they may potentially be associated with VOCs that discharge to the river (as evidenced by upwelling data) at RM 343-344; furthermore, none was detected in any Reference area samples. Both thallium and fluoride had statistically greater means in the 300 Area Sub-Area when compared to those of the Upriver Sub-Area. The remaining Reference COPCs include bis-2-ethylhexylphthalate, nondioxin PCBs, arsenic, chromium, and lithium. Study Area concentrations of these contaminants were lower than those of Reference/OCI areas.

- Sediment (Table 3-70) has five Study Area COPCs, all of which are inclusion list constituents that are present at levels higher than those of Reference/OCI areas. These include hexavalent chromium and four radionuclides: cobalt-60, europium-152, technetium-99, and tritium. All these classifications are based on qualitative analyses, as either there was an insufficient FOD for statistical tests to be valid or Reference/OCI data were not available (such as for tritium). Other constituents were present at levels lower than those of Reference/OCI areas.

- For island soils (Table 3-71), five constituents were identified as Study Area COPCs. These include arsenic, chromium, cobalt-60, europium-152, and strontium-90. Cesium-137, which may be potentially related to fallout from atmospheric testing, and the other nonradioactive metals were determined to be present at levels lower than or consistent with those of Reference areas. All of the radionuclides identified as Study Area COPCs were detected at relatively low frequencies.

- In fish tissue (all species combined; Tables 3-72 through 3-74 for individual tissue types, and Table 3-75 for a summary of all tissue types), heptachlor epoxide, lithium, mercury, uranium, and carbon-14 are identified as Study Area COPCs. In general, the concentrations observed in 300 Area fish tissue were similar to those of Reference areas. Fillet concentrations of pesticides and PCBs were lower in 300 Area fish samples relative to Upriver, but some of these constituents were higher in 300 Area liver/kidney samples. Arsenic, cadmium, and cobalt concentrations were similar to or lower than those of Upriver areas across all tissue types.

Study Area and Reference COPCs in all media in the 300 Area are summarized in Table 3-76.

3.8.2.3 Lake Wallula Sub-Area. The Study Area to Reference comparisons for the Lake Wallula Sub-Area was conducted for surface water, sediment, and fish tissue. There were no islands sampled from within Lake Wallula, and therefore no comparison was conducted for island soils. The surface water and sediment results are presented in Tables 3-77 and 3-78, respectively. Fish tissue results (all species) are presented in Tables 3-79 through 3-81 for fillet, carcass, and liver/kidney, respectively; Table 3-82 presents a summary of the Reference comparison results for fish across all tissue types. Results of these comparisons indicate the following:

- In surface water (Table 3-77), only two COPCs were identified as Study Area COPCs (i.e., not consistent with Reference): TPH (diesel-range) and plutonium-239/plutonium-240. TPH was identified as a Study Area COPC based on a qualitative evaluation; this constituent was not detected in any Reference/OCI samples.

  Plutonium-239/plutonium-240 is an inclusion list constituent and was classified as a Study Area COPC based on qualitative analyses, due to very low FOD; the single detection of plutonium-239/plutonium-240 in the Study Area (out of 19 samples) was 5 times higher than the single detect observed in the Reference area. It is intriguing that plutonium was also detected in the Reference Area, suggesting that atmospheric deposition from nuclear fallout may be a potential source. Additionally, because the single surface water sample in Lake Wallula, in which plutonium isotopes were detected (LW-2SW), was collected from the bottom of the water column near the river bed, it is likely that this detection in an unfiltered sample reflects suspended sediment rather than dissolved-phase plutonium.

  Bis-2-ethylhexyl phthalate, PCBs, and all metals were found to be present at lower concentrations in the Study Area relative to the Reference area and therefore are considered as Reference COPCs.

- In sediment (Table 3-78), there are only four identified Study Area COPCs considered as not consistent with Reference: hexavalent chromium, cobalt-60, europium-152, and europium-154. Note that a qualitative comparison was conducted for these four COPCs due to the limited number of detections. Europium-154, for example, is an inclusion list constituent and was detected in only 1 of 123 samples and at an activity equivalent to the maximum MDA of Reference area samples. The remaining COPCs are all consistent with Reference.

- The fish tissue comparison results are presented in Tables 3-79 through 3-81 and summarized in Table 3-82. As with the 300 Area Sub-Areas, nearly all chlorinated pesticides, except for heptachlor epoxide, and all PCBs were designated as Reference COPCs. Also similar to the 100 Area, heptachlor epoxide was detected only in liver/kidney, and at concentrations only marginally higher in Study Area samples. For metals, Study Area concentrations were lower than or consistent with those observed in Reference areas, with the exception of uranium. Uranium was identified as a Study Area COPC in fish carcass, because it was detected more frequently and at higher concentrations in Lake Wallula samples, as compared

to Upriver samples. Carbon-14 was the only radionuclide identified as a COPC in fish tissue; this constituent was categorized as a Study Area COPC.

Table 3-83 presents a summary of the Study Area to Reference comparison for Lake Wallula for all media. A pattern similar to the other sub-areas is seen in Lake Wallula COPCs—most metals are Reference COPCs across media, whereas most radionuclides are Study Area COPCs.

3.8.2.4 Individual Fish Species Evaluation. The comparison to Reference for six individual fish species was completed for fillet and carcass COPCs. (Because this evaluation was conducted for the "avid angler" scenario, further described in Section 4.0, and this receptor is assumed to ingest only fillet and carcass, no evaluation of liver/kidney data was required.) There are two tables for each species, for fillet and carcass: Tables 3-84 and 3-85 for bass; Tables 3-86 and 3-87 for carp; Tables 3-88 and 3-89 for sturgeon; Tables 3-90 and 3-91 for sucker; Tables 3-92 and 3-93 for walleye; and Tables 3-94 and 3-95 for whitefish. Because a relatively low number of tissue samples were available for the Reference area (generally about five samples per species), only a limited number of COPCs are classified based on statistical tests. The remaining COPCS are classified based on a qualitative evaluation. The method for classifying a COPC as Reference for a species with data for two different tissues was as follows. First, the quantitative or qualitative comparisons were conducted. If the COPC was present in fillet, then the Reference or Study Area designation for the fillet was applied to the carcass. If a carcass COPC was not present in the fillet, then the designation was based on the quantitative or qualitative evaluation for carcass. Table 3-96 presents a summary of the Reference comparison across all species and tissue types.

Table 3-97 summarizes the Reference and Study Area COPCs across all fish species. In sturgeon and walleye, all pesticides and PCBs are designated as Reference COPCs. Sucker is the only species for which PCBs are designated as Study Area COPCs. Beta-HCH is a Study Area COPC in bass and sucker; delta-HCH is a Study Area COPC in carp; and endrin is a Study Area COPC in whitefish. For the metals, all metals are Reference COPCs in bass.

Tin in carp is the only Study Area COPC for this species. The only Study Area metal COPC in sturgeon is mercury. Lithium is the only designated Study Area metal COPC in sucker. Whitefish has the most metal COPCs designated as Study Area: antimony, mercury, selenium, and tin. Carbon-14 is the only radionuclide COPC in fish tissue; it is designated as a Study Area COPC in carp, sucker, and whitefish.

3.8.3 Study Area to Reference Comparison Conclusions

The Study Area to Reference comparison was performed by sub-area for surface water, sediment, island soil, and fish tissue (all species, combined). Additionally, a Study Area to Reference comparison was conducted for individual fish species within the Hanford Site Study Area, as described above.

Table 3-98 presents a summary of the Study Area-Reference determinations across all media (soil, sediment, surface water and fish tissue, all species combined). This table shows how the

classification of a COPC as either a Reference or Study Area COPC varies across the various environmental media. The following observations can be made about the various contaminant classes:

- Chlorinated VOCs are Study Area COPCs in surface water in the 300 Area Sub-Area; these constituents are not COPCs in any other sub-area or medium. Note that chlorinated VOCs such as trichloroethene and tetrachloroethene have not been detected in the most recent rounds of surface water samples (i.e., 2008-2009).

- Chlorinated pesticides were identified as COPCs only in fish tissue; in the abiotic media, these constituents were infrequently detected and at low concentrations. With the exception of heptachlor epoxide, all other chlorinated pesticides in fish are identified as Reference COPCs.

- PCBs were identified as COPCs only in surface water and fish tissue (all sub-areas). In surface water, PCBs are classified as Reference COPCs. Although not COPCs in sediment, PCBs in this medium are present at levels consistent with reference conditions throughout the Study Area.

- Fluoride is a COPC in surface water and is present at levels consistent with Reference in Lake Wallula but not in either the 100 or 300 Area Sub-Areas.

- Numerous metals are COPCs in all media. With the exception of arsenic, chromium, hexavalent chromium, lithium, mercury, thallium, and uranium (elemental), all other metals are identified as Reference COPCs. Arsenic is a Study Area COPC only in island soil; it is a Reference COPC in other media. As discussed previously, because the arsenic concentrations observed in island soil samples are within the range of concentrations observed for sediment (which in turn are consistent with Reference sediment concentrations) and because the levels of arsenic in soil observed in island soils are consistent with background arsenic levels published by Ecology (1994), it is likely that arsenic in this island soil may potentially be naturally occurring or at least present at levels consistent with local conditions. However, the existing data set does not allow that conclusion to be made; therefore, arsenic was conservatively retained as a Study Area COPC in island soil.

- The following radionuclides have been identified as COPCs in various media and sub-areas: carbon-14, cesium-137, cobalt-60, europium-152, europium-154, plutonium-239/plutonium-240, strontium-90, technetium-99, and tritium. These radionuclides, with the exception of cesium-137, have been identified as Study Area COPCs. Cesium-137 was identified as a Reference COPC in all sub-areas and media. The presence of this radionuclide is assumed to be attributed primarily to atmospheric fallout from previous nuclear testing.

In summary, many of the COPCs in Columbia River media are identified as Reference COPCs, with only select metals, VOCs, and radionuclides identified as Study Area COPCs. These conclusions are consistent with the CSM, which indicates a number of natural and anthropogenic non-Hanford Site sources of contaminants to the river. Volatile organic compounds, chromium, and radionuclides, however, are documented contaminants at the Hanford Site.

In the subsequent sections of this HHRA, a COPC that has been classified as not consistent with Reference is referred to as a "Study Area COPC," and a COPC that has been classified as consistent with Reference is referred to as a "Reference COPC." Cumulative noncancer hazard and cancer risk associated with Study Area and Reference COPCs are further discussed in Section 6.0.

3.9 REFERENCES

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4.0 EXPOSURE ASSESSMENT

The objective of the exposure assessment is to estimate the types and magnitude of potential exposures to COPCs present at or migrating from the Hanford Site. Exposure is quantified for a subset of the human populations potentially exposed to contaminated media via specific exposure pathways based on current and likely future potential land use. The exposure estimates are first calculated using COPC-specific EPCs and receptor-specific exposure parameters and then combined with toxicity information to characterize the potential risk to human receptors. The approach for selection of these exposure scenarios and exposure parameters considered previously issued scoping documents (e.g., DOE/RL-2004-49); the RCBRA (DOE/RL-2007-21); and numerous meetings, workshops, and discussions with the Tri-Parties and various stakeholders during the development of the RI Work Plan (DOE/RL-2008-11).

The exposure assessment was conducted in a manner consistent with EPA risk assessment guidance (e.g., EPA/540/1-89/002, *Risk Assessment Guidance for Superfund, Volume I: Human Health Evaluation Manual [Part A], Interim Final*; EPA/600/P-95/002Fa, *Exposure Factors Handbook, Volume I*: *General Factors*; EPA/600/P-95/002Fb, *Exposure Factors Handbook, Volume II: Food Ingestion Factors*; EPA/600/P-95/002Fc, *Exposure Factors Handbook, Volume III: Activity Factors*; EPA/540/R-99/005, *Risk Assessment Guidance for Superfund, Volume I: Human Health Evaluation Manual [Part E, Supplemental Guidance for Dermal Risk Assessment], Final*). For each identified human receptor at each exposure point, complete or potentially complete exposure pathways were identified based on Hanford Site activities and uses, and the presence of COPCs in environmental media. Age groups that represented the longest or most intense exposure periods were selected to be adequately protective of all stages of a human receptor's life.

All but one scenario included evaluation of both central tendency exposures (CTEs) and reasonable maximum exposures (RMEs) to provide both central and upper-bound estimates of potential health risks. The EPA guidance recommends evaluation of the CTE and RME scenarios to provide information on the range of potential risks to each human receptor; however, the need for remedial action is typically based on the risks estimated under the RME scenario. Only one scenario based on upper-bound (i.e., RME) EPCs was evaluated for the Yakama Nation scenario. This approach, including most exposure assumptions, is in general agreement with the document *Yakama Nation Exposure Scenario for Hanford Site Risk Assessment, Richland, Washington* (Ridolfi 2007) and was conducted in accordance with the RI Work Plan (DOE/RL-2008-11).

The exposure assessment discusses the relevant exposure pathways and human receptors through a CSM. The CSM identifies relevant exposure points, representative data, and EPCs and presents the physiological exposure parameters, activity factors, and equations used to quantify exposures to COPCs in the various environmental media.

4.1 EXPOSURE SCENARIOS/CONCEPTUAL SITE MODEL

Complete and potentially complete exposure pathways are identified and quantitatively evaluated as part of the HHRA. A complete exposure pathway, which links COPCs in an environmental medium to a human receptor, consists of the following elements:

- A source and mechanism of chemical release
- A retention or transport medium
- A point of potential human contact (exposure point)
- An exposure route (e.g., dermal contact, ingestion, or inhalation).

Human exposure may be direct (i.e., the receptor contacts the COPC in the affected medium such as air, water, or soil) or it may be indirect, involving exposure to chemicals via the food chain (e.g., one may ingest COPCs via consumption of fish that have accumulated contaminants from surface water or sediments) or through external irradiation.

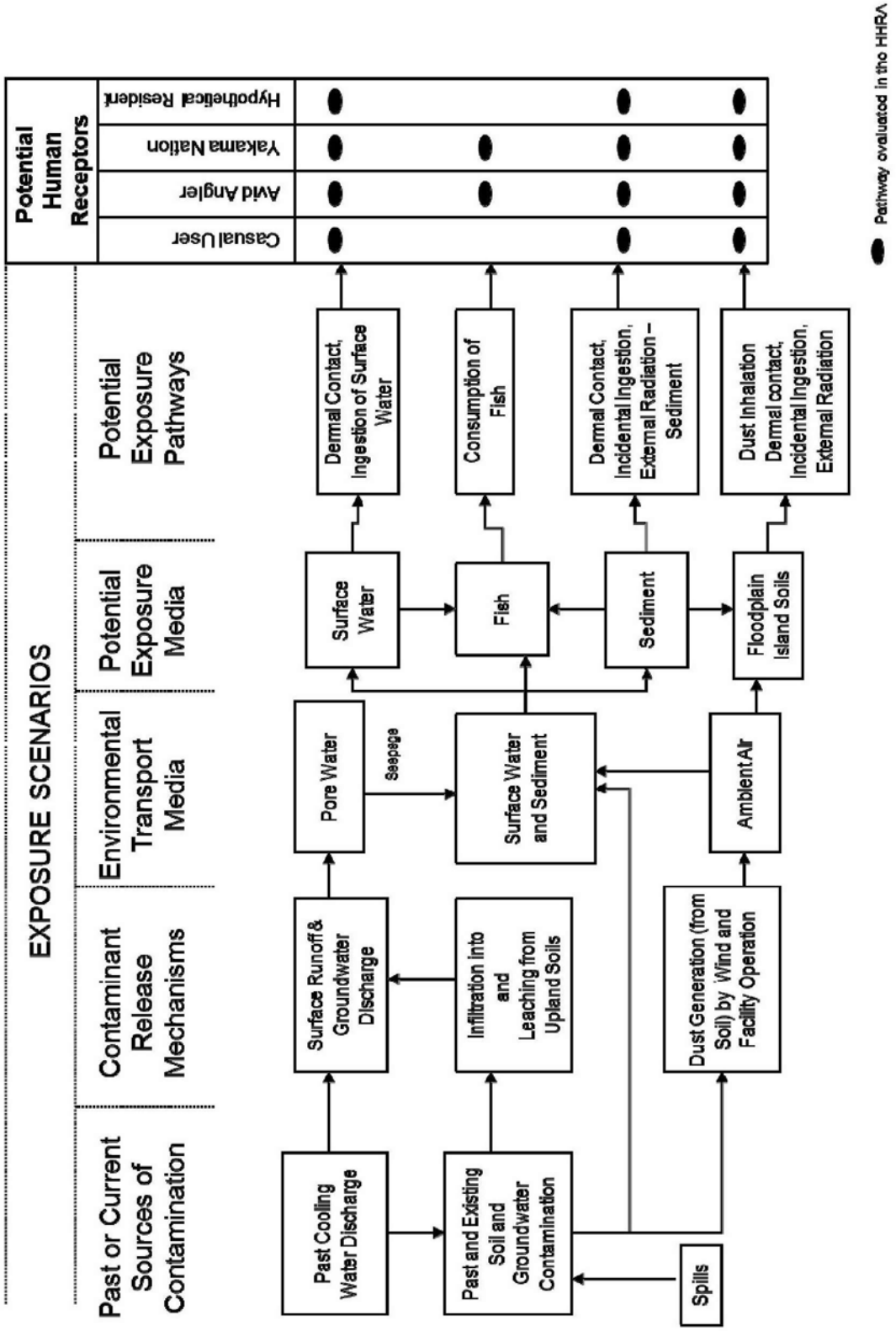
4.1.1 Conceptual Site Model

The CSM identifies the sources of contamination and the environmental transport and exposure pathways between contaminant sources and applicable receptors by using historical information and existing data. Figure 4-1 provides a CSM summary of contaminant sources, transport/migration pathways, potential human receptors, and potentially complete exposure pathways. The current CSM reflects a subset of historical information and available analytical data (discussed in Section 3.0), as well as discussions with federal and state regulators and other interested parties.

The primary media of concern in the Hanford Site Study Area include surface water, porewater, sediment, and island soil that have been impacted by both on- and off-site sources of contaminants, as well as naturally occurring elements. Some of the contaminants in these media have also accumulated in fish tissue. Hanford Site sources are primarily related to historical cooling water discharges and ongoing groundwater plume migration and discharge to the Columbia River. As discussed in Section 2.0, off-site sources of contamination that are unrelated to the Hanford Site are located upriver, within the Hanford Reach, and downriver of the Hanford Site and include both natural sources (such as the result of local geochemical conditions); discharges associated with various industrial discharges; agricultural run-off; and a variety of other nonpoint source discharges, such as roadway runoff, fugitive dust, atmospheric deposition, and discharges from commercial and recreational watercraft. Potential contaminants of concern include organic compounds, inorganic elements, and radionuclides.

The Columbia River is widely used for recreational purposes such as boating, wading, swimming, fishing, and water-skiing. Numerous beaches, boat ramps, and wildlife viewing areas are located throughout the Hanford Site Study Area. The Hanford Reach National Monument consists of a 77-km (48-mi) stretch of the Columbia River and federally owned riparian lands. Below the Monument's southern boundary, recreational use is widespread throughout the next 80 km (50 mi) of the McNary Dam impoundment (e.g., Lake Wallula).

Figure 4-1. Conceptual Exposure Model.



Numerous islands are located within the Hanford Site Study Area. Most of these islands are owned by federal or state agencies and are designated as conservation/recreation areas. Many of the islands (or portions of the islands) are entirely submerged during periods of high water and consequently subject to depositional/erosional forces.

In addition to recreational use, surface water of the Columbia River is used for river navigation/transportation; hydropower; and as a domestic, agricultural, and industrial water supply. The city of Richland also relies on filtered/treated river water as its source of public drinking water; the Richland Pumphouse, a primary treatment system, is located near RM 340. The river also provides essential habitat for a variety of resident and migratory fish and wildlife.

Based on regional land use and beneficial water use, the following exposure scenarios have been developed:

- **Avid Angler** scenario includes both adults and older children (i.e., older than 6 years of age) who engage in fishing activities.

- **Casual User** is an adult or child who uses the river for seasonal recreational purposes.

- **Yakama Nation** scenario includes local and regional tribes who have ties to the Hanford Reach and surrounding lands and use the river on a regular basis.[1]

- **Hypothetical Future Upland Resident** scenario in which a child and adult may be routinely exposed to sediments from only those portions of the Columbia River that currently have dredged channels and that, at some point in the future, may be placed in upland residential areas. This scenario also assumes that the hypothetical resident may use surface water as a potable water supply.

These scenarios reflect the receptors most likely to have the longest and/or most comprehensive exposures to any of the four river media relevant to human exposures: sediment, surface water, island soil, and fish tissue[2]. Accordingly, evaluation of these different receptor groups is assumed to be protective of other lesser exposed receptors, such as occasional visitors. The approach for selection of these receptors also considered previously issued scoping documents (e.g., DOE/RL-2004-49) and the RCBRA (DOE/RL-2007-21) as well as numerous meetings, workshops, and discussions with the Tri-Parties and various stakeholders.

As discussed, the HHRA for the Columbia River evaluates only riverine exposures and does not address potential exposure scenarios associated with upland areas of the Hanford Site. The HHRA does, however, include a screening-level evaluation of hypothetical residential exposure to sediments assumed to be dredged from the Columbia River channel and placed on upland

[1] The Confederated Tribes of the Umatilla Indian Reservation (CTUIR) scenario, although relevant to the Columbia River Corridor, is evaluated separately in the RCBRA.
[2] Although contaminants have been identified in porewater, this medium is not considered to be relevant to human exposure scenarios.

areas with no restrictions on use assumed[3], as requested by Ecology. In addition, the HHRA also includes a screening-level assessment of potable water use of (unfiltered/untreated) Columbia River surface water. This evaluation of hypothetical future residential use is provided in Appendix A. Each of these scenarios is described in more detail below.

4.1.1.1 Avid Angler Scenario. The Avid Angler scenario includes both adults and older children (older than age 6) who frequently engage in fishing activities in and along the Columbia River. The Avid Angler could potentially be exposed to contaminants through consumption of fish from the river. Other potential routes of exposure to contaminated sediment, island soil, and/or surface water include incidental ingestion, dermal contact, external irradiation, and dust inhalation while fishing, wading, and/or boating in the river. Because finfish will likely be brought home, fish ingestion for a young child (aged 1 to 7 years) was also evaluated in this HHRA; however, it is assumed that a young child would not likely be actively fishing.

4.1.1.2 Casual User Scenario. The Casual User is an adult or child who uses the Columbia River for seasonal recreational purposes. This scenario includes adults and children who may swim, waterski, boat, wade, camp, or participate in other similar types of recreational activities along the Columbia River. Potential routes of exposure to contaminated sediment, island soil, and/or surface water for this receptor include incidental ingestion, dermal contact, external irradiation, and dust inhalation during these recreational activities.

4.1.1.3 Yakama Nation Scenario. The Yakama Nation scenario includes local and regional tribes who have ties to the Hanford Reach of the Columbia River and surrounding lands. This scenario evaluates subsistence fishing-related exposures for the Yakama Nation. Potential routes of exposure to COPCs in contaminated sediment, island soil, and/or surface water include incidental ingestion, dermal contact, external irradiation, and dust inhalation during a variety of activities including boating, fishing, wading, or other cultural activities. Other potentially relevant routes of exposure, such as hunting and tribal use of sweat lodges, are being assessed as part of the RCBRA and are not evaluated in this report.

The Yakama Nation receptors could also potentially be exposed to COPCs through consumption of fish from the river. For this assessment, it is assumed that the majority of their daily diet consists of finfish (e.g., bass, walleye) caught from the Hanford Site Study Area in the Columbia River. This scenario is consistent with the *Yakama Nation Exposure Scenario for Hanford Site Risk Assessment, Richland, Washington* (Ridolfi 2007).

4.1.1.4 Hypothetical Future Resident (Upland Exposures) Scenario. At the request of Ecology, the HHRA evaluated a scenario in which a child and adult may be routinely exposed to dredged sediments removed from within existing navigational channels where the ACOE has authority to dredge (e.g., 14 ft [+2] mean low water) and placed in upland residentially zoned areas. These are the only receptors identified for which (hypothetical) exposure to dredge spoils (e.g., sediments) may occur. Potential routes of exposure for this scenario include dermal

[3] This assessment was performed via comparison of dredgeable sediment concentrations to WAC 173-340, "Model Toxics Control Act – Cleanup," and other potentially relevant benchmarks (see Appendix A).

contact with and incidental ingestion of sediment (as soil), as well as the inhalation of dust. Additionally, because constituents in the dredged sediment may leach into groundwater, there is the potential for hypothetical future residents to ingest or dermally contact (leachable) constituents that have migrated to groundwater.

It should be noted that past dredging projects in the Columbia River conducted by the ACOE required extensive permitting and evaluations of "beneficial use" of dredged sediments to ensure that the ultimate disposition of dredge spoils would not pose risks to future potential receptors/users of such materials, for both human and ecological receptors.

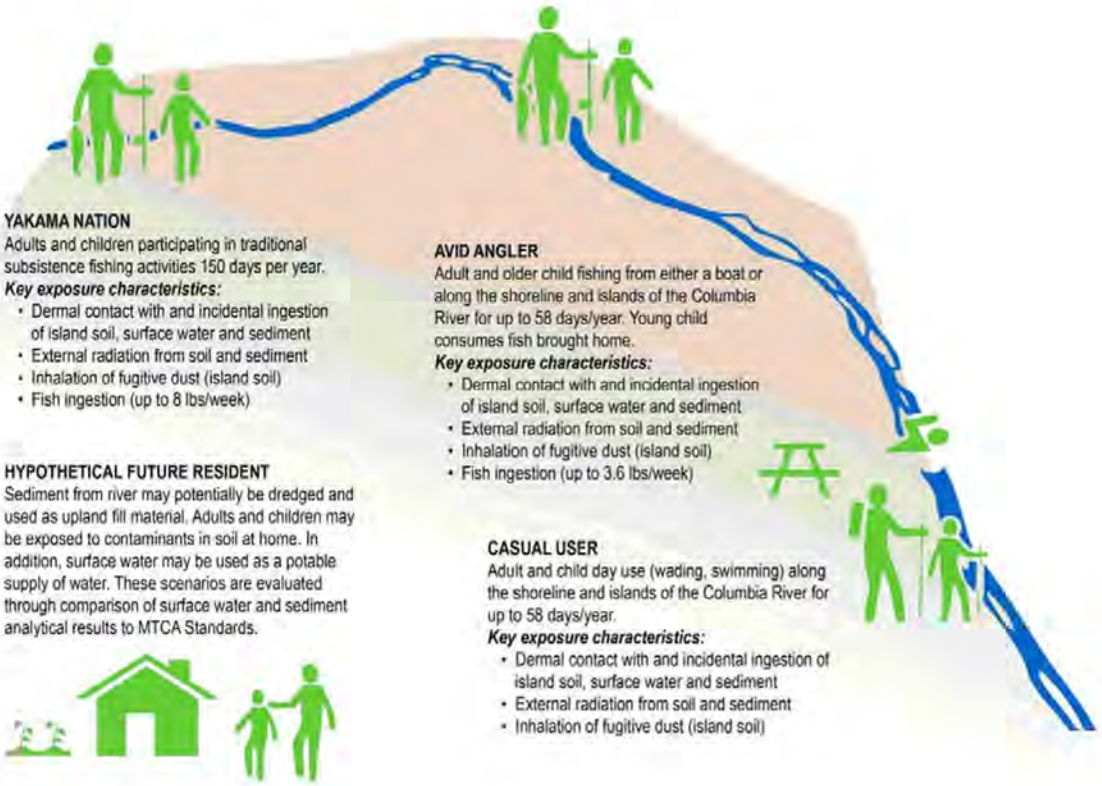
In addition to sediment exposures, this scenario included an evaluation of (hypothetical) surface water use for potable purposes. Although the Columbia River is currently used as a source of potable water for the City of Richland, filtered and treated water from the river is routinely monitored prior to its distribution and meets federal drinking water standards (MCLs), as required by the Safe Drinking Water Act. Therefore, although the river water within the Hanford Reach is not currently used as a potable supply, and the downstream user (City of Richland) filters, treats, and tests the water prior to its distribution and use, this hypothetical and very conservative scenario in which untreated river water is used as a residential drinking water supply was included.

Both the drinking water and dredged sediment exposure pathways (i.e., "Hypothetical Future Resident" receptor scenario) were evaluated through a simplified screening-level evaluation in the HHRA. Rather than generate cumulative risk estimates for this hypothetical future scenario (as was conducted for the recreational and Tribal scenarios), sediment and surface water data were directly compared to medium-specific benchmarks protective of residential exposure pathways. This evaluation is presented in Appendix A.

Table 4-1 summarizes the receptors and exposure pathways to be evaluated in the HHRA and indicates the type of analysis conducted (either comparative or quantitative). In addition, these receptors and relevant pathways are illustrated in Figure 4-2.

Although there is the potential for recreational users to engage in hunting activities along the Columbia River, an "Avid Hunter" scenario (i.e., waterfowl hunter and consumer) was not evaluated within this river-focused HHRA. The rationale for exclusion of this scenario is that this HHRA focuses exclusively on river-related exposure scenarios. Because of the anticipated relatively small risk presented by waterfowl hunting (and consumption) as compared to other pathways of exposure (such as fish ingestion), this waterfowl hunter exposure scenario was not evaluated in this HHRA. Furthermore, the Yakama Nation scenario evaluated herein assumed a protein diet subsisting almost completely of fish caught from the Hanford Site Study Area. A comprehensive "Avid Hunter" scenario is included in the RCBRA (DOE/RL-2007-21), which evaluates risk for broad-area, upland exposure scenarios. As described in the RCBRA, this receptor is assumed to hunt for and consume upland game (deer, gamebirds) and may also include waterfowl.

Figure 4-2. Summary of Human Receptors Evaluated in the Baseline Human Health Risk Assessment.



4.2 EXPOSURE POINTS AND EXPOSURE POINT CONCENTRATIONS

An exposure point is the distinct location and medium where a receptor may come into contact with contaminants. Exposure points are related to both the potential for exposure and the concentrations of contaminants in a medium. The Columbia River is used for a variety of purposes, especially recreational ones. There are a number of beaches, boat ramps, and parks located along the banks of the river in Lake Wallula and lower reach of the 300 Area and, consequently, some portions of the river are easily accessible by foot. The entire Hanford Site Study Area is accessible by boat. Therefore, there is a potential for exposure along much of the Hanford Site Study Area under the recreational exposure scenario.

An EPC is the concentration of a COPC representative of an exposure point and is used in conjunction with receptor-specific and chemical-specific parameters to quantify noncancer hazard and cancer risk. Exposure points and EPCs are discussed in the following subsections.

4.2.1 Evaluation of Distinct Exposure Points

Exposure point concentrations are calculated for each exposure point. It is important to understand the distribution of contaminant concentrations with respect to exposure potential

prior to calculating EPCs, in order to avoid diluting out "hot spots," which could potentially underestimate exposure and associated risk (see Section 7.0 for more discussion on under- and overestimating potential risk). However, it should be recognized that the EPCs used to estimate risk are typically designed to address the variability of the underlying analytical concentration; for example, the EPC for reasonable maximum exposures is the 95% UCLs of the mean (meaning that there is only a 5% probability that the true mean would be higher) or the maximum detected concentration. See Section 4.2.2 for further discussion of the calculation of EPCs.

As described previously, the Hanford Site Study Area was divided into three sub-areas as per the RI Work Plan, based on sources of contamination and overall contaminant distribution. These sub-areas include 100 Area (RM 387 to RM 366), 300 Area (RM 365 to RM 340), and Lake Wallula (RM 339 to RM 292)[4]. A number of environmental samples have been collected from these three sub-areas. Each sub-area encompasses approximately 24 km (15 mi) to more than 32 km (20 mi) of the Columbia River.

For this evaluation, the three sub-areas that make up the Hanford Site Study Area were further assessed to identify any unique exposure points, i.e., areas with relatively elevated contaminant concentrations and/or increased potential for exposure. The ultimate purpose of this evaluation is to determine whether it is appropriate to evaluate each sub-area as a single exposure point or whether areas of elevated contaminant concentrations exist within each sub-area, which would warrant that such points be evaluated as distinct exposure points.

The surface water, sediment, and island soil data described in Section 3.0 were reviewed by sub-area to determine if spatial patterns exist indicating a distinct exposure point (i.e., an area of elevated concentrations). Initially, analytical results were plotted by constituent from each medium and then patterns examined within and across media. The preliminary review consisted of addressing the following issues/questions:

- Are elevated concentrations present in a single sample or are there multiple samples with similar concentrations from the same or adjacent locations?
- Are elevated concentrations more than 10 times the mean concentration for that sub-area?
- Do locations of elevated concentrations occur for similar constituents in sediment and surface water?

[4] Although the Reference locations (i.e., Upriver Sub-Area and OCIs) are part of the RI study area and are used to evaluate Reference/OCI conditions in the river (as discussed in Section 3.8), the focus of the HHRA is on potential health risks associated with exposures within the Hanford Site Study Area (RM 292 to RM 387). Therefore, risks estimates were not evaluated for Reference areas. However, and as described in more detail in Section 3.8, risks for both "Reference" and "Study Area" COPCs were quantified for each of the three sub-areas within the Hanford Site Study Area.

- Do any sample locations represent an area with unique exposure potential? For example, is the area locally accessible (such as a beach) or does it currently have limited access (e.g., continuously inundated)?

These criteria were developed with two purposes in mind: first, to be relevant to potential human exposure; and second, to capture a representative picture of the spatial distribution of COPCs. This evaluation resulted in the decision to divide each sub-area into two separate exposure points, designated as either A or B (e.g., 100-A and 100-B). For all sub-areas, the "B" designation is applied to the portion of the sub-area where elevated concentrations meeting the criteria described above were identified. In other words, the "B" designation represents a potential "hot spot" or area of increased contaminant levels, whereas the "A" designation encompasses all other locations within that sub-area. The results of this evaluation and the rationale for identification of each exposure point are presented in the following subsections.

Many analytes in surface water, sediment, and soil were examined for consistency with the criteria presented above along the entire length of the Study Area. In surface water, the COPC analytes included metals such as arsenic, chromium, hexavalent chromium and thallium, PAHs, and radionuclides. Except for chromium, throughout the entire length of the Study Area the concentrations of all other analytes varied by less than an order of magnitude and frequently by less than 50%.

For sediment, the COPC analytes evaluated included metals such as arsenic, chromium, hexavalent chromium, and uranium; and radionuclides such as cesium-137, cobalt-60, europium-152, and europium-154. Of this array of compounds, only four COPCs exhibited significant spatial variation: chromium, hexavalent chromium, cesium-137, and europium-152. Concentrations of the remaining compounds, similar to surface water, were either fairly consistent, varying by much less than an order of magnitude, or had very few samples detected above the reporting limit. Contaminants of potential concern examined in island soil included metals such as arsenic and cadmium; and radionuclides such as carbon-14, cesium-137, cobalt-60, europium-152, and strontium-90. Only one analyte, europium-152, exhibited substantial spatial variation; concentrations of all others varied by less than an order of magnitude or were infrequently detected above the reporting limit.

Each sub-area and the analytes that exhibited spatial variation meeting the criteria discussed above are presented below.

4.2.1.1 100 Area Sub-Area. The data from sediment, surface water, and island soils in the 100 Area Sub-Area were evaluated for spatial anomalies. For this portion of the Hanford Site Study Area, the distinct exposure point is based on sediment and surface water data, since the island soil data showed no discernible spatial variation.

In the sediment data set, the only constituents showing a spatial variation that meet the criteria identified above (Section 4.2.1) are total chromium, hexavalent chromium, and the radionuclides cesium-137 and europium-152. Total chromium has the highest concentrations in the 100 Area Sub-Area between RM 378 and RM 369, with three samples ranging in concentration from

122 to 275 mg/kg. The three elevated total chromium results are clearly shown in Figure 4-3. The value at RM 369 (275 mg/kg) exceeds by a factor of 10 the mean concentration for the 100 Area Sub-Area (25.8 mg/kg). Hexavalent chromium concentrations are approximately four to eight times higher in two samples at RM 378 and RM 373 relative to other samples. Figure 4-4 illustrates the hexavalent chromium results from the 100 and 300 Area Sub-Areas with the highest reported concentrations.

In surface water (of the 12 metals identified using the criteria presented above), only total chromium and nickel revealed a different pattern of concentration across the Hanford Site Study Area. The highest chromium concentrations are at RM 378 in the 100 Area Sub-Area. These results are shown in Figure 4-5. This location corresponds to the same location of elevated chromium concentrations in the sediment data. The maximum concentration of nickel is at RM 378 in the 100 Area Sub-Area, tracking with the chromium maximum concentration; however, it does not exceed the human health benchmark.

Based on location of the elevated chromium concentrations in sediment and surface water and nickel in surface water, RM 378 to RM 369 is considered a distinct exposure point and is designated exposure point 100-B. The remaining portion of the 100 Area Sub-Area, RM 387 to RM 379 and RM 368 to RM 366, is designated as exposure point 100-A. For each distinct exposure point, the EPCs for sediment, surface water, and island soil are based on data from sampling locations within this range. The location of the samples used in the calculation of EPCs for exposure point 100-A along with the associated river mile are presented in Table 4-2; Table 4-3 shows the same information for exposure point 100-B.

4.2.1.2 300 Area Sub-Area. In the 300 Area Sub-Area, sediment, surface water, and island soil data were reviewed. In this sub-area, the distinct exposure point is based on island soil, specifically, Johnson Island.

In the sediment data set, there is a single elevated value for hexavalent chromium of 17.3 mg/kg at RM 357. This sampling location is in a slough off the main river channel and on the east bank (opposite the Hanford Site) of the river. All other hexavalent chromium samples in the 300 Area Sub-Area are generally less than 2 mg/kg. Concentrations of total chromium in the 300 Area Sub-Area show no areas of elevated concentrations, with levels consistently between 9 and 30 mg/kg. Therefore, this singular sample at RM 357 is certainly elevated, but is not considered a distinct exposure point.

Across the 300 Area Sub-Area, the surface water data did not exhibit spatial variability meeting the criteria described in Section 4.2.1. The metal data were either low concentrations or were of low FOD. For the inorganic constituents nitrate, nitrite, and sulfate, there was not a distinct pattern to the data. Similarly, no hot spots were identified for radionuclides.

In island soils, the distribution of concentrations of metals was identified using the criteria presented above. None of the metal results exhibited a distinct spatial pattern.

Figure 4-3. Sediment – Total Chromium.

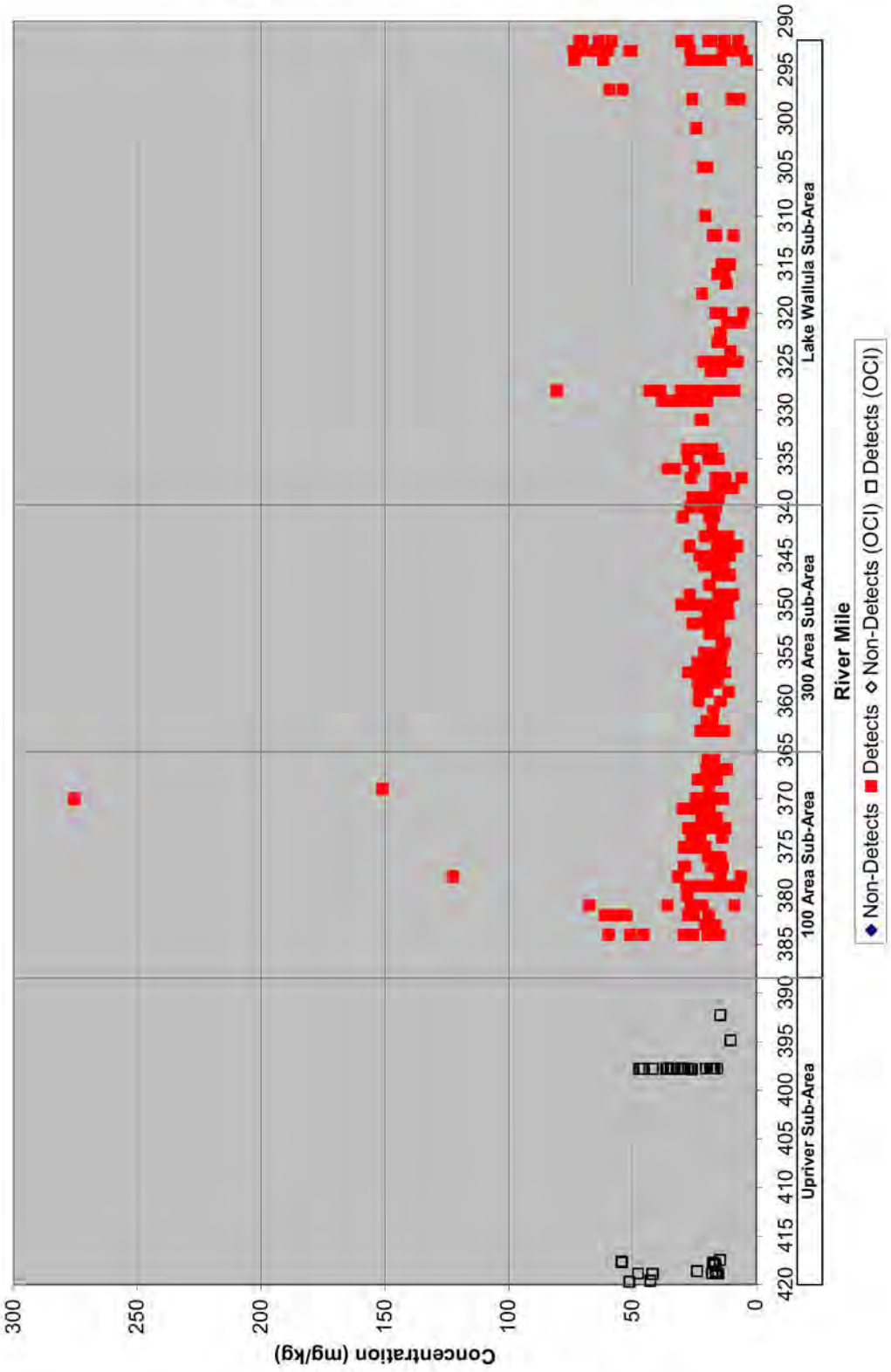


Figure 4-4. Sediment – Hexavalent Chromium.

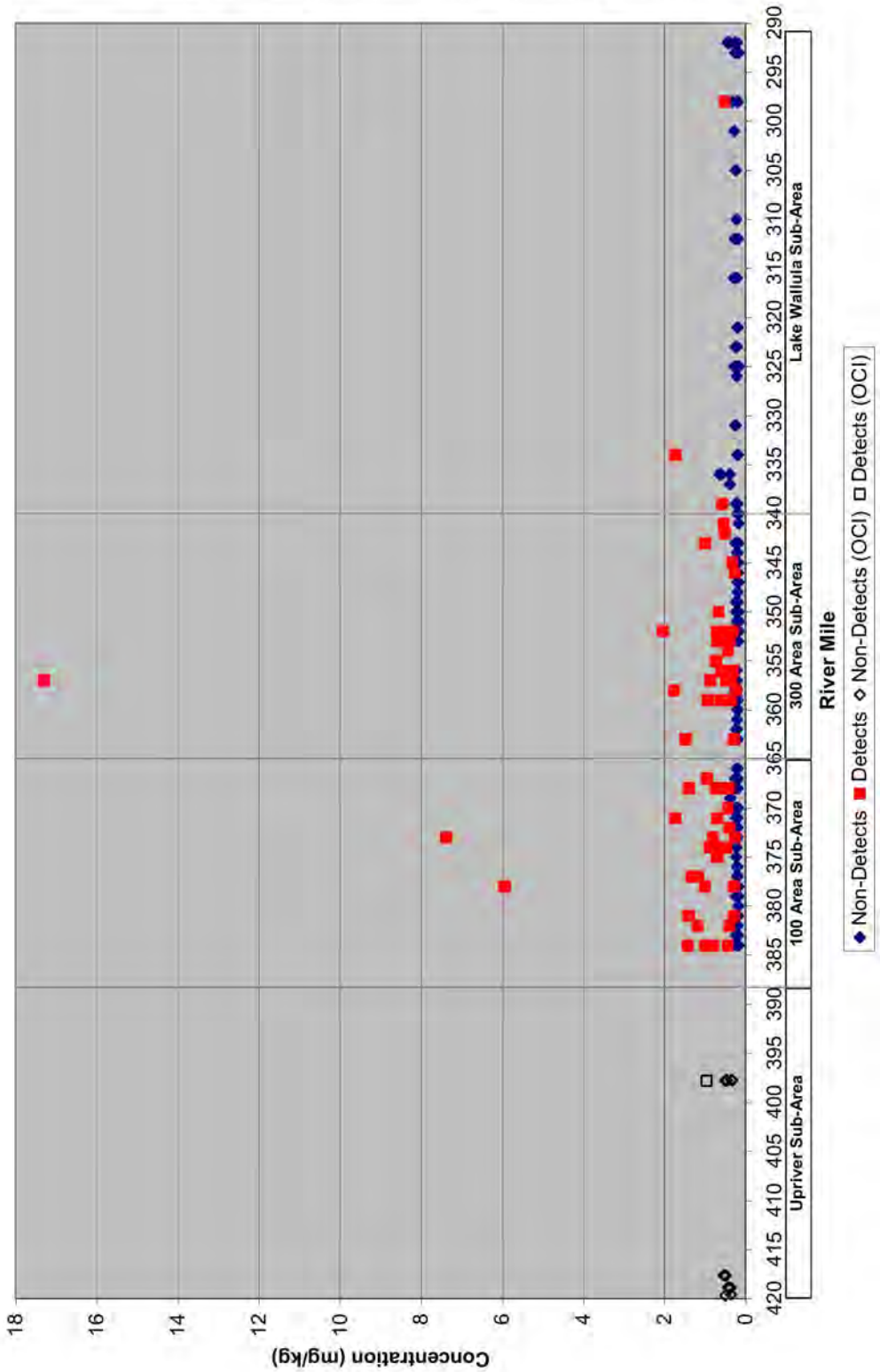
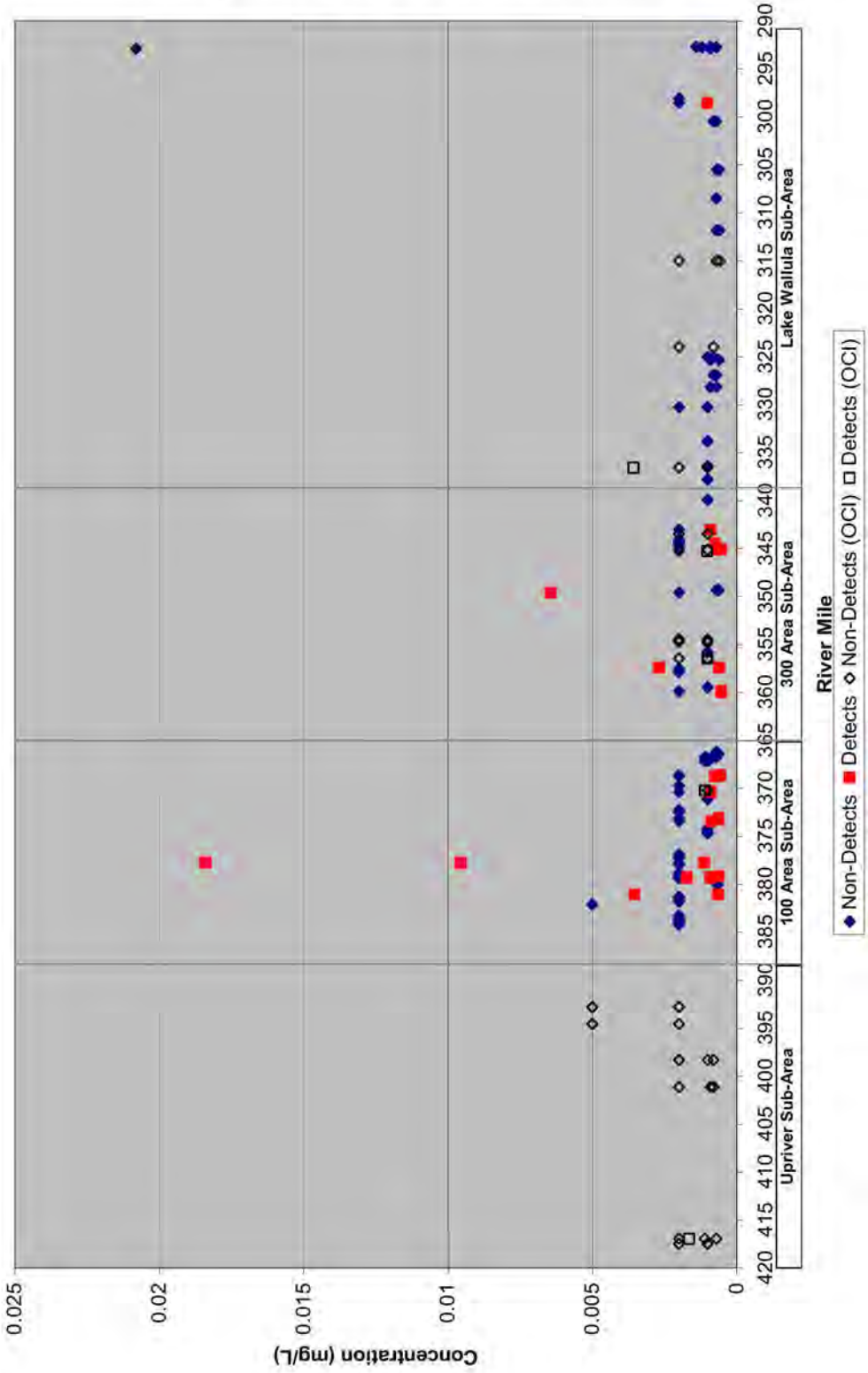


Figure 4-5. Surface Water – Chromium.



When reviewing the radionuclides, only europium-152 showed a strong distinct pattern in the 300 Area Sub-Area at RM 345 (Johnson Island; see Figure 4-6). Uranium-233/uranium-234 also had an elevated concentration at this same location. None of the SVOCs (pesticides, TPH, PCBs) resulted in a distinct pattern.

Johnson Island is considered a distinct exposure point location based on the elevated soil concentrations of cadmium and europium-152 and is labeled exposure point 300-B. In creating this distinct exposure point, sediment data and soil collected from Johnson Island are used to calculate the EPCs for 300-B. Exposure point 300-A covers the remainder of the 300 Area Sub-Area from RM 365 to RM 340, with the exclusion of Johnson Island. For surface water, there are fewer sampling locations adjacent to Johnson Island, and therefore an EPC for this area adjacent to the island was not calculated. Surface water sampling locations are used to calculate surface water EPCs for both 300-A and 300-B. Tables 4-4 and 4-5 present the sampling locations used in the calculation of EPCs for exposure points 300-A and 300-B, respectively.

4.2.1.3 Lake Wallula Sub-Area. Surface water and sediment data were evaluated for spatial variability in the Lake Wallula Sub-Area. (The Lake Wallula Sub-Area does not contain any islands that were sampled.) The shoreline (shallow) sediment data from Lake Wallula show higher concentrations than either the 100 or 300 Area Sub-Areas for two metals, mercury and strontium, and two radionuclides, cesium-137 and europium-152. The elevated radionuclide concentrations occur at the farthest end of Lake Wallula, adjacent to McNary Dam, RM 296 to RM 292. Both cesium-137 and europium-152 concentrations meet the criteria for a distinct exposure point. These results are presented in Figures 4-7 and 4-8. The surface water data showed minimal variation with sampling location. Thus, RM 296 to RM 292 of Lake Wallula is designated as exposure point B (LW-B), and the remainder of Lake Wallula (RM 339 to RM 297) is designated as exposure point A (LW-A). The sampling locations that represent exposure points LW-A and LW-B are shown in Table 4-6 and Table 4-7, respectively.

4.2.1.4 Summary of Exposure Points. Figures 4-9 through 4-11 show the exposure points identified as a result of evaluation of spatial trends in contaminant presence and concentration. Two distinct exposure points were identified for each sub-area:

- **100 Area:** Exposure point 100-A represents much of this sub-area. Exposure point 100-B represents elevated chromium concentrations in sediment and surface water at RM 378 to RM 369.

- **300 Area:** Exposure point 300-A represents most of this sub-area. Exposure point 300-B represents elevated radionuclide and metal concentrations in soils on Johnson Island and the near-shore sediments surrounding this island.

- **Lake Wallula:** Exposure point LW-A represents RM 339 to RM 297. Exposure point LW-B represents elevated radionuclide concentrations in shallow (upper 0.2 m [0.5 ft]) sediment between RM 296 and RM 292.

Figure 4-6. Soils – Europium-152.

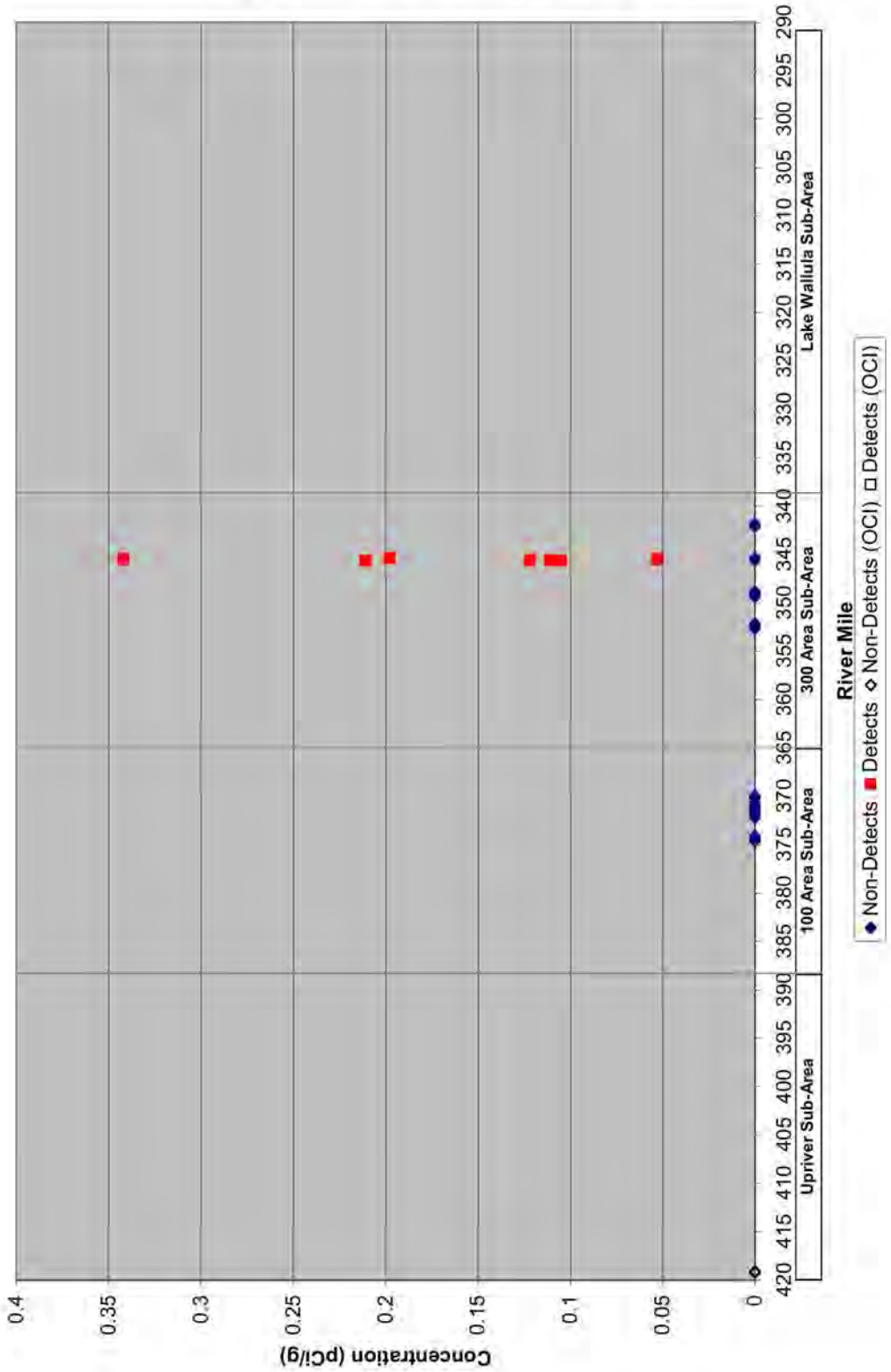


Figure 4-7. Sediment – Cesium-137.

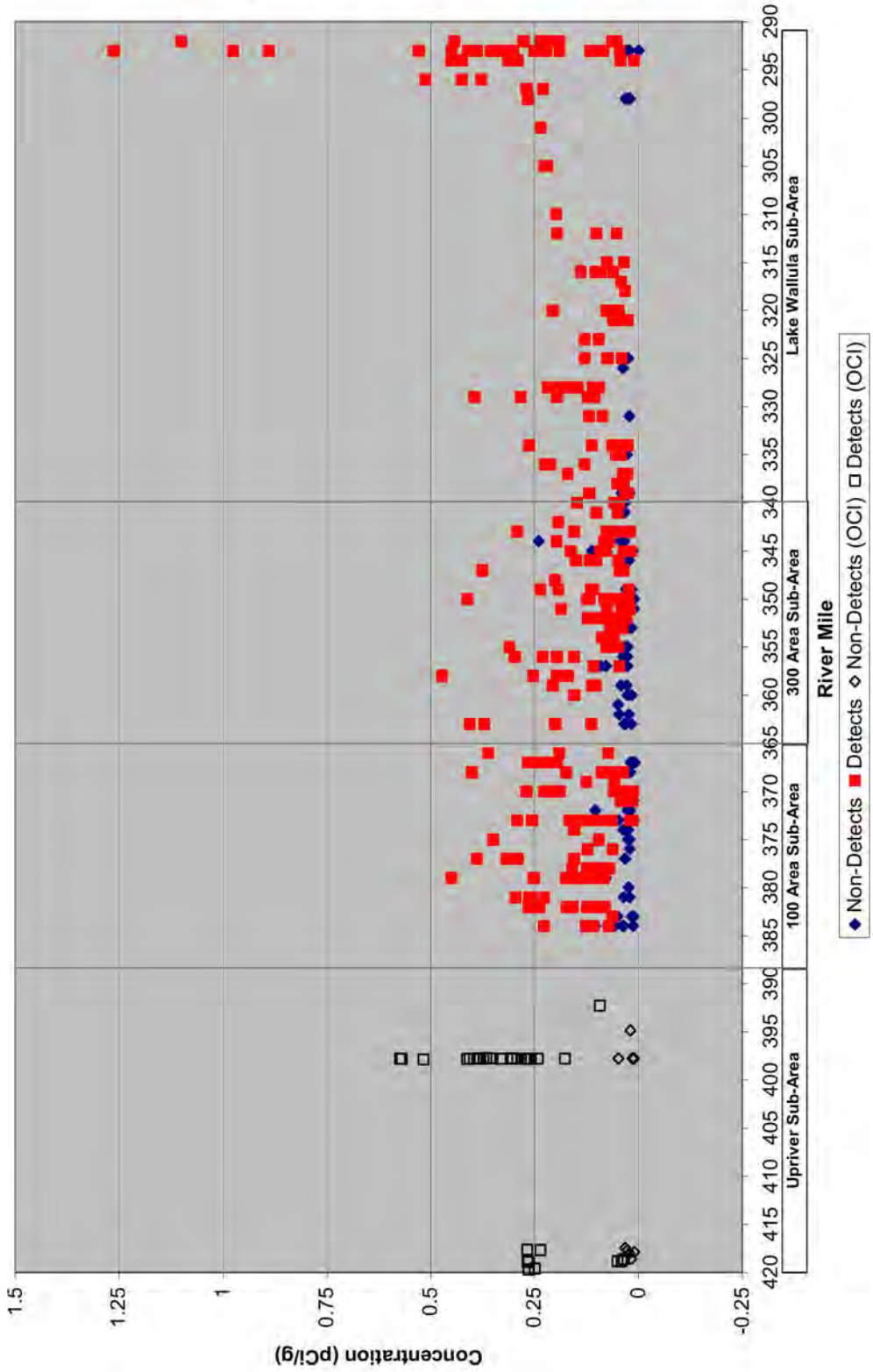


Figure 4-8. Sediment – Europium-152.

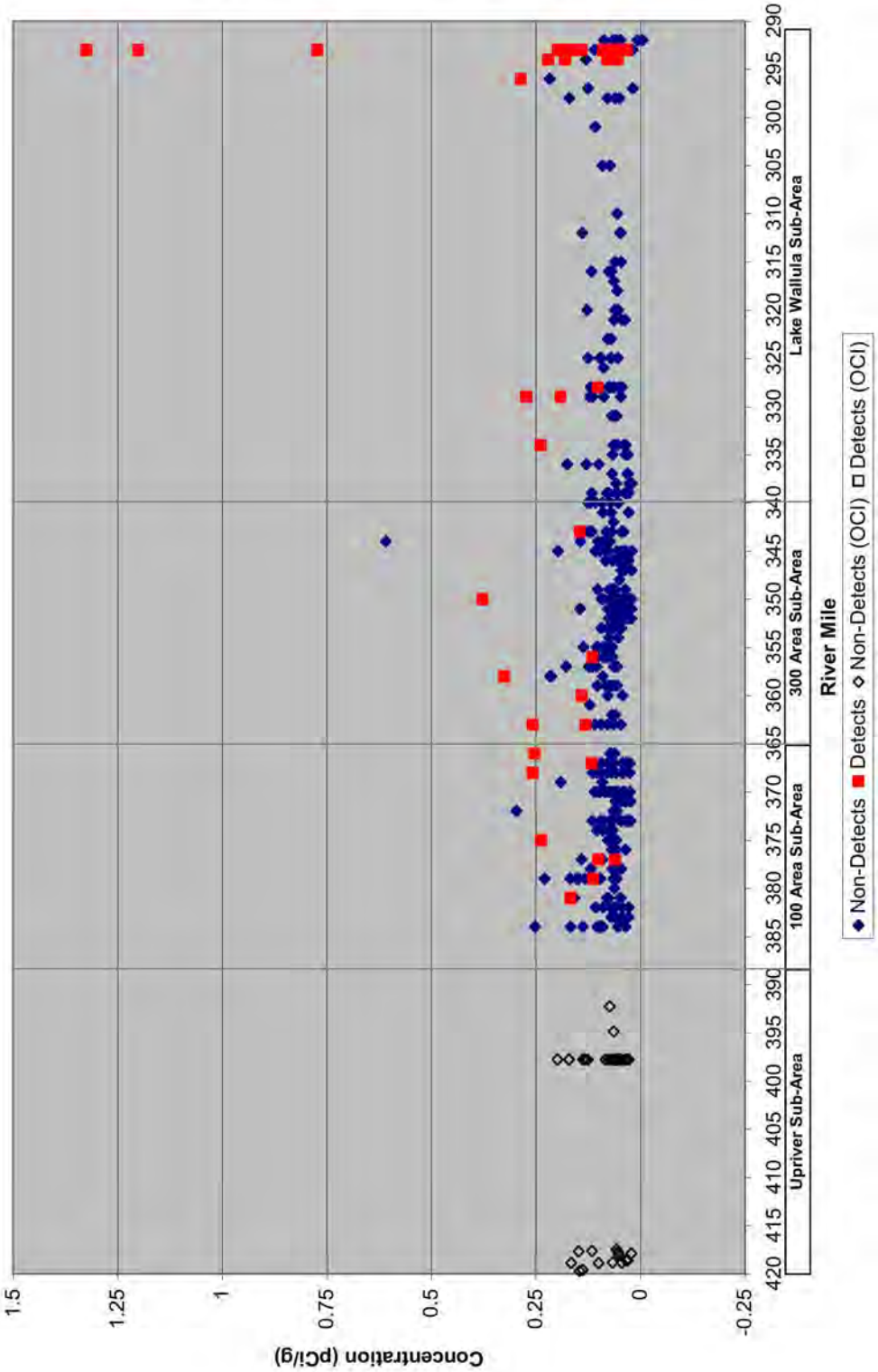


Figure 4-9. Map of Distinct Exposure Points, 100 Area A and B.

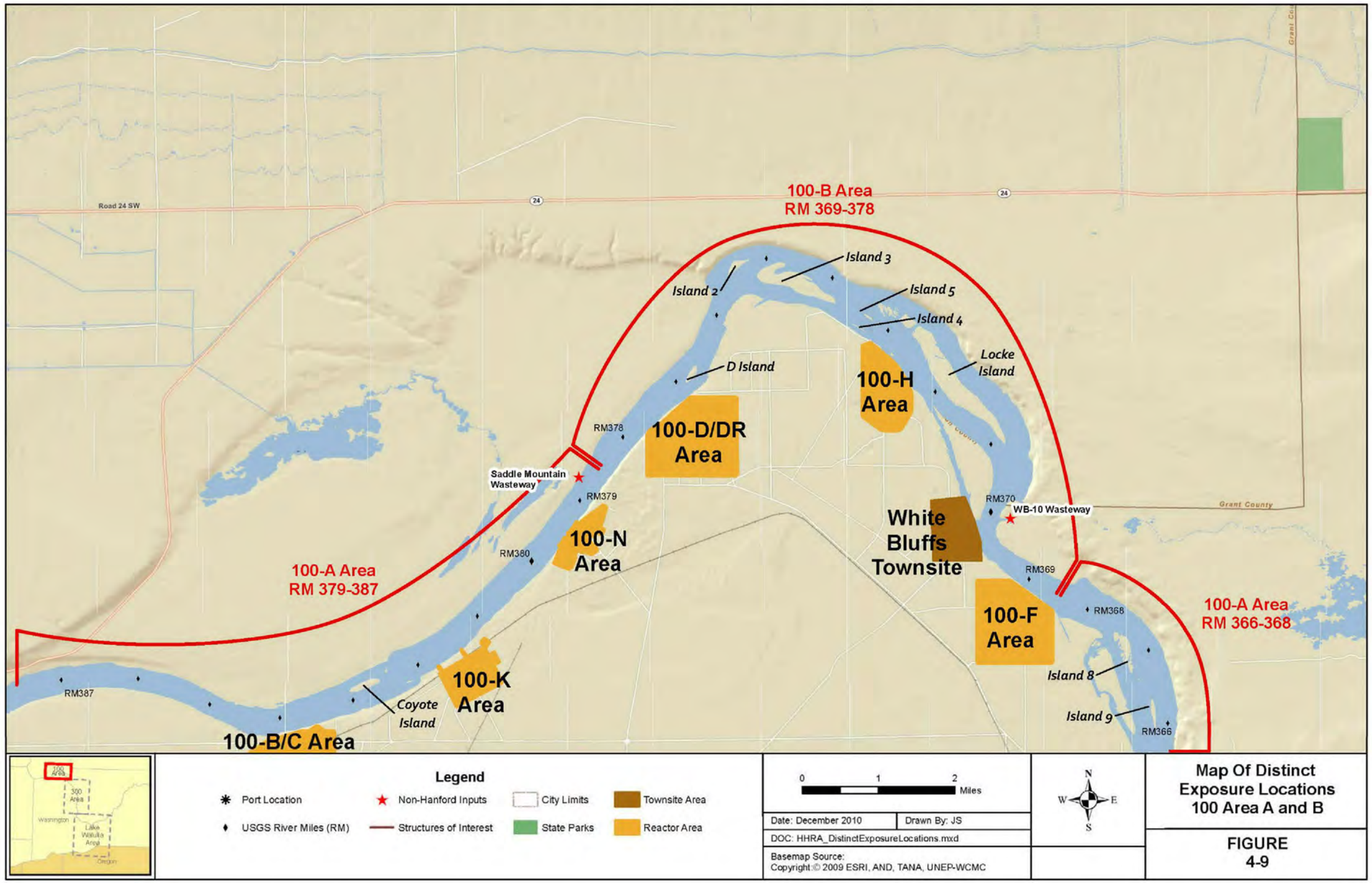


Figure 4-10. Map of Distinct Exposure Points, 300 Area A and B.

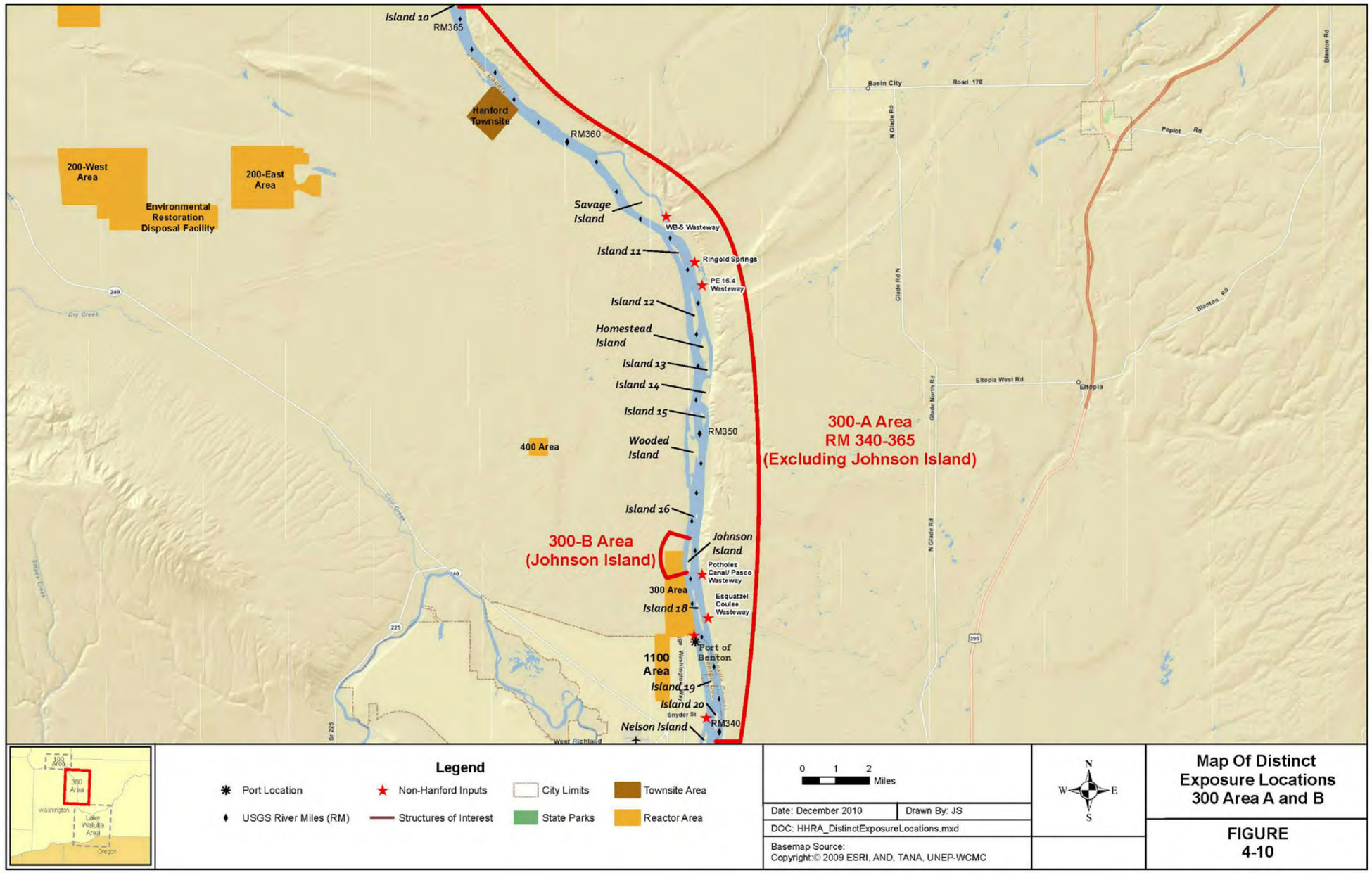
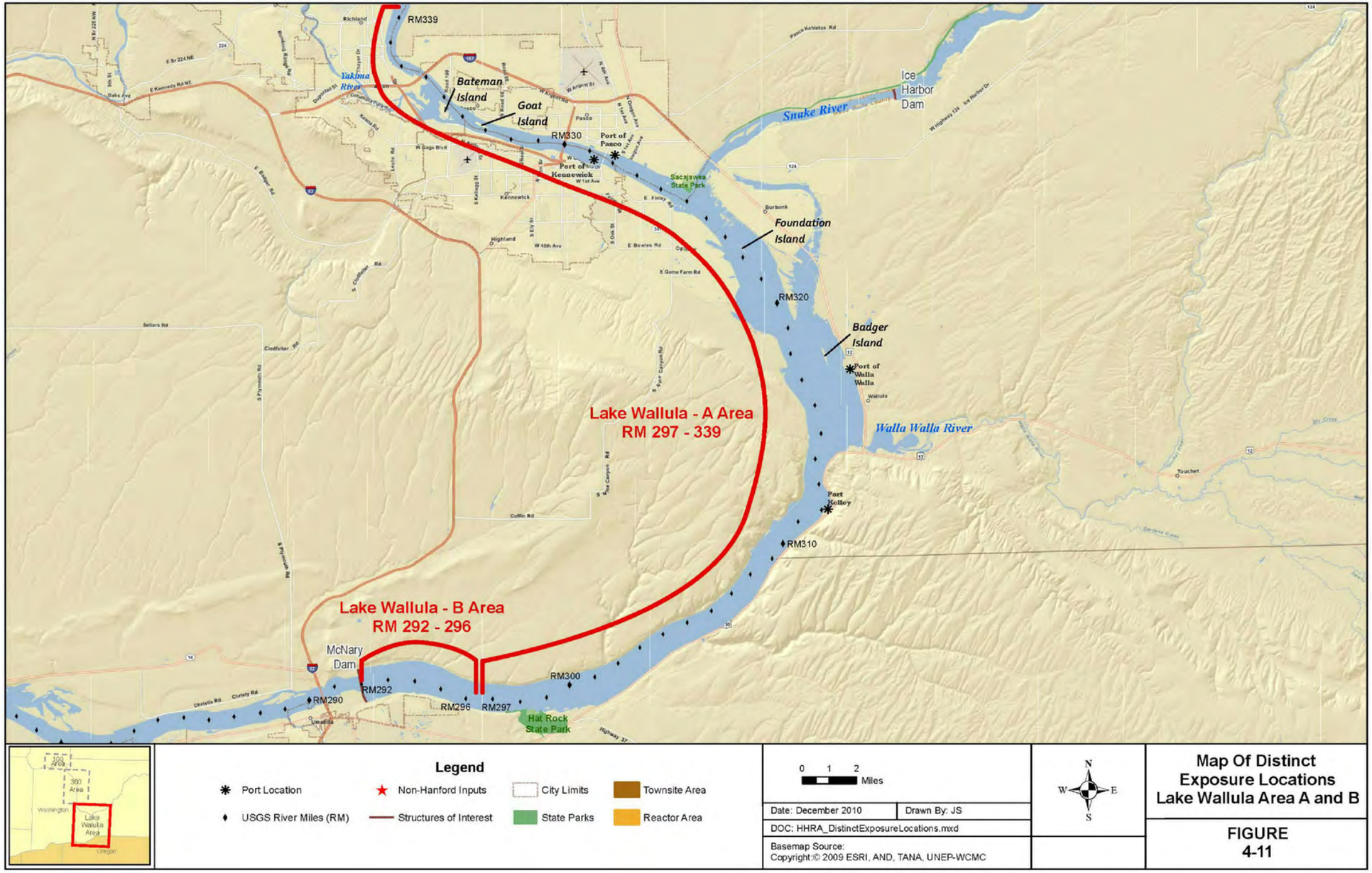


Figure 4-11. Map of Distinct Exposure Points, Lake Wallula Area A and B.



4.2.2 Exposure Point Concentrations

Exposure point concentrations are estimates of the chemical concentrations in environmental media at an exposure point to which a potential receptor is likely to be exposed under current and reasonable foreseeable future activities and uses. Exposure point concentrations were calculated for each exposure point identified within the Hanford Site Study Area using the relevant data within the HHRA data set, as previously discussed. This data set includes surface water, sediment, island soil, and fish tissue data up through the 2010 sampling events.

As discussed, this HHRA includes an evaluation of two conditions of exposure: CTE and RME. The CTE condition is representative of the average member of the exposed population and the average (arithmetic mean) concentration of the COPCs. The RME is representative of that portion of the population that experiences the greatest potential for exposure, based on characteristic behaviors and upper-bound concentrations of COPCs. The same toxicity values were used to assess both CTE and RME scenarios; refer to Section 5.0 for a description and the basis of these data. The CTE and RME estimates of risk provide the risk manager with a range of risk estimates to help capture and illustrate the potential variability in the estimation of risk. As previously described, only the one condition (based on RME EPCs and assumptions provided in Ridolfi 2007) is provided for the Yakama Nation receptor group.

In general, the process for deriving EPCs, as described in the following sections, follows EPA guidance (e.g., EPA/540/1-89/002; OSWER 9285.6-10, *Calculating Upper Confidence Limits for Exposure Point Concentrations at Hazardous Waste Sites*; and EPA/600/R-07/038, *ProUCL Version 4.00.05 User Guide*). When determining the appropriate metric to use as the EPC, the factors considered included the number of available sample results per exposure unit, prevalence of censored data, and available and appropriate statistical method(s) that would provide reasonable estimates of mean and upper-bound exposures.

The decision logic for choosing an appropriate statistical method was based on the number of detected samples and the statistical distribution of the available results for the spatial scale of interest. In general, the arithmetic mean concentration was used as the EPC for CTE and the 95% UCL of the arithmetic mean concentration was used as the EPC for RME scenarios, in accordance with the RI Work Plan (DOE/RL-2008-11). Use of the mean and upper bound on the mean provides a reasonable estimate for exposures anticipated to occur on a chronic basis in each exposure point and captures uncertainties inherent with estimating a "true" average (EPA/540/1-89/002).

As previously discussed in Section 3.5.8, the EPA software ProUCL (Version 4.00.05) (EPA/600/R-07/038) was used to calculate means and 95% UCLs. For analytes with a small number of detected sample sizes (n'), calculation of the mean and the 95% UCL can be problematic, however, because of the low number of samples and corresponding statistical variability. Therefore, for COPCs with detected concentrations (n') less than four, the CTE and RME were selected according to the following conditions:

- n' = 1; the detected result was used as the EPC for both CTE and RME
- n' = 2; the maximum detected result was used as EPC for both CTE and RME
- n' = 3 or 4; the mean was used as EPC for CTE, and the maximum detected concentration was used as the EPC for RME.

Constituents not detected within a particular exposure point were not considered to be relevant COPCs for that exposure point; therefore, EPCs were not generated for those constituents.

When the number of detected concentrations was five or greater and the total number of samples exceeded the number of detected samples, the following method was used. Analytical results that are below the concentration which the laboratory considers the reliable lower limit of the method are referred to as nondetects. For those samples, the laboratory reports the lower limit of detection and assigns a qualifier ("U") to the result. These nondetect results are referred to as censored data. The true value of the nondetect result may range from nearly zero (i.e., the constituent is absent in a sample) up to laboratory's detection limit (i.e., the constituent is present but at a level that cannot be quantified).

When calculating the mean or the UCL, it is necessary to estimate a value within that range so that the mean and UCL are not biased by the detection limits of nondetect results. One method recommended by EPA to calculate substitute values for the detection limits is the KM)estimation technique (EPA/600/R-07/038). This is a nonparametric approach for estimating the mean and standard deviation. Information on the equations for this calculation can be found in EPA/600/R-07/038. For all data sets with less than 100% FOD, the KM technique was used for the mean and standard error. These parameter estimates were then used in the calculation of a UCL.

ProUCL performs various distribution fitting evaluations of the data set and then computes a UCL in accordance with the best fit distribution. Occasionally, the data do not fit any distribution particularly well. In such cases, ProUCL computes UCLs for different distributional assumptions and/or parametric or bootstrap resampling algorithms. In such cases, the maximum UCL computed was used for the RME estimate.

The following sections describe the data sets and present the EPCs for each medium. For all media and analytes, the tables contain information on FOD (number of detected results and total number of samples), the CTE EPC, and the basis for that estimate; and for the RME estimate, the maximum detected concentration, UCL value, and basis of the UCL from the ProUCL output.

The following discussion presents the EPCs for each medium and distinct exposure point. The backup for these data are the ProUCL outputs, contained in Appendix D.

4.2.2.1 Sediment. The sediment data set used to calculate EPCs is for shallow surface sediments, assumed to be representative of sediments that individuals using the Columbia River could logically be exposed to during recreational or fishing-related activities. Sediment samples were generally collected from 0 to 30 cm (0 to 12 in.) below the surface, as described in Section 3.0. In cases where specific depth of a sample was unavailable, this sample was assumed to be representative of shallow sediment.

This sediment data set also includes samples representative of varying water depths. Sediment data from areas of deep water (i.e., over 1.8 m [6 ft]) were included within the EPC data set, assuming that, even though under current conditions a receptor may not be exposed to such sediments, there is the potential that scouring, mixing, and redeposition of sediments from these areas may result in potential exposure in the future.

The CTE and RME concentrations were calculated for each distinct exposure point, as previously described. Table 4-8 presents the CTE and RME estimates for the COPC in exposure point 100-A, and Table 4-9 presents the same data for exposure point 100-B. Table 4-10 contains the sediment EPC data for exposure point 300-A, which encompasses all sediment sampling locations on either bank of the Columbia River. The exposure point 300-B is Johnson Island, and the sediment samples are only from that island and are shown in Table 4-11. Sediment EPCs for the Lake Wallula A and B exposure points are presented in Tables 4-12 and 4-13, respectively.

4.2.2.2 Surface Water. Surface water EPCs are presented in Tables 4-14 through 4-18. Tables 4-14 and 4-15 are for the 100-A and 100-B exposure points. As the 300-B exposure point consists of soils on Johnson Island, it was decided that a separate EPC for river water at this exposure point was not appropriate because of the limited number of surface water samples collected directly adjacent to the island. Therefore, the surface water EPC is used for the dose calculations for exposure points 300-A and 300-B. Table 4-16 contains the surface water EPCs for the 300 Area Sub-Area based on all samples collected within that section of the river. The EPCs for the "A" and "B" exposure points of Lake Wallula Sub-Area are presented in Tables 4-17 and 4-18, respectively.

4.2.2.3 Island Soils. The following three exposure points within the Hanford Site Study Area contain islands: 100-B, 300-A, and 300-B. The 100-B and 300-A exposure points contain multiple islands, so the EPC is based on samples collected on different islands (although not all islands in a sub-section were sampled). These EPCs are presented in Tables 4-19 and 4-20, respectively. The 300-B EPCs (Table 4-21) are based solely on samples from Johnson Island.

4.2.2.4 Fish Tissue. Fish ingestions risks were evaluated using two different approaches. First, fish tissue data from all six species were aggregated together to generate EPCs for fish as an exposure medium. This approach made no assumptions about preferential consumption of individual species types and was intended to evaluate general health risks associated with fish ingestion in each of the three sub-areas of the Hanford Site Study Area.

The second approach entailed calculation of separate fish EPCs for each individual species as a means of evaluating comparative risk among the different fish species. This second approach was applied to only the Avid Angler scenario (note that this is *in addition* to use of the first approach). Because risk is directly proportional to exposure, the magnitude of difference in fish ingestion risks among the species for one receptor (i.e., the angler) may be applied to other receptors (i.e., the members of the Yakama Nation). Thus, evaluation of ingestion risk related to consumption of individual fish species for only the Avid Angler scenario will streamline the HHRA while also allowing application of the results to the Yakama Nation exposure scenario. Note that although the fillet comprises most of the consumable portion of fish, there is potential for receptors to consume other portions such as organ meat, skin, and small bones. Therefore, assumptions about the consumable fraction of each tissue type were made in this HHRA for the Avid Angler and Yakama Nation exposure scenarios.

Each of these approaches is discussed further below. Additionally, Sections 4.2.2.4.3 and 4.2.2.4.4 provide a discussion of the treatment of arsenic and mercury speciation in fish tissue EPCs.

4.2.2.4.1 Fish Exposure Point Concentrations: All Species Combined. For this approach, analytical results for all species were combined to generate fish tissue EPCs in each sub-area. As discussed in Section 3.0, six species of fish were collected in each of the three sub-areas: bass, carp, sturgeon, sucker, walleye, and whitefish. These species were selected during RI Work Plan development as species representative of those frequently caught and consumed in the Study Area. Note that salmon species comprise the majority of fish caught and consumed, particularly for Native American groups. However, due to their anadromous nature (i.e., spending most of their lives in the ocean), salmonids were not included in the fish collection program.

It is recognized that not all human receptors will consume all six species of fish, and that preferential consumption of certain fish species (such as bass or walleye) is likely to occur. However, pooling species increased the statistical sample size so that more robust statistical methods could be used. Furthermore, evaluation of analytical results from fish tissue does not suggest that COPC levels are consistently elevated in one fish species with respect to another (see Section 3.6.4).

Fish EPCs were generated for each of the three sub-areas. There are two reasons for this: first, fish are highly mobile; thus, it is not possible to say that fish caught at one location are only representative of the environmental conditions of that location. Second, the number of fish collected in each of the three sub-areas (100 Area, 300 Area, and Lake Wallula) was not of sufficient number to allow sub-dividing them based on collection location. Thus, analytical results from all fish samples collected within each sub-area are used in the calculation of the EPCs for that sub-area (i.e., 100 Area, 300 Area, and Lake Wallula). The fish tissue samples for the 100 Area, 300 Area, and Lake Wallula Sub-Areas are shown in Table 4-22. Although it is possible for fish to swim among the three sub-areas, fish EPCs were developed for each sub-area to retain general consistency with the approach used in generating EPCs for other media. It is thus possible that the fish EPCs may not necessarily reflect an assigned sub-area. However, given the size of each sub-area (length varies from approximately 40 to 80 km [25 to 50 mi]),

fish home range area, and preference for certain feeding holes for the species evaluated, it is likely that fish EPCs are representative of tissue concentrations within each sub-area.

To generate fish EPCs (six species combined), the analytical results for the six fish species were first separated by tissue type (fillet, carcass, and liver and kidney). Representative tissue concentrations were then calculated for each tissue type in each sub-area. These representative concentrations were defined as either the 95% UCL or maximum detected concentration, depending on number of detected results (see Section 3.5.8). Therefore, representative concentrations were calculated for fillet (which includes skin, except for sturgeon), carcass, and combined liver and kidney. Individual results from liver and kidney samples for the sturgeon and carp, for which individual liver and kidney sample results were collected (rather than combined, as was conducted for other species), were pooled with the combined liver/kidney sample results from other fish species, since there are insufficient results for these individual organs to treat them separately. Representative fish tissue concentrations for different tissue types are shown in different tables for each sub-area. The representative concentrations for the 100 Area Sub-Area are shown in Tables 4-23 through 4-25, for the 300 Area Sub-Area in Tables 4-26 through 4-28, and for the Lake Wallula Sub-Area in Tables 4-29 through 4-31.

From these representative concentrations, separate fish tissue EPCs were calculated for the two human receptors assumed to consume fish: the avid angler (adult, youth, and child) and members of the Yakama Nation (adult and child). As discussed, it is assumed that each receptor will consume some fraction of each tissue type (fillet, liver/kidney, and carcass). For each of these receptors, the EPC was therefore weighted by differing percentages of the fish tissue components, assuming that each human receptor ingests different portions of a fish.

For the Yakama Nation scenario, it was assumed that 90% of total fish diet consisted of fillet, based on recommendations in *Exposure Scenario for CTUIR Traditional Subsistence Lifeways* (Harris and Harper 2004) and "A Native American Exposure Scenario" (Harris and Harper 1997). The remaining 10% of the fish diet was assumed to consist of organ meat and carcass. This remaining fraction was divided equally between these two nonfillet tissue components, assuming that 5% of the diet consisted of organ tissue and 5% consisted of carcass. For the avid angler, it was assumed that the fish diet consisted primarily of fillet (95%), with a small fraction (5%) consisting of carcass (to account for incidental ingestion of pin bones and the like) and that no organ meat would be ingested. For both receptors, the fraction of tissue ingested was assumed to be consistent among all age groups evaluated (i.e., child, youth, and/or adult).

Fish tissue EPCs for combined species are shown in Table 4-32 for the CTE and Table 4-33 for the RME. Each table contains the EPC for all three sub-areas. Note that the final EPC for fish ingestion is the same for each receptor within an individual sub-area rather than exposure point; for example, the fish ingestion EPC for a receptor in the 100-A exposure point is the same as that for a receptor in the 100-B exposure point. As discussed, fish EPCs for combined species were generated based on sub-area and not individual exposure points, as was done for the abiotic media.

4.2.2.4.2 Fish Exposure Point Concentrations: Individual Species. As discussed in the previous section, fish EPCs were derived for individual fish species, in addition to EPCs derived for all species, combined. This approach was used for evaluation of only the avid angler RME scenario, as discussed previously. For this second approach, EPCs were generated for individual fish species to evaluate relative fish ingestion risks among the six species analyzed.

Under the assumption that a receptor may catch and consume fish from anywhere along the Hanford Site Study Area, the fish tissue data for each species were aggregated across all three sub-areas. Aggregation of fish tissue data increased the number of analytical results used to calculate EPCs by species, reducing intraspecies variability and providing a more robust estimate of the 95% UCL.

For each fish species, a representative concentration (either a 95% UCL or maximum, depending on the number of detected results; see Section 3.5.8) was first calculated for each body part (i.e., fillet and carcass), as in the methodology described in Section 4.2.2.4.1. Because EPCs for individual species are used to evaluate only the Avid Angler scenario, in which a receptor is assumed to consume fillet and carcass, representative concentrations were generated for only these two tissue types. Representative concentrations for fish tissue by tissue type and species are summarized in Tables 4-34 through 4-45. A weighted species-specific fish tissue EPC was then calculated based on the assumption that the avid angler consumed 95% fillet and 5% carcass. Table 4-46 summarizes fish tissue EPCs (RME scenario) for each species. Note that the COPCs for individual species may not mirror the COPCs for all species combined; this is because constituents were not consistently detected in all fish species.

4.2.2.4.3 Arsenic Exposure Point Concentrations in Fish Tissue. Arsenic exists in fish tissue in two forms: inorganic and organic (such as arsenobetaine). The common organic forms of arsenic in tissue are generally not considered toxic, unlike inorganic forms (e.g., As^{3+}, As^{5+}) of arsenic. Therefore, in order to understand speciation of this metalloid in fish tissue, select fish tissue samples were analyzed for total arsenic and TIAS.

Total arsenic was detected infrequently in fillet, carcass, and liver/kidney samples, and only in sturgeon and whitefish. Arsenic concentrations in carcass and liver/kidney samples were generally consistent with those observed in fillet. Liver/kidney concentrations ranged from 0.2 to 1.6 mg/kg, with the highest concentration observed in a sturgeon sample collected from the 100 Area Sub-Area.

Total inorganic arsenic was analyzed for in most fish species evaluated and detected at a greater frequency and in more fish species than total arsenic, due to lower LRLs as a result of differences in analytical methodologies between the two analyses. Results show that inorganic arsenic was detected most often in carp, sucker, and walleye; less often in sturgeon and bass; and not detected in whitefish. Sturgeon, however, is the only fish species in which both total arsenic and TIAS were consistently analyzed and detected. In fillet samples, in which these two parameters were detected, TIAS comprised less than 1% of the total arsenic concentration, as shown in Table 4-47.

These sturgeon TIAS results are lower than published values for TIAS in sturgeon in the lower Columbia River, in which TIAS comprised approximately 2% to 18% of total arsenic in sturgeon (Lorenzana et al. 2009, "Arsenic in Seafood: Speciation Issues for Human Health Risk Assessment"). Studies of arsenic in fish from Lake Roosevelt, located upriver of the Hanford Reach showed that up to 30% of total arsenic in rainbow trout (whole body) and up to 20% in fillet was in the inorganic form (CH2MHILL 2007).

Liver/kidney samples were not analyzed for TIAS. Therefore, TIAS concentrations were estimated from total arsenic values based on carcass and fillet results. For this tissue type, it was assumed that TIAS comprised 1% of the total arsenic concentration for purposes of calculating arsenic fish tissue EPCs. This assumption is supported by the analytical results. Across all fish species, the mean TIAS concentration in fillet and carcass (combined) is 0.003 mg/kg, approximately 0.7% of the mean total arsenic concentration of 0.4 mg/kg (Table 3-6).

4.2.2.4.4 Mercury Exposure Point Concentrations in Fish Tissue. Most if not all of mercury in fish tissue is in organic form. In the aqueous environment, inorganic mercury in sediments or adsorbed to particulate matter suspended in the water column is methylated by anaerobic bacteria into methyl mercury, the most bioavailable and toxic form of mercury. It is this form of mercury that preferentially accumulates in fish tissue (ATSDR 1999, *Toxicological Profile for Mercury*). Total mercury was analyzed for and detected in nearly 100% of all fish tissue samples.

Methyl mercury was analyzed in six sturgeon fillet and carcass samples within the Upriver, 300 Area, and Lake Wallula Sub-Areas. Methyl mercury concentrations in fillet samples were compared to total mercury concentrations in corresponding sturgeon fillet samples in Table 4-48.

Methyl mercury comprises a subset of total mercury concentrations. However, because different analytical methods are used to analyze total and methyl mercury, methyl mercury concentrations may exceed total mercury concentrations, yielding a ratio greater than 100%. As shown in Table 4-8, methyl mercury accounts for most or all of the total mercury concentrations in sturgeon fillet tissue.

Therefore, it was assumed that methyl mercury comprised 100% of the total mercury concentration in all fish tissue and the methyl mercury toxicity value was used to assess risk, as further described in Section 5.4.2. Because total mercury was analyzed for in all fish species and tissue types, total mercury data rather than methyl mercury data were used to develop the fish tissue EPCs for this COPC.

4.3 QUANTITATION OF EXPOSURE

The quantitative exposure assessment describes a conservative estimate of exposure to a representative individual within a subpopulation (receptor group) based on the defined exposure scenarios. The exposure dose therefore represents the amount of a COPC to which an individual receptor may come into contact. It is a function of receptor-specific exposure assumptions and chemical-specific exposure parameters. The material that reaches the receptor's absorption barrier (such as the skin, lung, or gastrointestinal tract) is referred to as the applied dose (for

ingestion and inhalation exposures), while the absorbed (or internal) dose is defined as the amount of material that actually crosses the receptor's exchange boundary.

Exposure doses (or intakes) for chemical constituents are calculated as the daily amount of the chemical taken into the body per unit body weight per unit time (mg/kg-day; EPA540/1-89-002). The general equation used to estimate average daily doses (ADDs; for noncancer effects) and lifetime average daily dose (LADDs; for carcinogenic effects) is as follows:

$$\text{ADD (or LADD)} = \frac{\text{Total amount of COPC contacted/ingested}}{\text{Body weight} * \text{averaging period}}$$

Exposure intakes are normalized to a receptor's averaging period (in days). Note that averaging time for noncarcinogenic compounds is equivalent to the exposure duration, whereas the averaging time for carcinogens is always equivalent to a 70-year (25,550-day) lifetime (EPA/540/1-89/002).

For inhalation exposures, an average daily exposure (ADE; noncancer) and lifetime average daily exposure (LADE; carcinogenic effects) is used instead of intake (EPA/540-R-070-002). Average daily exposures or LADEs are calculated, instead of ADDs or LADDs, to make them compatible with the inhalation dose-response values presented as Reference concentrations or unit risks (URs) (expressed in units of mg/m^3 and $[mg/m^3]^{-1}$, respectively). Exposures are then estimated by normalizing fugitive dust or vapor EPCs with averaging times as follows:

$$\text{ADE or LADE} = \frac{\text{Time-adjusted exposure concentration for airborne chemicals}}{\text{Averaging time}}$$

The general intake equation for radiation intake is analogous to that for chemical exposures, except that averaging time and body weight are omitted and the dose is presented in units of activity (pCi) (EPA/540/1-89/002).

Average daily doses, or LADDS for carcinogenic COPCs, are based on conservative exposure assumptions and factors developed in accordance with EPA risk assessment guidelines, Hanford Site-specific information, and other relevant guidance.

Exposure doses were calculated using receptor-specific exposure variables and chemical-specific exposure parameters (e.g., the appropriate EPCs, anatomical and physiological parameters, absorption adjustment factors, skin permeability [Kp] coefficients) to calculate the chemical-specific doses or exposures for each receptor and pathway.

All intake and exposure equations and parameters are provided in the following subsections.

4.3.1 Sources of Information for Exposure Parameters

Exposure parameters describe physiological or behavioral aspects of each target receptor and will represent a mix of CTE and upper-bound exposure assumptions and recommendations from EPA risk assessment guidance, as well as professional judgment.

For the purposes of this HHRA, both CTE and RME scenarios for each receptor group, with the exception of the Yakama Nation scenario, were evaluated. As per the RI Work Plan (DOE/RL-2008-11, Section 4.6.5) and in accordance with Ridolfi (2007), only one scenario was evaluated for the Yakama Nation scenario, based on exposure parameters.

Standard physiological exposure parameters, such as skin surface areas, body weights, and inhalation rates, were based on a receptor's age range for both males and females and generally were the values recommended by EPA, as indicated in various guidance documents (e.g., EPA/600/P-95/002Fa, EPA/540/R-99/005).

Several different resources were consulted for the casual user and avid angler scenarios. River usage parameters (e.g., time spent fishing or swimming) were taken from PNNL-13840, *2001 Columbia River Recreation Survey – Implications for the Hanford Site Integrated Assessment*. For these recreational receptors, activity factors specific to the Columbia River recreational areas were used to estimate exposures. For the casual user, exposure parameters specific to swimming, wading, and waterskiing were considered, since these are the types of activities frequently observed on the river. For the Avid Angler scenario, the majority of anglers are assumed to participate in boat fishing (PNNL-13840) or fishing from banks; therefore, exposure factors reflect these types of activities.

The Yakama Nation exposure parameters were taken primarily from the white paper titled *Yakama Nation Exposure Scenario for Hanford Site Risk Assessment, Richland, Washington* (Ridolfi 2007). Values from DOE/RL-96-16, *Screening Assessment and Requirements for a Comprehensive Assessment: Columbia River Comprehensive Impact Assessment*, and Harris and Harper (2004) were used for parameters not specifically included in the Yakama Nation report.

The specific exposure parameters for all receptors proposed for quantitative evaluation in this baseline risk assessment are summarized in Table 4-49 for CTE and Table 4-50 for RME. Specific exposure parameters are further discussed by exposure pathway in Section 4.3.2.

4.3.2 Calculation of Intake and Exposure

The following subsections present the equations and parameters used to calculate chemical and radiological intakes/exposures. Parameters unique to each exposure pathway are also described. Dose calculations for all receptors, as well as the equations and parameters used in calculating doses, are provided in Appendices F through L.

4.3.2.1 Soil and Sediment Ingestion. Chemical intake via soil or sediment ingestion is calculated using the following equation:

$$ADD/LADD\ (mg/kg\text{-}day) = (EPC_s * IR_s * EF * ED * C1)/(BW * AT)$$

The equation for radionuclide intake (RI) excludes both body weight and averaging time, or

$$RI\ (pCi) = (EPC * IR_s * EF * ED * C2)$$

where:

EPC_s = soil or sediment EPC (mg/kg or pCi/g)
IR_s = daily soil/sediment ingestion rate (mg of soil/day)
EF = exposure frequency (days/yr)
ED = exposure duration (yr)
BW = body weight (kg)
AT = averaging time (days)
C1 = units conversion factor, 1E-06 kg/mg
C2 = units conversion factor, 0.001 g/mg.

Soil and sediment EPCs were described in Section 4.2 and generally represent the arithmetic mean (for CTE) or 95th UCL of the mean concentration (for RME).

4.3.2.1.1 Soil Ingestion Rate. It is assumed that a small amount of sediment or soil is inadvertently swallowed during wading, fishing, or swimming in the river. For example, soil may adhere to hands, and then soil is transferred to the mouth while a receptor is eating. Soil ingestion rates are typically higher in younger children than in older children and adults, due to increased hand-mouth activities.

The EPA-recommended daily soil ingestion rates (EPA/540/1-89/002) were used for the Avid Angler and Casual User scenarios. It was assumed that these rates were suitable for both soil and sediment exposures. Children (<7) were assumed to ingest 100 mg (for CTE) or 200 mg (for RME) per day, whereas older children and adults were assumed to ingest 50 mg/day or 100 mg/day (for CTE and RME, respectively) for recreational scenarios (Avid Angler and Casual User). These values are EPA-recommended soil ingestion rates for adults and children. The recommended soil ingestion rates (200 mg/day for the adult and 400 mg/day for the child) from Ridolfi (2007) were used to evaluate the Yakama Nation scenario. These enhanced soil/sediment ingestion rates are intended to reflect a more active lifestyle.

For exposure points where a receptor was assumed to be exposed on a daily basis to both island soil and sediment, these upper-bound daily ingestion rates were split between these two media, so that on each day of exposure, one-half of the ingestion rate (e.g., 200 mg/day for the Yakama Nation child receptor) was assumed to be derived from soil exposure, and the other half was assumed to be derived from sediment exposure. This approach was taken because the soil ingestion rates assume that a given amount of soil (regardless of source) is ingested on a

daily basis. Therefore, assuming the full soil ingestion rate for both media would essentially result in the "double-counting" of risks from incidental ingestion of these media.

With the exception described above, it was assumed that the entire daily ingestion rate of both sediment and soil ingestion was derived from each exposure point. This is a reasonable although likely conservative assumption, given the active nature of recreational activities that are anticipated to occur and the length of time of each exposure event.

4.3.2.1.2 Exposure Frequency. Exposure frequency describes how many exposure events occur per a given time (in this case, a year). For the CTE scenarios, the exposure frequency value of 47 days/yr was used for the Casual User and Avid Angler scenarios; this value is based on the average number of visits per year to the Columbia River, as reported in PNNL-13840, Table 4.4. For the RME scenario, an exposure frequency of 58 days/yr was applied to both Casual User and Avid Angler receptors. This value is based on the survey results published by PNNL-13840 (in Table 4.4) and represents the maximum number of visits per year to the Columbia River among various Washington counties and other areas. For members of the Yakama Nation (both children and adults), the exposure frequency of 150 days/yr is that used in DOE/RL-96-16.

4.3.2.1.3 Exposure Duration. The exposure duration describes the length of time over which the receptor comes into contact with contaminants. The child Casual User and child Avid Angler exposure duration of six years represents a child 1 < 7 years; the youth Avid Angler exposure duration represents an older child ages 7 < 14 years. These exposure duration values were used to assess both CTE and RME for these receptors. For the adult Avid Angler and Casual User, exposure durations of 9 and 30 years were used to evaluate CTE and RME, respectively. These values are EPA-recommended values for residents (EPA/600/P-95/002Fc). The exposure duration values of 6 years for the Yakama Nation child (1 < 7 years) and 70 years for the Yakama Nation adult scenarios are those values recommended by Ridolfi (2007).

4.3.2.1.4 Body Weight. The body weight of 16.6 kg for the child Casual User and Angler scenarios is based on the mean body weight of male and female children, ages 1 < 7 years (EPA/600/P-95/002Fa, Tables 7-6 and 7-7). For the Avid Angler youth, the body weight of 37 kg is the mean weight of males/females ages 7 < 14 years (EPA/600/P-95/002Fa). The body weights for child members of the Yakama Nation (16 kg) are the EPA recommended value presented in OSWER 9285.6-03, *Risk Assessment Guidance for Superfund Volume 1: Human Health Evaluation Manual, Supplemental Guidance, "Standard Default Exposure Factors" Interim Final*, and Ridolfi (2007). For all adult receptors, the default mean body weight of 70 kg was used (EPA/540/1-89/002).

4.3.2.2 Water Ingestion. Chemical intake via water ingestion is calculated in a manner similar to that of soil/sediment, but includes an adjustment based on a receptor's time spent in an activity. Whereas soil/sediment ingestion is assumed to occur on a daily basis, water intake for recreational and fishing scenarios is assumed to occur on an hourly basis.

In this HHRA, the quantitative assessment of water intake is associated with incidental ingestion of surface water while swimming, wading, waterskiing, or engaging in other similar recreational activities in the river. Water intake is described using the following equation:

$$ADD/LADD\ (mg/kg\text{-}day) = (EPC_w * IR_w * ET * EF * ED * EV)/(BW * AT)$$

The equation for radionuclide intake excludes both body weight and averaging time, or

$$Radionuclide\ intake\ (pCi) = (EPC_w * IR_w * ET * EF * ED * EV)$$

where:

EPC_w = surface water EPC (mg/L or pCi/L)
IR_w = water ingestion rate (L/hr)
EF = exposure frequency (days/yr)
ED = exposure duration (yr)
ET = exposure time (hr/event)
EV = event frequency (1 event/day)
BW = body weight (kg)
AT = averaging time (days).

Exposure frequency and duration, body weights, and averaging times are the same values as those used to assess soil and sediment ingestion, as described in Section 4.3.2.1.

Exposure time values of 4 hr/day and 6.1 hr/day, for CTE and RME, respectively, were used for the Casual User scenario (adult and child). These values are based on the survey data from PNNL-13840 (Table 4.6) and represent the amount of time spent per trip when the primary activity is waterskiing and secondary activities are swimming and boating. The CTE value represents the amount of time spent engaged in waterskiing, the primary activity, whereas the RME value is the total amount of time per trip. It is likely that this "trip" time includes nonriverine activities such as walking to and from the car, unloading boats or other sporting equipment, potentially eating on upland areas on shore, etc.

For the Avid Angler scenario, exposure time values of 6.1 hr/day (CTE) and 6.7 hr/day (RME) are based on survey data from PNNL-13840 (Table 4.6) and represent the amount of time spent engaging in fishing from a boat. The CTE value represents the time spent fishing from a boat and the RME value represents total time spent per trip.

For the Yakama Nation (both children and adults), the exposure time (7 hr/day) is that recommended by Ridolfi (2007).

4.3.2.2.1 Water Ingestion Rate. It is assumed that a small amount of river water is inadvertently swallowed during wading, fishing, or swimming in the river. The ingestion rate for surface water for the Casual User and the Yakama Nation receptors is the default value for swimming (0.05 L/hr), as recommended by EPA/540/1-89/002, Exhibit 6-12. The water ingestion rate for the Avid Angler is one-half the EPA default values (i.e., 0.025 L/hr). As the

angler is engaged in fishing activities, this receptor is assumed to incidentally ingest only a minimal amount of river water while fishing. These ingestion rates were used to evaluate both CTE and RME scenarios.

4.3.2.3 Fish Ingestion. Fish tissue intakes are calculated in a manner analogous to that of soil/sediment. Chemical intake via fish ingestion is calculated using the following equation:

$$\text{ADD/LADD (mg/kg-day)} = (EPC_f * IR_f * EF * ED * C1)/(BW * AT)$$

The equation for radionuclide intake excludes both body weight and averaging time, or

$$\text{Radionuclide intake (pCi)} = (EPC_f * IR_f * EF * ED * C2)$$

where:

EPC_f = fish EPC (mg/kg or pCi/g)
IR_f = daily fish ingestion rate (mg of fish per day)
EF = exposure frequency (days/yr)
ED = exposure duration (yr)
BW = body weight (kg)
AT = averaging time (days)
C1 = units conversion factor (1E-06 kg/mg)
C2 = units conversion factor (0.001 g/mg).

Exposure duration, body weights, and averaging times are the same values as those used to assess soil and sediment ingestion, as described in Section 4.3.2.1. Unlike soil and water ingestion pathways, the ingestion rates for fish consumption are based on year-round, daily consumption. Therefore, an exposure frequency of 365 days/yr was used for all receptors for evaluation of the fish ingestion pathway.

For the Avid Angler CTE scenario, the daily fish ingestion rates reflect the 50th percentile value for "consumer only" intake of fish (0.443 g/kg-d, or 31 g/day, assuming a 70-kg body weight) in the western United States (EPA/600/P-95/002Fb, Table 13-27). For the Avid Angler RME scenario, the daily fish ingestion rates reflect the 95th percentile value (3.73 g/kg-d, or 261 g/day) for consumer only intake of fish in the western United States (EPA/600/P-95/002Fb, Table 13-27). This RME ingestion rate is almost 10 times the CTE ingestion rate for this receptor. Fish ingestion rates for the Avid Angler scenario (CT and RME) incorporate an 11% preparation loss during cooking, as recommended by EPA/600/P-95/002Fb. Age-specific fish ingestion rates are then calculated by multiplying the estimated rate (mg/kg-day) by the appropriate body weight for the receptor.

The Avid Angler fish consumption rates used in this HHRA were designed to be consistent with those used in the RCBRA. However, recent EPA, Ecology, and Oregon Department of Environmental Quality (DEQ) guidance has been reviewed to determine whether the fish ingestion rates are consistent with updated guidelines. These documents include the following:

- EPA/600/R-090/052F, *Exposure Factors Handbook*: *2011 Edition*
- Ecology, 2012, *Focus on Fish Consumption Rates: Reducing Toxics in Fish, Sediments and Water*, Ecology Publication Number 12-10-005
- OAR 340-041-0033, "Toxic Substances," Oregon Administrative Rules, Chapter 340, Division 41, Rule 0033.

Table 10-1 of the 2011 EPA *Exposure Factors Handbook* (EPA/600/R-090/052F) values for the general US population are higher for the CTE condition (45.5 g/d, or 0.65 g/kg-day) and lower for the RME (147 g/d, or 2.1 kg-g/day) condition. The Ecology preliminary fish consumption rate range is 157 to 267 g/day. The CTE value used in the HHRA is lower than the lower end of the Ecology range, but the RME in the HHRA is very close to the upper end of the range. The Oregon DEQ has also recently revised the fish ingestion rate (175 g/day) used in deriving their water quality standard development (i.e., OAR 340-041-0033). This rate is less than the RME value used in the HHRA, but higher than the CTE value.

These results suggest that the CTE value in the HHRA may potentially underestimate fish consumption exposure for general fish consumption. However, the HHRA assumes that 100% of the fish consumed originates from the study area (which is likely a very conservative estimate) whereas both the EPA and Ecology values reflect fish consumption from a variety of sources, both recreational and commercial, for the general population. The EPA (Table 10-5 of EPA/600/R-090/052F, *Exposure Factors Handbook*) also provides mean and 95th percentile rates of 10 and 42 g/day for the state of Washington, based on recreational consumption of fish, suggesting that the values used in the HHRA are highly conservative.

The daily fish ingestion rates for members of the Yakama Nation (both children and adults) are those recommended in Ridolfi (2007). These values reflect upper-bound values for the U.S. population for each age range and equate to approximately 363 and 519 g/day for the child and adult receptors, respectively.

4.3.2.4 Dermal Contact with Soil or Sediment. Absorption of a COPC via dermal contact with soil or sediment is calculated using the following equation:

$$ADD/LADD\ (mg/kg\text{-}day) = (EPC_s * SA_s * AF * EF * ED * ABS_d * C1)/(BW * AT)$$

where:

EPC_s = soil or sediment EPC (mg/kg)
SA_s = skin surface area for soil/sediment exposures (cm^2)
AF = skin-soil adherence factor (mg/cm^2-day)

EF = exposure frequency (days/yr)
ED = exposure duration (yr)
ABS_d = dermal absorption fraction (unitless)
BW = body weight (kg)
AT = averaging time (days)
C1 = units conversion factor (1E-06 kg/mg).

Exposure frequency and duration, body weights, and averaging times are the same values as those used to assess sediment and soil ingestion, as previously described in Section 4.2.3.1. Dermal absorption of radionuclides from soil/sediment is not considered a significant pathway by EPA (EPA540/1-89-002); instead, external radiation from a ground source is evaluated, as further described in Section 4.3.2.7.

4.3.2.4.1 Skin Surface Area and Soil/Sediment Adherence. Parameters unique to the dermal pathway include skin surface area and soil/sediment adherence factors (AF_s).

The soil and sediment skin surface area (SA_s) value for all receptors assumes sediment or soil will contact exposed skin on the face, hands, forearms, lower legs, and feet (EPA/600/P-95/002Fa). These are the areas of the body assumed to be exposed (not covered by clothing) and in contact with island soils or near-shore sediments. Values are calculated according to age of receptor. The EPA-recommended residential SA_s of 2,800 cm^2 and 5,700 cm^2 were used to evaluate soil and sediment dermal contact for children and adults (respectively) in all scenarios (EPA/540/R-99/005, Exhibit 3-5). These values reflect the 50th percentile value for males and females, within the specific age group, for exposed skin on the face, hands, forearms, lower legs, and feet. Because the youth angler receptor encompasses a different age range than those reflected in the EPA default values, a SA_s of 4,015 cm^2 was calculated for this receptor. This value is the mean SA_s based on 50^{th} percentile values for males and females 7 <15 years old, for exposed skin on the face, hands, forearms, lower legs, and feet (EPA/600/P-95/002Fa, Table 6-4). Skin surface area values for CTE were also used for RME, consistent with EPA guidance (EPA/540/R-99/005).

Adherence factors describe the amount of soil or sediment that adheres to the skin following contact. Adherence factors vary depending on the activity as well as soil type; adherence would be expected to be higher, for example, for high-intensity activities (such as playing in the dirt) than for passive recreational activities such as walking through a park. Adherence factors are also weighted according to the area of skin exposed.

The soil adherence factors for the Casual User child (0.04 mg/cm^2-CTE; 0.2 mg/cm^2-RME) and adult (0.01 mg/cm^2-CTE and 0.07 mg/cm^2-RME) used in this HHRA are the recommended AFs for residential settings (EPA/540/R-99/005, Exhibit 3-5).

Because it is anticipated that a greater amount of sediment would adhere to skin, relative to (presumably drier) upland soil, higher AFs were used to evaluate sediment exposures. The sediment AF of 0.2 mg/cm^2 for the Casual User child, Yakama Nation child, and Avid Angler youth reflects the geometric mean AF for "children playing in wet soil" (EPA/540/R-99/005, Exhibit C-3). This value was selected for both CTE and RME because the Columbia River

sediments are of a predominantly sandy matrix that does not absorb water like soils with a high organic content. The AF of 0.1 mg/cm^2 for "gardeners" was used for the adult Casual User and Avid Angler receptors (EPA/540/R-99/005, Exhibit C-3). For the Yakama Nation adult receptor, the AF of 0.3 mg/cm^2 for "reed gatherers" was used, since this was believed to be most analogous to the types of culturally based activities in which this receptor may engage (EPA/540/R-99/005, Exhibit C-3). These dermal exposure values were applied to both CTE and RME scenarios.

4.3.2.4.2 Dermal Absorption Fraction. The routes of exposure and the exposure matrices upon which toxicological studies and resultant toxicity values are based are often different from the route of exposure and exposure matrix of a chemical at a particular disposal site. This may result in different absorption rates and efficiencies. The dermal absorption fraction from soil (ABS_d) is used to account for these differences in the absorption of a chemical and allows for quantification of absorbed dose. This assessment relied on chemical-specific dermal ABS_d provided in Exhibit 3-4 of EPA/540/R-99/005. For SVOCs without published ABS_d (e.g., TPH, pesticides), a value of 0.1 (10%) was assumed, in accordance with EPA 2004 guidance (Page 6-1, EPA/540/R-99/005). EPA does not recommend quantifying dermal exposure to constituents lacking ABS_d. Therefore, for VOCs and inorganic COPCs lacking published ABS_d values, absorbed dose was not quantified. Instead, this lack of absorption data and its impact on the HHRA conclusions is discussed in the uncertainty analysis (Section 7.0). Dermal absorption fractions are summarized in Table 4-51.

4.3.2.5 Dermal Contact with Water. For surface water exposures, EPA-recommended equations (EPA/540/R-99/005) were used to estimate dermal absorption of COPCs for each exposure scenario. Dermal absorption from water is a function of the concentration of the COPC, the chemical/physical properties of a COPC, as well as the receptor's exposure time and skin surface area. The general equation used to estimate dermal contact with COPCs in water is as follows:

$$\text{ADD/LADD (mg/kg-day)} = (DA_{event} * SA_w * EF * ED * EV)/(BW * AT)$$

where:

DA_{event} = absorbed dose per event (mg/cm^2-event)
SA_w = skin surface area for water exposures (cm^2)
EF = exposure frequency (days/yr)
ED = exposure duration (yr)
EV = event frequency (1 event/day)
BW = body weight (kg)
AT = averaging time (days).

For radionuclides, dermal absorption from water is generally not a significant exposure pathway relative to other pathways, with the exception of tritiated water vapor (EPA/540/1-89/002, Chapter 10). However, tritium was not identified as a surface water COPC. Therefore, radiation dose and intake from this pathway was not quantified in the HHRA.

As an intermediate step in the calculation of dermal groundwater exposures, the absorbed dose per event (DA_{event}) is first calculated. This value takes into account physical properties of the chemical as well as the exposure time unique to each exposure scenario. The equations used to calculate DA_{event} are those presented in EPA/540/R-99/005 (Equations 3.2, 3.3, and 3.4) and are provided in the dose calculations presented in Appendices F through K. The Kp is a key parameter in estimating dermal absorption of chemicals in water. Kp (cm/hr) represents the permeability of a chemical from an unspecified (aqueous) vehicle (such as groundwater) through the skin. Published literature on experimentally measured or estimated values of Kp were used for constituents in groundwater (EPA/540/R-99/005). Table 4-52 provides a summary of Kp values as well as other chemical-specific constituents used in the calculation of DA_{event}.

Some constituents, such as PCBs and other lipophilic organics, have Kp values outside the effective prediction domain of the model used to estimate this parameter. In such instances, EPA guidance suggests applying a fraction absorbed value to the Kp, which accounts for loss of a constituent due to desquamation of the skin (EPA/540/R-99/005). Where a published fraction absorbed value existed and quantitative assessment was recommended by EPA (as indicated in Exhibit B-3 of EPA/540/R-99/005), this value was used to quantify aqueous dermal exposures. The EPA does not recommend quantifying dermal aqueous exposures for PCBs, dioxins, or PAHs (as indicated in Exhibit B-3 of EPA/540/R-99/005). Therefore, dermally absorbed dose was not estimated for COPCs in water that belong to these classes of chemicals.

4.3.2.5.1 Skin Surface Areas. The skin surface area (SA_w) values of 6,600 cm^2 and 18,000 cm^2 used for the child and adult Casual User and Yakama Nation receptors, respectively, reflect the 50th percentile value for males and females, according to age range, for exposed skin on the entire body (EPA/540/R-99/005, Exhibit C-1). The SA_w for the Casual User and Yakama Nation scenarios assumes whole body immersion; thus, it is a total body value. The SAw value for Avid Angler receptor reflects the 50th percentile value for males and females, for exposed skin on the forearms, hands, face, lower legs, and feet (EPA/540/R-99/005, Exhibit C-1), analogous to what was assumed to be exposed to soil or sediment since this receptor spends the vast majority of time per trip on a boat. Youth Avid Angler values are based on the mean SA values for male and female children ages 7<14 years (EPA/540/R-99/005, Exhibit C-1).

Exposure frequency, time and duration, body weights, and averaging times are the same values as those used to assess water ingestion, as described in Section 4.3.2.2.

4.3.2.6 Inhalation of Dust. For inhalation pathways (i.e., inhalation of dust), a time-averaged concentration in air is used to estimate exposure and calculate risk, and so age-specific physiological parameters are not included in the calculation (EPA-540-R-070-002, *Risk Assessment Guidance for Superfund, Volume I: Human Health Evaluation Manual, Part F, Supplemental Guidance for Inhalation Risk Assessment, Final*). The equation for calculating dust inhalation exposure is as follows:

$$\text{ADE or LADE } (mg/m^3) = (EPC_s * ET * EF * ED)/(PEF * AT * C3)$$

where:

EPC_s = soil EPC (mg/kg)
ET = exposure time (hr/day)
EF = exposure frequency (day/yr)
ED = exposure duration (yr)
PEF = particulate emission factor (m^3/kg)
AT = averaging time (days)
C3 = units conversion factor (24 hr/day).

Exposure frequency, time, duration, and averaging times are the same values as those used to assess other pathways, as described in previous sections. The particulate emission factor of 1.36×10^9 m^3/kg is the default value provided in Equation 4-5 of OSWER 9355.4-24, *Supplemental Guidance for Developing Soil Screening Levels for Superfund Sites*.

For radionuclides, inhalation risk uses an inhalation rate specific to the age range evaluated:

$$\text{Radionuclide intake (pCi)} = (EPC_s * InhR * ET * EF * ED)/(PEF * C3 * C4)$$

where:

EPC_s = soil EPC (pCi/g)
InhR = inhalation rate (m^3/d)
ET = exposure time (hr/day)
EF = exposure frequency (day/yr)
ED = exposure duration (yr)
PEF = particulate emission factor (m^3/kg)
C3 = units conversion factor (24 hr/day).
C4 = units conversion factor (0.001 kg/g).

Inhalation of soil-borne (fugitive) dust is assumed to occur while the receptors may be visiting islands. In reality, the majority of island soils are likely covered with some type of vegetation, which would likely limit the amount of (dry) soil that becomes airborne due to wind or other mechanical disturbance. The inhalation rates for the casual user child (7.6 m^3/day) and avid angler youth (14.4 m^3/day) are age-weighted recommended inhalation rates from Table 5-23 of EPA/600/P-95/002Fa. The inhalation rate for casual user and avid angler adults (13.25 m^3/day) is the EPA-recommended inhalation rate for adults (EPA/600/P-95/002Fa). The inhalation rates for Yakama Nation child (16 m^3/day) and adult (26 m^3/day) are the inhalation rates specified in Table 3 of Ridolfi (2007).

4.3.2.7 External Radiation. Dermal absorption of radionuclide COPCs is not a significant exposure pathway for radionuclides; rather, external radiation from a ground source is evaluated. The amount of radiation is a function of a receptor's exposure time. External irradiation is calculated according to the following equation:

$$\text{External radiation exposure (pCi-yr/g)} = EPC_s * ET * EF * ED/ (C3 * C5)$$

where:

EPC_s = soil EPC (pCi/g)
ET = exposure time (hr/day)
EF = exposure frequency (day/yr)
ED = exposure duration (yr)
C3 = units conversion factor (24 hr/day)
C5 = units conversion factor (365 days/yr).

Exposure frequency, time, and duration are the same values as those used to assess other pathways, as described in previous sections.

4.3.3 Calculation of Radiation Dose

The intakes and exposures described in the previous section for radionuclides are used to assess cancer risk. In addition to risk, an annual radionuclide dose was calculated for each receptor. (Note that CERCLA is not a dose-based program; dose is calculated in this HHRA only to be consistent with past Hanford Site risk assessments and in keeping with applicable or relevant and appropriate requirements.)

Radionuclide dose is calculated in a manner similar to intake for radionuclides, although it excludes the exposure duration term and is multiplied by a dose conversion factor (DCF):

$$\text{Internal dose-ingestion pathways (mrem/yr)} = EPC_{s,f,w} * IR_{s,f,w} * EF * DCF_{ing}$$

$$\text{Internal dose-inhalation pathways (mrem/yr)} = (EPC_s * InhR * EF * ET * DCF_{inh})/(PEFC4)$$

$$\text{External irradiation exposure (mrem/yr)} = (EPC_s * ET * EF * DCF_{ext}) / (C3 * C5)$$

where:

EPC = Exposure point concentration in soil or sediment (s; pCi/g), fish (f; pCi/g) or water (w, pCi/L)
$IR_{s,f,w}$ = Ingestion rate of soil, sediment, fish (g/day) or water (L/hour)
ET = exposure time (hr/day)
EF = exposure frequency (day/yr)
Inh = inhalation rate (m^3/hr)
PEF = particulate emission factor (m^3/kg)
DCF_{ing} = dose conversion factor-ingestion (mrem/pCi)
DCF_{inh} = dose conversion factor-inhalation (mrem/pCi)
DCF_{ext} = dose conversion factor-external irradiation (mrem/yr per pCi/g)
C3 = units conversion factor (24 hr/day)
C4 = units conversion factor (0.001 kg/g)
C5 = units conversion factor (365 days/yr).

Dose conversion factors are used to convert a radionuclide concentration (activity per mass or volume) into a radiation dose. Radionuclide DCFs, presented in Table 4-53, are both pathway

(inhalation, ingestion, external irradiation) and age-specific (1 year, 5 years, 10 years, 15 years, and adult). These DCFs were obtained from the DOE's RESidual RADioactivity (RESRAD) software (Version 6.5; October 2009) and are based on values provided in EPA-402-R-93-081, *External Exposure to Radionuclides in Air, Water, and Soil, Federal Guidance Report No. 12*; EPA 402-R-99-001, *Cancer Risk Coefficients for Environmental Exposure to Radionuclides, Federal Guidance Report No. 13*; and International Commission on Radiological Protection (ICRP) Publication 72, *Age-Dependent Doses to the Members of the Public from Intake of Radionuclides Part 5, Compilation of Ingestion and Inhalation Coefficients*. The adult DCFs were used for all adult receptors. For nonadult receptors (child and youth), age-weighted DCFs were calculated from the age-specific DCFs in accordance with EPA/402/R-99/001. However, the child/youth receptors evaluated in this HHRA represent various ages that span different DCFs. Therefore, to reflect the entire age range of each nonadult receptor, ICRP DCFs were age-weighted using the age ranges represented by each DCF category. These weighting factors are summarized in the table below.

Dose Conversion Factor Age Group and Corresponding Age Range	Child 1 to 7 Years Weighting Factor	Youth 7 to 14 Years Weighting Factor
Infant, 1: Ages 1 to 2	1	0
Child, 5: Ages 2 to 7	5	0
Older Child, 10: Ages 7 to 12	0	5
Teen, 15: Ages 12 to 14	0	2

For external irradiation, an effective dose equivalent is calculated. For ingestion and inhalation pathways, a committed effective dose equivalent is calculated. Dose is calculated for each pathway and COPC and summed to generate a cumulative annual dose per receptor. This cumulative annual radiation dose, termed total effective dose equivalent (TEDE), is then compared to an annual dose limit, as further discussed in Section 6.3. Calculation of radiation dose is provided in Appendices F through L; radiation dose results are discussed in Section 6.0.

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5.0 TOXICITY ASSESSMENT

5.1 INTRODUCTION

The toxicity assessment describes the relationship between the level of exposure and the likelihood and/or severity of an adverse effect. In other words, the toxicity assessment quantifies the toxicity of each COPC using information obtained from published literature describing epidemiologic or toxicological studies. The products of the toxicity assessment are the toxicity values used to predict the likelihood of adverse health effects in identified receptors at site-specific exposure levels.

Toxicity information used in the HHRA was obtained for carcinogenic (i.e., cancer-causing) and/or noncarcinogenic (i.e., systemic) effects. For each of the COPCs, toxicity values for the relevant exposure periods (i.e., chronic and/or lifetime) were selected according to the following hierarchy of references, as recommended by EPA (OSWER 9285.7-53, "Human Health Toxicity Values in Superfund Risk Assessments") (see complete listing in Section 5.5):

- Tier 1: EPA Integrated Risk Information System (IRIS) On-Line Database (EPA 2012)
- Tier 2: EPA Provisional Peer Reviewed Toxicity Values, as provided by the EPA Superfund Health Risk Technical Support Center
- Tier 3: Other sources, including EPA/540/R-97/036, *Health Effects Assessment Summary Tables, FY 1997 Update*; California Environmental Protection Agency (CalEPA 2011); Agency for Toxic Substance Disease Registry (ATSDR); and other EPA regional and state hazardous waste site programs.

Radionuclide cancer slope factors (CSFs) were obtained from the Health Effects Assessment Summary Table (HEAST): Radionuclides (EPA 2001) and EPA 402-R-99-001, *Cancer Risk Coefficients for Environmental Exposure to Radionuclides, Federal Guidance Report No. 13.*

Tables 5-1 through 5-6 summarize the toxicological values used in this HHRA. In these tables, the sources of toxicological information for each COPC have been documented. If no toxicity information was available for a particular COPC, a structurally similar compound was identified as a surrogate for that COPC, as appropriate, and the surrogate's toxicity values were used to quantify risks. Where appropriate surrogate compounds were not identified, risks for that particular constituent were not quantified but rather addressed qualitatively in the uncertainty analysis (refer to Section 7.0). Toxicity values were available for most COPCs. The uncertainties associated with the toxicity values and surrogates employed in this risk assessment are further discussed in the uncertainty analysis (Section 7.0).

Toxicity information is divided into three major categories: (1) toxicity data associated with threshold (noncarcinogenic) effects, (2) toxicity data concerning carcinogenicity, and (3) the

absorption adjustment factors used to relate toxicity information identified from the literature to the exposure pathways evaluated for the Hanford Site.

5.2 TOXICITY CRITERIA FOR NONCARCINOGENIC EFFECTS

Noncarcinogenic effects, such as organ damage or reproductive effects, are evaluated by reference doses (RfDs) or RfCs. Reference doses and RfCs are values developed by EPA or other entities and are based upon the assumption that there exists a threshold dose or concentration below which there will be minimal risk, if any, for adverse health effects. These values provide a benchmark for the daily dose (or concentration) to which humans may be subjected without an appreciable risk of deleterious effects during a given period of exposure. Reference doses and RfCs also incorporate modifying and/or uncertainty factors to ensure they are protective even for sensitive subpopulations.

Reference doses for oral and dermal exposure are presented in units of milligrams of contaminant per kilogram body weight per day (mg/kg-day), and RfCs for inhalation exposure are presented in milligrams of contaminant per cubic meter of air (mg/m^3). The chronic RfD and RfCs are conservative estimates of concentrations below which no adverse noncancer effects are expected to occur over long periods of exposure. Subchronic RfDs and RfCs are designed to be protective of shorter duration exposures ranging from days to less than or equal to 7 years. For this evaluation, chronic RfDs/RfCs were used for each receptor, regardless of the exposure period. Use of the chronic value is a conservative approach for receptors with shorter term exposures, such as children or occasional recreational users.

Medium-specific RfDs for food, soil, and water are available for manganese. The pathway-specific RfD was applied as appropriate for each pathway evaluated.

Table 5-1 provides a summary of the oral RfDs for each COPC at the Hanford Site. Inhalation RfCs are provided in Table 5-2. These tables also provide information on the species used and critical effects observed in the studies that formed the basis of the RfD or RfC, as well as uncertainty and modification factors that were applied in the derivation of the toxicity value.

5.3 TOXICITY CRITERIA FOR CARCINOGENIC EFFECTS

Previously, the EPA had developed a classification system for constituents based upon the strength of evidence that a constituent is a human carcinogen. The classification system was defined as follows:

- Group A - Human carcinogen
- Group B - Probable human carcinogen
- Group B1 - Limited human data are available
- Group B2 - Sufficient evidence in animals and inadequate or no evidence in humans
- Group C - Possible human carcinogen

- Group D - Not classifiable as to human carcinogenicity
- Group E - Evidence of noncarcinogenicity for humans.

In 2005, EPA identified a new method for classifying carcinogens by a weight-of-evidence narrative (EPA/630/P-03/001F, *Guidelines for Carcinogen Risk Assessment*). Because EPA has not updated the classification system in its IRIS database for all COPCs at this time, the previous weight-of-evidence classification was retained for this report to maintain internal consistency.

The EPA's Carcinogen Assessment Group reviews human, animal, and in vitro data regarding suspected chemical carcinogens and derives oral CSFs and inhalation unit risks (URs) for those chemicals determined to be known, probable, or possible carcinogens (Groups A, B, or C; however, a CSF or UR may not necessarily be derived for all of these known/probable/possible carcinogens). Cancer slope factors are upper-bound estimates of the excess risk of developing cancer as a result of a period of continuous exposure to a chemical averaged throughout the course of a 70-year lifetime and are developed based on the assumption that there is no threshold level of exposure below which adverse effects will not be seen. Cancer slope factors are generally derived using data from animal bioassays, although human data are used when available. The excess carcinogenic risk for an experimental animal is then extrapolated to an expected excess carcinogenic risk for humans. The resulting values are more likely to overestimate than to underestimate the potential risk. A CSF has units of cancer risk per milligrams of chemical per kilogram of body weight per day [1/(mg chemical/kg body weight-day)] or 1/(mg/kg-day). Table 5-3 summarizes oral CSFs for the COPCs identified at the site.

The inhalation UR is the 95% UCL of the mean incremental lifetime cancer risk (ILCR) estimated to result from lifetime exposure to a contaminant if it is in the air at a concentration of 1 $\mu g/m^3$. Inhalation UR values are used in lieu of the chemical's slope factor when an estimate of a lifetime average concentration of the chemical is available. Inhalation UR values are summarized in Table 5-4.

5.3.1 Mutagenic Mode of Action Carcinogens

Cancer risk typically has been associated with aging, resulting from extended exposure durations and prolonged latency periods. However, exposures early in life can also result in the development of cancer. As described in EPA/630/R-03/003F, *Supplemental Guidance for Assessing Susceptibility from Early-Life Exposure to Carcinogens*, evidence suggests that chemicals with a mutagenic mode of action, which would be expected to cause irreversible changes to DNA, would exhibit a greater effect in early-life versus later-life exposures.

For carcinogens that are known to have a mutagenic mode of action, an age-dependent adjustment factor (ADAF) is applied to early-life exposures (EPA/630/R-03/003F). The ADAF accounts for susceptibility differences between early- and later-life exposures and is applied to the CSF or inhalation UR. For children under the age of 2, an ADAF of 10 is applied to cancer toxicity values; for children ages 2 through 15, an ADAF of 3 is applied, in accordance with EPA guidance.

For this site, ADAFs are applicable only to the child and youth scenarios, each of which encompasses age groups younger than 16 years. Mutagenic COPCs identified for the site include hexavalent chromium, which is carcinogenic via the inhalation route of exposure. Therefore, age-weighted ADAFs for child and youth receptors were applied to the inhalation UR for hexavalent chromium in assessing risk due to inhalation of dust.

Supporting calculations for ADAF-adjusted cancer risk estimates are presented along with the risk estimates generated for each exposure scenario, as discussed in the Section 6.0.

5.3.2 Cancer Slope Factors for Radionuclides

Cancer risk related to radionuclide exposure is evaluated using a CSF, which, like the CSF for chemical constituents, represents the average estimate of the lifetime risk of cancer associated with exposure to a specific concentration (or for radionuclides, activity) of a carcinogen in an environmental medium (EPA 402-R-99-001). Cancer slope factors for radionuclides are available for different exposure pathways (ingestion, inhalation) and media (soil, water, food). A radionuclide CSF has units of cancer risk per activity $(pCi)^{-1}$. For the external irradiation pathway, the radionuclide CSF is presented in units of cancer risk per year per picocuries per gram. Radionuclide CSFs, presented in Table 5-5, were obtained from HEAST: Radionuclides (EPA 2001), which are based on values provided in EPA 402-R-99-001.

5.4 DERMAL TOXICITY VALUES

Toxicity values provided by Tier 1, 2, and 3 sources are typically based on an administered (e.g., oral) dose. For dermal exposure pathways (i.e., contact with soil, sediment, or water), the absorbed dose is most relevant; however, the use of oral toxicity values without modification may potentially underestimate the potential risk. Therefore, EPA recommends that oral toxicity values be adjusted where adequate information is available on gastrointestinal absorption efficiency, so that the dermal toxicity values reflect toxicity related to an absorbed rather than administered dose.

Dermal toxicity values were derived from oral RfDs and oral CSFs using the gastrointestinal absorption fraction (ABS) values (ABSgi) and adjustment equations recommended in EPA/540/R-99/005, *Risk Assessment Guidance for Superfund, Volume I: Human Health Evaluation Manual (Part E, Supplemental Guidance for Dermal Risk Assessment), Final.* Where no ABSgi was recommended for a particular COPC, an ABSgi of 100% was assumed (i.e., no adjustment was made), and the dermal RfDs and CSFs are estimated to be the same as the oral values. A summary of ABSgi values and the derivation of dermal toxicity values are presented in Table 5-6.

5.5 CONSTITUENT-SPECIFIC TOXICITY VALUES

The toxicity values used to evaluate several of the COPCs are worth noting individually, as discussed in the following subsections.

5.5.1 Arsenic in Fish Tissue

Environmental samples were analyzed for total arsenic. Additionally, many of the fish tissue samples collected between 2009 and 2010 were analyzed for TIAS. Because the forms of arsenic play a significant role in determining toxicity, it is important to understand the contribution of each chemical form of this COPC.

Arsenic (As) exists in multiple forms in the environment: in inorganic forms as arsenite (As+3) and arsenate (As+5), or in various organic forms. Generally, As+3 is more toxic than As+5, and both inorganic forms are more toxic than organic forms of arsenic, according to the Agency for Toxic Substances and Disease Registry's *Toxicological Profile for Arsenic* (ATSDR 2007).

Once accumulated in organisms such as fish, arsenic is methylated and converted into organic forms such as arsenobetaine, monomethylarsonic acid (MMA), dimethylarsenic acid (DMA), and arsenocholine (collectively referred to as "fish arsenic").

In fish, more than 80% of total arsenic may be in an organic form (Lorenzana et al. 2009). Unlike mercury, which is highly toxic in its methylated form, the methylated forms of arsenic in fish tissue have generally been considered to be relatively nontoxic and are rapidly excreted (ATSDR 2007). However, there are recent studies suggesting that the trivalent form of intermediate arsenic metabolites (specifically, MMA3+ and DMA3+) may be more toxic than inorganic forms (Yamanaka et al. 2004, Klaasen 2008). Although IRIS provides toxicity criteria (RfD and CSF) for total arsenic (based on inorganic forms), none of the Tier 1-3 sources specifies toxicity criteria for organic forms of arsenic; therefore, the risk associated with organic arsenic may potentially be underestimated.

Fish tissue data collected under the RI indicate that TIAS comprises only a very small fraction of total arsenic. As discussed in Section 4.2.2.4.3, the arsenic EPC was calculated using TIAS data when available. For liver/kidney, for which only total arsenic data are available, the assumption that 1% of total arsenic was in inorganic form was based on both literature values and site-specific data. Uncertainties associated with this assumption are further discussed in Section 7.2. The Tier 1 IRIS oral RfD and CSF values were used in conjunction with the TIAS EPCs to evaluate potential noncancer hazard and cancer risk from ingestion of TIAS in fish tissue.

5.5.2 Mercury in Fish Tissue

Mercury enters the environment typically in an inorganic form and is methylated by microorganisms once in soil or sediment. This methylated form is preferentially accumulated by organisms and is considered to be more toxic than inorganic mercury. Site-specific fish tissue data show that methyl mercury comprises nearly all of the total mercury load in fish tissue.

Therefore, the Tier 1 IRIS oral RfD for methyl mercury of 0.0001 mg/kg-day was used to evaluate health risks related to the fish ingestion pathway.

5.5.3 Polychlorinated Biphenyls

Polychlorinated biphenyls in all media were analyzed for either via Aroclor analysis (EPA Method 8082) or via congener analysis (EPA Method 1668). Congener data are available for all media and were preferentially used over Aroclor data in this HHRA, because congener analysis potentially provides a more accurate quantification of total PCB concentrations.

Polychlorinated biphenyls are represented in this HHRA by two individual calculated values: "total dioxin-like PCBs" and "total nondioxin PCBs." Derivation of these values was previously discussed in Section 3.0.

The IRIS RfD and Tier 3 California Environmental Protection Agency RfC, CSF, and inhalation UR values for 2,3,7,8-TCDD were used to evaluate the "total dioxin-like PCBs," whereas total PCB or Aroclor toxicity values presented in IRIS were used to evaluate "total non-dioxin PCBs," as noted in Tables 5-1 through 5-4.

5.5.4 Uranium

A Tier 1 IRIS RfD of 0.003 mg/kg-day (based on soluble uranium salts) is available for uranium (EPA 2012). This value was last revised in IRIS in 1989. However, the RfD for uranium was recently reevaluated under the EPA's Drinking Water Program in support of updating the MCL for this element as demonstrated in *Radionuclides Notice of Data Availability Technical Support Document* (EPA and USGS 2000). This value of 0.0006 mg/kg is five times more stringent than the IRIS value. This revised RfD was used conservatively to evaluate noncancer hazard from uranium exposures.

5.5.5 Medium-Specific Toxicity Values for Cadmium and Manganese

The IRIS provides separate food and water/soil RfDs for cadmium and manganese. These values were applied to the relevant exposure pathways (soil/sediment ingestion, water (incidental) ingestion) as appropriate.

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6.0 RISK CHARACTERIZATION

Characterization of risk to human health is the estimation of the incidence and severity of the adverse effects that may potentially occur in a human population due to chemical and radionuclide exposures, expressed as risk estimates. Risk estimates are based on the comparison of the results generated through integration of the exposure assessment and the toxicity assessment to relevant risk management criteria (e.g., EPA risk limits) and are indicative of the likelihood for adverse effects to occur. The purpose of a risk characterization is to present numerical estimates of risk (of both cancer and noncancer effects) in a context that can be used to make remedial decisions. Additionally, annual radiation dose is presented in this section for consistency with other Hanford Site risk assessments. The results of the risk characterization are used to inform risk management decisions regarding the future need for remedial actions.

Calculation of cumulative cancer risk and noncancer hazard estimates for each receptor and the relevant exposure pathways are summarized and compared to EPA risk limits (and for radionuclides, the annual radiation dose threshold), as described further in the following sections:

- Evaluation of risks under both CTE and RME conditions
- Identification of primary risk drivers (both COPCs in a specific medium and specific areas/locations)
- Discussion of risks associated with Study Area COPCs relative to Reference COPCs.

Evaluation of both the CTE and RME for a particular receptor permits a greater understanding of the potential range of exposures and risks that may occur for a variable population. However, as previously indicated only one condition, based on RME EPCs, was evaluated for the Yakama Nation scenario.

6.1 NONCANCER HAZARD

Exposure to contaminants may potentially affect organ systems and developmental, reproductive, neurobehavioral, and other physiological functions. Unlike potential cancer effects, these effects are assumed to have a threshold (or "safe") dose, below which no effects are expected. The potential for noncarcinogenic health effects is characterized by the HQ, which is the ratio of the estimated ADD (or exposure concentration, for inhalation pathways) and a toxicity value considered to be the level below which adverse health effects would not be observed (i.e., RfD or RfC):

$$HQ = ADD/RfD \text{ (oral, dermal pathways)}$$

$$HQ = ADE/RfC \text{ (inhalation pathways)}$$

To account for exposures that a receptor may receive from multiple chemicals and exposure routes, the cumulative noncancer hazard, known as the hazard index (HI), is calculated as the sum of the chemical-specific HQs, under the global assumption that effects from individual COPCs are additive. As shown in the following two equations, the cumulative HI for a receptor is calculated by summing the route-specific HIs. Route-specific HIs are calculated as the sum of all chemical-specific HQs:

$$\text{Total } HI_{\text{route-specific}} = \sum HQ_{\text{chemical-specific}}$$

$$\text{Cumulative } HI_{\text{receptor}} = \sum HI_{\text{route-specific}}$$

Route-specific HIs may also be broken down further by summing the cumulative risks for each target organ or adverse effect, if warranted, for cumulative HIs exceeding the noncancer hazard threshold.[1] Separate doses/exposures and the resultant HIs are calculated for each receptor age group evaluated (i.e., young child, older child, or adult), since the averaging time over which noncancer effects are assessed is equivalent to the exposure duration, and thus these noncancer hazards are not summed across age groups (EPA/540/1-89/002, *Risk Assessment Guidance for Superfund, Volume 1, Human Health Evaluation Manual [Part A] [Interim Final]*).

The cumulative HI for each receptor age group evaluated is then compared with a noncancer hazard threshold of 1, as per EPA guidance (OSWER Directive 9355.0-30, *Role of the Baseline Risk Assessment in Superfund Remedy Selection Decisions*) and Washington State Department of Ecology Model Toxics Control Act (MTCA) Cleanup Regulations (Ch.173-340-708). If the HI for the RME condition is less than or equal to 1, then it is assumed that chemical concentrations of COPCs do not pose a risk of harm to human health, i.e., there is little concern that potential noncancer health effects will occur as a result of exposure, and that further response actions are not warranted.

For this assessment, in addition to calculating cumulative noncancer hazard (i.e., summed risks for all COPCs across all pathways for each receptor scenario), noncancer hazards attributable to Reference COPCs are segregated from noncancer hazards attributed to Study Area COPCs. Differentiating Reference COPC risks from Study Area COPC risks is consistent with EPA guidance (OSWER 9285.6-07P, *Role of Background in the CERCLA Cleanup Program*) and the approved RI Work Plan (DOE/RL-2008-11), and will be used to focus remedial efforts. Study Area and Reference COPCs were identified and discussed in detail in Section 3.8. Briefly, a Reference COPC is identified as a constituent (chemical or radiological) that is present at concentrations (or activity levels) consistent with or lower than the concentrations (or activity levels) observed in reference/OCI areas, whereas a Study Area COPC is identified as a constituent present at concentrations higher than those observed in reference/OCI areas.

[1] Segregation of noncancer hazard by target organ was not conducted for this HHRA due to the magnitude of threshold exceedances by individual COPCs/pathways. Because hazard from individual COPCs exceeds the threshold of one, there is no benefit to segregation of hazard indices by target organ.

6.2 CANCER RISK

The potential for carcinogenic health effects is characterized as the ILCR. The ILCR represents the incremental probability of an individual developing cancer over a lifetime as a result of exposure to a potential carcinogenic COPC and is calculated for carcinogenic chemicals as well as radioisotopes. For a given constituent, the ILCR is the product of the quantified exposure and the measure of carcinogenic potency (i.e., CSF or UR):

$$\text{ILCR} = \text{LADD} \times \text{CSF (oral and dermal pathways)}$$

$$\text{ILCR} = \text{LADE} \times \text{UR (inhalation pathways)}$$

The ILCR, which represents the probability of developing cancer related to potential exposures to carcinogenic COPCs evaluated in the risk assessment (distinct from the "background incidence" of cancer in the general population), is presented in scientific notation. For example, the ILCR of a specific chemical might be expressed as 1×10^{-6} or one in one million, which means that the probability of an individual developing cancer due to lifetime exposure to that potentially carcinogenic COPC is one in one million.

To account for exposures that a receptor may receive from multiple chemicals and radioisotopes the ILCRs for all COPCs are summed to calculate a route-specific ILCR (e.g., for incidental ingestion of surface water). Analogous to the noncancer hazard described above, the cumulative ILCR for a receptor is then calculated by summing all of the route-specific ILCRs across relevant environmental media for each type of exposure, as demonstrated by the following equations:

$$\text{Total ILCR}_{\text{route-specific}} = \sum \text{ILCR}_{\text{COPC-specific}}$$

$$\text{Cumulative ILCR}_{\text{receptor}} = \sum \text{ILCR}_{\text{route-specific}}$$

Because cancer risk is expressed as a probability averaged over a lifetime of exposure, the cancer risks for each receptor age group evaluated within a scenario (i.e., child and adult) are added together to calculate a cumulative lifetime cancer risk.

The cumulative ILCR for a receptor is compared to EPA's cumulative receptor cancer risk range of 10^{-6} to 10^{-4} (OSWER Directive 9355.0-30). A cumulative risk limit of 1×10^{-5}, which is the midpoint of EPA's target risk range, has been promulgated as risk management criteria by the State of Washington (WAC 173-340-705 [4]). Reasonable maximum exposure cancer risks that fall within or below this low-probability risk range are considered to be "*de minimis*" and essentially nonobservable relative to the background incidence of being eventually diagnosed with cancer in a population (which for the United States is approximately one out of two [or 50%] for all sexes/races [SEER 2010]). Remedial action is generally not warranted for cumulative cancer risks below 10^{-4} (OSWER Directive 9355.0-30). Cumulative cancer risks exceeding the upper end of the target EPA ILCR range, however, may require a risk management decision point to determine if remedial action is warranted.

In Section 6.4, cancer risks attributable to radionuclides are discussed separately from chemical cancer risks, primarily due to differences in development of CSFs (EPA/540/1-89/002). However, cumulative cancer risks across both chemical and radiological COPCs are presented per receptor and compared to the EPA cancer risk range, in accordance with EPA guidance (OSWER 9200.4-18, *Establishment of Cleanup Levels for CERCLA Sites with Radioactive Contamination*).

Similar to the process described above for noncarcinogenic hazard, cumulative cancer risks were also segregated with respect to Reference and Study Area COPCs in order to assess the relative contribution of risk resulting from local conditions within the Columbia River.

The cancer risks presented in Section 6.5 are cumulative over a lifetime. Risks are summed for each age subgroup within a receptor category. Thus, the avid angler carcinogenic risk represents the sum of the child[2], youth, and adult estimated risks, and provides an estimate of risks over a lifetime. Risks for all individual receptors are presented in Appendix F.

6.3 RADIONUCLIDE DOSE

Although not considered a "risk" estimate, cumulative radiation doses were calculated for each receptor/exposure scenario as an additional risk endpoint to be evaluated. Radiation doses for each exposure route (ingestion, inhalation, and external irradiation) and radionuclide COPC were summed to calculate the annual TEDE to an individual. This radiation dose was then compared to a radiation dose threshold of 15 mrem/yr, in accordance with the RI Work Plan (DOE/RL-2008-11).

The origin of this dose threshold was in guidelines published by the EPA for establishing cleanup levels for radionuclides under CERCLA that stated that 15 mrem/yr above background levels should generally be the maximum dose limit for humans (OSWER 9200.4-18). Current EPA policy, however, states that cancer risk be used as a basis for CERCLA cleanup levels rather than radiation dose. The DOE has also published health and safety orders related to identification of a radiation dose threshold, of which DOE Order 5400.5, *Radiation Protection of the Public and the Environment*, is most pertinent. DOE Order 5400.5 requires the reduction of all DOE-source radiation doses to a level as low as reasonably achievable, below a primary dose threshold of 100 mrem/yr above background.

Results of the baseline HHRA are presented by receptor in the following sections.

[2] Child avid angler receptor is evaluated for only fish ingestion, as previously discussed in Section 4.0 of this report.

6.4 QUANTIFICATION OF CARCINOGENIC RISK AND NONCARCINOGENIC HAZARD ESTIMATES

Cumulative risk estimates were calculated for all evaluated receptors, by medium and exposure pathway, and these cumulative noncancer hazard and cancer risk estimates were compared to the relevant EPA risk limits. Both CTE and RME scenarios were evaluated for each receptor (with the exception of the Yakama Nation scenario, for which only one condition was evaluated). Cumulative noncancer HIs were compared to the EPA noncancer threshold of 1; cumulative ILCRs were compared to the EPA target risk range of 10^{-6} to 10^{-4}. The annual TEDE was compared to the radiation dose threshold of 15 mrem/yr. As discussed, the HI of 1 and the ILCR of 10^{-4}, based on the RME condition, are the bases for determining whether remedial actions are required (OSWER Directive 9355.0-30).

Calculation of risk estimates for each receptor and the relevant exposure pathways is provided in Appendices F through L in electronic format. Tables within these appendices (arranged by exposure point) show for each receptor the risk and hazard calculations by exposure route (i.e., dermal contact, ingestion, and dust inhalation) and by exposure medium (i.e., soil, sediment, surface water, and fish tissue). Risk/hazard is then summed across media to derive cumulative risk and hazard. Cumulative cancer risks and noncancer hazards for each receptor, as well as radiation dose, are discussed in the following sections and summarized in Tables 6-1 through 6-84. Table 6-85 presents a comprehensive summary of cumulative noncancer hazard, cancer risk, and radionuclide dose for all receptors and exposure scenarios.

As described in the RI Work Plan (DOE/RL-2008-11), three types of cumulative risks are presented in the risk tables: total risk, Reference risk, and Study Area risk. The "total" risk number reflects risks posed by both "Study Area COPCs," which are those constituents present in media within the Hanford Site Study Area at levels higher than those in reference/OCI areas, and "Reference COPCs," which are those constituents identified in media within the Hanford Site Study Area at levels consistent with or lower than reference/OCI conditions. Section 3.8 discusses the evaluation of Reference concentrations and identification of Study Area and Reference COPCs for each medium.

For each receptor, the COPCs and exposure pathways that contribute to the majority of risk at each exposure point are discussed. For scenarios where cumulative hazard/risk exceeds the EPA risk management criteria (HI of 1 and/or ILCR of 10^{-4}), "risk drivers" are also identified. Risk drivers are those individual COPCs with concentrations resulting in a cumulative noncancer HI greater than 1 or a cumulative ILCR greater than 1×10^{-6}.

6.5 SUMMARY OF CUMULATIVE NONCANCER HAZARD AND CANCER RISK

The following discussion presents the estimated noncancer hazards and cancer risks for all receptors exposed to a portion of the Columbia River identified as the Hanford Site Study Area,

which includes the 100 Area, 300 Area, and Lake Wallula Sub-Areas, located between Vernita Bridge and McNary Dam. The receptors evaluated included the following:

- **Casual User:** This receptor represents a child and adult recreational user exposed to surface water, sediment, and island soil while engaged in various recreational activities (such as swimming, wading, or waterskiing).

- **Avid Angler:** This receptor represents a youth and adult avid angler exposed to surface water, island soil, and sediment while fishing, and a young child, youth, and adult who consume fish brought home.

- **Yakama Nation:** This receptor represents a Yakama Nation child and adult exposed to surface water, island soil, and sediment while engaged in fishing activities and consume fish from the Columbia River.

Section 4.1 presented a detailed description of each of these three scenarios.

As discussed in Section 4.2, each of the three sub-areas (100 Area, 300 Area, and Lake Wallula) was divided into two unique exposure points based on sections of the sub-areas that were identified as having elevated concentrations of certain COPCs. Exposure points within the sub-area are identified as either "A" or "B"; for example, 100-A or 100-B. Across all sub-areas, the "B" designation is used for the exposure point that was identified as having relatively elevated concentrations. Separate noncancer hazard, chemical and radiation cancer risk, and radiation dose was calculated for each of the above human receptors at each exposure point; these calculations are presented in the following appendices:

- Appendix F: 100-A
- Appendix G: 100-B
- Appendix H: 300-A
- Appendix I: 300-B
- Appendix J: LW-A
- Appendix K: LW-B.

An index is presented at the beginning of each of these appendices indicating the sets of tables relevant to each receptor. Tables within each appendix are arranged by receptor scenario (e.g., 100-A child casual RME; 100-A adult casual user RME). Within a particular scenario, tables are presented in which are calculated the following:

- The dose/intake or exposure
- Noncancer hazard and cancer risk by pathway
- Noncancer hazard and cancer risk by medium
- Annual radiation dose by pathway (if applicable)
- Annual radiation dose by medium (if applicable).

Additionally, dose calculations, cancer risk, and noncancer hazard estimates for the consumption of individual fish species for the Avid Angler RME scenario are presented in Appendix L.

For each exposure scenario, summaries of cumulative noncancer hazard, chemical cancer risk, radiation cancer risk, and annual TEDE are presented in Tables 6-1 through 6-84. These tables show the individual pathway risks for the four types of endpoints evaluated (i.e., chemical HI and ILCR, radiological ILCR and annual TEDE). Hazard, risk and dose results are presented by receptor for all exposure points under the RME assumption, followed by results for the CTE assumption, if applicable. Table 6-85 presents a summary of HI, ILCR (chemical and radiological), and annual radiation dose for each receptor and exposure point.

Hazard, risk and annual TEDE are also summarized in the following subsections across the six different exposure points in bar charts. A bar chart is provided for each receptor and scenario (i.e., RME and CTE conditions) for each endpoint evaluated in the HHRA: noncancer hazard, chemical cancer risk, radiation cancer risk, and annual TEDE. When cumulative risk levels exceed risk management criteria, the "risk drivers" (i.e., constituents that comprise the majority of cumulative hazard or risk) are discussed in detail.

Noncancer hazards are presented for the youngest age range evaluated in each scenario (e.g., child or youth), because younger receptors have a relatively higher level of exposure, due to proportionately higher skin surface areas, ingestion, and/or inhalation rates relative to body weight. Therefore, evaluation of noncancer hazard for the younger age groups (e.g., child or youth) is conservatively protective of older age groups (e.g., youth or adult). The calculated HIs for child or youth receptors exceed those calculated for adults, and because of this, only child or youth HI values are presented in this section. However, as stated in the beginning of this section, noncancer HIs were calculated for each receptor age group evaluated. These values are presented in Tables 6-1 through 6-83 for reference.

Cancer risk and radiation dose estimates are presented across the full exposure duration of each scenario; therefore, results for individual receptor age ranges evaluated within an exposure scenario are added together (e.g., child and adult, or child, youth, and adult). Chemical and radiation cancer risks are also summed together to calculate a cumulative ILCR for each receptor.

This section also presents and discusses risks associated with each relevant exposure route (e.g., dermal contact) and medium (soil, sediment, surface water, or fish tissue) and identifies risk drivers, which are the COPCs and exposure pathways that comprise a majority of the cumulative hazard/risk. This HHRA also segregates risks attributable to Reference COPCs (identified in the data evaluation, Section 3.0) from risks attributed to Study Area COPCs. Differentiating Reference COPC risks from Study Area COPC risks aids in focusing remedial action efforts, if warranted.

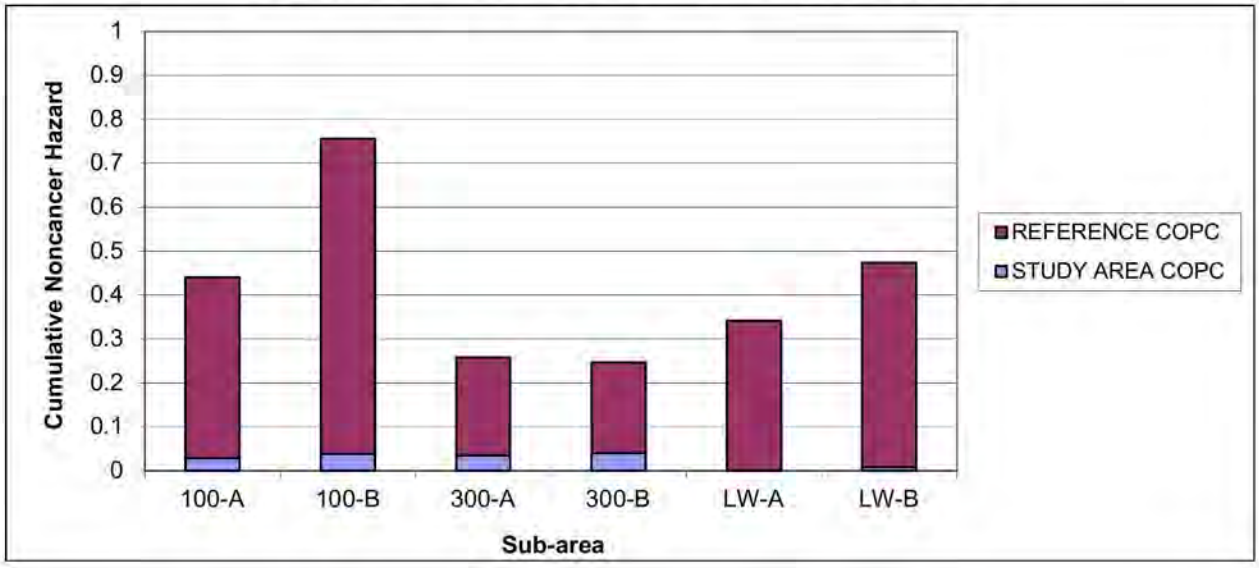
6.5.1 Casual User

The Casual User scenario represents adults and children who frequent the shorelines and islands of the Columbia River for recreational purposes and engage in activities such as swimming, picnicking, wading, and/or waterskiing. Relevant exposure pathways include direct contact with and incidental ingestion of sediment, surface water, and/or island soil. In addition, inhalation of island soil (fugitive dust) was included in this evaluation. As noted, island soil is a relevant exposure medium only in the 100-B, 300-A, and 300-B exposure points. Both RME and CTE scenarios were evaluated for this receptor. Sections 4.1 through 4.3 provide additional details on exposure pathways and parameters unique to the Casual User scenario.

6.5.1.1 Noncancer Hazard. HI calculations for the Casual User RME scenario are provided in Appendices F through L. Cumulative RME HIs are presented for this receptor in Table 6-1 for 100-A, Table 6-2 for 100-B, Table 6-3 for 300-A, Table 6-4 for 300-B, Table 6-5 for LW-A, and Table 6-6 for LW-B. As discussed, noncancer hazards are presented for only the child Casual User, since the HIs for this age range are higher than those estimated for the adult.

Figure 6-1 shows the cumulative noncancer HI for the child Casual User across the six different exposure points for the RME scenario. Noncancer hazard, represented by the HI, account for cumulative exposures across all relevant exposure pathways, media, and COPCs. This figure also shows the contribution from Study Area COPCs and Reference COPCs that were previously identified in Section 3.8.

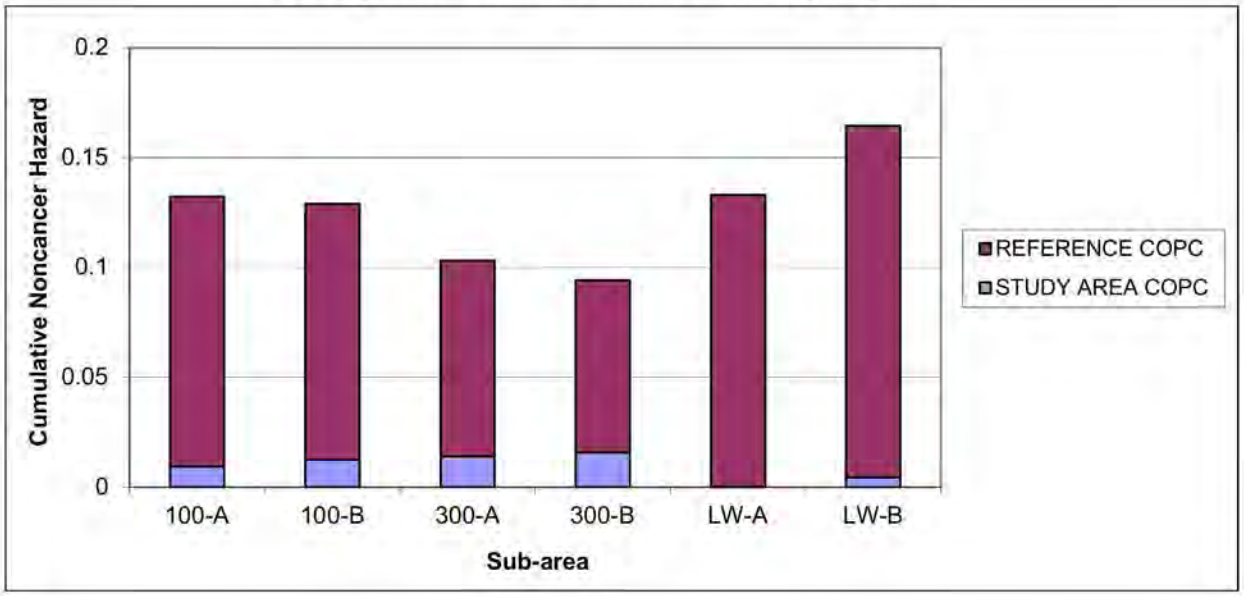
Figure 6-1. Cumulative Noncancer Hazard for the Casual User Child, All Exposure Media: Reasonable Maximum Exposure.



As indicated in Figure 6-1, the cumulative HIs in the 100 Area Sub-Area, 300 Area Sub-Area, and Lake Wallula Sub-Area exposure points are below the noncancer EPA and MTCA hazard threshold of 1, indicating that exposure to COPCs in sediment, island soil, and surface water is unlikely to result in adverse noncancer health effects. Most of the hazard is related to several metals in surface water, sediment, and soil (see Appendices F through L). Note that the metals that contribute most to cumulative hazard, which include arsenic, cobalt, thallium, lithium, and/or iron, are present at levels consistent with those of reference areas (i.e., are Reference COPCs).

Central tendency exposure HIs for the casual user child are summarized in Tables 6-7 through 6-12 for the six exposure points. Figure 6-2 summarizes cumulative noncancer hazard across exposure points.

Figure 6-2. Cumulative Noncancer Hazard for the Casual User Child, All Exposure Media: Central Tendency Exposure.



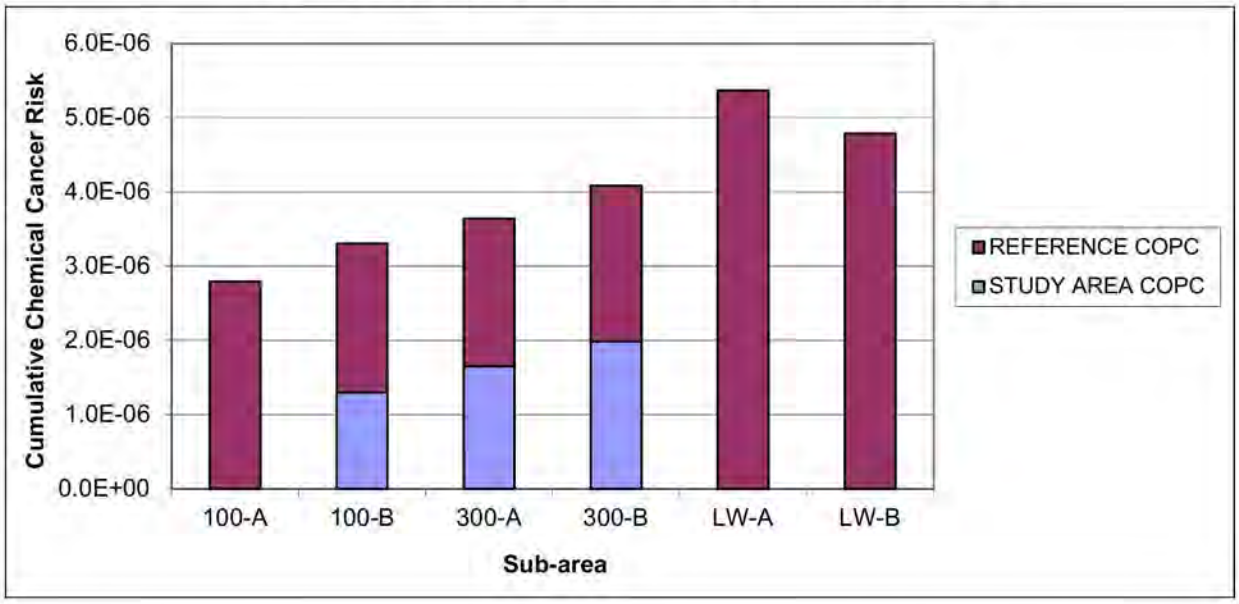
The HIs calculated for the CTE scenario are approximately two to three times lower than those estimated for the RME scenario. None of the HIs estimated for individual exposure points exceeds the noncancer hazard threshold of 1, indicating that exposure to COPCs in impacted media is not anticipated to result in adverse health effects to recreational users. Similar to the RME scenario, reference-related heavy metals in surface water, island soil and sediment are the largest contributors to noncancer hazard in all sub-areas.

6.5.1.2 Cancer Risk. Cumulative ILCRs were calculated for both chemical and radiological COPCs. Although risks from both types of COPCs are ultimately summed together for a given receptor, risks due to chemical and radiological COPCs are presented and discussed separately in

the following subsections. Cumulative cancer risks for chemical and radionuclide COPCs for the Casual User are then presented in Section 6.5.1.2.3.

6.5.1.2.1 Chemical Cancer Risk. Tables 6-1 through 6-6 present cumulative ILCRs at each exposure point for the child Casual User, RME, whereas Tables 6-13 through 6-18 present ILCRs for the adult Casual User, RME. The Casual User scenario, however, reflects multiple age groups, including a child ages 1 through 6 years as well as an adult. Therefore, cumulative lifetime cancer risks for chemical (i.e., nonradiological) COPCs estimated for the adult and child were added together. Figure 6-3 depicts cumulative cancer risk, represented by the ILCR, for the Casual User (all ages, combined) across all exposure points, exposure pathways, and COPCs.

Figure 6-3. Cumulative Chemical Cancer Risk for the Casual User, All Exposure Media: Reasonable Maximum Exposure.



As indicated in Figure 6-3, ILCRs estimated for all exposure points for the RME condition are within the EPA risk range of 10^{-6} to 10^{-4} and below the MTCA cumulative risk limit of 1×10^{-5}. Highest cancer risk is in the LW-A exposure point (Tables 6-5 and 6-17), with nearly all of the risk attributed to arsenic in sediment (in which it is a Reference COPC). Likewise, in the LW-B exposure point (Tables 6-6 and 6-18 for the child and adult), arsenic in both surface water and sediment comprises 100% of cumulative chemical cancer risk (see Appendix G). The levels of arsenic in the Lake Wallula Sub-Area are consistent with Reference conditions.

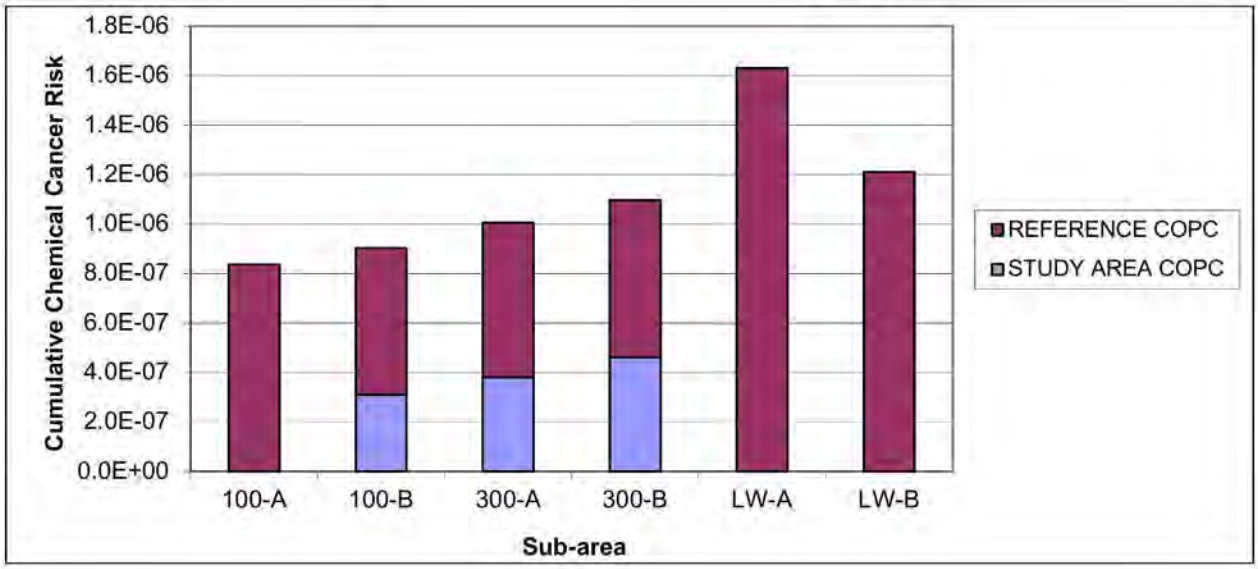
In the 100-A exposure point (Tables 6-1 and 6-13 for the child and adult, respectively; see Appendix F), none of the carcinogenic COPCs is a Study Area COPC. Risks at this exposure point are primarily due to arsenic in sediment and, to a lesser extent, arsenic and PCBs in surface water; this exposure point had the lowest estimated cancer risks. In the 100-B exposure point

(Tables 6-2 and 6-14; see Appendix G), approximately 65% of the cumulative chemical cancer risk is also related to arsenic in surface water and sediment. Arsenic in each of these media is considered reference related. However, the remainder of cumulative cancer risk in the 100-B exposure point is attributable to arsenic in island soil. Arsenic is a Study Area COPC in island soil in this exposure point.

In the 300 Area Sub-Area (both 300-A and 300-B exposure points; see Tables 6-3, 6-4, 6-15, and 6-16), arsenic is also the primary contributor to cumulative risk in surface water, island soil, and sediment for the Casual User scenario (Appendices H and I). Note that, with the exception of the 100 Area and 300 Area Sub-Areas, in which arsenic in island soil is a Study Area COPC, arsenic in other areas and media is reference related. However, none of the cancer risks estimated for this receptor exceeds the EPA cancer risk range or MTCA cumulative cancer risk limit.

The ILCRs for the CTE child Casual User scenario are summarized in Tables 6-7 through 6-12; ILCRs for the CTE adult Casual User scenario are summarized in Tables 6-19 through 6-24. Figure 6-4 depicts cumulative ILCRs across exposure points for the Casual User CTE scenario.

Figure 6-4. Cumulative Chemical Cancer Risk for the Casual User, All Exposure Media: Central Tendency Exposure.

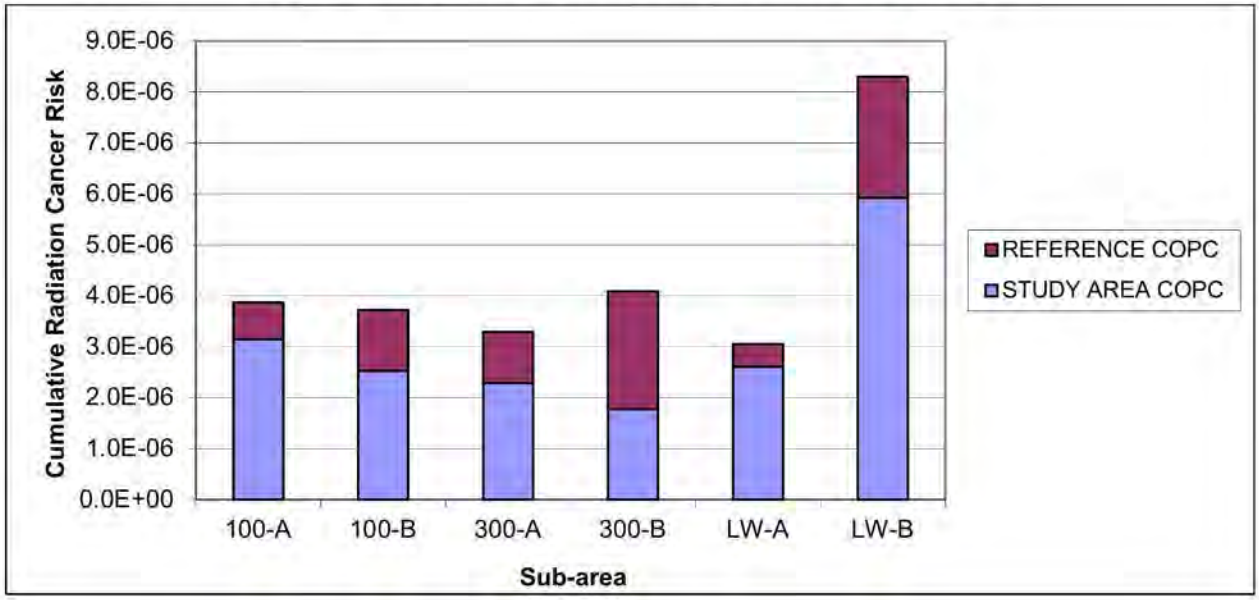


As shown in this figure, the relative magnitude of risks among exposure points and the contribution from Reference COPCs is similar to that of the RME condition, although the risks for CTE are approximately three-fold lower. Most ILCRs are slightly at or above the lower (i.e., more stringent) end of the EPA risk range (10^{-6}). Absent risks from Reference COPCs, the cumulative ILCRs for all exposure points, as related to Study Area COPCs, would fall below the lower end of the EPA risk range (10^{-6}).

6.5.1.2.2 Radiation Cancer Risk. Radiation cancer risks, like chemical cancer risks, are presented as cumulative risk across multiple age groups. Therefore, radiation cancer risks presented for the casual user in this section reflect both the child and adult receptor.

Tables 6-1 through 6-6 and Tables 6-19 through 6-24 present radiation cancer risks calculated for the child and adult Casual User RME scenario. Cumulative radiation cancer risks (adult and child) are summarized in Figure 6-5. All calculated radiation ILCRs across exposure points are within the EPA risk range of 10^{-6} to 10^{-4}.

Figure 6-5. Cumulative Radiation Cancer Risk for the Casual User, All Exposure Media: Reasonable Maximum Exposure.



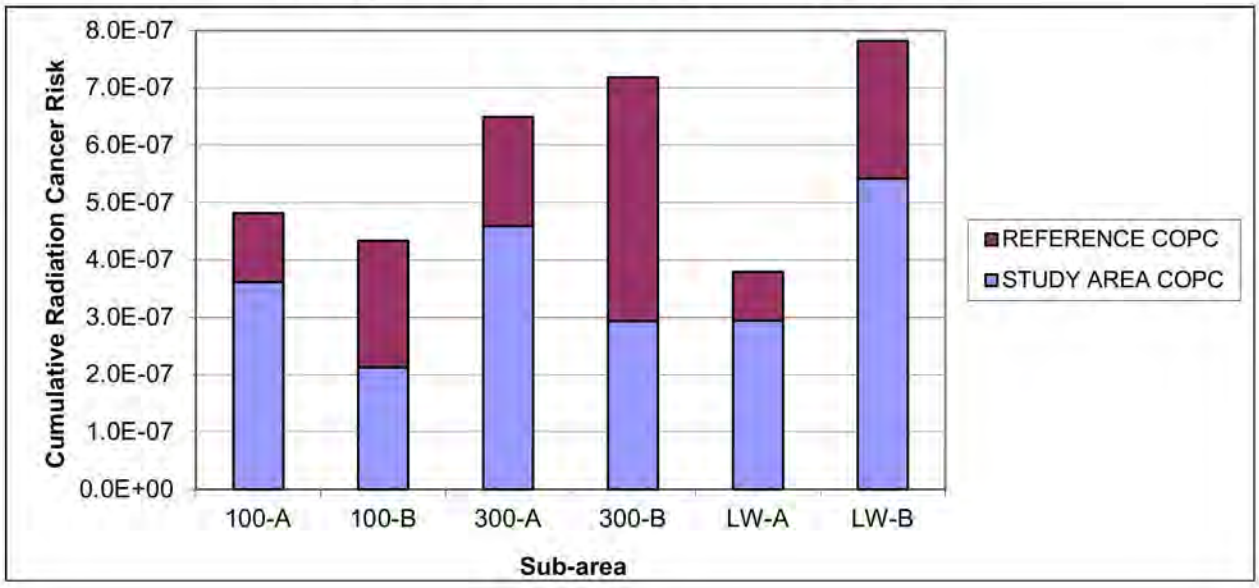
For most of the exposure points (excluding 300-B), Study Area radionuclide COPCs constitute the majority of cumulative radiation cancer risk. In the 100-A exposure point (Tables 6-1 and 6-13 for the child and adult, respectively), cobalt-60 and europium-152 (both Study Area COPCs) in sediment comprise the majority of total risk (Appendix F). These two radionuclides, in addition to cesium-137 (a Reference COPC) in soil, are also primary contributors to risk from sediment within the 100-B exposure point (Tables 6-2 and 6-14 for the child and adult; also see Appendix G).

In the 300-A exposure point, Study Area COPCs cobalt-60 and europium-152 in sediment contribute the most to cumulative radiation cancer risk. In the 300-B exposure point, however, Reference COPCs, in particular cesium-137, entirely comprise cumulative risk in sediment (see Tables 6-3 and 6-4 for the child and Tables 6-15 and 6-16 for the adult; Appendices H and I present risk calculations for 300-A and 300-B). In 300-B island soil (primarily Johnson Island), the Study Area COPCs cobalt-60, europium-152, and strontium-90 also contribute to cumulative cancer risk.

Europium-152 and cobalt-60 in the Lake Wallula Sub-Area (both LW-A and LW-B) are Study Area COPCs in sediment and collectively constitute the majority of the cumulative radiation risk in this sub area. (See Tables 6-5 and 6-6 for the child, and Tables 6-17 and 6-18 for the adult; see Appendices J and K for risk calculations for LW-A and LW-B, respectively.) However, absent all reference-related risks, the risks attributable to the Study Area-related radionuclides in all exposure points are below the upper end of EPA's target risk range (10^{-4}).

Central tendency exposure risks for the casual user are summarized across exposure points in Figure 6-6. Cancer risks for the child are presented in Tables 6-7 through 6-12. Cancer risks for the adult are presented in Tables 6-13 through 6-24. Individual risk calculations for each exposure point are presented in Appendices F through K.

Figure 6-6. Cumulative Radiation Cancer Risk for the Casual User, All Exposure Media: Central Tendency Exposure.



As indicated in Figure 6-6, none of the radiation ILCRs calculated for each exposure point exceeds the lower end of the EPA cancer risk range (10^{-6}). COPCs that contribute to most of the cumulative risk under the CTE scenario are similar to those identified for the RME scenario. In the 100-A and 100-B exposure points, most of the risk is attributed to Study Area COPCs and due to exposure to europium-152 and cobalt-60 in sediment. In the 300 Area and Lake Wallula Sub-Areas, these radionuclides contribute to the majority of risk. Cesium-137, a Reference COPC, also contributes to cumulative risk throughout all exposure points.

6.5.1.2.3 Cumulative Cancer Risk. Although radiation risks are described separately from chemical cancer risks, cumulative overall risk from both radionuclides and chemical cancer risks for the Casual User scenario are presented below to depict overall cancer risk for the Hanford Site Study Area, in accordance with EPA guidance (OSWER 9200.4-18). These cumulative cancer risks (rounded to one significant figure) are presented in the table below for both RME and CTE scenarios.

Cumulative Cancer Risk, Casual User Scenario - RME	100-A	100-B	300-A	300-B	LW-A	LW-B
Total ILCR-RME, all COPCs	7.E-06	7.E-06	7.E-06	8.E-06	8.E-06	1.E-05
Total ILCR-RME, Study Area COPCs	3.E-06	4.E-06	4.E-06	4.E-06	3.E-06	6.E-06

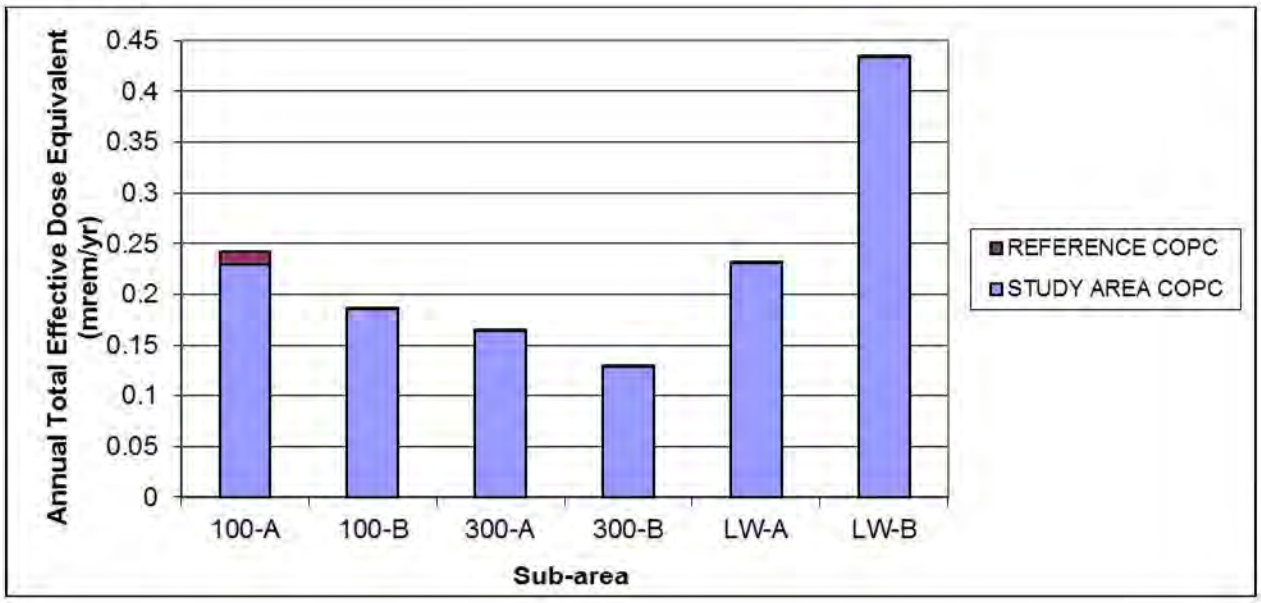
Cumulative Cancer Risk, Casual User Scenario - CTE	100-A	100-B	300-A	300-B	LW-A	LW-B
Total ILCR-CTE, all COPCs	1.E-06	1.E-06	2.E-06	2.E-06	2.E-06	2.E-06
Total ILCR-CTE, Study Area COPCs	4.E-07	5.E-07	9.E-07	8.E-07	3.E-07	5.E-07

Across all exposure points, chemical COPCs comprise the majority of cumulative cancer risk (approximately 55% to 70%). In the 300 Area Sub-Area, chemical and radionuclide COPCs contribute approximately the equivalent amount to total risk. In the LW-A exposure point, approximately 60% of the risk is attributable to chemical COPCs, whereas in LW-B, 60% of the risk is due to radionuclides.

The summed radiation and chemical cancer risks for both the RME and CTE scenario at each exposure point are above the lower/more stringent end of the EPA target risk range (10^{-6}), but do not exceed the upper end of the EPA target range of 10^{-4}. The CTE cumulative cancer risks are either at or slightly exceed the more stringent end of the EPA target risk range (10^{-6}). Cumulative chemical cancer risks do not exceed the MTCA cumulative risk limit of 1×10^{-5}.

6.5.1.3 Annual Total Effective Dose Equivalent. Dose calculations for the Casual User scenario are presented in Appendices F through K, and doses by exposure medium are summarized in Tables 6-1 through 6-6 for the child and Tables 6-13 through 6-18 for the adult. Annual TEDEs for the casual user, RME scenario, across all exposure points are summed in Figure 6-7. These doses were compared to a 15 mrem/yr radiation dose threshold described in Section 6.3.

Figure 6-7. Annual Total Effective Dose Equivalent for the Casual User, All Exposure Media: Reasonable Maximum Exposure.

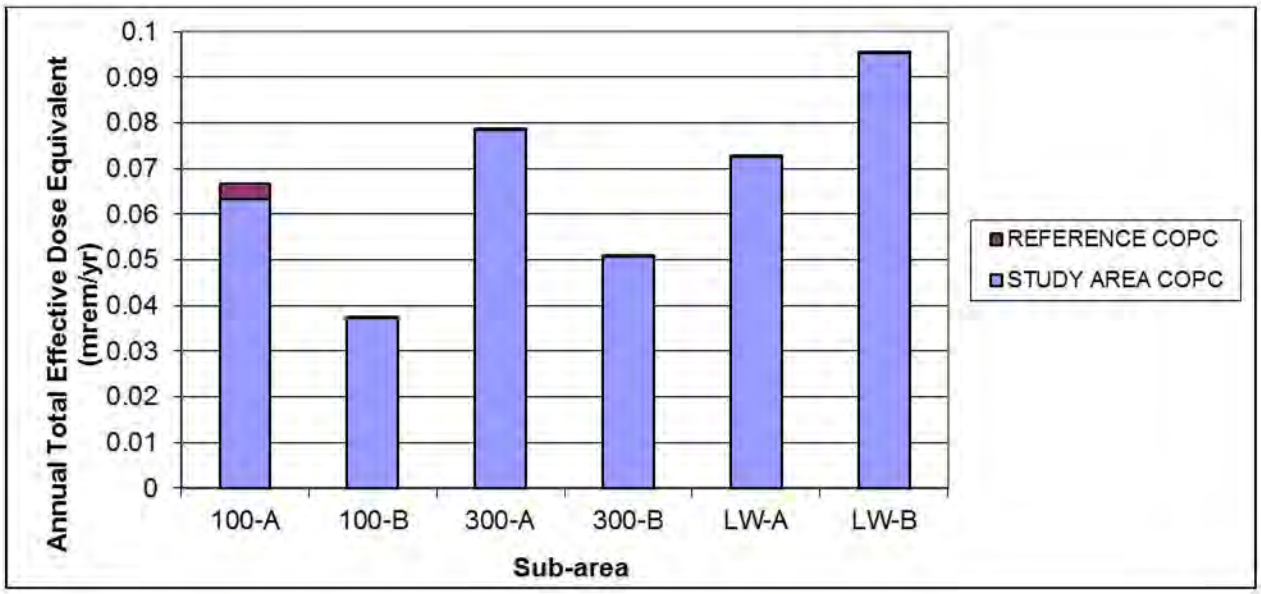


As indicated in Figure 6-7, annual TEDE is well below the 15 mrem/yr dose threshold in all exposure points, with the maximum annual TEDE at LW-B. In all exposure point areas except 300-B, Study Area radionuclide COPCs in sediment (europium-152, cobalt-60, etc.) contribute to the majority of annual TEDE. In 300-B (Johnson Island), europium-152 in island soil (a Study Area COPC) contributes to the majority (approximately 90%) of the annual TEDE, whereas carbon-14 and cesium-137 contribute to the majority of cumulative radiation dose from sediment. Both carbon-14 and cesium-137 are Reference COPCs in sediment.

Radiation doses for the CTE Casual User scenario are presented in Tables 6-7 through 6-12 for the child and Tables 6-19 through 6-24 for the adult. Annual TEDE is summarized across exposure points in Figure 6-8.

Similar to the RME scenario, the annualTEDE for each subarea is well below the radiation dose threshold of 15 mrem/yr, with over half of the dose accounted for by Study Area radionuclide COPCs. Central tendency exposure doses are approximately two to four times lower than those estimated for the RME scenario.

Figure 6-8. Annual Total Effective Dose Equivalent for the Casual User, All Exposure Media: Central Tendency Exposure.



6.5.2 Avid Angler

The Avid Angler receptor is intended to represent individuals who use the Hanford Site Study Area for fishing and who subsequently bring home and consume their catch. Risks were evaluated for three age ranges: adult, youth (age 7 through 13 years), and child (age 1 through 6 years). Both the adults and youths were assumed to be exposed to surface water, sediment, and island soil (where present) in each of the sub-areas, as well as to consume fish caught in these areas. The (young) child of the Avid Angler is assumed not to engage in fishing activities but to only consume the fish that is caught and is thus not exposed to the other (abiotic) media. The adult and youth receptors are assumed to be on the river the same number of days per year and hours per day; thus, the difference in risks is due to the different exposure parameters related to body size and intake rates.

As discussed, the Avid Angler scenario evaluated risk from fish consumption using two approaches. The first approach evaluated risk/hazard assuming that all six fish species (for which analytical results are available) were consumed. The second approach evaluated risk/hazard for the six individual fish species. Results of both approaches are discussed in this section.

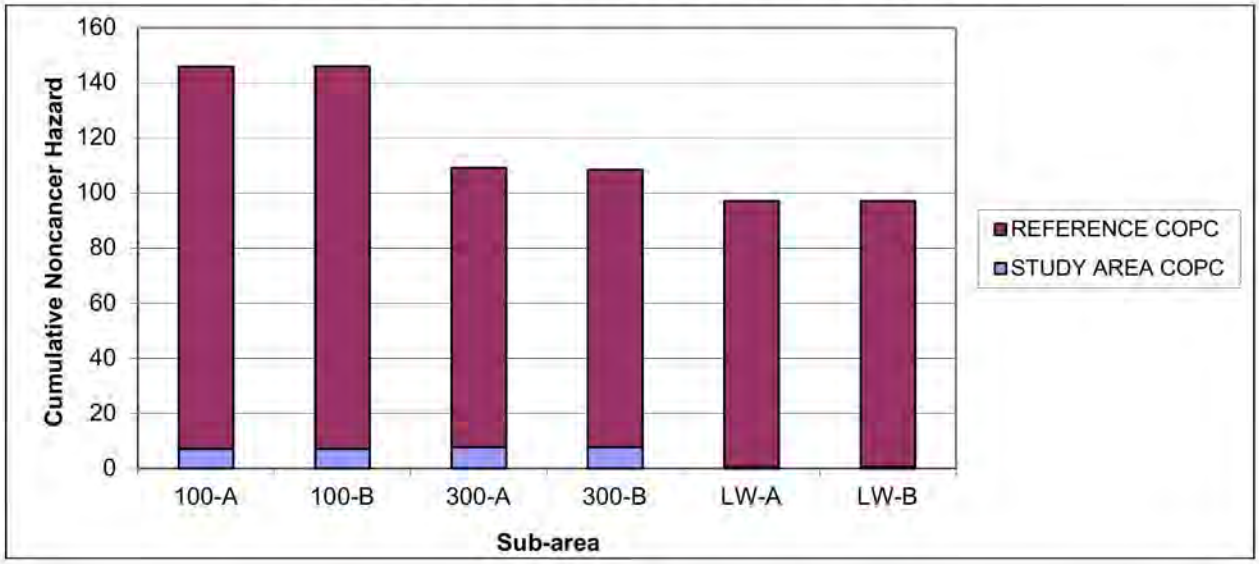
Both RME and CTE scenarios were evaluated for this receptor. Sections 4.1 through 4.3 provide additional details on exposure pathways and parameters unique to the Avid Angler scenario. Tables 6-25 through 6-27 present noncancer hazard, chemical, and radiation cancer risk and annual TEDE estimates for the child. Results for the Avid Angler youth are presented in Tables 6-28 through 6-33. Adult Avid Angler results are presented in Tables 6-34 through 6-39.

The CTE summary risk summary tables for each receptor/exposure point follow the same order as that for the RME and are presented in Tables 6-40 through 6-54.

6.5.2.1 Noncancer Hazard. For noncarcinogenic hazards associated with exposure to all media, the youth has the highest exposure potential due to the lower body weight relative to other exposure parameters (such as intake rates). Therefore, noncancer hazard is presented in this section for only the youth Avid Angler instead of the adult receptor. Additionally, noncancer hazards are presented for the younger child who is only exposed via fish ingestion, since this receptor is assumed to not be exposed to abiotic media.

The cumulative RME noncancer hazard for the Avid Angler youth receptor is shown in the following chart (Figure 6-9). Noncancer hazard is presented in Tables 6-28 through 6-33 for each of the different exposure points.

Figure 6-9. Cumulative Noncancer Hazard for the Avid Angler Youth, All Exposure Media: Reasonable Maximum Exposure.



The cumulative HIs across all exposure points exceed the risk management criteria of 1 by a factor of 10 to 100. Reference COPCs account for the vast majority of total hazard for all exposure points, and the fish ingestion pathway (all species, combined) accounts for approximately 99% of the HI.

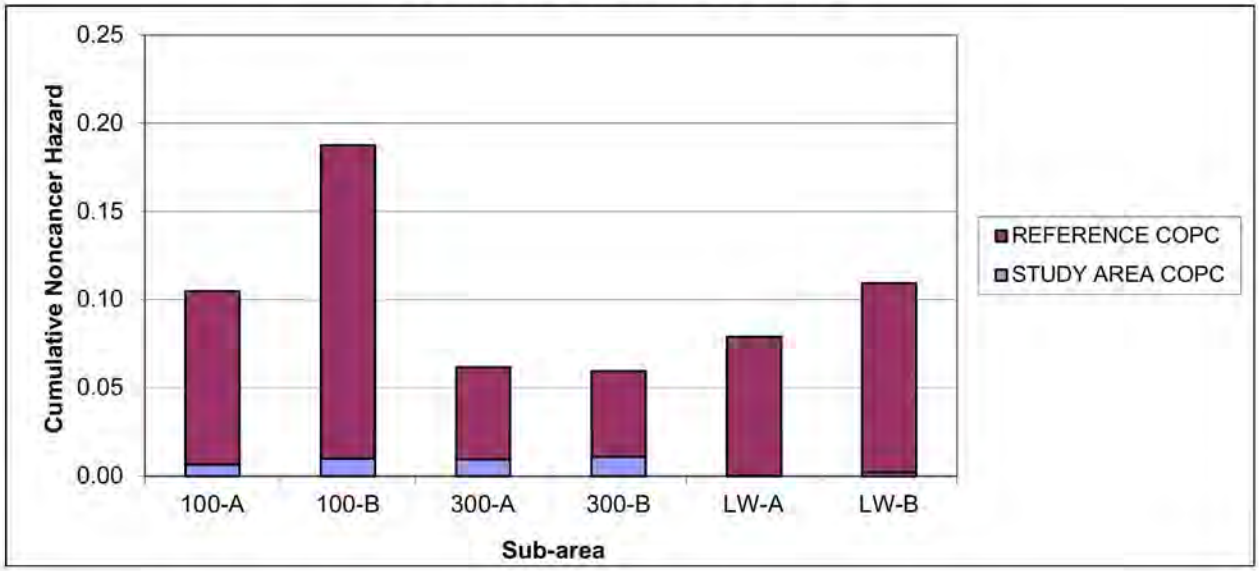
As discussed in Section 4.0, due to the mobile nature of fish, the fish tissue EPC is calculated for an entire sub-area (i.e., 100 Area) rather than an individual exposure point within the sub-area; thus, there is no difference in the HI between the "A" and "B" exposure points for this pathway, which is why the cumulative noncancer HIs are virtually the same within each sub-area. In all sub-areas, the COPCs that are the primary risk drivers, accounting for the vast majority

(over 98%) of the HI from fish ingestion, include PCBs, beta-HCH, dieldrin, cobalt, and lithium, which are all Reference COPCs, and mercury, a Study Area COPC for the 100 Area sub-area. Approximately 60-80% of the cumulative HI results from ingestion of PCBs (both dioxin and non-dioxin-like) in fish tissue. Across all three sub-areas, dioxin-like PCBs contribute the most hazard (HQ of 64, or 44% for the 100 Area, HQ of 47, or 43% for the 300 Area, and HQ of 34, or 35% for Lake Wallula).

The other COPC that comprises a substantial fraction of cumulative hazard is cobalt, which results in a HQ of 20 in the 100 Area, 13 in the 300 Area and 25 in Lake Wallula. Cobalt is a Reference COPC across all three subareas.

The picture for the youth and adult Avid Angler receptor changes dramatically when the abiotic (i.e., sediment, soil, and surface water) exposure pathways are examined separately from fish ingestion, as shown in Figure 6-10 (note differences in scale relative to Figure 6-9).

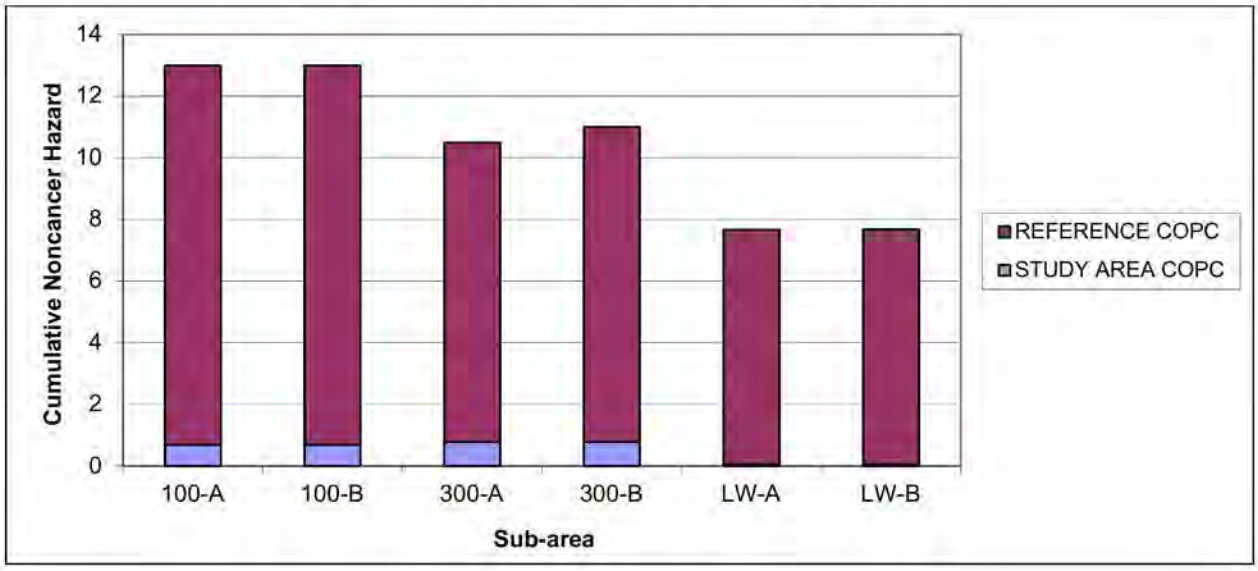
Figure 6-10. Cumulative Noncancer Hazard for the Avid Angler Youth, All Exposure Media Excluding Fish Tissue: Reasonable Maximum Exposure.



The cumulative HIs decrease by up to two orders of magnitude when fish ingestion is excluded, to a HI of approximately 0.1 or less across all sub-areas.

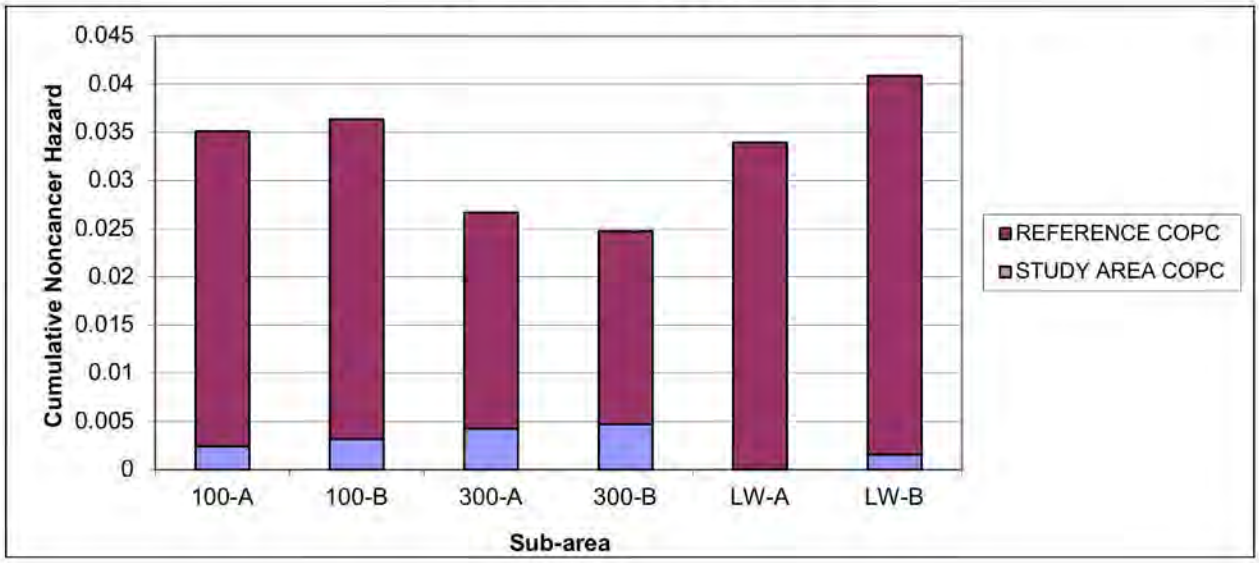
The CTE cumulative HIs for the Avid Angler youth for all media are shown in Figure 6-11. (See risk result Tables 6-43 through 6-48 for the Avid Angler youth receptor.) As with the RME assumption, noncancer hazard is greater than 1 in all sub-areas and attributable almost entirely to fish ingestion; however, the magnitude of the exceedance is much less. The greatest total CTE HI is approximately 13 for the 100-A and 100-B exposure points, roughly an order of magnitude less than the HI calculated under the RME assumption for this area. Fish tissue again accounts for 99% of the HI (see Tables 6-28 through 6-33) and PCBs (dioxin-like and nondioxin; both Reference COPCs in the 100 and 300 Areas) again account for approximately 80% of the HI. These two constituents are the only COPCs that result in a HQ greater than 1.

Figure 6-11. Cumulative Noncancer Hazard for the Avid Angler Youth, All Exposure Media: Central Tendency Exposure.



When pathways other than fish are reviewed under the CTE assumption, no HIs exceed the noncancer hazard threshold of 1 in any sub-area, as shown in Figure 6-12. These results indicate that exposure to COPCs in abiotic media is not anticipated to cause adverse noncancer health effects.

Figure 6-12. Cumulative Noncancer Hazard for the Avid Angler Youth, All Exposure Media Excluding Fish Tissue: Central Tendency Exposure.

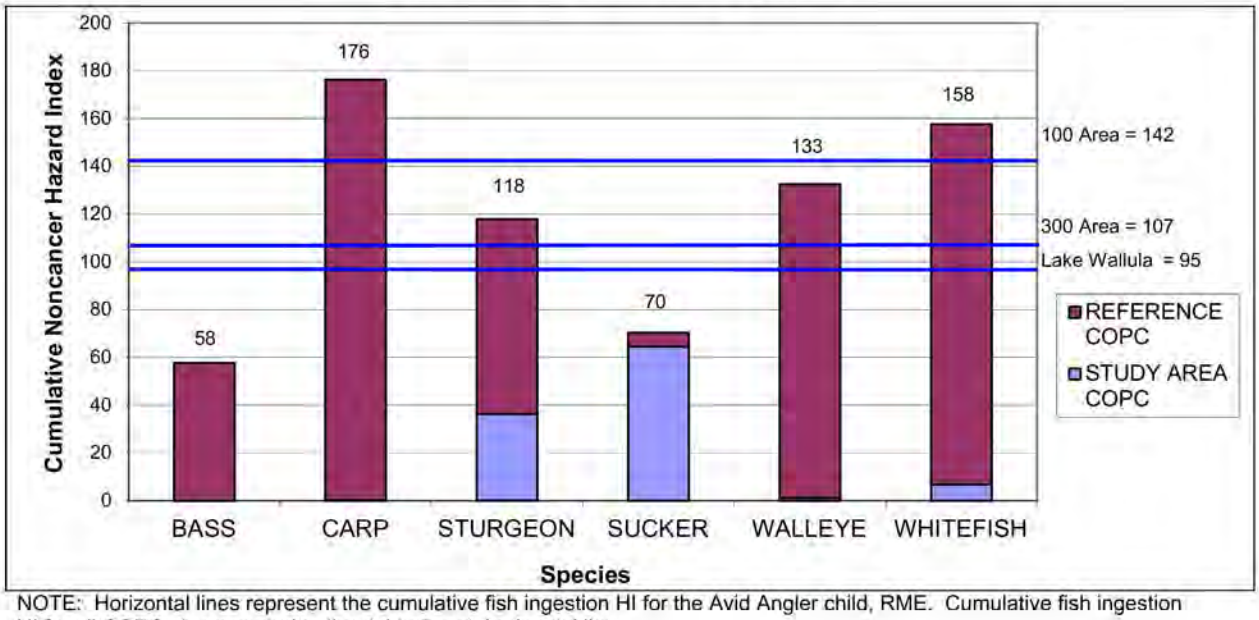


As seen in Figures 6-9 through 6-12, the fish ingestion pathway comprises the vast majority of cumulative noncancer hazard. As discussed, the hazards estimated for this pathway reflect consumption of all six fish species (combined) evaluated in this HHRA.

As previously discussed, a second approach was used to evaluate risk from the fish consumption pathway. Noncancer hazards were evaluated for each individual fish species to understand interspecies variability and corresponding hazards associated with consumption of a particular species. Fish EPCs for individual species are based on analytical results from the Hanford Site Study Area (i.e., 100 Area, 300 Area, and Lake Wallula Sub-Areas combined), as discussed previously in Section 4.2.2.4.

Results of this analysis are presented in Tables 6-25 through 6-27 for all exposure points and summarized in Figure 6-13. The cumulative noncancer hazards for each species (for the entire Hanford Site Study Area), are presented in this figure, represented by vertical bars. Also included in this figure are the cumulative fish ingestion HIs estimated for all species (combined) for each sub-area for the child Avid Angler, as represented by horizontal lines (see Tables 6-25 through 6-27).

Figure 6-13. Cumulative Noncancer Hazard for the Avid Angler Child for Consumption of Individual Fish Species: Reasonable Maximum Exposure.



NOTE: Horizontal lines represent the cumulative fish ingestion HI for the Avid Angler child, RME. Cumulative fish ingestion HI for all COPCs is presented to the right of each horizontal line.

COPC = contaminant of potential concern
HI = hazard index
RME = reasonable maximum exposure

Cumulative HI is lowest for bass and highest for carp, as shown in Figure 6-13, and exceeds the HI threshold of one for all species. Relative to the hazards estimated for consumption of all species in each of the sub-areas, results for individual species were varied (HI of 58 to 176), with three of the six species resulting in hazard within or above the range of HIs estimated for the combined species analysis (in which the HIs ranged from 95 to 142). These results suggest that the hazards estimated for consumption of all species combined may potentially underestimate hazard associated with ingestion of certain individual species such as carp or whitefish, but may also overestimate the hazard for other species such as bass, sturgeon, or sucker.

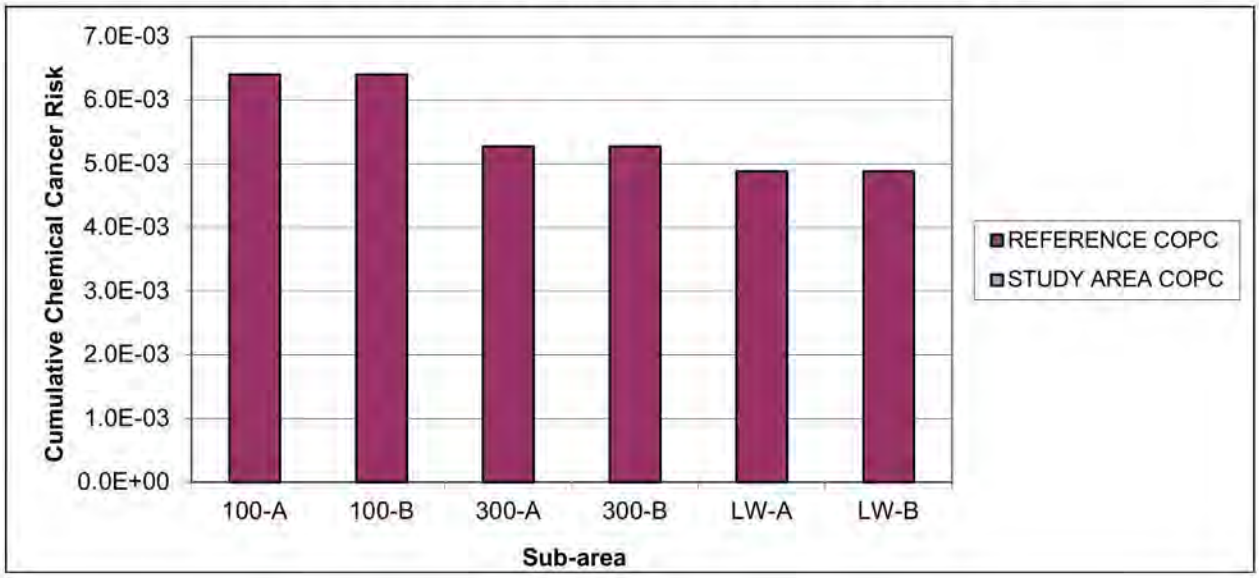
Polychlorinated biphenyls, chlorinated pesticides (notably, dieldrin), cobalt, lithium and mercury were risk drivers across all species. Most of these constituents are Reference COPCs across the six species. Of the Study Area risk drivers, cobalt and mercury contribute to a HI of 37 in sturgeon. In sucker, primary Study Area risk drivers include PCBs (HI of 64). In whitefish, antimony, dieldrin and mercury are the primary Study Area risk drivers.

As seen for the combined species analysis, PCBs account for the majority of cumulative hazard, with the dioxin-like PCBs comprising most of the hazard from all PCB congeners, except in whitefish. This species was the only fish species that had a higher HQ for non-dioxin-like PCBs (71 versus a HQ of 67 for dioxin-like PCBs). Hazard quotients for dioxin-like PCBs range from 23 in bass (versus a total PCB HQ of 40) to 82 in carp (versus a total PCB HQ of 142).

6.5.2.2 Cancer Risk.

6.5.2.2.1 Chemical Cancer Risk. The cumulative lifetime chemical cancer risks for the Avid Angler (sum of child, youth, and adult ILCRs) are presented in Figure 6-14. (For a summary of risk results, see Tables 6-25 through 6-39.)

Figure 6-14. Cumulative Chemical Cancer Risk for the Avid Angler, All Exposure Media: Reasonable Maximum Exposure.

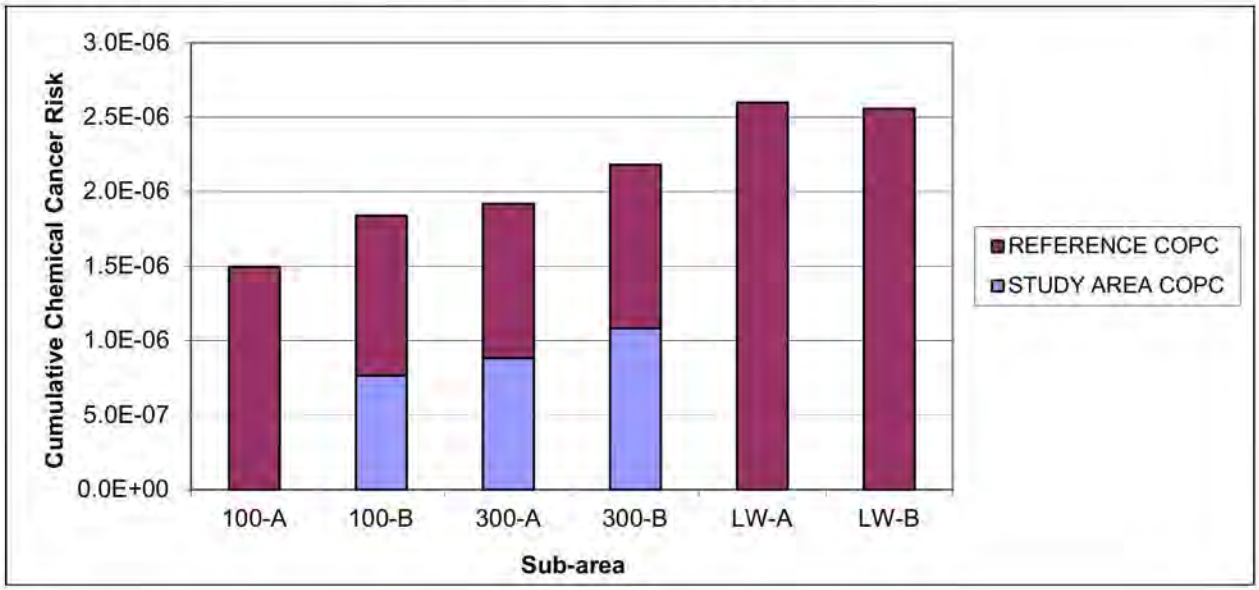


Across all exposure locations, the EPA upper risk value of 10^{-4} is exceeded by an order of magnitude. The relative proportion of total risk from Reference COPCs follows the same pattern as was seen for the noncancer HIs. Specifically, Reference COPCs account for the vast majority of risk across all exposure points, with the fish ingestion pathway accounting for over 99% of the cumulative ILCR.

Polychlorinated biphenyls, chlorinated pesticides, and arsenic are the COPCs that are the cancer risk drivers for the fish ingestion pathway across all sub-areas; these constituents are Reference COPCs across all three sub-areas. Polychlorinated biphenyls collectively account for approximately 70% of the total cancer risk from fish in the 100 and 300 Area Sub-Areas and approximately 60% in the Lake Wallula Sub-Area. Dioxin-like PCBs constitute approximately 70-80% of cumulative risk from all PCB congeners, with ILCRs ranging from 2×10^{-3} in Lake Wallula to 3×10^{-3} in both the 100 and 300 Area sub-areas.

Cancer risks without the fish ingestion pathway are shown in Figure 6-15. This figure is based on the sum of the youth and adult receptor only, since the child receptor is assumed to be exposed via only the ingestion of fish. As with noncancer hazard, the cumulative cancer risks decrease dramatically when the fish ingestion pathway is excluded, as shown in Figure 6-15.

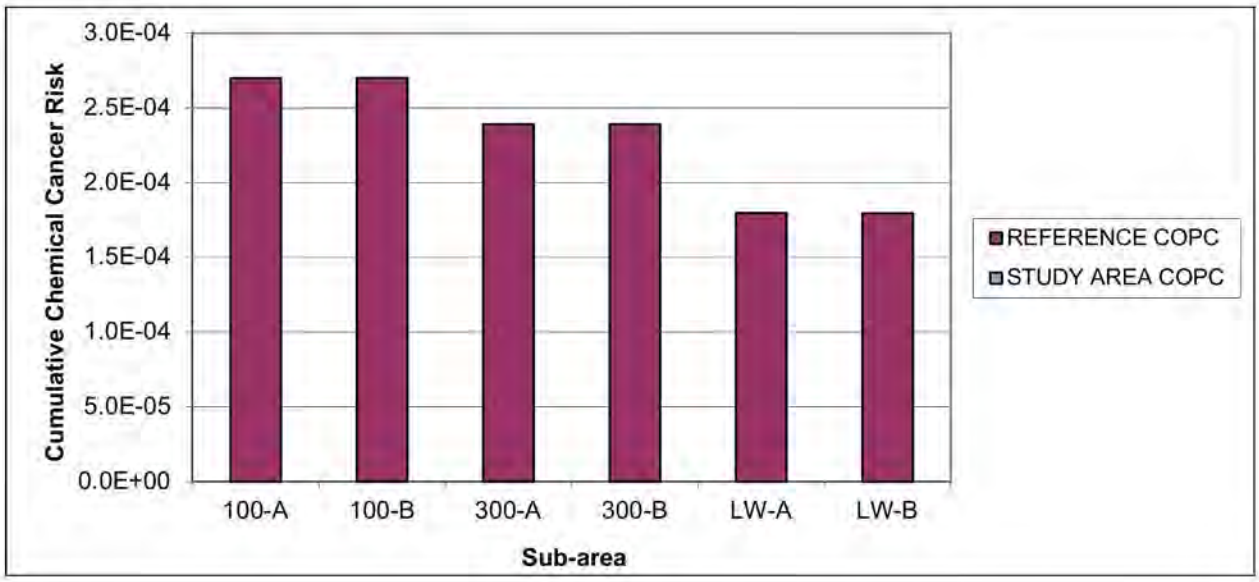
Figure 6-15. Cumulative Chemical Cancer Risk for the Avid Angler, All Exposure Media Excluding Fish Tissue: Reasonable Maximum Exposure.



The abiotic exposure pathway risks are all within the EPA's target risk range of 10^{-6} to 10^{-4} and below the MTCA cumulative risk limit of $1x10^{-5}$. When fish consumption is excluded, the majority of the cumulative chemical cancer risk is associated with reference-related (versus Study Area-related) COPCs at most exposure points. In the 100-A exposure location, arsenic in surface water and sediment (Reference COPC in each medium) accounts for nearly all of the cancer risk for abiotic media. In the 100-B exposure location, the only Study Area COPC-related risk is from arsenic in island soil, and the reference-related risk is due to arsenic in sediment. The 300-A risks are almost equally divided between island soil and sediment, whereas in 300-B, island soil contributes to slightly more risk. Arsenic accounts for the majority of risk in both media and locations and is a Study Area COPC in island soil and a Reference COPC in sediment. In Lake Wallula, arsenic (Reference COPC) in sediment accounts for the most risk.

The CTE cumulative cancer risks for all media are shown in Figure 6-16. Calculated cancer risks exceed the MTCA risk limit of $1x10^{-5}$ as well as the upper end of EPA's target risk range of 10^{-4} in all exposure points.

Figure 6-16. Cumulative Chemical Cancer Risk for the Avid Angler, All Exposure Media: Central Tendency Exposure.



As under the RME assumption, risks from fish ingestion far exceed those of other media, with Reference COPCs in fish tissue accounting for the majority of risk. The risk drivers under the CTE assumption are the same as for the RME assumption: PCBs, chlorinated pesticides, and arsenic in fish tissue. Each of these is a Reference COPC in all Sub-Areas.

Figure 6-17 shows the CTE cumulative cancer risks for all other media, excluding fish tissue. These cumulative risks for soil, sediment, and surface water are all below the MTCA cumulative risk limit as well as the lower end of EPA's target cancer risk range (10^{-6}). For the 100-B, 300-A, and 300-B exposure locations, island soil, sediment, and surface water exposure pathways contribute similar levels of risk. In Lake Wallula, surface water and sediment exposure pathways also contribute approximately equal amounts of risk. Reference COPCs, most frequently arsenic in sediment, account for the majority of risk in all sub-areas.

Figure 6-17. Cumulative Chemical Cancer Risk for the Avid Angler, All Exposure Media Excluding Fish Tissue: Central Tendency Exposure.

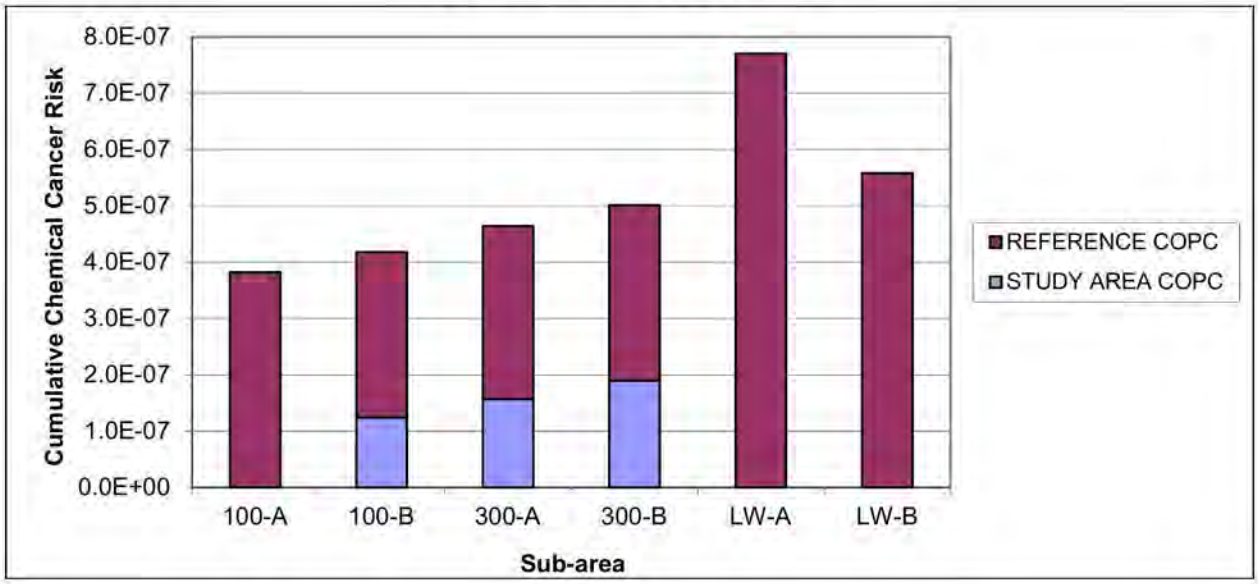


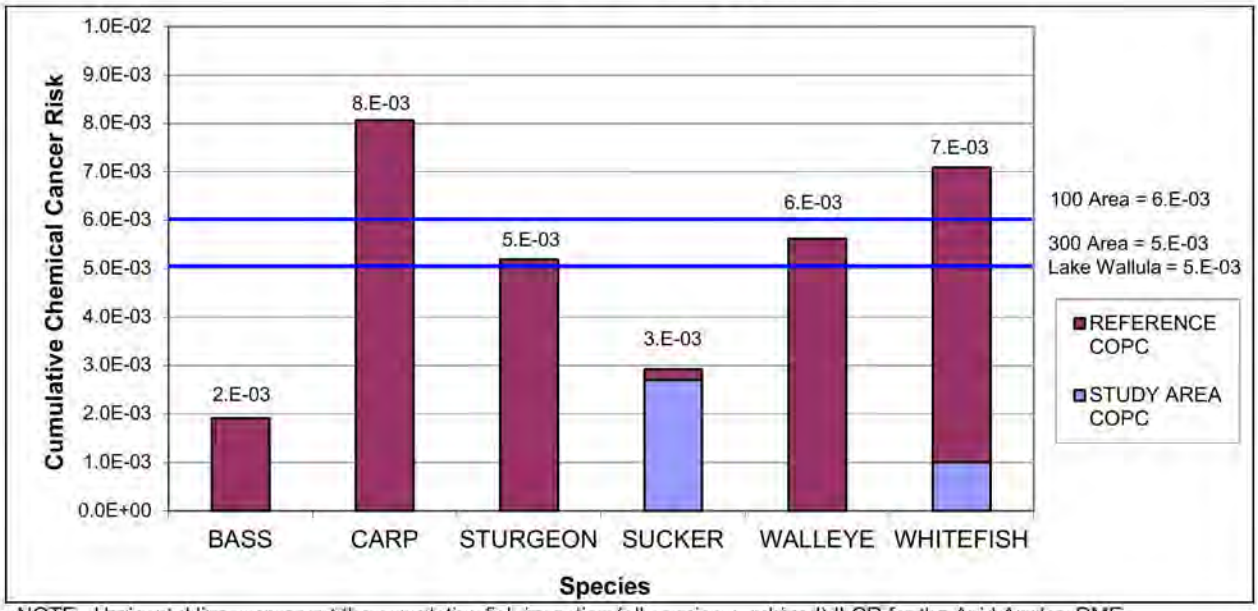
Figure 6-18 presents chemical cancer risk for the Avid Angler (all age groups), RME, for consumption of each individual fish species. Fish (all species combined) ingestion risks from each of the three sub-areas are also presented in this figure for comparison.

On a species-specific basis, chemical cancer risk associated with fish consumption varies within a factor of four. Similar to that observed for the noncancer hazard (Figure 6-13), cancer risk was highest for carp, sturgeon, walleye, and whitefish, and lowest for bass and sucker, with ILCRs ranging from 2×10^{-3} to 8×10^{-3}. In all cases, cumulative cancer risk exceeded the MTCA cumulative risk limit as well as the upper end of EPA's cancer risk range of 10^{-4}. With the exception of sucker, Reference COPCs (primarily PCBs and pesticides) contributed to most, if not all, of the cancer risk across species. For sucker, PCBs were identified as Study Area COPCs and comprise the majority of cumulative cancer risk (ILCR of 2.7×10^{-3}, or 93%).

Polychlorinated biphenyls are the biggest cancer risk drivers among all species, with total ICLR from all congeners ranging from 1.5×10^{-3} in bass to 5×10^{-3} in carp. Across all species, dioxin-like PCBs account for 70-80% of cancer risk related to all PCB congeners. ILCR from dioxin-like PCBs ranged from 1×10^{-3} in bass to 4×10^{-3} in carp.

Compared to risks derived for consumption of all species, the cancer risks calculated for individual species was within or below the range of ICLRs from across all three sub-areas (ILCRs of 6×10^{-3} to 1×10^{-2}). This indicates that the cumulative approach, wherein fish ingestion was evaluated for all species combined but assessed on a sub-area basis, may overestimate or underestimate potential carcinogenic risks for consumption of individual species.

Figure 6-18. Cumulative Chemical Cancer Risk for the Avid Angler for Consumption of Individual Fish Species: Reasonable Maximum Exposure.



NOTE: Horizontal lines represent the cumulative fish ingestion (all species combined) ILCR for the Avid Angler, RME. Cumulative fish ingestion ILCR for all COPCs is presented to the right of each horizontal line. Note that in this case, the cumulative ILCRs for the 300 Area and Lake Wallula Sub-Area are both 5E-03.

COPC = contaminant of potential concern
ILCR = incremental lifetime cancer risk
RME = reasonable maximum exposure

6.5.2.2.2 Radiation Cancer Risk. The cumulative radiological risks for the avid angler summed across all age groups and all media (abiotic and fish tissue) for the RME assumption are shown in Figure 6-19.

Radiological cancer risks are below the upper end of EPA's target risk range of 10^{-4} for all six exposure points.

In the 100 Area sub-area, carbon-14 in fish tissue contributes to the majority of cumulative cancer risk, with an ILCR of 4×10^{-5}, approximately 98% of the cumulative ILCR. Risk attributed to carbon-14 in the 300 Area is similar to those presented by abiotic media. In Lake Wallula, carbon-14 in fish tissue constitutes approximately 75% of the cumulative radiation cancer risk.

Of the abiotic media, the radionuclides that contribute to the majority of cumulative radiation risk in sediment in the 100 and 300 Area Sub-Areas include cobalt-60 and europium-152, which are both Study Area COPCs. In Lake Wallula, europium-152 in sediment (Study Area COPC) constitutes the majority of risk at both exposure points. Throughout the Hanford Site Study Area, cesium-137, a Reference COPC in island soil and sediment, also contributes to overall radiation cancer risk.

Figure 6-19. Cumulative Radiation Cancer Risk for the Avid Angler, All Exposure Media: Reasonable Maximum Exposure.

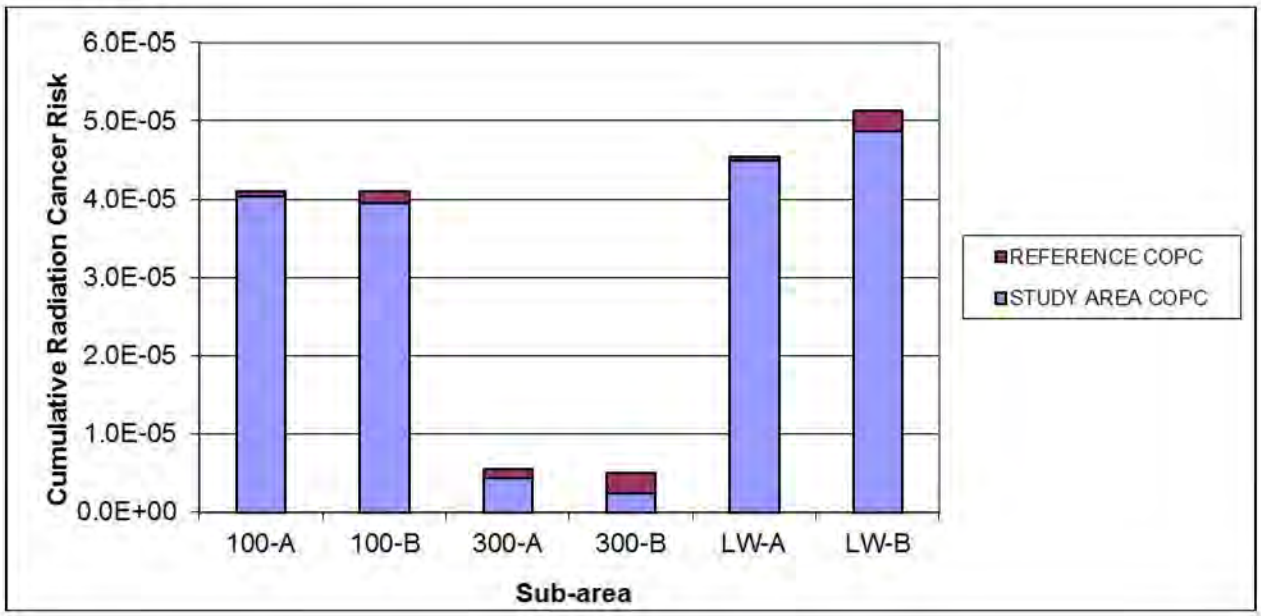


Figure 6-20 depicts cumulative radiation cancer risk from only abiotic media. As indicated on this figure, cumulative ILCRs across all exposure points are below the MTCA cumulative risk limit and within the EPA risk range.

Figure 6-20. Cumulative Radiation Cancer Risk for the Avid Angler, Abiotic Media: Reasonable Maximum Exposure.

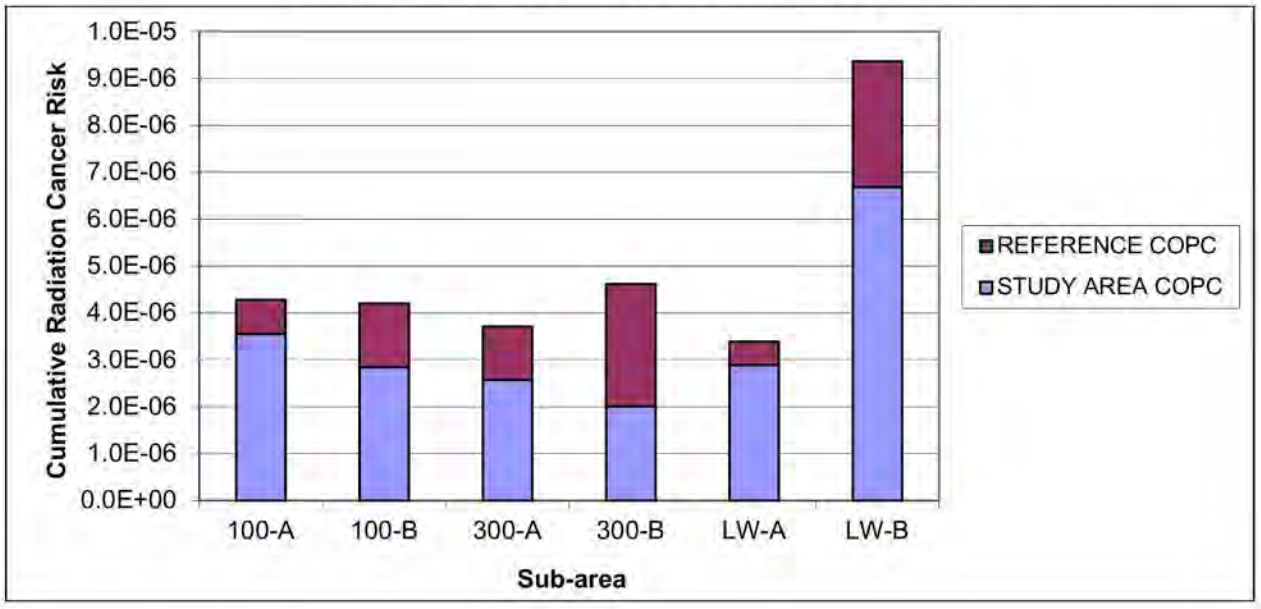
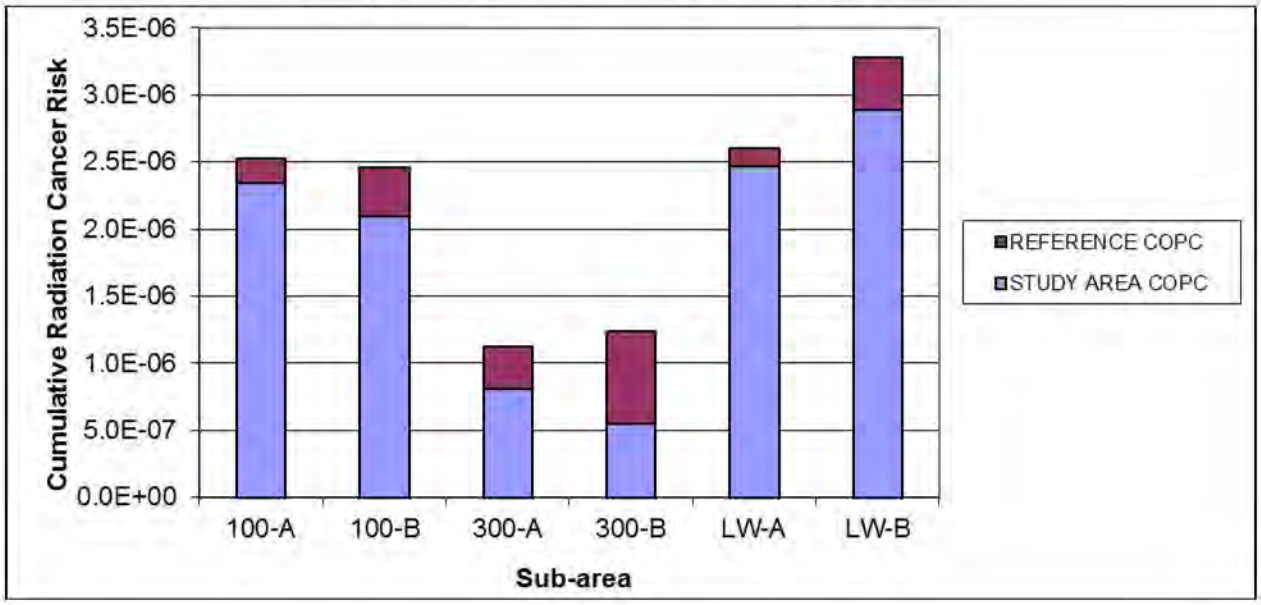


Figure 6-21 presents the cumulative radiation cancer risks under the CTE assumptions for exposure to all media.

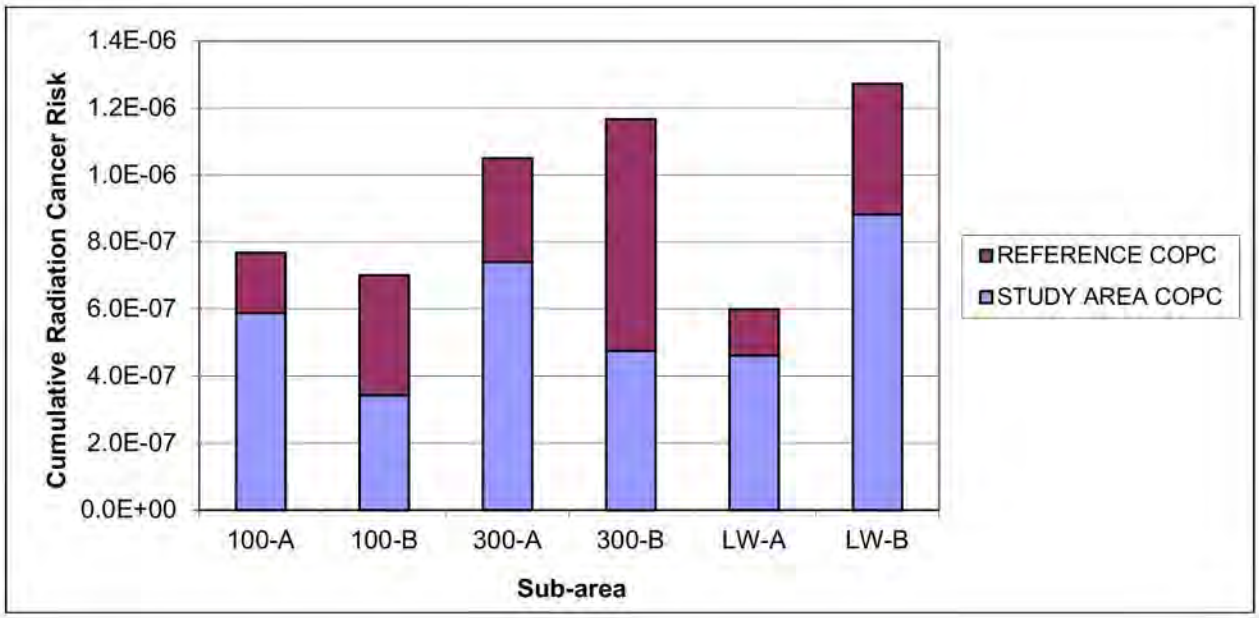
Figure 6-21. Cumulative Radiation Cancer Risk for the Avid Angler, All Exposure Media: Central Tendency Exposure.



As with the RME scenario, the figure above depicts cumulative risk from both fish and abiotic media, and carbon-14 in fish tissue contributes to the majority of radiation cancer risk in both the 100 Area and Lake Wallula sub-areas. The CTE cumulative cancer risks are approximately an order of magnitude less than those calculated for the RME assumption. Radiation cancer risk at all exposure points is within EPA's target risk range.

Figure 6-22 depicts cumulative radiation cancer risk from only abiotic media.

Figure 6-22. Cumulative Radiation Cancer Risk for the Avid Angler, Abiotic Media: Central Tendency Exposure.



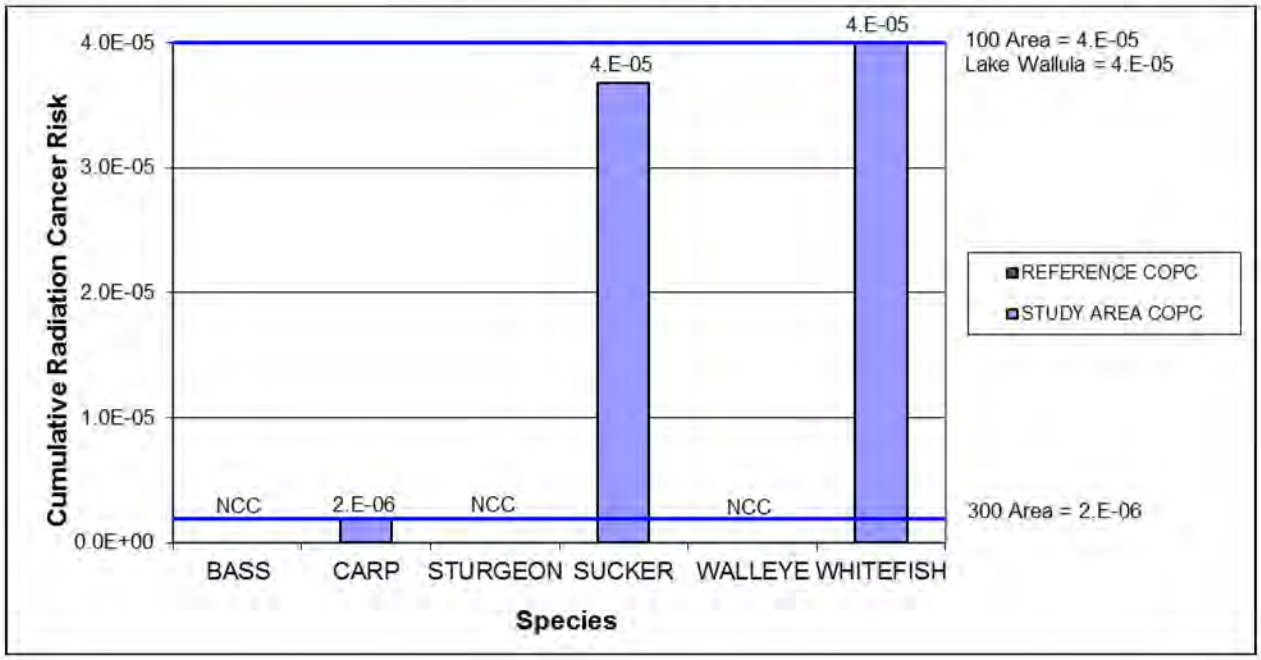
Study Area COPCs account for the majority of risk at most exposure points. In the 100-A, 100-B, and 300-A exposure points, carbon-14, cobalt-60 and europium-152 contribute to most of the cumulative radiation risk, similar to the RME assumption. In the 300-B exposure point, cesium-137, a Reference COPC in sediment and soil, drives the majority of radiation cancer risk. Europium-152 (a Study Area COPC) also contributes to cumulative risk from island soil.

Cobalt-60, europium-154, and europium-152 in sediment are the COPCs that constitute most of the radiation risk in Lake Wallula. These radionuclides are Study Area COPCs. In LW-B, cesium-137, a reference COPC in sediment, also contributes to cumulative risk.

At all exposure points, cumulative cancer risk from Study Area COPCs is less than the lower end of the EPA target risk range (10^{-6}).

Figure 6-23 presents radiation cancer risk for the Avid Angler (all age groups), RME, for consumption of each individual fish species. Fish (all species combined) ingestion risks are also presented in this figure for comparison.

Figure 6-23. Cumulative Radiation Cancer Risk for the Avid Angler for Consumption of Individual Fish Species: Reasonable Maximum Exposure.



Carbon-14 was not detected in bass, sturgeon or walleye. Radiation cancer risk from this COPC ranged from 2 x 10^{-6} in carp to approximately 4 x 10^{-5} in sucker and whitefish. Radiation risks estimated for individual species were within or below the range of radiation cancer risks for the combined species analysis.

6.5.2.2.3 Total Cancer Risk. The total chemical and radiation cancer risks for the Avid Angler scenario, by exposure point location, are shown in the following table for the RME assumption.

Cumulative Cancer Risk, Avid Angler: RME	100-A	100-B	300-A	300-B	LW-A	LW-B
Total - fish ingestion pathway	**6 x 10^{-3}**	**6 x 10^{-3}**	**5 x 10^{-3}**	**5 x 10^{-3}**	**5 x 10^{-3}**	**5 x 10^{-3}**
Total - abiotic pathways	6 x 10^{-6}	6 x 10^{-6}	6 x 10^{-6}	7 x 10^{-6}	6 x 10^{-6}	1 x 10^{-5}
Total ILCR, all COPCs	**6 x 10^{-3}**	**6 x 10^{-3}**	**5 x 10^{-3}**	**5 x 10^{-3}**	**5 x 10^{-3}**	**5 x 10^{-3}**
Total ILCR, study area COPCs	**4 x 10^{-5}**	**4 x 10^{-5}**	**5 x 10^{-6}**	**3 x 10^{-6}**	**4 x 10^{-5}**	**5 x 10^{-5}**

For the RME assumption, cancer risks across all sub-areas exceed EPA's target risk range and the MTCA cumulative risk limit by up to two orders of magnitude. As discussed, nearly all of the risk is attributed to PCBs, pesticides, and carbon-14 for the fish consumption pathway. Cumulative risk for only abiotic exposure pathways, however, is within the EPA target risk range and below the MTCA risk limit.

For the CTE assumption, total cancer risks for chemical and radionuclide COPCs are shown in the table below.

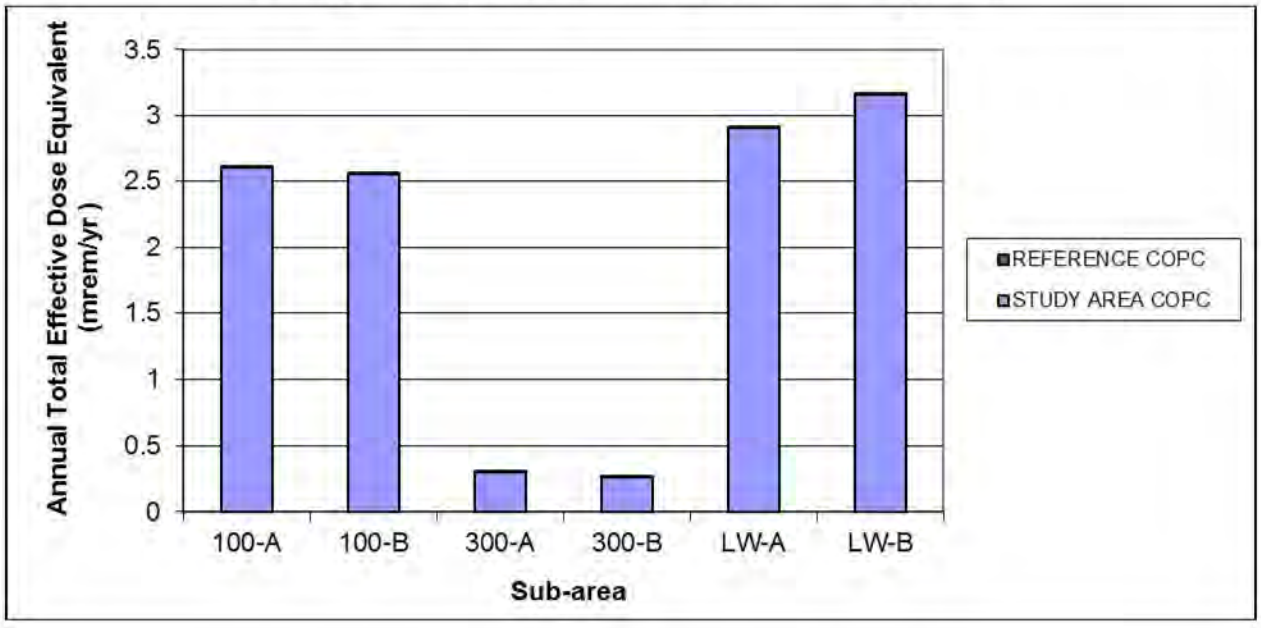
Cumulative Cancer Risk, Avid Angler: CTE	100-A	100-B	300-A	300-B	LW-A	LW-B
Total - fish ingestion pathway	3×10^{-4}	3×10^{-4}	2×10^{-4}	2×10^{-4}	2×10^{-4}	2×10^{-4}
Total - abiotic pathways	1×10^{-6}	1×10^{-6}	2×10^{-6}	2×10^{-6}	1×10^{-6}	2×10^{-6}
Total ILCR, all COPCs	3×10^{-4}	3×10^{-4}	2×10^{-4}	2×10^{-4}	2×10^{-4}	2×10^{-4}
Total ILCR, study area COPCs	2 x 10-6	2×10^{-6}	1×10^{-6}	7×10^{-7}	2×10^{-6}	3×10^{-6}

The CTE risks slightly exceed the upper bound of EPA's target risk range (10^{-4} and are an order of magnitude higher than the MTCA cumulative cancer risk limit. Absent the fish consumption pathway, cumulative risk for the CTE condition for abiotic pathways is within the EPA target risk range and below the MTCA risk limit.

For both RME and CTE assumptions, chemical cancer risks associated with fish consumption account for nearly all of the cumulative risk for all exposure points, mainly due to PCBs and pesticides in fish tissue.

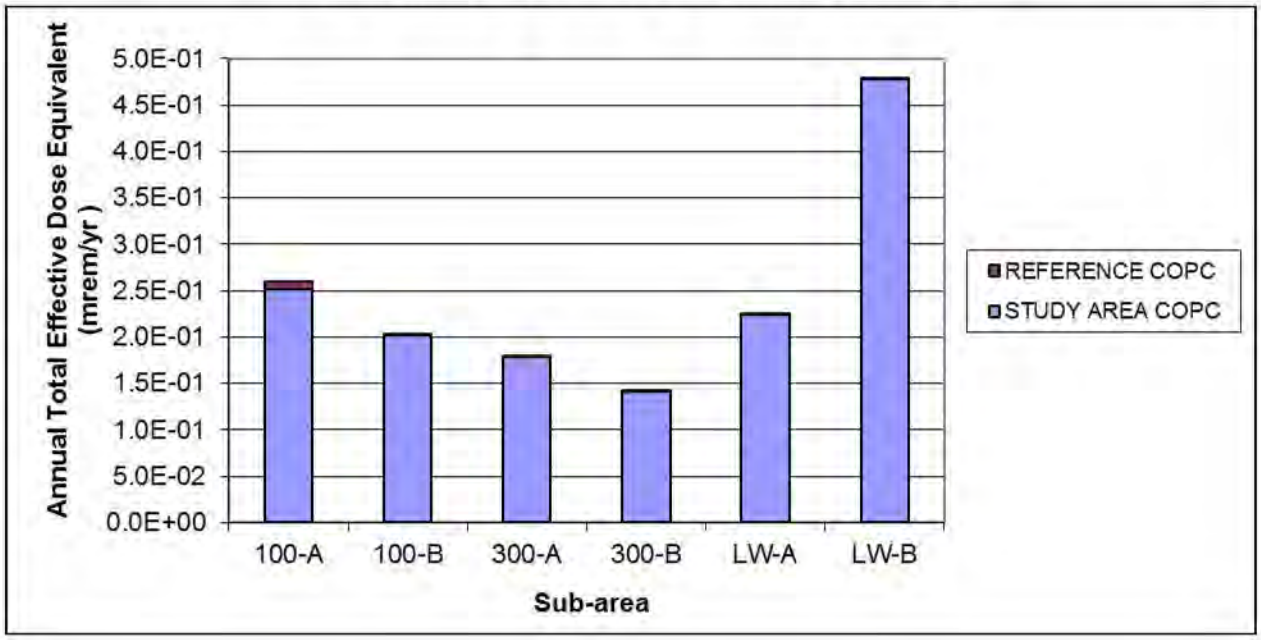
6.5.2.3 Annual Total Effective Dose Equivalent. The cumulative annual TEDE for the Avid Angler receptor is shown in Figure 6-24.

Figure 6-24. Annual Total Effective Dose Equivalent for the Avid Angler, All Exposure Media: Reasonable Maximum Exposure.



Throughout the Hanford Site Study Area, all annual TEDEs are well below the annual radiation dose limit of 15 mrem/yr. In the 100 Area and 300 Area sub-areas, most of the annual TEDE is attributed to carbon-14 in fish tissue. The annual TEDEs for abiotic media are presented in Figure 6-25 below. The annual TEDE for abiotic media in the 100 Area is approximately an order of magnitude lower than that for all media, including fish tissue.

Figure 6-25. Annual Total Effective Dose Equivalent for the Avid Angler, Abiotic Media: Reasonable Maximum Exposure.



Likewise, the cumulative annual TEDE at each exposure point for the CTE assumption, shown in Figure 6-26 for all media and Figure 6-27 for abiotic media, does not exceeds the radiation dose threshold of 15 mrem/yr.

All annual TEDEs are less than 0.5, approximately 30 times below the 15 mrem/yr radiation dose threshold.

Figure 6-26. Annual Total Effective Dose Equivalent for the Avid Angler, All Exposure Media: Central Tendency Exposure.

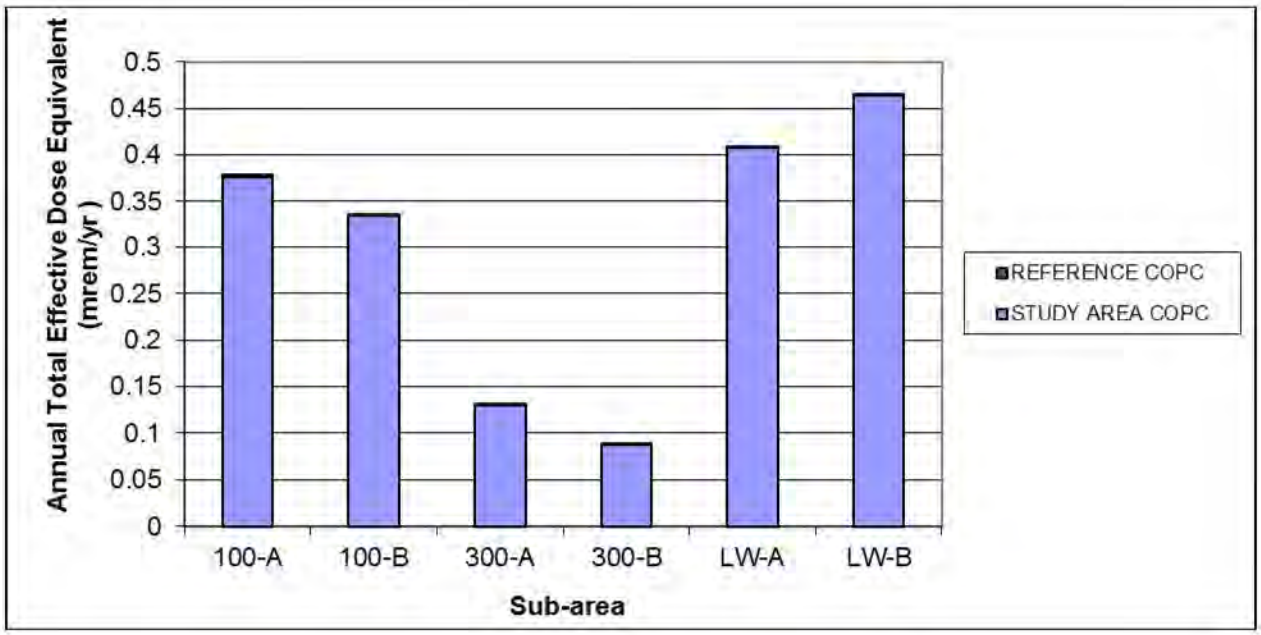


Figure 6-27. Annual Total Effective Dose Equivalent for the Avid Angler, Abiotic Media: Central Tendency Exposure.

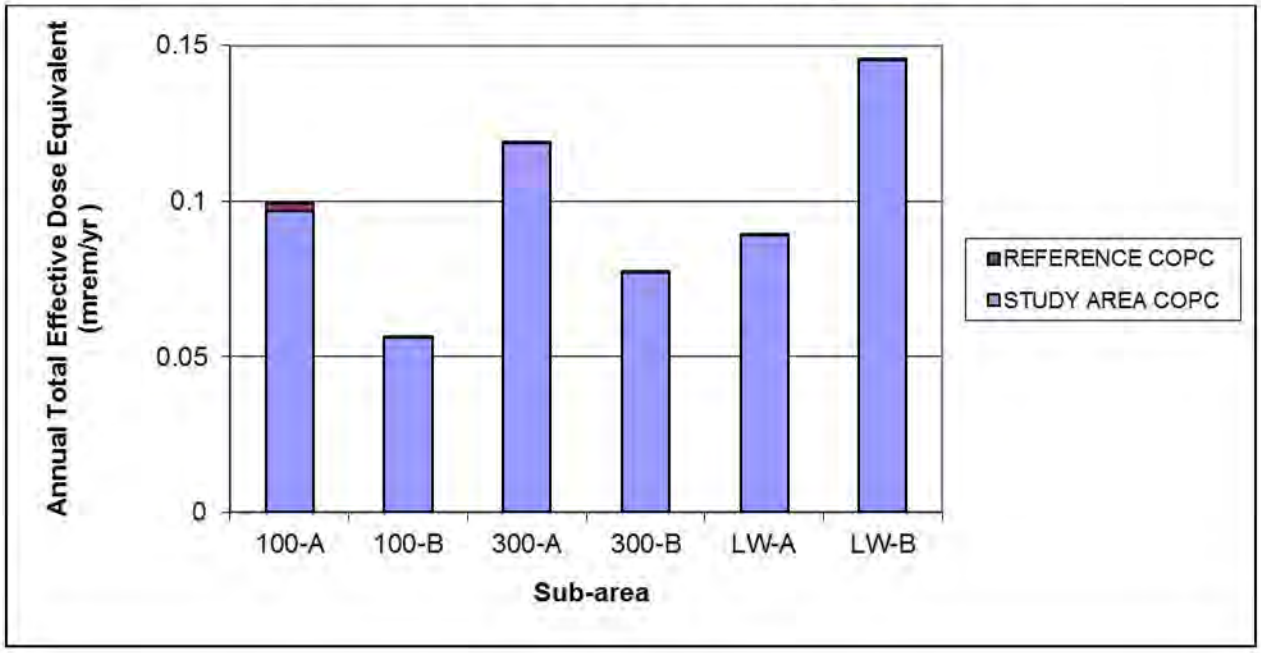
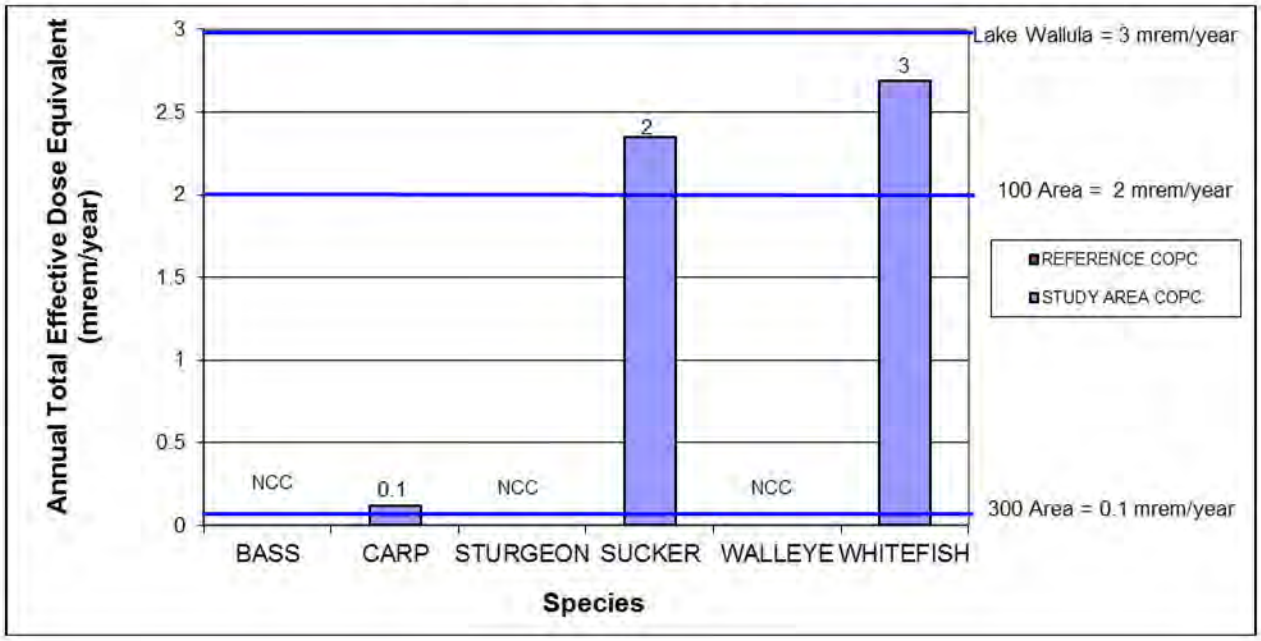


Figure 6-28 presents the annual TEDE for the Avid Angler (all age groups), RME, for consumption of each individual fish species. Fish (all species combined) ingestion risks are also presented in this figure for comparison.

Figure 6-28. Annual Total Effective Dose Equivalent for the Avid Angler for Consumption of Individual Fish Species: Reasonable Maximum Exposure.



As shown in this figure, the annual TEDE related to ingestion of each evaluated species is well below the 15 mrem/yr radiation dose threshold.

6.5.3 Yakama Nation

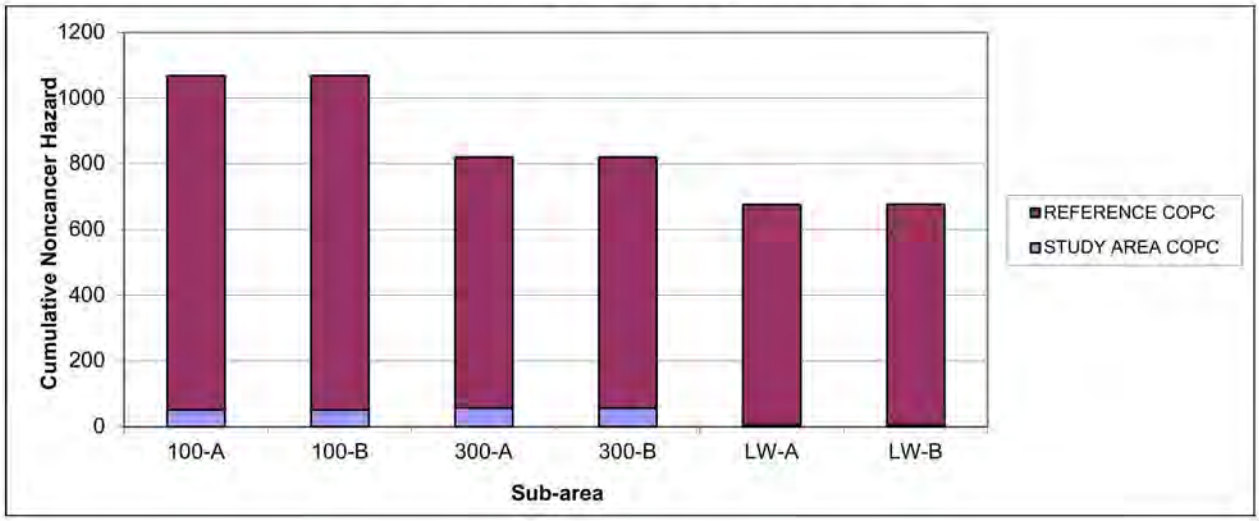
The Yakama Nation scenario represents a hypothetical member of the Yakama Nation who routinely engages in activities related to subsistence fishing. The Yakama receptor is assumed to have direct contact with surface water, island soil, and sediment while fishing, and is assumed to eat fish caught from the Hanford Site Study Area in the Columbia River on a routine basis. As described in more detail in Section 4.3, the fish ingestion rate used for this scenario assumes that fish comprise the majority of this receptor's diet. This scenario, for which only one condition based on RME EPCs was evaluated, does not reflect residential or agricultural exposures and is intended to address only fishing-related exposures. Additional details on the Yakama Nation scenario are presented in Sections 4.1 through 4.3.

6.5.3.1 Noncancer Hazard. Hazard index calculations for the Yakama Nation scenario are provided in Appendices F through L. Cumulative HIs are presented for the Yakama Nation child

in Table 6-73 for 100-A, Table 6-74 for 100-B, Table 6-75 for 300-A, Table 6-76 for 300-B, Table 6-77 for LW-A, and Table 6-78 for LW-B. As conducted for other receptors, noncancer hazards are presented for only the child age group, since the HIs for this age range (1 through 6 years) are higher than those estimated for the adult.

Figure 6-29 shows the cumulative noncancer HI for the Yakama Nation child across the six different exposure points. Noncancer hazard accounts for cumulative exposures across all relevant exposure pathways, media and COPCs. This figure also shows the contribution from Study Area COPCs and Reference COPCs that were previously identified in Section 3.8.

Figure 6-29. Cumulative Noncancer Hazard for the Yakama Nation Child, All Exposure Media.

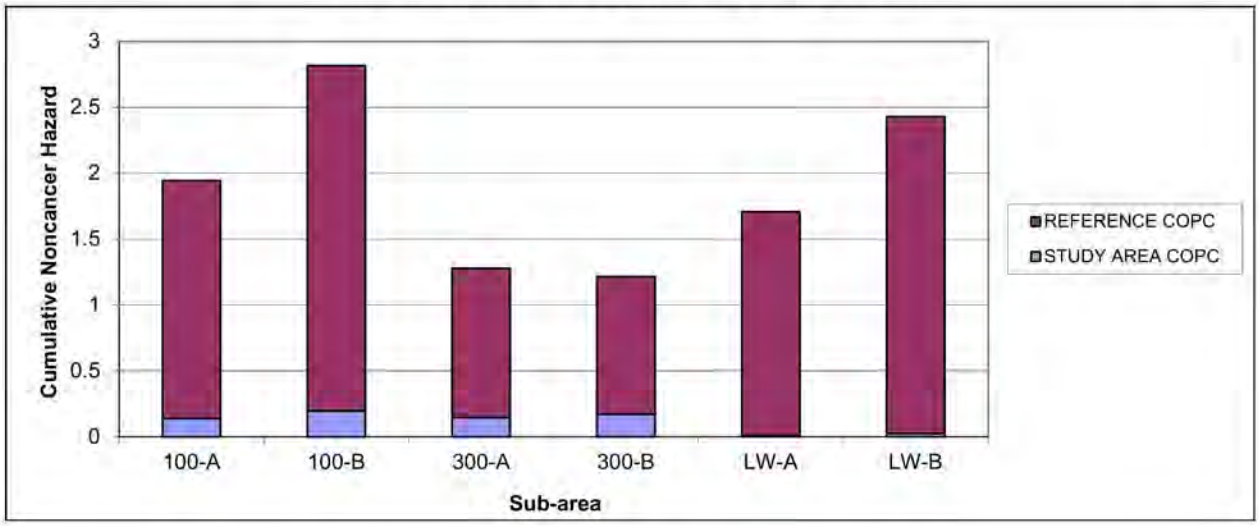


The cumulative HI for all exposure points exceeds the threshold HI of 1, with HIs ranging from approximately 600 in the Lake Wallula Sub-Area to over 1,000 in the 100 Area Sub-Area (see Tables 6-73 through 6-78). Nearly all of the hazard (>99%) is attributable to fish ingestion alone, and within that pathway, the majority of the total HI is due to PCBs (both dioxin-like and nondioxin-like) and cobalt, both of which are Reference COPCs in fish tissue across all exposure points. Cumulative hazard resulting from PCBs alone resulted in a HI of 900 in the 100 Area sub-area, with smaller hazard in 300 Area and Lake Wallula (HQ of approximately 600 and 400, respectively). As seen for the Avid Angler scenario, dioxin-like PCBs account for more than half of the cumulative HI from all PCB congeners, with the highest hazard (HQ of 469) present in the 100 Area, and the lowest (HQ of 240) in Lake Wallula.

Ingestion of cobalt in fish resulted in a hazard quotient of 100 in the 100 Area, 80 in 300 Area and 200 in Lake Wallula. Fish ingestion of most COPCs, however, generally resulted in a hazard quotient of 1 or greater at all exposure points.

Similar to that observed for the Avid Angler scenario, noncancer hazards related to the fish ingestion pathway alone dwarf noncancer hazard from other pathways by almost two orders of magnitude. Risks from other media contributed very little to overall noncancer hazard. Figure 6-30 shows the cumulative HI for the Yakama Nation child for surface water, sediment, and island soil (where applicable), excluding the fish ingestion pathway.

Figure 6-30. Cumulative Noncancer Hazard for the Yakama Nation Child, All Exposure Media Excluding Fish Tissue.



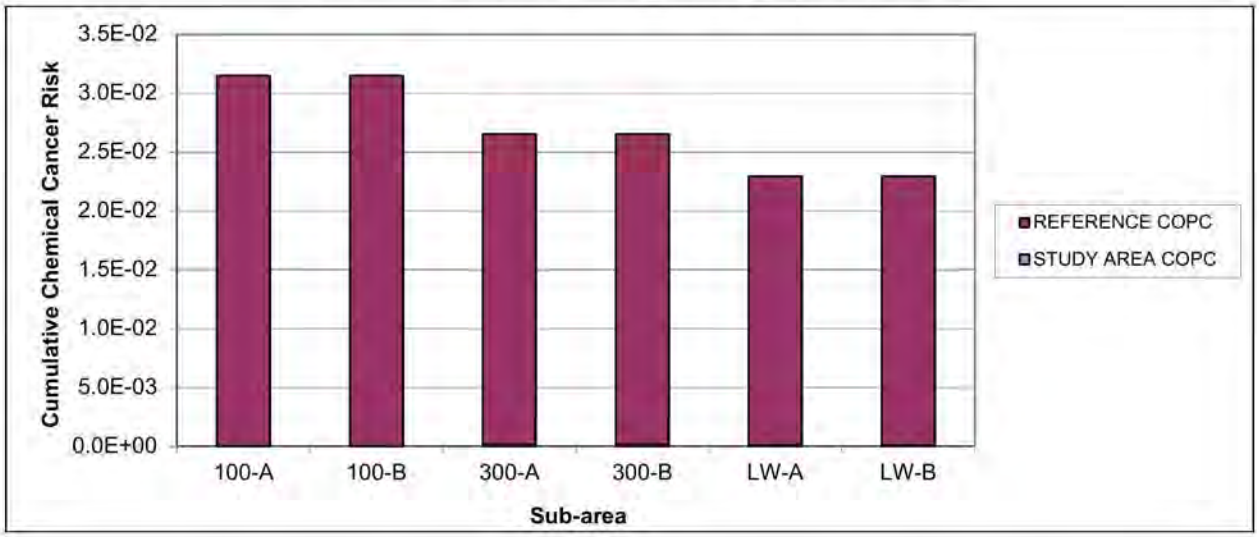
As indicated in this figure, the HIs among all three sub-areas are similar, ranging from approximately 1 to 3. Most of the risk in these areas is attributable to Reference COPCs (mainly cobalt, thallium, iron, and/or arsenic) in sediment and/or island soil (arsenic is a Study Area COPC in soil in the 300 Area Sub-Area only; as discussed, the 100-A, LW-A, and LW-B areas do not contain any islands). Note that the cumulative HIs associated with Study Area COPCs are below 1 at all exposure points.

6.5.3.2 Cancer Risk. Cumulative ILCRs were calculated for both chemical and radiological COPCs. Although risks from both types of COPCs are summed together for a given receptor, risks due to chemical and radiological COPCs are presented and discussed separately in the following subsections. Cumulative cancer risks from both chemical and radiological COPCs for the Yakama Nation receptor are discussed in Section 6.5.3.2.3.

6.5.3.2.1 Chemical Cancer Risk. Tables 6-73 through 6-78 present cumulative ILCRs at each exposure point for the Yakama Nation child and Tables 6-79 through 6-84 present ILCRs for the adult receptor. Cancer risk calculations for individual exposure pathways and exposure media are presented in Appendices F through K. Because the Yakama Nation scenario encompasses both child and adult exposures, cancer risks from each age group are added together to derive

cumulative lifetime cancer risks. Figure 6-31 depicts cumulative ILCRs for the Yakama Nation scenario across all exposure points, exposure pathways, and COPCs.

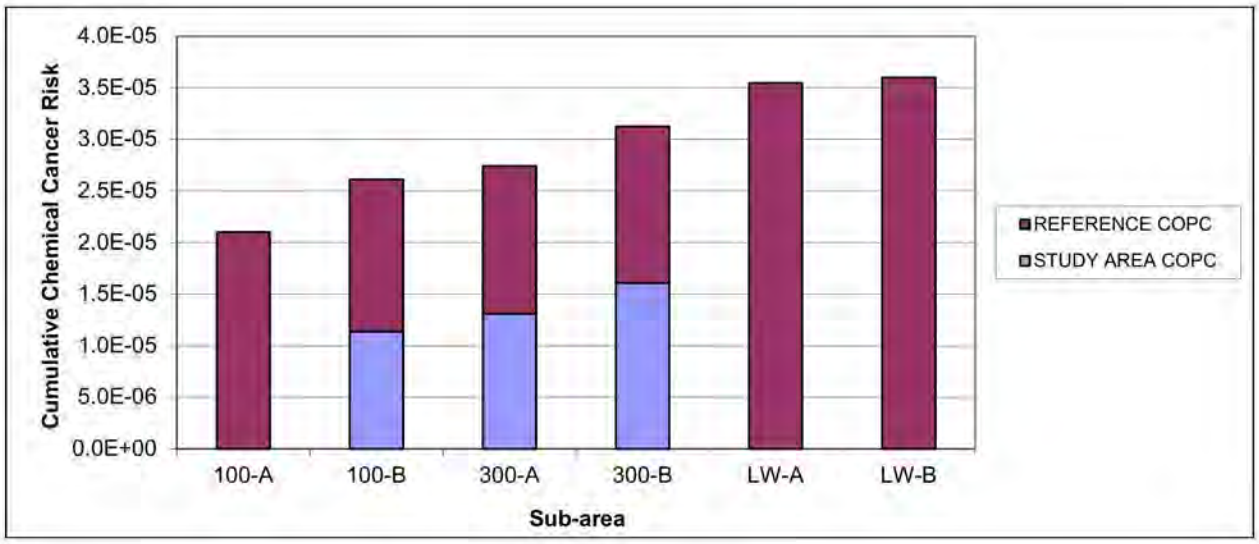
Figure 6-31. Cumulative Chemical Cancer Risk for the Yakama Nation Scenario, All Exposure Media.



Cancer risks across exposure points exceed the MTCA cumulative cancer risk limit of 1×10^{-5} and the upper end of the EPA cancer risk range 10^{-4}. Similar to noncancer hazard, cumulative ILCRs are almost entirely related to fish ingestion; this pathway accounts for approximately 99% of cumulative cancer risk. Primary risk drivers in fish tissue include PCBs, chlorinated pesticides such as dieldrin and DDE, and arsenic. As previously described and as shown in the figure above, virtually all of this cancer risk is associated with Reference COPCs in fish tissue; of these, approximately 50% to 80% of the cumulative ILCR is related to PCBs, with the highest PCB-associated ILCR in the 100 Area sub-area. Of the PCBs, dioxin-like PCBs accounted for the majority (70% to 80%) of the cumulative ILCR from all PCBs, resulting in an ILCR of 1×10^{-2} for the 100 and 300 Area sub-areas and 9×10^{-3} for Lake Wallula.

Other exposure pathways contribute to approximately 1% or less excess cancer risk. Figure 6-32 depicts cumulative cancer risk related to soil, sediment, and surface water exposures, excluding risks related to fish ingestion.

Figure 6-32. Cumulative Chemical Cancer Risk for the Yakama Nation Scenario, All Exposure Media Excluding Fish Tissue.

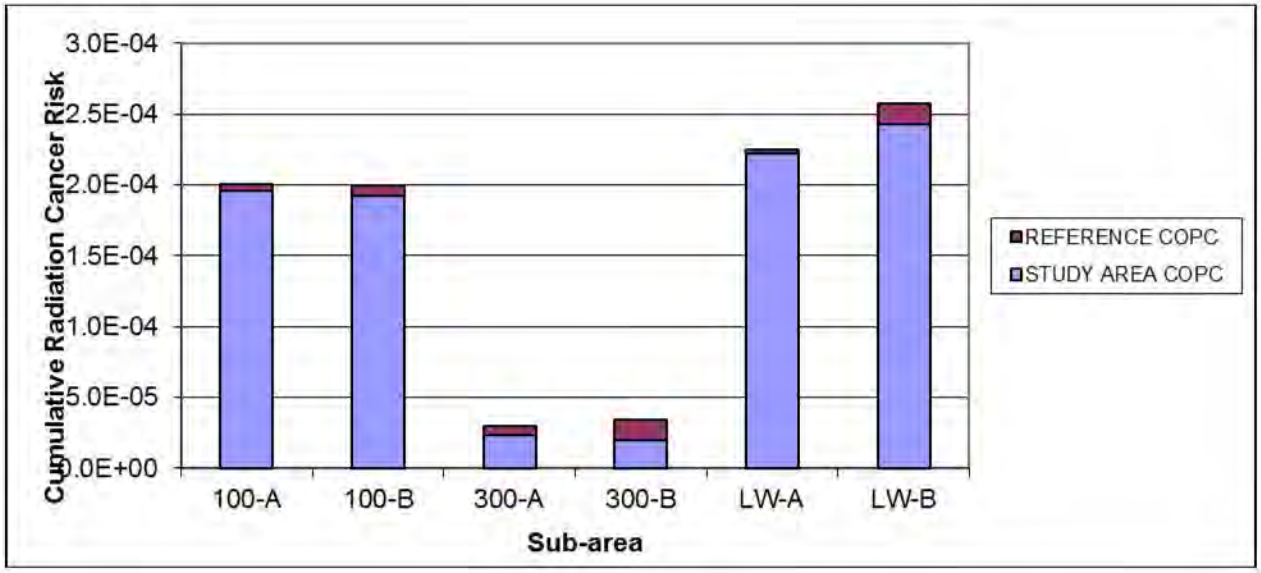


As seen in Figure 6-32, cancer risk related to surface water, sediment, and/or soil exposures is at a level of 2 x 10^{-5} or greater in all exposure points, higher than the MTCA cumulative risk limit, with the highest cancer risk in the LW-A exposure point area. However, no estimated cancer risk at any exposure point exceeds the upper end of EPA's target risk range (10^{-4}). Cancer risk in the 100-A exposure point area and the Lake Wallula Sub-Area is entirely attributed to Reference COPCs; in particular, arsenic in sediment (see Tables 6-73, 6-77, and 6-78 for the child and Tables 6-79, 6-83, and 6-84 for the adult). In the 100-B exposure point area, arsenic in soil comprises nearly all of the Study Area COPC risk for this exposure point. In the 300 Area Sub-Area, chloroform in surface water and arsenic in island soil constitute the vast majority of Study Area COPC-related risks, whereas arsenic in sediment and surface water comprise nearly all of the Reference COPC-related risk.

6.5.3.2.2 Radiation Cancer Risk. Radiation cancer risks are presented as cumulative risk across multiple age groups. For the Yakama Nation scenario, radiation cancer risks reflect both the child and adult receptor.

Summaries of radiation cancer risks across media are presented in Tables 6-73 through 6-78 for the Yakama Nation child and Tables 6-79 through 6-84 for the Yakama Nation adult. Appendices F through K contain by exposure point the radiation cancer risk calculations by exposure pathway and medium. Cumulative radiation cancer risks for both the adult and child are summarized in Figure 6-33.

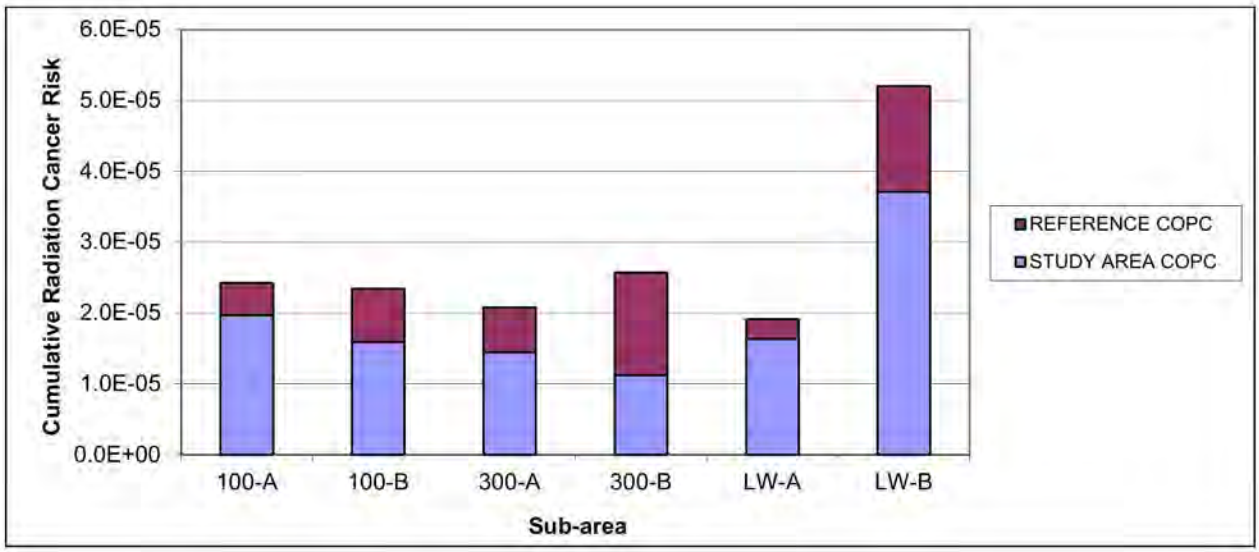
Figure 6-33. Cumulative Radiation Cancer Risk for the Yakama Nation Scenario, All Exposure Media.



In the 100-A and 100-B exposure points, cumulative ICLRs exceed the upper end of the EPA target risk range. Most of the radiation cancer risk is associated with ingestion of carbon-14 in fish tissue. Carbon-14 in fish is also a primary risk driver for the 300-A and 300-B exposure points. All calculated ILCRs across exposure points in the 300 Area and Lake Wallula sub-areas are within the EPA target risk range, ranging from approximately 2×10^{-5} in the LW-A and 300-A exposure points to over 5×10^{-5} in the LW-B exposure point.

Most of the cumulative radiation ILCR in abiotic media (see Figure 6-34 below) is attributable to various radionuclides in sediment, although radionuclides in island soil in the 300-B exposure point area (i.e., Johnson Island) are also significant risk drivers. In the 100 Area exposure points, cobalt-60 and europium-152, both Study Area COPCs in sediment, contribute to the majority of cumulative risk. Within the 300 Area, cesium-137 in soil contributes to approximately 50% to 80% of the total risk related to island soil exposure pathways (ingestion, inhalation, and external irradiation). Cesium-137 is a Reference COPC in soil. In Lake Wallula, europium-152, europium-154, and cobalt-60 (Study Area COPCs) in sediment and cesium-137 (a Reference COPC) comprise all of the total risk for this medium.

Figure 6-34. Cumulative Radiation Cancer Risk for the Yakama Nation Scenario, All Exposure Media Excluding Fish Tissue.



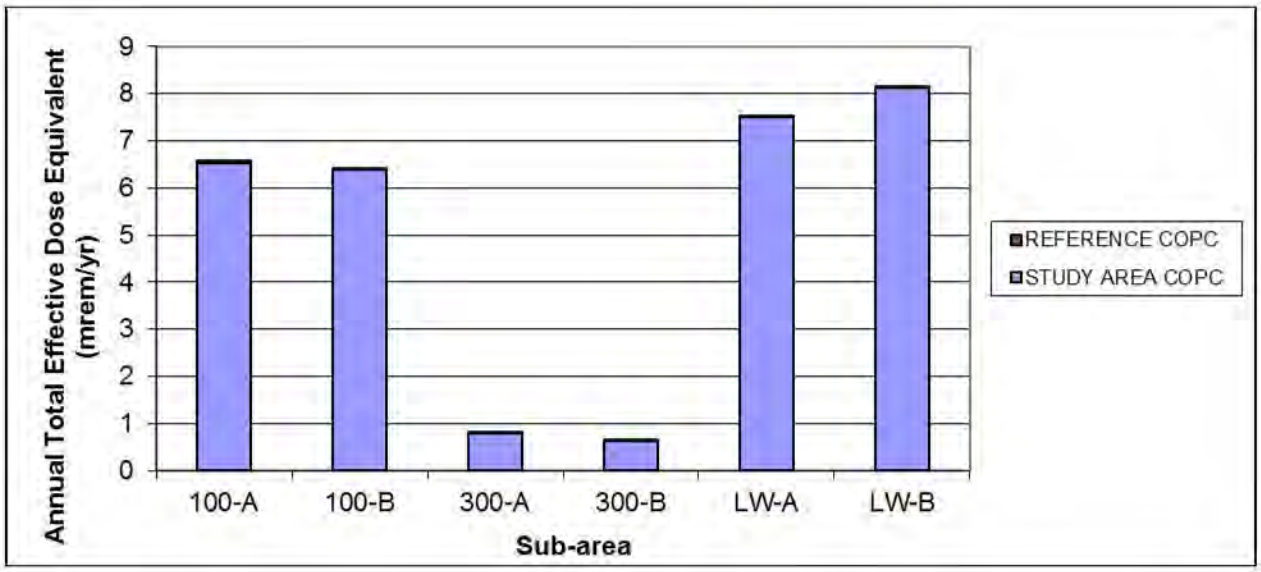
6.5.3.2.3 Cumulative Cancer Risk. Although radiation risks are described separately from chemical cancer risks, cumulative overall risk from both radionuclides and chemical cancer risks for the Yakama Nation scenario are presented below to depict overall cancer risk for the Hanford Site Study Area, in accordance with EPA guidance (OSWER 9200.4-18). These cumulative cancer risks are presented in the table below for the Yakama scenario.

Cumulative ILCR, Yakama Nation	100-A	100-B	300-A	300-B	LW-A	LW-B
Total - fish ingestion pathway	3×10^{-2}	3×10^{-2}	3×10^{-2}	3×10^{-2}	2×10^{-2}	2×10^{-2}
Total - abiotic pathways	5×10^{-5}	5×10^{-5}	5×10^{-5}	6×10^{-5}	5×10^{-5}	1×10^{-4}
Total ILCR - Yakama Nation, all COPCs	3×10^{-2}	3×10^{-2}	3×10^{-2}	3×10^{-2}	2×10^{-2}	2×10^{-2}
Total ILCR - Yakama Nation, study area COPCs	$\mathbf{2 \times 10^{-4}}$	$\mathbf{2 \times 10^{-4}}$	$\mathbf{1 \times 10^{-4}}$	$\mathbf{1 \times 10^{-4}}$	$\mathbf{3 \times 10^{-4}}$	$\mathbf{3 \times 10^{-4}}$

Cumulative cancer risk for all COPCs exceeds the upper end of the EPA cancer risk range of 10^{-4}. As previously described, all of the cumulative chemical cancer risk in fish tissue is attributable to Reference COPCs; however, most of the radiation cancer risk is related to carbon-14, a Study Area COPC in fish tissue. Reference COPCs also generally account for the majority of cancer risk for abiotic media. Absent the fish consumption pathway, cumulative cancer risks for abiotic media are within the EPA target risk range of 10^{-6} to 10^{-4} at each exposure point, although exceed the MTCA risk limit of 1×10^{-5}, with the majority of risk related to radionuclides and arsenic in island soil and sediment.

6.5.3.2.4 Total Effective Dose Equivalent. Dose calculations are presented by exposure point in Appendices F through K, and radiation doses by exposure medium are summarized in Tables 6-73 through 6-78 for the child and Tables 6-79 through 6-84 for the adult. Annual TEDEs for the Yakama Nation scenario across all exposure points are summed in Figure 6-35.

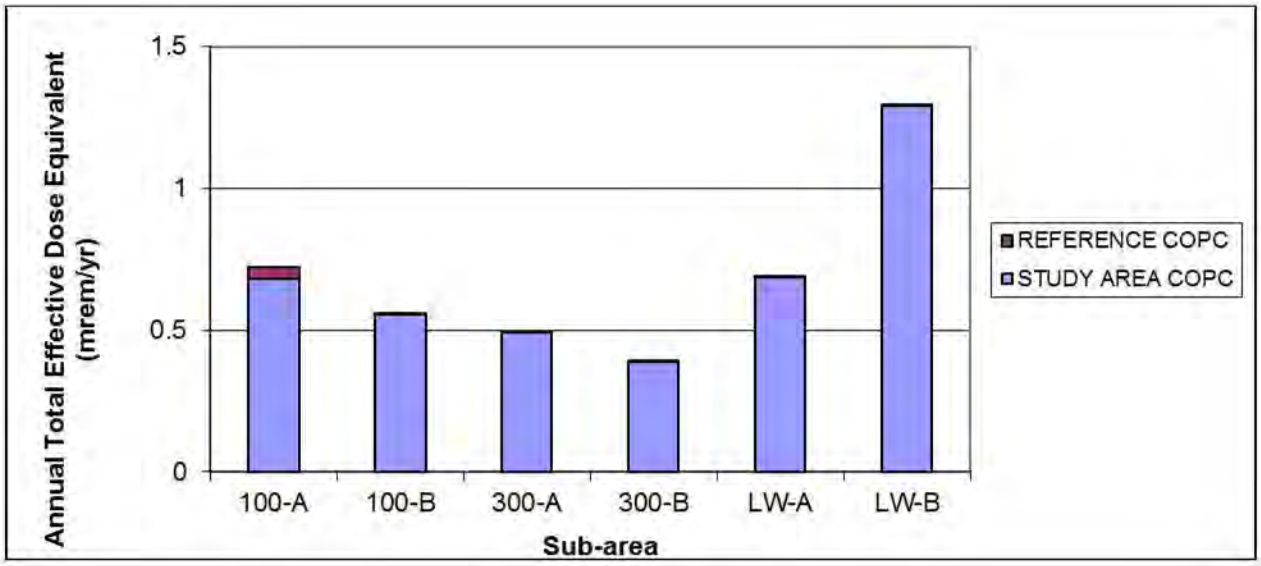
Figure 6-35. Annual Total Effective Dose Equivalent for the Yakama Nation Scenario, All Exposure Media.



Annual TEDE from all media is below the 15 mrem/yr radiation dose threshold at all exposure points. In the 100-A/B exposure points, most of the annual TEDE is related to ingestion of carbon-14 in fish tissue. Carbon-14 in fish tissue is attributed to approximately 50% of the annual TEDE in the 300-A/B exposure points.

Figure 6-36 shows annual TEDE associated with only abiotic media. Most of the annual TEDE in abiotic media is related to europium-152 and cobalt-60 in sediment, with the exception of exposure point 300-B. In this exposure point, most of the annual TEDE is related to inadvertent ingestion of island soil radionuclides on Johnson Island (annual dose of 0.4 mrem/yr for soil versus an annual TEDE of 0.6 mrem/yr for all media).

Figure 6-36. Annual Total Effective Dose Equivalent for the Yakama Nation Scenario, All Exposure Media Excluding Fish Tissue.



6.6 RISK CHARACTERIZATION SUMMARY

Noncancer hazard and cancer risks evaluated for the three exposure scenarios (Casual User, Avid Angler, and Yakama Nation) reflect varying levels of exposures for typical activities that would be expected to occur along the Columbia River. The Casual User represents recreational activities that primarily relate to swimming, picnicking, and wading along beaches. The Avid Angler scenario reflects a fishing scenario where exposure to sediment, soil, and surface water is relatively minor, but fish consumption rates may be high with respect to that consumed by an average U.S. citizen that occasionally catches and consumes fish. For example, the RME fish ingestion rate for the Avid Angler is approximately eight times higher than the CTE fish ingestion rate. The Yakama Nation scenario reflects an even higher level of fishing-related activities than the Avid Angler (approximately 19 times higher than the CTE consumption rate and 2 times higher than the Avid Angler RME rate).

Both RME and CTE were evaluated for the Casual User and Avid Angler scenarios, and there is, in general, an approximate two- to three-fold difference in estimated risk between these two conditions. Primary differences relate to the frequency and duration of exposure, amount of soil or sediment inadvertently ingested, concentration in the exposure medium and, for the Avid Angler scenario, the amount of fish ingested. These differences reflect variability in the population, but in both cases are intended to represent central tendency and upper-bound exposure potential. Only one condition was evaluated for the Yakama Nation scenario, as provided by the Yakama Nation and in accordance with the RI Work Plan (DOE/RL-2008-11).

Estimated risks for both the Avid Angler and Yakama Nation scenarios were dominated by the fish ingestion pathway, which accounted for approximately 99% of the cumulative cancer risk

and noncancer hazard. Both RME and CTE risks estimated for these two receptors exceeded EPA risk limits by up to three orders of magnitude when all exposure pathways were included. Polychlorinated biphenyls, chlorinated pesticides, arsenic, cobalt, and mercury are the primary risk drivers within fish tissue. Most of these constituents are Reference COPC across all exposure points, indicating that the concentrations of contaminants observed in the various fish species are, in general, similar to those observed in areas upstream of the Hanford Site. Relative to risk associated with fish ingestion, the contribution to cumulative risk from Study Area COPCs is minor, especially for abiotic media.

When comparing the results of the two approaches used to evaluate fish consumption risks, some patterns were observed related to risks estimated for individual fish species. For chemical cancer risk, results indicate that cancer risk was lowest for bass and sucker and highest in carp. For both chemical cancer risk and noncancer hazard in all species, risk was primarily due to PCBs, which are Reference COPCs across all exposure points.

Notwithstanding the fish ingestion exposure route, risks from other abiotic media resulted in a noncancer hazard below the EPA threshold of 1 for the RME and CTE Recreational User and Avid Angler scenarios in all exposure points. Cancer risks for these scenarios were also within the EPA cancer risk range of 10^{-6} to 10^{-4} and below the MTCA cumulative risk limit of 1×10^{-5} in all areas. Annual TEDE for all media was below the annual TEDE threshold of 15 mrem/yr.

For the Yakama Nation scenario, noncancer hazards for all media other than fish tissue slightly exceeded the noncancer hazard threshold in all exposure points, with the highest HIs in the Lake Wallula Sub-Area. Nearly the entire hazard at each exposure point, however, was attributable to Reference COPCs (primarily metals in sediment); for all exposure points, cumulative hazard from Study Area COPCs was below the threshold of one. Cumulative cancer risk, exclusive of fish tissue, ranged from approximately 2×10^{-5} to 4×10^{-5}; these risks are within EPA's target risk range. Nearly all of the cancer risk was attributable to arsenic in sediment, surface water, and soil. Arsenic is a Reference COPC in sediment and surface water, but a Study Area COPC in island soil.

Both CTE and RME cancer risks for the Casual User scenario were within or below the EPA cancer risk range of 10^{-6} to 10^{-4}. Noncancer hazard for the RME and CTE scenarios for this receptor were also below the threshold of one. Radiation dose for the Casual User (both RME and CTE) was below the annual TEDE threshold of 15 mrem/yr.

Results from the risk characterization indicate that the risks related to exposure to surface water, sediment, and island soil are very small relative to that from fish ingestion, and the cumulative cancer risk associated with Study Area COPCs, for all receptors and at all exposure points, was within the EPA target cancer risk range of 10^{-6} to 10^{-4} at all exposure points. As previously stated, fish ingestion comprised most of the cumulative risk; the primary risk drivers for fish ingestion included PCBs, chlorinated pesticides, arsenic, cobalt, and mercury. PCBs, most pesticides and many heavy metals are Reference COPCs in fish tissue in all sub-areas and are present at levels similar to those in reference locations areas beyond or upstream of the Hanford Site Study Area. For abiotic media, Reference COPCs account for the majority of noncancer

hazard and, in most cases, both chemical and radiation cancer risk in all sub-areas. Arsenic in sediment within most of the exposure points accounted for over half of the cumulative cancer risk. Of the radionuclides, carbon-14, cobalt-60 and europium-152, which are Study Area COPCs, constitute the majority of radiation cancer risk, although reference-related cesium-137, ubiquitous in all abiotic media, also contributed to cumulative cancer risk.

Table 6-85 presents a summary of HI, ILCR, and annual TEDE for each receptor and exposure point.

6.7 SCREENING-LEVEL ASSESSMENT: HYPOTHETICAL FUTURE RESIDENTIAL SCENARIO

As a component of the HHRA, and at the request of Ecology, sediment data from areas of the Columbia River that could potentially be dredged in the future were compared to MTCA and other residential soil screening levels. A discussion of this scenario was presented in Section 4.0. These screening-level comparisons are presented in Appendix A.

Results indicate that arsenic, chromium, hexavalent chromium, lithium, fluoride, nondioxin PCBs, and several chlorinated VOCs exceeded human health surface water benchmarks in one or more sub-areas of the river. A comparison of sediment EPCs to selected human health benchmarks for sediment in the Columbia River shows that EPCs of aluminum, arsenic, chromium, cobalt, iron, manganese, thallium, vanadium, cesium-137, and europium-152 exceed one or more sediment benchmarks.

Many of the COPCs in surface water and sediment that exceeded benchmarks have also been identified as risk drivers in the baseline cancer risks and noncancer hazards calculated in Section 6.5 for the various human receptors evaluated in the HHRA. As demonstrated in Section 6.5, arsenic, cesium-137, and europium-152 in shallow sediment and arsenic in surface water are primary risk drivers. Of these constituents, only europium-152 is a Study Area COPC; the other constituents are Reference COPCs.

6.8 REFERENCES

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7.0 UNCERTAINTY ANALYSIS

The uncertainty analysis is an important component of all risk assessments, because many of the input parameters may be highly variable and/or difficult to quantify. This introduces uncertainty in the baseline risk assessment. The uncertainty analysis identifies and evaluates the uncertainties associated with key parameters in the risk assessment, including the environmental concentrations, toxicity values, and exposure assumptions used to estimate the magnitude of exposure and to quantify health risks. In general, assumptions are selected and intended to be conservative by design and therefore protective of human health. However, because of numerous assumptions that are difficult to quantify, uncertainties in this baseline risk assessment may bias the risk result to either overestimate or underestimate risk to humans.

Many assumptions incorporated into this HHRA are inherently conservative (i.e., protective of human health). Therefore, the risk estimates presented in this report are generally more likely to overestimate rather than underestimate the potential risk. A discussion of the uncertainty and conservatism associated with this HHRA is provided for each of the four risk assessment components (i.e., data evaluation, exposure assessment, toxicity assessment, and risk characterization) to facilitate an understanding of the inherent limitations and uncertainties associated with this HHRA.

7.1 DATA EVALUATION

In general, uncertainties associated with data evaluation include the methodologies used to collect the samples, the analyses conducted on samples collected, the overall number of samples that are collected, the COPC selection process, and the identification of Study Area and Reference COPCs. Specific uncertainties relevant to the data evaluation are identified in Table 7-1. Primary sources of uncertainty pertinent to this component of the risk assessment process are discussed in the following subsections.

7.1.1 Analytical Data used in the HHRA

Overall, there is a large body of environmental data available for the Hanford Site Study Area with which to estimate risk. Specific to this HHRA, primary uncertainties related to the data set used in the baseline risk assessment include the combining of historical sediment and surface water data (from 2000 to 2007) with data collected in 2008 to 2010. It is possible that the older data are not representative of current conditions, and if these data have higher or lower concentrations, then they could bias the EPCs accordingly. Sediments in the Columbia River are continually being scavenged and redeposited along the entire length of the study area. Historical surface water data have the greatest uncertainty, as this dynamic medium is always changing due to differing flow conditions and changing conditions of the watershed. However, a review of the historical data used in this HHRA (collected between 2000 and 2007) suggests that surface water and sediment quality has not significantly changed over the past decade.

Some samples were collected from areas where historical information and/or modeling data indicated elevated concentrations of contaminants, such as groundwater upwelling areas. The analytical results from these focused samples may potentially introduce a conservative bias (e.g., overestimate risk) into the overall dataset.

Seasonal input of contaminants, particularly pesticides, from major tributaries may also influence contaminant concentrations in the Columbia River. For example, concentrations of pesticides and PCBs could potentially increase during the summer and early fall months due to increased runoff and soil erosion from agricultural land or seasonally low water levels. Overall, most of the surface water and sediment samples in the Lake Wallula Sub-Area, which receives sediment and surface water input from the Snake and Yakima Rivers as well as the contaminant load from upstream areas, were collected during the spring, summer, and fall months, suggesting that these data represent seasonal changes in surface water and sediment chemistry. Likewise, sampling limitations for fish (such as mobility and/or habitat and feeding preferences of species, number of samples obtained, fish age/length, etc.) can introduce uncertainty into the risk assessment with regard to representativeness of the dataset. Because Reference and Study Area samples were collected from similar timeframes, this seasonal influence is unlikely to impact the conclusion of the Study Area-Reference comparison.

There is also some uncertainty associated with analytical results for radionuclides and other constituents that were detected relatively infrequently, particularly in fish tissue, where radionuclides were detected in only a few fish tissue samples, at a frequency of less than 1%. In most cases, there was no pattern observed in detection with respect to tissue type, location, or species, and there are instances where it is possible that the positive result is actually a false-positive due to laboratory contamination or instrument error, as discussed in Section 3.6.4. Although exclusion of these radionuclides in the quantitative HHRA may potentially underestimate cumulative risk, this bias is anticipated to be relatively low, given that infrequently detected constituents are not anticipated to have a significant impact on risk for long-term exposures.

In some instances, the low FOD may be related in part to elevated LRLs for samples. Reporting limits are unique to a sample and may be influenced by matrix interferences and other issues. Elevated reporting limits may potentially represent false-negatives; i.e., the constituent is not detected but is actually present in the medium at an undetectable concentration. False-negatives could result in underestimating the number of true positive results. Because FOD was used as one means of selecting COPCs and evaluating Reference conditions, the presence of false-negatives could potentially influence the outcome of these processes. Some of the uncertainty associated with elevated reporting limits is reduced by inclusion of censored results in estimation of EPCs.

7.1.2 Selection of Contaminants of Potential Concern

Uncertainty is associated with the selection of COPCs from the total list of detected analytes. In the screening process, as described in Section 3.0, chemicals meeting specified criteria were not carried through (e.g., excluded from) the quantitative HHRA process. Selection of COPCs

streamlines the risk assessment process by focusing the HHRA on potentially significant risk contributors or drivers and thus provides the most useful information for making remedial action decisions. The COPC selection process was consistent with EPA guidance pertaining to selection of COPCs for risk assessment (EPA/540/1-89/002, *Risk Assessment Guidance for Superfund, Volume I: Human Health Evaluation Manual [Part A], Interim Final*) and followed the approach specified in the RI Work Plan (DOE/RL-2008-11). The COPC refinement process included a number of complementary steps and criteria, including consideration of a pre-selected list of contaminants that were either not excluded or included, evaluation of spatial distribution and concentration, and an evaluation of potential toxicity through a comparison of concentrations to conservative risk-based screening criteria. However, the exclusion of constituents as COPCs could cause the risk estimates to be biased low, although the magnitude of the bias is likely to be relatively small.

Many of the constituents that were detected at low frequency (less than 5% if more than 20 samples were analyzed) were eliminated as COPCs for soil, sediment, and surface water. Overall, the number of constituents eliminated by this process was small and not anticipated to significantly underestimate risk. In most instances, the constituent was detected in only one sample. Table 7-2 summarizes constituents in abiotic media and fish fillet that were eliminated due to low-frequency status (i.e., less than 5% detects of 20 or more sample results).

Undetected analytes (i.e., never detected in a medium) were also excluded as COPCs. To evaluate the potential magnitude/impact of the exclusion of these constituents, the analytical results (i.e., reporting limits) of nondetect constituents were evaluated with respect to human health screening criteria. This comparison is presented in Appendix M, Table M-1 for surface water, Table M-2 for sediment, Table M-3 for island soil, and Table M-4 for fish tissue. Results from this comparison indicate that the instances where the majority of reporting limits exceeded benchmarks in abiotic media, the constituent was either a SVOC (particularly phenols and PAHs), a chlorinated VOC, or a pesticide. As described below, due to their low prevalence, exclusion of these nondetected analytes is not anticipated to significantly bias risk estimates.

Results are summarized as follows.

- In sediment and soil, relatively few constituents had a high frequency of reporting limits exceeding benchmarks. Constituents with reporting limits that consistently exceeded benchmarks mainly included SVOCs, such as PAHs. Note that PAHs that were detected in these media were detected in only a few of the several hundred samples analyzed.

- In sediment and soil, VOCs are unlikely to be retained in sediment due to the low organic carbon content of the sandy sediment substrate and high solubility and/or volatility of this class of compounds. Therefore, the elevated reporting limits for sediment and soil are unlikely to produce a false-negative result (i.e., result is reported as nondetect when the constituent is actually present).

- In soil, only europium-152 had reporting limits consistently higher than benchmarks. This radionuclide is an inclusion list constituent. Its isotope, europium-154, was detected infrequently in sediment samples.

- In surface water, SVOCs and several chlorinated VOCs had numerous reporting limits exceeding benchmarks. However, none of these analytes are inclusion list analytes, and the general low prevalence of VOCs/SVOCs in surface water samples suggest that surface water results do not underestimate the presence of these constituents. As a class of constituents, SVOCs (including pesticides) are not expected to be detected frequently in surface water due to their low water solubility.

- In fish tissue, beryllium, toxaphene, and several radionuclides had reporting limits exceeding benchmarks 100% of the time. Of the radionuclides, cobalt-60, europium-152, and uranium-235 are inclusion list constituents.

For detected constituents, concentrations were also screened against various medium-specific benchmarks as a means of selecting COPCs. The benchmarks used in this analysis for surface water, sediment, and soil are based on residential exposure scenarios (e.g., residential soil screening levels were used for sediment recreational exposures) and are generally conservative values to use as screening tools. However, the assumptions underlying the fish tissue benchmarks (which use the consumption rate unique to the Avid Angler scenario) may be less conservative for the Yakama Nation scenario evaluated in this HHRA, which encompasses a relatively high fish ingestion rate (approximately twice that of the Avid Angler scenario) and assumes that, in addition to fillet, organ meat is also consumed. The use of the Avid Angler ingestion rate for fish tissue benchmarks results in the exclusion of a few additional constituents as COPCs, which could potentially underestimate cumulative hazard and risk for the Yakama Nation scenario. However, their exclusion does not change the outcome of the risk assessment, as the primary risk drivers in fish tissue (e.g., PCBs, mercury, pesticides) result in noncancer hazard and cancer risk exceeding risk management criteria.

As provided for in the EPA risk guidance (EPA/540/1-89/002), the COPC selection process resulted in a number of constituents eliminated as COPCs. Some of these constituents at individual locations may be present at concentrations greater than Reference/OCI areas and, in some instances, greater than human health screening levels.

In surface water, several contaminants not retained as COPCs in Table 3-20 have elevated concentrations in the surface water but were not included in the risk calculations. These contaminants have elevated concentrations in specific locations, relative to upriver locations. Nonradiological and radiological surface water contaminants are summarized in Table 7-3. These results are noted as examples, because the concentrations at these locations either exceed risk-based levels for drinking water or surface water; or the concentrations are high enough that, when added (as risk or hazard quotient) with that of other contaminants having similar targets, could potentially influence the total risk and/or hazard levels. Table 7-4 summarizes constituents in fish tissue that are also noted as examples because these contaminants have elevated

concentrations relative to upriver locations, and could potentially influence the total risk and/or hazard levels associated with fish consumption.

The risk assessment, however, does not characterize risk for all contaminants detected on a point-by-point basis, but instead relies on estimates of exposure based on the 95% UCL, which takes into account point locations with elevated concentrations. The UCL represents, with 95% confidence, the upper bound of true mean concentration, which is appropriate for characterizing long-term, chronic exposures for a population. For surface water, which is a dynamic medium, characterization of risk based on a single sample result is not appropriate for evaluating long-term exposures. Therefore, although elimination of some detected contaminants as COPCs may underestimate risk, the overall effect is expected to be low and not influence the conclusions of the HHRA. This allows the HHRA to focus on those constituents that present the greatest potential risk.

7.1.3 Comparison to Reference Conditions

Following the COPC selection process, COPCs from each of the sub-areas (100 Area, 300 Area, and Lake Wallula) were evaluated and identified as either "Reference COPCs" or "Study Area COPCs." It is worth reiterating that this evaluation determined only whether a COPC was present at (1) concentrations indistinguishable from Reference/OCI areas ("Reference COPC") or (2) concentrations were elevated with respect to reference/OCI areas ("Study Area COPC"). Accordingly, the designation as a "Study Area COPC" does not necessarily indicate that the presence of that chemical is directly related to Hanford Site releases, but simply that the concentration was higher in that sub-area relative to RfCs. This is described in more detail below. Furthermore, these comparisons did not eliminate any COPCs from further evaluation in the baseline risk assessment, but were used to distinguish COPCs such that appropriate recommendations could be made for their further evaluation if they were found to be associated with excess risk.

Contaminant concentrations from the three sub-areas of the Hanford Site Study Area were compared to contaminant concentrations from reference areas not impacted by the Hanford Site activities (i.e., Reference/OCI locations). These locations were either upriver or were from OCIs such as tributaries and wasteways/irrigation ditches. Two-tailed statistical tests were used for this comparison (see Section 3.8) to assess whether contaminant concentrations in the Study Area were higher or lower than those in Reference areas. A one-tailed test could have been employed to assess only whether Study Area concentrations were higher than those of Reference. One-tailed tests generate p-values that are one-half that of the equivalent two-tailed test. Therefore, certain comparisons that resulted in a p-value of between 0.05 and 0.1 that were previously classified as Reference COPCs would have instead been classified as Study Area COPCs, as the p-value would now fall below the alpha threshold of 0.05. This would potentially increase the cumulative risk/hazard attributed to Study Area COPCs. A review of the statistical comparison tables presented in Section 3.0 indicates that the number of Reference COPCs with (1) a p-value between 0.05 and 0.1, and (2) mean concentrations higher in the Study Area (but not a statistically significant difference), but that would have been classified as Study Area

COPCs using a one-tailed test, is very small, and all of the occurrences are in fish tissue (mainly in carp carcass in the 300 Area and Lake Wallula). These occurrences include the following:

- Mercury: 300 Area fish carcass, carp fillet and carp carcass
- Selenium: Lake Wallula fish (all species combined) fillet
- Beta-HCH: Lake Wallula fish (all species combined) carcass
- Total nondioxin PCBs: carp carcass
- Total dioxin-like PCBs: sucker carcass

For COPCs with a low FOD or a low overall number of samples, qualitative analyses were performed to evaluate COPCs that are consistent with reference conditions. Where use of statistical comparisons was not supported and there was no clear determination of whether a constituent was consistent with reference conditions, the OCI and Reference data were evaluated in detail and a protocol adopted to assign a constituent as either a Study Area or Reference COPC. This protocol resulted in the determination that most COPCs were "consistent with Reference" and thus defined as "Reference COPCs." Although the assignment of a COPC to the Study Area or Reference category does not affect the total risk values (which reflect cumulative risk across all Study Area and Reference COPCs), this assignment could impact risk management decisions. Misclassification of a COPC as a Reference COPC may result in elimination of a contaminant from future remedial actions. Conversely, misclassification of a COPC as a Study Area COPC may potentially result in costly and ineffective remedial decisions.

A case in point is PCBs, which were categorized as a "Reference COPC" throughout the Hanford Site Study Area in surface water, sediment, and fish, with the exception of sucker and 100 Area and 300 Area nonfillet tissues. It is widely accepted that PCBs are present globally, even in remote rural areas. The data appear to support this conclusion for the Hanford Site Study Area.

For example, PCBs were identified as a major risk driver in fish tissue. However, evaluation of the distribution of PCB congener data (discussed in Section 3.5.7) showed that PCB composition was very similar among all sub-areas including OCIs and other Reference areas and that there was a relatively narrow range of detected concentrations along the entire river. The conclusion was that PCBs in fish tissue and other media in the 100, 300, and Lake Wallula Area Sub-Areas were present at concentrations consistent with those in Reference/OCI areas.

Because there are numerous sources, both natural and anthropogenic, of contaminants to the Columbia River, it is important to stress that a "Study Area COPC" may not be directly related to a Hanford Site release, but that the constituent is present only at relatively higher concentrations than in other areas of the river. In some instances, few data were available to support standard comparative statistics, and so professional judgment was employed to identify a COPC as either "Study Area" or "Reference." (Refer to Section 3.8 for a more detailed discussion of COPC selection.) This often occurred for pesticides in fish tissue that were infrequently detected, and often the magnitude of difference between Upriver and Study Area concentrations was very small.

Based on this assessment, the existing data set adequately characterizes potential exposures within the Hanford Site Study Area. In this HHRA, uncertainty due to limited sample results or detection status has been offset by incorporating conservative assumptions into the exposure assessment and risk characterization when possible.

7.2 EXPOSURE ASSESSMENT

In general, estimation of EPCs (including calculation of arithmetic mean and UCL concentrations), characterization of current and reasonably foreseeable site activities and uses, and calculation of ADDs contribute most to the uncertainty in the exposure assessment component of the risk characterization. To counter this uncertainty, health-protective exposure assumptions based on either site-specific information or conservative default values provided in EPA and other guidance were used to quantitatively evaluate potential risks posed by the Hanford Site.

Perhaps one of the largest areas of uncertainty is that associated with estimating activity patterns for human recreational exposures. In this HHRA, it is assumed that a receptor is exposed to an entire exposure point, which in this case may encompass an area much broader than what is typically encountered (such as a small beach or boat ramp) during each exposure event. Furthermore, although this HHRA relied on Columbia River survey data (PNNL-13840, *2001 Columbia River Recreation Survey – Implications for the Hanford Site Integrated Assessment*) for estimating recreational activity exposure factors (such as number of hours spent at the beach or number of trips made), these results may not always accurately reflect activity patterns for a specific portion of the population or for a discrete location such as a beach.

Exposure point concentrations estimated for the Hanford Site Study Area represented a broad area of exposure, and so there is some uncertainty related to whether the EPC adequately characterizes the level of a COPC to which a receptor is routinely exposed. However, each sub-area was divided into separate exposure points to allow distinct evaluation of areas with relatively elevated concentrations of COPCs. Furthermore, for RME exposures, 95% UCLs or maximum concentrations were used as EPCs. These upper-bound metrics are intended to conservatively estimate exposure throughout an area with variable levels of contamination. In some instances, EPCs were based on only a few detected concentrations, among many nondetect results. Overall, EPCs used in this assessment are conservative and likely overestimate risk.

There is also some uncertainty in estimating EPCs in a dynamic system such as a river, where the nature, extent, and level of contamination may change over time as surface water continuously moves downstream and sediments are transported or buried. Comparison of historical data to the data used in this HHRA suggests that concentrations of many contaminants like metals have decreased over the past two decades, and as some sources of contaminants, such as upstream mines, have been removed, one would anticipate this trend to continue. Radionuclides degrade with time, and many of the radionuclide COPCs have a half-life of 30 years or less. Over a 30-year exposure duration, one would therefore expect the activity of radionuclides in surface water and sediment to decrease considerably. Examples of this are shown in Table 7-5, which

presents the percentage of radionuclide COPCs remaining after 30 years, which is the exposure duration for the Casual User and Avid Angler scenarios. Thus, use of current data to estimate EPCs for long-term scenarios likely overestimates risk.

This HHRA is deterministic, relying on point estimates of exposure. In general, a mixture of conservative and mid-range exposure assumptions were used in order to derive realistic, yet protective, estimates of exposure. This risk analysis included evaluation of the RME for each receptor as well as evaluation of CTE conditions. The RME exposure assumptions (including receptor-specific variables, such as ingestion rate of fish, soil, sediment, or water) reflect upper-bound or maximum values and are intended to be conservative, thus likely overstating risks for most of the general population. Typically, the upper-bound assumptions used to quantify doses and risks for the RME scenario make it unlikely to underestimate risks for the evaluated receptors.

Quantification of risk for both RME and CTE scenarios also provides insight on the variability in exposure that may be experienced by a particular receptor. However, reliance on single-point estimates can potentially over- or underpredict exposure and, hence, estimated risk.

Specific examples of uncertainty in the exposure assessment are presented in Table 7-6.

7.3 TOXICITY ASSESSMENT

The primary sources of uncertainty in the toxicity assessment are associated with toxicity values used to quantify risks. These uncertainties include (1) extrapolation of toxicity information from effects observed at high doses to predict adverse effects at low concentrations/activity levels anticipated for human exposure to environmental contaminants, (2) use of toxicity information compiled from short-term exposure studies to predict the effects associated with long-term exposures (and vice-versa), (3) use of toxicity information from animal studies to predict likely effects in humans, and (4) use of toxicity information based on homogeneous animal populations or healthy human populations to predict the effects that are likely to be observed in the general population (including sensitive subgroups). Human variability in response to chemical exposures may be dependent on numerous factors, and risks estimated for one population may not necessarily be protective or indicative of risks in a different population.

However, the toxicity values used in the calculation of noncancer hazards and cancer risk estimates are, for many of the COPCs, very conservative values. Reference doses and RfCs are derived using a number of safety factors (e.g., up to several thousand) and are developed in order to protect sensitive populations. Therefore, the actual dose or concentration associated with a health effect is likely to be higher than the dose or concentration established by EPA for evaluating risk in most groups within the general population.

However, toxicity values for other COPCs such as arsenic, hexavalent chromium, methylmercury, and radionuclides may be less conservative due to a number of factors. In some instances, smaller uncertainty factors (which are protective factors applied to benchmark doses to

account for uncertainties related to use of animal data, subchronic studies, and other factors) are used in the development of toxicity values. In the case of methylmercury RfD (0.0001 mg/kg-day; IRIS 2012), this value was derived based on human epidemiological data using an uncertainty factor of ten to account for toxicokinetic variability in ingested dose estimation and pharmacodynamic variability and uncertainty (IRIS 2012; NRC 2000; Rice et al. 2003). Although a lower uncertainty factor reflects a higher confidence in the toxicity value, less protective assumptions used in the RfD's derivation may result in a lower bias for estimates of noncancer hazard.

In other cases, more recent studies suggest that a more conservative toxicity value is warranted. For example, the CSF for arsenic used in this HHRA (1.5 per mg/kg-day) is based on the IRIS review dated 1998 (EPA 2012). However, more recent studies indicate that this value likely underestimates risk of internal cancers by approximately an order of magnitude (EPA/635/R-10/001; NRC 1999, 2001). The IRIS carcinogenicity assessment is currently under development by EPA, which has proposed a potency factor of 25.7 per mg/kg-day, approximately 20 times more conservative than the current CSF.

Radionuclide CSFs are based on human population data, dosimetry, and biokinetic models (EPA 2001) and represent central estimates of the mean, rather than upper-bound estimates characteristic of slope factors for nonradionuclides. The linear dose-response used for chemical carcinogens was adapted from radiogenic exposures on human populations (EPA/402-R-99-001, 1999 and 2002); therefore, there is less uncertainty associated with these cancer potency values, but these values may consequently be less conservative. Additional uncertainties associated with the toxicity values used in this HHRA are summarized in Table 7-7.

7.4 RISK CHARACTERIZATION

Specific uncertainties in the risk characterization are summarized in Table 7-8. The primary source of uncertainty in the risk characterization section is the assumption of simple additivity of toxicity when calculating cumulative risk across COPCs. Equal weight is given to the toxicity of each COPC, even though the basis of the toxicity values may vary considerably, particularly for RfDs/RfCs that may have different confidence levels, different endpoints, and different modes of action. In actuality, chemical mixtures may result in additive, synergistic, or antagonistic effects, or the toxicity of a constituent may even be independent of that of other constituents. For chemical CSFs, which are based on 95% UCLs, simple additivity across COPCs may result in overestimates of risk.

7.4.1 Uncertainty Related to Identification of Study Area and Reference Contaminants of Potential Concern

There is some uncertainty associated with quantification of cumulative hazard and risk associated with Study Area and Reference COPCs. As discussed in Section 3.8, all COPCs were designated as either Study Area or Reference COPCs, based on either a qualitative or quantitative evaluation of analytical results (i.e., the "reference comparison"). Cumulative risk

was evaluated for all COPCs (both Study Area and Reference, combined), but was also discussed separately for Study Area and Reference COPCs.

For the fish ingestion pathway, which evaluated consumption of three different tissue types (fillet, carcass, and liver/kidney), the reference comparisons sometimes yielded conflicting results among the multiple tissue types. That is, Study Area concentrations of a constituent in one tissue type could be consistent with RfCs, while the concentrations in the two other tissue types may have been higher in the study area samples (e.g., see Tables 3-67, 3-75, and 3-82). To resolve these discrepancies, the final COPC designation for fish tissue (considered as one complete exposure medium) was based on fillet results, since this portion of the fish is assumed to comprise the bulk of the diet (see Section 3.8.1.6). This approach resulted in the categorization and evaluation of select constituents in fish carcass and liver/kidney as "Reference COPCs," although they would otherwise be considered to be "Study Area COPCs" based on their concentrations in individual tissue types. This approach may potentially result in an underestimate of the cumulative risk attributed to Study Area COPCs and an overestimate of risk attributed to Reference COPCs. (This approach had no impact on overall cumulative risk, however.) However, as indicated above, the relative consumption rate of these tissue types compared to fillet, is small (5% or less).

The constituents for which this bias may exist (pesticides, PCBs, and metals) are highlighted in Tables 3-67, 3-75, and 3-82 for the 100 Area, 300 Area, and Lake Wallula, respectively. Note that these constituents were not identified as Study Area COPCs in either sediment or surface water. PCBs and pesticides are considered "legacy" pesticides that are persistent and ubiquitous at low levels throughout the world as a result of their past use in industrial and agricultural settings, such as those that exist adjacent to/along the Columbia River.

For each of the three sub-areas, the following COPCs were designated as "Study Area COPCs" in carcass and/or liver and kidney, although not in fillet:

- 100 Area: PCBs (nondioxin and dioxin-like), pesticides (beta and gamma-HCH, dieldren, and heptachlor), arsenic, cadmium, and cobalt. The result of the reference comparison was based on a qualitative evaluation for most of the pesticides, arsenic. and cobalt.

- 300 Area: Pesticides (delta- and gamma-HCH, dieldrin) and dioxin-like PCBs. The result of the reference comparison for all of the pesticides was based on a qualitative evaluation.

- Lake Wallula: Beta- and gamma-HCH, dieldren, and arsenic. The results of the reference comparison for all but beta-HCH were based on a qualitative evaluation.

To evaluate how the decision to classify these compounds as Reference COPCs (based on fillet concentrations) might bias cumulative risk for Study Area COPCs, hazard and risk related to fish consumption were calculated for both the Avid Angler and Yakama Nation scenarios for these specific COPCs in carcass and liver/kidney. These calculations are presented in Appendix O, and the resulting hazard and chemical cancer risks are summarized in Table 7-9.

Cumulative hazard and risk from all Study Area and Reference COPCs (as identified in Section 3.8) are also summarized on this table for comparison.

This evaluation indicates that the cumulative hazard and risk from *Study Area COPCs*[1] may potentially be underestimated in the 100 Area and the 300 Area, primarily as a result of treating the identified PCBs, pesticides and metals in liver, kidney and carcass as Reference COPCs. Cumulative hazard from Study Area COPCs in Lake Wallula is only slightly underestimated as a result of classifying dieldrin and beta-HCH in liver/kidney as Reference COPCs. However, when compared to cumulative risk from all COPCs (i.e., both Study Area and Reference), the contribution from these compounds in nonfillet tissue is relatively small (i.e., less than 15%). PCBs and pesticides (as Reference COPCs) in fillet comprise the majority of the hazard and chemical cancer risk. Therefore, compared to overall cumulative risk, the decision to categorize several constituents in nonfillet tissues as Reference COPCs, despite their having higher concentrations in the Study Area, does not have an overall impact on the overall conclusions of the risk assessment.

7.4.2 Uncertainty Related to Elimination of Detected Constituents as Contaminants of Potential Concern

There is also some uncertainty associated with estimation of cumulative hazard and risk, due to elimination of detected constituents from the COPC selection process. Cumulative risk addresses only COPCs that were quantitatively evaluated in the risk assessment. Other detected constituents that were ruled out during the data evaluation process (due to low frequency or concentration below human health-based benchmarks, or presence on the exclusion list, for example) were not assessed as COPCs, and so risk related to these constituents is not accounted for.

However, as illustrated in the discussion of risk in Section 6.0, typically only one or two contaminants (e.g., PCBs in fish, arsenic in soil/sediment) typically represent 80% or more of the cumulative hazard or risk. Therefore, it is likely that excluding low-frequency or low-toxicity constituents will have little overall effect on the estimated risk value. However, the risk characterization does focus on those constituents identified early on in the risk assessment that are prevalent, highly toxic, and/or most likely to pose a potential human health risk.

Reliability of analytical results may also introduce uncertainty into assessing cumulative risk. In some instances, suspected laboratory-induced data quality issues (e.g., cross-contamination) led to the elimination of five detected radionuclides (tritium, plutonium-239/240, strontium-90, cesium-137, and technetium-99) from the HHRA. Six radionuclides were detected in fish tissue; of these, only carbon-14 was carried through as a COPC, because the analytical results for the other five radionuclides were suspected to be false-positives and so were deemed unusable for the HHRA (see Section 3.6.4.4). If these radionuclides are actually present in fish tissue, however, their exclusion as COPCs could potentially underestimate cumulative cancer risk.

[1] It is important to note that the reference comparison results do not impact the *overall cumulative hazard or risk,* which is calculated for all receptors and exposure points based on both Study Area and Reference COPCs.

Radiation cancer risk associated with consumption of these five radionuclides in fish tissue is calculated in Appendix O and is summarized in Table 7-10. This table also summarizes the cancer risk associated with ingestion of carbon-14 in fish, as well as the cumulative cancer risk from all COPCs (chemical and radiological) evaluated in the HHRA.

The ingestion of tritium, plutonium-239/240, strontium-90, cesium-137, and technetium-99 in fish tissue results in a cumulative cancer risk ranging from 1×10^{-5} in Lake Wallula to 4×10^{-3} in the 100 Area, based on the Yakama Nation scenario. In the 100 Area, plutonium and strontium-90 in fillet are the primary risk drivers; and cesium-137 and tritium are the major risk drivers in the 300 Area and Lake Wallula, respectively. These cumulative cancer risks are similar to or higher than those attributed to carbon-14 in fish tissue. Relative to cumulative risk from all COPCs, however, risk from these radionuclides is much lower than that associated with PCBs and pesticides in all sub-areas. Furthermore, it should be noted that these risks are in many instances based on maximum detected concentrations (or activities), where a constituent was rarely detected, and therefore may not realistically characterize long-term exposures.

Additional uncertainties in the risk characterization are summarized in Table 7-6.

7.5 UNCERTAINTY ANALYSIS SUMMARY

As previously discussed in Sections 7.1 through 7.4, many of the assumptions and parameters used in this HHRA are intended to be conservative and therefore overestimate potential health risk. As indicated in Section 6.0, the fish ingestion pathway contributes to nearly all of the risk for the Avid Angler and Yakama Nation scenarios. Given the uncertainty regarding consumption rates specific to the Columbia River, use of upper-bound estimates of fish ingestion used in the HHRA likely overestimates risk for the Hanford Site Study Area. Polychlorinated biphenyls, mercury, and other metals, and chlorinated pesticides in fish tissue are primary risk drivers. These types of contaminants are prevalent in fish tissue in many waterbodies, due to their widespread historical use, atmospheric deposition, and, consequently, high prevalence in abiotic media. Releases from the Hanford Site are not a major contributor of these COPCs, if at all.

For other media, assumptions regarding how often a human receptor visits the river, and in what types of activities a human receptor would be anticipated to participate, largely influence the level of exposure and estimated risk. Because this HHRA uses a deterministic approach (rather than evaluating a distribution of exposure) in assessing risk, the risks generated do not take into account all members of a population. However, the point estimates of different physiological attributes, activity patterns, toxicity values, and EPCs used to characterize exposure and toxicity are designed to be protective of sensitive subpopulations or members of a population who are likely to have an enhanced exposure potential. Also, both CTE and RME scenarios were evaluated for receptors other than the Yakama Nation scenario, for which only one scenario was provided. Assessment of both CTE and RME helps to evaluate the potential range of health risk estimates, even though a deterministic approach was used in each. Furthermore, when using high-dose toxicity data to predict risks for a chemical exposure at low doses or using animal data

to extrapolate to human populations, there is considerable uncertainty whether any disease will actually occur due to exposure, and may result in significant overestimate of potential risks.

In light of these uncertainties, the risks calculated are *estimated* risks. Therefore, it is emphasized that the risks generated in this evaluation are *hypothetical*, not actual, and are by design intended to be conservative (tend to overestimate actual risks). By using this conservative approach in developing risk estimates, it is expected that the calculated risk estimates are likely to result in upper-bound estimates of actual Study Area-related risks/hazards. Consequently, these estimates should be used to highlight areas of potential concern and to assist in providing practical risk management information rather than be considered as absolute estimates of health risks.

7.6 REFERENCES

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8.0 CONCLUSIONS

The purpose of this baseline HHRA was to characterize the nature and magnitude of risk associated with exposure to chemical and radiological COPCs[1] in surface water, sediment, island soil, and/or fish tissue, and to identify risk drivers that are attributed to the Hanford Site. The Columbia River is widely used for various recreational and tribal purposes as well as a source of potable water for the City of Richland (following filtration and treatment). Therefore, it is important to understand whether the presence of contaminants in river media poses a potential health risk to the human receptors that may be exposed at a level exceeding established EPA and MTCA risk management criteria. In this HHRA, cumulative noncancer hazards and cancer risks were evaluated for three of the four receptor scenarios. These included the following:

- The Casual User scenario, which includes both adults and children who use the river for seasonal recreational purposes;
- The Avid Angler scenario, which includes both adults and older children who engage in fishing activities and younger children who consume fish that are brought home; and
- The Yakama Nation scenario, which includes local and regional tribes who have ties to the Hanford Site Study Area and surrounding lands.

Additionally, at the request of Ecology, a comparative analysis was conducted in which sediment and surface water EPCs were compared to risk-based residential benchmarks. This analysis was used to evaluate a hypothetical future residential scenario, which addressed hypothetical residential exposures to possibly dredgeable sediments from within Lake Wallula as well as exposures to river water (unfiltered and untreated) assuming its potential future use as a potable water source.

8.1 RECREATIONAL AND TRIBAL SCENARIOS: QUANTITATIVE RISK ASSESSMENT RESULTS

For the Casual User and Avid Angler scenarios, both RME and CTE risks were evaluated in accordance with EPA guidance. Only one scenario, based on exposure assumptions provided by *Yakama Nation Exposure Scenario for Hanford Site Risk Assessment, Richland, Washington* (Ridolfi 2007) and upper-bound EPCs, was estimated for the Yakama Nation scenario. The risks evaluated for these three exposure scenarios – Casual User, Avid Angler, and Yakama Nation – reflect varying levels of exposures for typical activities that would be expected to occur along the Columbia River, ranging from swimming, picnicking, and water sports to subsistence fishing.

[1] As described in this HHRA (see Section 3.0), this assessment included quantitative evaluation of potential health risks for COPCs that were classified as both "Study Area COPCs" (if the constituent concentrations within the Hanford Site Study Area were higher than those in Reference/OCI areas) and "Reference COPCs" (if constituent concentrations within the Hanford Site Study Area were consistent with or lower than those of Reference/OCI areas).

Potential noncancer hazards, chemical and radiation cancer risks, and annual TEDE were quantified for these three scenarios. In addition to quantifying the cumulative hazard/risk for COPCs, the HHRA also separately evaluated the noncancer hazard and cancer risk associated with Study Area and Reference COPCs. As previously discussed, because there are numerous sources of contaminants both natural and anthropogenic, a Study Area COPC may not necessarily be directly related to a Hanford Site release. A Study Area COPC designation indicates only that the constituent is present at a concentration higher than that in reference/OCI areas of the river, and does not assign a specific source to the reported constituent.

Table 6-85 summarized cumulative noncancer hazard, cancer risk, and radiation dose for RME and CTE conditions (if applicable) in the three evaluated receptors. Noncancer hazards and cancer risks were compared to the following EPA and MTCA risk management criteria:

- EPA and MTCA cumulative HI of 1
- The EPA cancer risk range of 10^{-6} to 10^{-4}
- The MTCA cancer risk limit of 1×10^{-5}.

Additionally, annual TEDE was compared to a radiation dose threshold of 15 mrem/year. For scenarios where cumulative hazard, risk or dose exceeded these risk management criteria, risk drivers were identified. Risk drivers are the COPCs that resulted in an HI greater than 1 or an ILCR greater than 1×10^{-6}.

Results of the HHRA indicate that ingestion of certain recreational finfish species may pose a health risk exceeding both EPA and Ecology risk management criteria, and that noncancer hazards and cancer risks are primarily due to PCBs, pesticides, and several metals, which were detected in most, if not all fish tissue samples. As discussed in Section 6.0, risk from the fish ingestion pathway alone contributed more than 99% of the cumulative risk for both the Avid Angler and Yakama Nation scenarios[2]. The annual TEDE for these receptors (i.e., Avid Angler and Yakama Nation) was below the radiation dose threshold of 15 mrem/yr.

In the three sub-areas (i.e., 100 Area, 300 Area, and Lake Wallula), the risk drivers in fish tissue that produced the highest risk estimates include PCBs, chlorinated pesticides, arsenic, cobalt, mercury, and carbon-14. Among these, PCBs contributed up to 80% of the cumulative cancer risk. Polychlorinated biphenyls were identified as Reference COPCs (as discussed in Section 3.8) across all sub-areas (for the combined species analysis) and in most of the individual fish species (except for sucker). With a few exceptions, pesticides were also identified as Reference COPCs. Mercury was identified as a Study Area COPC for the 100 Area and 300 Area (combined species analysis) and in sturgeon and whitefish (individual species analysis).

[2] The Casual User scenario did not include ingestion of fish, focusing instead on contact with sediment, surface water, and island soil during recreational activities.

Table 8-1 summarizes the Study Area COPCs that are risk drivers in fish tissue. Carbon-14 was the only radionuclide consistently detected among fish tissue samples, although at an overall low frequency; carbon-14 was also only sporadically detected in abiotic media. For other Study Area COPCs, such as mercury and PCBs, although mean concentrations of these types of constituents were statistically higher in the Study Area relative to upriver concentrations, this difference was oftentimes relatively small (e.g., the Study Area mean was marginally greater than that of Upriver, and the detected concentrations in the Study Area were within the range of detected Upriver concentrations). Factors, such as small sample sizes or low FOD contribute considerable uncertainty into determining whether these contaminants in the Study Area are truly different from Upriver areas, or even regional concentrations. The major risk drivers in fish tissue (particularly PCBs, pesticides, and mercury), and magnitude of risks, are similar to those identified in other risk assessments conducted along the Columbia River (see Section 3.6.4) as well as across the United States, reflecting the ubiquitous occurrence, environmental persistence, and prevalence of these types of contaminants.

Noncancer hazards and cancer risks from abiotic media (i.e., surface water, sediment, and island soil) were substantially lower than those identified for the fish ingestion pathway for all receptors. For the Casual User and Avid Angler scenarios, both CTE and RME cancer risks for abiotic exposure pathways were within or below the EPA cancer risk range of 10^{-6} to 10^{-4} and below the MTCA cumulative cancer risk limit of 1×10^{-5}. Noncancer hazard for the RME scenario for these receptors was at or below the EPA and MTCA noncancer HI of 1 in all sub-areas. Radiation dose for these receptors was also below the radiation dose threshold of 15 mrem/yr.

The Yakama Nation receptor is assumed to have a higher level of exposure to river media than either the Avid Angler or Casual User receptors, and the estimated cumulative noncancer hazard and cancer risk for the Yakama Nation receptor are thus higher. Excluding the fish ingestion pathway, noncancer hazard related to sediment, soil, and surface water exposures across all sub-areas was above the EPA target risk HI of 1, primarily due to cobalt, thallium, iron, and arsenic in sediment and/or island soil. With the exception of arsenic in island soil, these metals are Reference COPCs in both sediment and soil. Cumulative cancer risk from abiotic media exceeded the MTCA cumulative cancer risk limit at each exposure point for the Yakama Nation scenario, although was within the EPA cancer risk range of 10^{-6} to 10^{-4}. For most exposure points, risk was primarily attributed to Reference COPCs, primarily associated with arsenic and cesium-137 in sediment. Radiation dose for the Yakama Nation scenario was below the radiation dose threshold of 15 mrem/yr at all exposure points.

Study Area COPCs that are identified as risk drivers in abiotic media are summarized in Table 8-2. These risk drivers are based on the Yakama Nation scenario, which has the highest level of exposure among the three scenarios evaluated.

The COPCs in abiotic media that contributed to the majority of risk for all receptors evaluated in this HHRA included arsenic in sediment and island soil, as well as several radionuclides including cesium-137, cobalt-60, and europium-152. Cesium-137 is a Reference COPC in all media and in all sub-areas. Arsenic is a Reference COPC in sediment, prevalent in all media at

relatively consistent concentrations throughout the Hanford Site Study Area as well as in Reference areas. However, arsenic is categorized as a Study Area COPC in island soil in the 100-B and 300-A/B exposure points, due to a higher concentration of this constituent on islands within the Hanford Site Study Area, relative to Reference locations. This condition may be related to the fact that a relatively few Upriver (i.e., Reference) island soil samples were collected, preventing a more robust statistical analysis as well as potential dissimilarity of island soil types between Reference and the Hanford Site Study Area. Hanford Site Study Area island soils are characteristic of reworked sediments, while the Reference Area soils appear to be more similar to upland soil as opposed to river sediments. The concentrations of arsenic observed in island soil samples from 100 Area and 300 Area islands are similar to published background arsenic concentrations for the state of Washington (Ecology 1994), as well as to the concentrations observed in sediment throughout the Hanford Site Study Area and Reference locations.

Cobalt-60 and europium-152, both Study Area COPCs, were reported at a relatively low FOD and at variable concentrations in sediment and island soils throughout the Hanford Site Study Area. These findings may reflect the presence of minute, random particles from historical Hanford Site operations.

8.2 HYPOTHETICAL FUTURE RESIDENTIAL SCENARIO: SCREENING-LEVEL EVALUATION RESULTS

As part of this HHRA, EPCs were compared to Ecology and EPA surface/groundwater benchmarks, which included federal drinking water standards and human health risk-based screening levels for surface water. This evaluation is presented in Appendix A. The results of this comparison indicate that no COPC had an EPC exceeding federal drinking water standards, although all surface water COPCs had EPCs exceeding one or more of the other screening levels used in this comparison (such as EPA Regional Screening Levels for tap water). The 300 Area exposure point had the most benchmark exceedances.

At the request of Ecology, sediments within Lake Wallula shipping channels that may potentially be dredged in the future were also evaluated with respect to residential soil screening criteria, assuming that dredged sediments could be placed in upland areas. The results of the screening level comparison are, in general, consistent with the findings of the quantitative risk assessment (i.e., COPCs that contributed to the majority of the risk, as identified in Section 6.0, were often the COPCs that exceeded residential soil benchmarks).

As described in Appendix A, the benchmarks used in these comparisons were very conservative, not consistent with current or likely uses of these (untreated) surface water or sediments, and should not be inferred to represent the magnitude of the potential for human health risk. Surface water that is used for potable purposes is treated prior to distribution by the cities in the immediate vicinity, and treated water meets federal drinking water standards. Furthermore, any dredging activity within the Columbia River is highly regulated by both the ACOE and the state

of Washington. Because of this, it is unlikely that contaminated sediments from the Hanford Site Study Area would be used in residential settings.

8.3 UNCERTAINTIES IN THE RISK ASSESSMENT

As discussed in Section 7.0, there are a number of uncertainties inherent in the analytical data, exposure assumptions, and toxicity values used to quantify human health risks. In general, many of the assumptions and parameters used in this HHRA are intended to be conservative and therefore overestimate potential human health risk, particularly for the RME scenarios. Perhaps one of the largest uncertainties is that related to fish consumption. These uncertainties include the amount of fish caught within the Hanford Site Study Area and its consumption, the types of fish consumed, and portions (i.e., fillet, organs, or whole fish) of the fish typically prepared and eaten.

8.4 OVERALL CONCLUSIONS

This baseline HHRA provided a comprehensive assessment of potential health risks associated with recreational and Tribal exposures to surface water, sediment, island soils, and fish tissue within the Hanford Site Study Area.

As previously detailed in Sections 8.2 and 8.3, many of the COPCs that comprise the majority of cumulative risk in this HHRA are Reference-related; in other words, these COPCs, which primarily include arsenic and cesium-137 in surface water and sediment, and PCBs, heavy metals, and pesticides in fish tissue, are present at concentrations within the Hanford Site Study Area that are comparable to those in other portions of the Columbia River that have not been impacted by Hanford Site releases. Analytical results have demonstrated that these types of constituents are ubiquitous in the environment.

Overall, the results of the quantitative risk characterization for the Recreational User and Avid Angler scenarios indicate that exposure to Study Area COPCs in surface water, island soil, and sediment does not result in risk/hazard exceeding MTCA or EPA risk management criteria, and that for the fish ingestion pathway, the overall contribution to cumulative risk from the Hanford Site is relatively minor when compared to risk related to Reference COPCs. However, the cancer risk attributed to Study Area COPCs in abiotic media exceeds the MTCA risk limit for the Yakama Nation scenario, mainly due to arsenic and radionuclides (cobalt-60, europium-152) in soil and sediment.

8.5 REFERENCES

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Ridolfi, 2007, *Yakama Nation Exposure Scenario for Hanford Site Risk Assessment, Richland, Washington*, Prepared for the Yakama Nation Environmental Restoration and Waste Management Program by Ridolfi, Inc. Available at http://www5.hanford.gov/arpir/?content=findpage&AKey=DA06587583.

APPENDIX OVERVIEW

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Due to the size and content of most of the appendices associated with this document, a number of them are contained only on the CD attached to the back cover. For clarity, see the list below for an explanation of what can be found as either hard copy or electronic copy.

APPENDICES

A SCREENING LEVEL ASSESSMENT: HYPOTHETICAL FUTURE RESIDENTIAL SCENARIO **(On CD only)**

B HUMAN HEALTH RISK ASSESSMENT DATABASE **(On CD only)**
Hard copy Introduction; see folder for Excel Users Guide and Access database file

C ABIOTIC AND FISH TISSUE SAMPLE LOCATION MAPS (**Hard copy and on CD**)

D PRO-UCL OUTPUT TABLES FOR SURFACE WATER, SEDIMENT, ISLAND SOIL, AND FISH TISSUE DATA SETS (95% UPPER CONFIDENCE LIMITS) **(On CD only)**

E BACKGROUND COMPARISON STATISTICAL RESULTS **(On CD only)**

F DOSE CALCULATIONS AND NONCANCER HAZARD/CANCER RISK ESTIMATES FOR 100-A EXPOSURE POINT **(On CD only)**

G DOSE CALCULATIONS AND NONCANCER HAZARD/CANCER RISK ESTIMATES FOR 100-B EXPOSURE POINT **(On CD only)**

H DOSE CALCULATIONS AND NONCANCER HAZARD/CANCER RISK ESTIMATES FOR 300-A EXPOSURE POINT **(On CD only)**

I DOSE CALCULATIONS AND NONCANCER HAZARD/CANCER RISK ESTIMATES FOR 300-B EXPOSURE POINT **(On CD only)**

J DOSE CALCULATIONS AND NONCANCER HAZARD/CANCER RISK ESTIMATES FOR LW-A EXPOSURE POINT **(On CD only)**

K DOSE CALCULATIONS AND NONCANCER HAZARD/CANCER RISK ESTIMATES FOR LW-B EXPOSURE POINT **(On CD only)**

L DOSE CALCULATIONS AND NONCANCER HAZARD/CANCER RISK ESTIMATES FOR INGESTION OF INDIVIDUAL FISH SPECIES **(On CD only)**

M COMPARISON OF DETECTION LIMITS FOR NON-DETECT CONSTITUENTS TO BENCHMARKS **(On CD only)**

N WASTEWAY ANALYSIS **(On CD only)**

O DOSE CALCULATIONS AND NONCANCER HAZARD/CANCER RISK ESTIMATES FOR UNCERTAINTY ANALYSIS **(On CD only)**