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River Corridor Baseline Risk Assessment

Volume II:
Human Health Risk Assessment

For External Review

United States
Department of Energy

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FORWARD

Hanford Site Cleanup Completion Framework

The following information presents a summary of the Hanford Site Cleanup Completion Framework (DOE/RL-2009-10.)[1] *The river corridor baseline risk assessment fits into this framework by providing risk information that will be used to support development of final cleanup decisions in the River Corridor.*

Cleanup of the Hanford Site is a complex and challenging undertaking. The framework document provides a comprehensive overview for completing Hanford's cleanup, including the transition to post-cleanup activities. The framework describes three major components of cleanup – River Corridor, Central Plateau, and Tank Waste. It provides the context for individual cleanup actions by describing the key challenges and approaches for the decisions needed to complete cleanup.

The U.S. Department of Energy (DOE), as regulated by the U.S. Environmental Protection Agency (EPA) and Washington State Department of Ecology (Ecology), is implementing a strategy to achieve final cleanup decisions for the River Corridor portion of the Hanford Site. The DOE Richland Operations Office and DOE Office of River Protection have prepared the framework document to describe that strategy and to begin developing the approach for making cleanup decisions for the remainder of the Hanford Site.

While it is important to understand what this overview document is, it is just as important to understand what it is not. The framework document does not make or replace any regulatory decision nor is it a *Comprehensive Environmental Response, Compensation, and Liability Act* (CERCLA) or *Resource Conservation and Recovery Act* (RCRA) document. The framework document does not substitute for, nor preempt, the regulatory decision processes as set forth in the *Hanford Federal Facility Agreement and Consent Order* (Ecology et al. 1989),[2] also known as the Tri-Party Agreement, and applicable laws, regulations, and other legal requirements. The DOE's intent is that this document will facilitate dialogue among the Tri-Parties and with Hanford's diverse interest groups, including Tribal Nations, State of Oregon, Hanford Advisory Board, Natural Resource Trustees, and the public. Future cleanup decisions will be enhanced by an improved understanding of the challenges facing cleanup and a common understanding of the goals and approaches for cleanup completion.

The overarching goals for cleanup are stated in Figure F-1. These goals embody more than 20 years of dialogue among the Tri-Party Agencies, Tribal Nations, State of Oregon,

[1] DOE/RL-2009-10, 2010, *Hanford Site Cleanup Completion Framework*, Rev. 0, U.S. Department of Energy, Richland Operations Office, Richland, Washington.

[2] Ecology, EPA, and DOE, 1989, *Hanford Federal Facility Agreement and Consent Order*, 2 vols., as amended, Washington State Department of Ecology, U.S. Environmental Protection Agency, and U.S. Department of Energy, Olympia, Washington.

stakeholders, and the public. They carry forward key values captured in forums such as the Hanford Future Site Uses Working Group, Tank Waste Task Force, Hanford Summits, and Hanford Advisory Board Exposure Scenario Workshops, as well as more than 200 advice letters issued by the Hanford Advisory Board (http://www.hanford.gov/page.cfm/hab). These goals help guide all aspects of Hanford Site cleanup. Cleanup activities at various areas of the site support the achievement of one or more of these goals. These goals help set priorities to apply resources and sequence cleanup efforts for the greatest benefit.

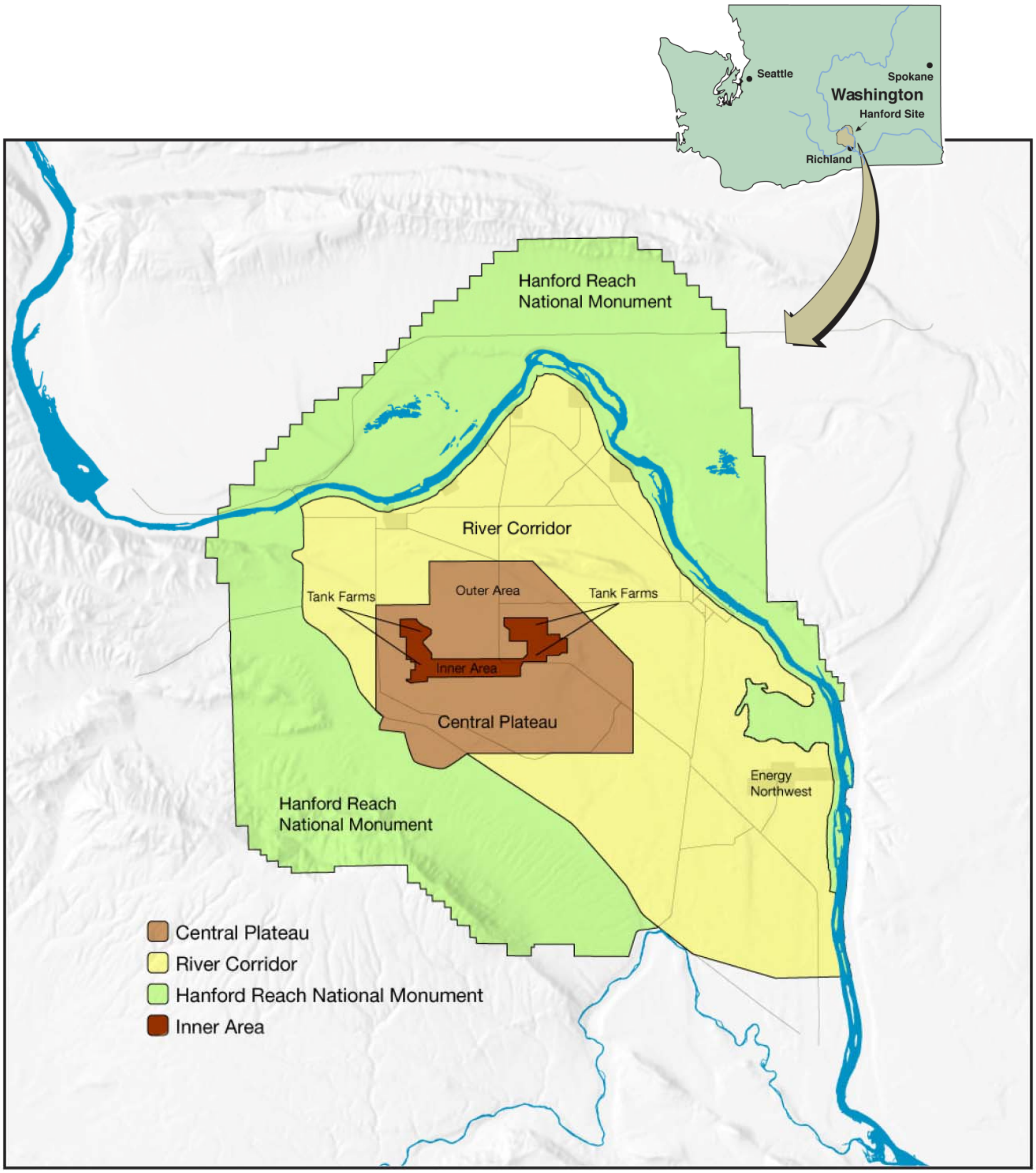
Figure F-1. Goals for Cleanup.

Goals for Cleanup	
Goal 1:	Protect the Columbia River.
Goal 2:	Restore groundwater to its beneficial use to protect human health, the environment, and the Columbia River.
Goal 3:	Clean up River Corridor waste sites and facilities to: • Protect groundwater and the Columbia River. • Shrink the active cleanup footprint to the Central Plateau. • Support anticipated future land uses.
Goal 4:	Clean up Central Plateau waste sites, tank farms, and facilities to: • Protect groundwater. • Minimize the footprint of areas requiring long-term waste management activities. • Support anticipated future land uses.
Goal 5:	Safely manage and transfer legacy materials scheduled for off-site disposition including special nuclear material (including plutonium), spent nuclear fuel, transuranic waste, and immobilized high-level waste.
Goal 6:	Consolidate waste treatment, storage, and disposal operations on the Central Plateau.
Goal 7:	Develop and implement institutional controls and long-term stewardship activities that protect human health, the environment, and Hanford's unique cultural, historical, and ecological resources after cleanup activities are completed.

These goals reflect DOE's recognition that the Columbia River is a critical resource for the people and ecology of the Pacific Northwest. The 50-mile stretch of the river known as the Hanford Reach is the last free-flowing section of the river in the United States. As one of the largest rivers in North America, its waters support a multitude of uses that are vital to the economic and environmental well being of the region, and it is particularly important in sustaining the culture of Native Americans. Cleanup actions must protect this river.

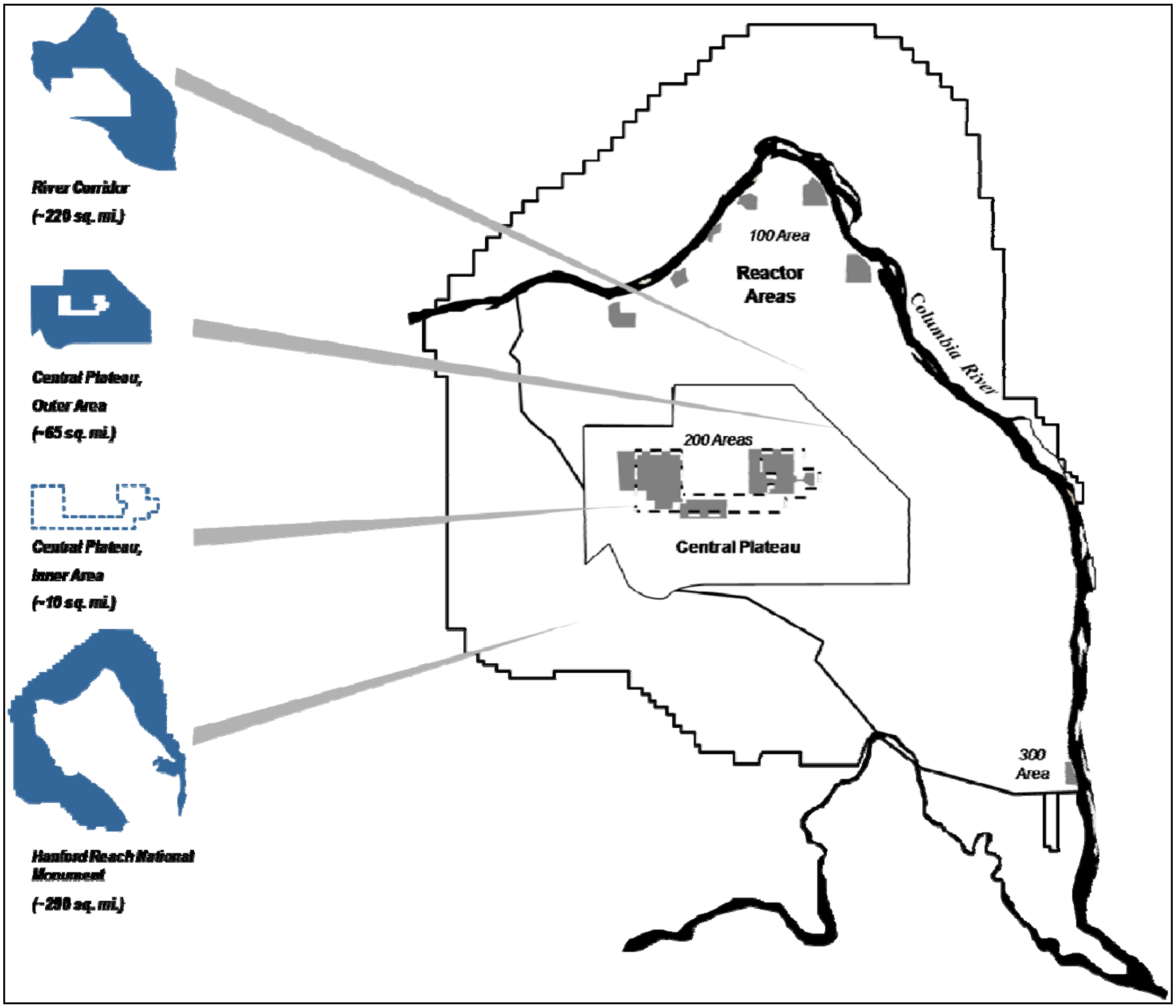
The Hanford Site cleanup consists of three major components: (1) River Corridor, (2) Central Plateau, and (3) Tank Waste (note that the Tank Waste component is contained within the geographic boundaries of the Central Plateau). Each component of cleanup is in itself a complex and challenging undertaking involving multiple projects and contractors and requiring many years and billions of dollars to complete. These components are shown in Figure F-2.

Figure F-2. Principal Components of Hanford's Cleanup Completion Framework: River Corridor, Central Plateau, and Tank Waste. (Note: River Corridor cleanup includes the south shore of the river that is part of the Hanford Reach National Monument.)



Active Cleanup Footprint Reduction. Figure F-3 illustrates the principal components of active cleanup footprint reduction. The Hanford Reach National Monument lands (~290 square miles) surround the Hanford Site. These lands are primarily managed to preserve natural and cultural resources. A portion of the monument along the south shore of the Columbia River is included in cleanup of the River Corridor. DOE expects to complete cleanup of the other portions of the national monument in fiscal year 2011. The following sections describe the components of active cleanup footprint reduction that will occur beyond the footprint reduction due to the national monument: the River Corridor, Central Plateau, and Tank Waste.

Figure F-3. Principal Components of Active Cleanup Footprint Reduction at Hanford.



Source: DOE/RL-2009-10, 2010, *Hanford Site Cleanup Completion Framework*, Rev. 0, U.S. Department of Energy, Richland Operations Office, Richland, Washington.

River Corridor Cleanup. Cleanup of the River Corridor has been one of Hanford's top priorities since the early 1990s. This urgency is due to the proximity of hundreds of waste sites to the Columbia River. In addition, removal of the sludge from K West Basin, which is near the river, remains a high priority. (Refer to Section 3.0 for details about River Corridor cleanup.)

This component of cleanup includes approximately 220 square miles of the Hanford Site as shown in Figure F-2. The River Corridor portion of the Hanford Site includes the 100 and 300 Areas along the south shore of the Columbia River:

- The 100 Areas contains nine retired plutonium production reactors. These areas are also the location of numerous support facilities and solid and liquid waste disposal sites that have contaminated groundwater and soil.

- The 300 Area, located north of the city of Richland, contains fuel fabrication facilities, nuclear research and development facilities, and their associated solid and liquid waste disposal sites that have contaminated groundwater and soil.

For purposes of this completion framework and to ensure that cleanup actions address all threats to human health and the environment, the River Corridor includes the adjacent areas that extend from the 100 Areas and 300 Area to the Central Plateau.

For sites in the River Corridor, remedial actions are expected to restore groundwater to drinking water standards and to ensure that the aquatic life in the Columbia River is protected by achieving ambient water quality standards in the river. It is intended that these objectives be achieved, unless technically impracticable, within a reasonable time frame. In those instances where remedial action objectives are not achievable in a reasonable time frame, or are determined to be technically impracticable, programs will be implemented to contain the plume, prevent exposure to contaminated groundwater, and evaluate further risk reduction opportunities as new technologies become available. River Corridor cleanup work also removes potential sources of contamination, which are close to the Columbia River, to the Central Plateau for final disposal. The intent is to shrink the footprint of active cleanup to within the 75-square-mile area of the Central Plateau by removing excess facilities and remediating waste sites. Cleanup actions will support anticipated future land uses consistent with the Hanford Reach National Monument, where applicable, and DOE/EIS-0222-F, *Hanford Comprehensive Land-Use Plan Environmental Impact Statement*.[3]

The River Corridor has been divided into six geographic decision areas to achieve source and groundwater remedy decisions. These decisions will provide comprehensive coverage for all areas within the River Corridor and will incorporate ongoing interim action cleanup activities. Cleanup levels will be achieved that support the anticipated land uses of conservation and preservation for most of this area and industrial use for the 300 Area. At the conclusion of cleanup actions, the federal government will retain ownership of land in the River Corridor and

[3] DOE/EIS-0222-F, *Hanford Comprehensive Land-Use Plan Environmental Impact Statement*, September 1999, U.S. Department of Energy, Washington, D.C.

will implement long-term stewardship activities to ensure protection of human health and the environment.

Central Plateau Cleanup. The Central Plateau component of cleanup includes approximately 75 square miles in the central portion of the Hanford Site as shown in Figure S-2. This component includes the Inner Area (~10 square miles) that contains the major nuclear fuel processing, waste management, and disposal facilities. This Inner Area is anticipated to be the final footprint of Hanford (see Figure F-3), and will be dedicated to long-term waste management and containment of residual contamination. The Outer Area (~65 square miles) is that portion of the Central Plateau outside the boundary of the Inner Area. The Outer Area waste sites are being cleaned up to a level comparable to that achieved for River Corridor waste sites. Cleanup of the Outer Area is planned to be completed in the 2015 to 2020 time period. Completing cleanup of the Outer Area will shrink the footprint of active cleanup by an additional 65 square miles leaving just the Inner Area remaining. (Refer to Section 4.0 for details about Central Plateau cleanup.)

Cleanup of the Central Plateau is a highly complex activity because of the large number of waste sites, surplus facilities, active treatment and disposal facilities, and areas of deep soil contamination. Past discharges of more than 450 billion gallons of liquid waste and cooling water to the soil have resulted in about 60 square miles of contaminated groundwater. Today, some plumes extend far beyond the plateau. Containing and remediating these plumes remains a high priority. For areas of groundwater contamination in the Central Plateau, the goal is to restore the aquifer to achieve drinking water standards. In those instances where remediation goals are not achievable in a reasonable time frame, programs will be implemented to contain the plumes, prevent exposure to contaminated groundwater, and evaluate further risk reduction opportunities as new technologies become available. Near-term actions will be taken to control plume migration until remediation goals are achieved.

At the completion of cleanup efforts, residual hazardous and radioactive contamination will remain, both in surface disposal facilities and in subsurface media within portions of the Central Plateau. It is DOE’s intent to minimize the area requiring long-term institutional controls for protection of human health and the environment. However, some areas of the Central Plateau will require long-term waste management activities. For the foreseeable future, it is expected that the Inner Area of the plateau will remain a waste management area.

The Central Plateau cleanup strategy includes the following elements:

- Implement groundwater treatment systems to contain contaminant plumes within the footprint of the Central Plateau, thereby protecting the Columbia River.
- Implement groundwater treatment systems to restore the groundwater.
- Develop a geographic cleanup strategy, analogous to the geographic strategy for the River Corridor.

- Develop and apply deep vadose zone treatment technologies to protect the groundwater.

- Implement cleanup decisions that are protective of human health and the environment and that support anticipated future land use.

- Remediate the outer portion of the Central Plateau to further reduce the active cleanup footprint of the Hanford Site.

- Remediate the inner portion of the plateau to minimize the area requiring long-term waste management activities.

- Regularly evaluate new and improved cleanup technologies to assess their potential to improve cleanup effectiveness and to allow for greater footprint reduction.

Tank Waste Cleanup. This component of cleanup lies within the Central Plateau and is one of Hanford's most challenging legacies. The tank farms contain approximately 53 million gallons of radioactive waste stored in 177 underground tanks. Sixty-seven of these tanks have or are suspected to have leaked up to 1 million gallons of waste. Releases from some single-shell tank farms have reached groundwater. DOE expects these impacts to increase in the future unless prompt actions are taken.

Today, actions are being taken to slow the movement of those contaminants. DOE is also containing and recovering those contaminants once they reach groundwater. A key step in fixing this problem is to retrieve as much waste from single-shell tanks as possible and put it into double-shell tanks. Then, the waste must be fed to the Waste Treatment Plant for processing and placed into solid glass waste forms.

The tasks of tank waste cleanup are to retrieve and treat Hanford's tank waste and close or remediate the tank farms within the Inner Area of the Central Plateau area (see Figure S-2). Retrieval and treatment of tank waste will remain the most important and difficult task facing completion of cleanup for several decades to come. However, these efforts will protect the groundwater on the Central Plateau, thereby protecting the Columbia River.

The tank waste cleanup strategy includes the following elements:

- Complete construction of the Waste Treatment Plant.

- Provide treatment capacity to enable mission completion.

- Treat tank waste and retrieve tank waste at a rate that supports treatment capacity.

- Store tank waste safely until it is retrieved for treatment.

- Safely store immobilized high-level waste pending ultimate disposition.

- Implement remedies that protect the groundwater and environment from past tank farm releases – in conjunction with surrounding waste sites and groundwater operable units.

- Complete closure of tank farms in coordination with, and consistent with, the Central Plateau cleanup completion strategy.

Long-Term Stewardship. Following the completion Hanford Site cleanup actions, there will be disposal facilities and other areas that will necessitate long-term management activities. The DOE Richland Operations Office has established a Hanford Long-Term Stewardship Program to ensure continued protectiveness of cleanup remedies, as defined by CERCLA and RCRA cleanup decision documents, and to ensure protection of natural resources, the environment, and human health. Long-term stewardship will include monitoring and maintenance activities to ensure continued protectiveness.

DOE is committed to maintaining the protection of human health and the environment and to meeting its long-term, post-cleanup obligations in a safe and cost-effective manner. The completion of cleanup and the transition to long-term stewardship are approaching. Therefore, cleanup actions are being considered and taken to mitigate natural resource concerns and ensure long-term stewardship considerations are incorporated into the cleanup decisions.

EXECUTIVE SUMMARY

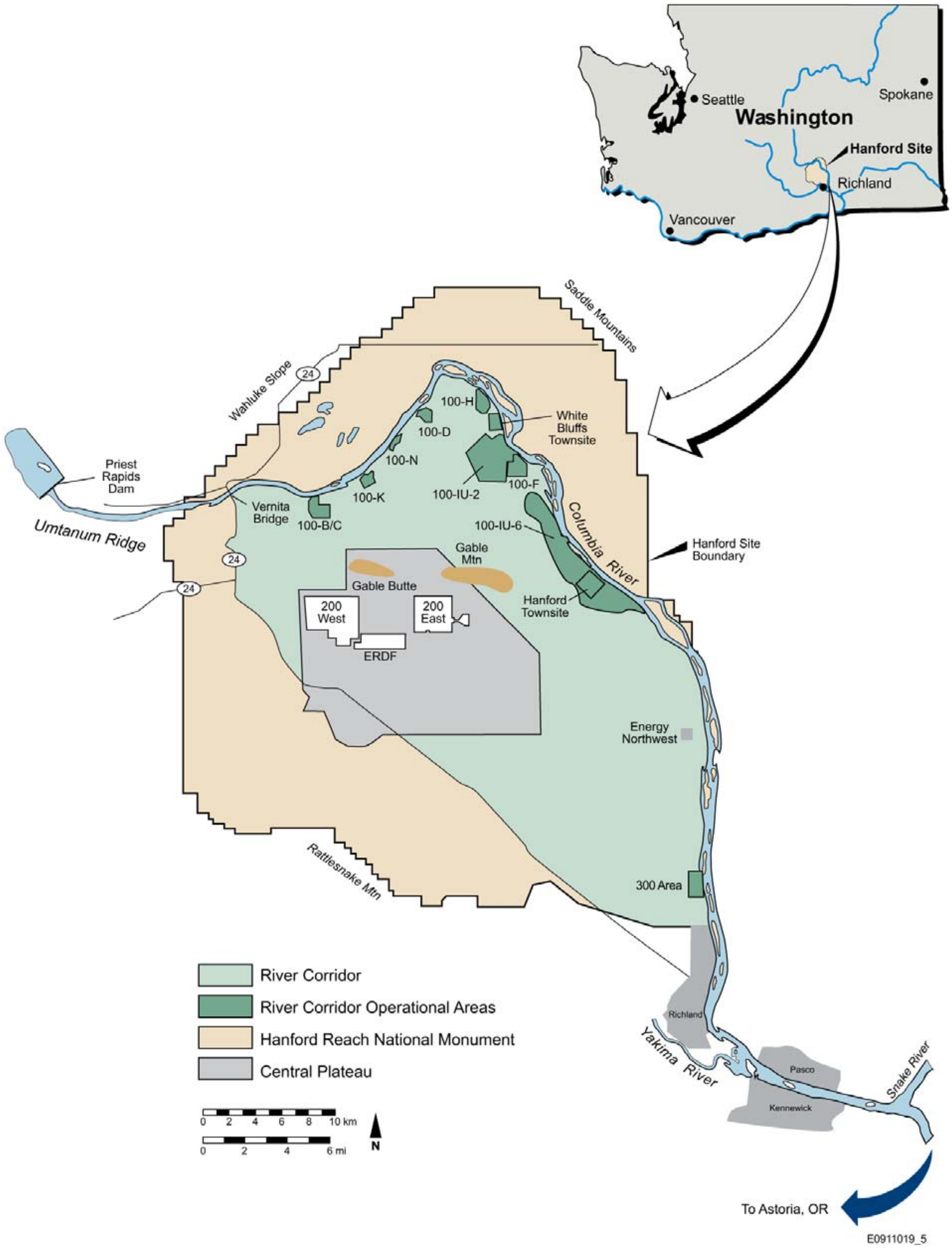
This volume (Volume II) of the River Corridor Baseline Risk Assessment (RCBRA) presents a comprehensive human health risk assessment for the River Corridor, considering relevant sources of contamination, exposure pathways, and contaminants to evaluate current and potential future risks posed by hazardous substance releases. Volume II will be used, along with a complementary ecological risk assessment (Volume I), to support final cleanup decisions for the River Corridor. Risk managers will use the results from this baseline risk assessment in conjunction with other information to develop final cleanup decisions that will be protective of human health and the environment. Final cleanup decisions applying to all portions of the River Corridor will be identified in proposed plans that will undergo public review and documented in records of decision (RODs).

SITE DESCRIPTION AND HISTORY

The U.S. Department of Energy's (DOE) 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. The Columbia River flows through the northern part of the Hanford Site and, turning south, forms part of the site’s eastern boundary. The general area and major features of the Hanford Site are shown in Figure ES-1.

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. The Hanford Site’s production mission continued until the late 1980s, when the mission changed to cleaning up the radioactive and hazardous wastes that had been generated during production in the previous decades.

Figure ES-1. Major Features of the Hanford Site and Surrounding Area.



In 1989, the 100 Area (encompassing the former plutonium production reactors along the Columbia River) and the 300 Area (which housed reactor fuel fabrication plants as well as many research and development facilities) were two of four areas at the Hanford Site 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. Together, the 100 and 300 Areas comprise a 570-km^2 (220-mi^2) portion of the Hanford Site that is called the River Corridor (see Figure ES-1).

In order to allow cleanup to begin as soon as possible, the DOE, the U.S. Environmental Protection Agency (EPA), and the Washington State Department of Ecology (collectively called the Tri-Parties) developed an approach for expedited remediation of the River Corridor in 1991. Cleanup decisions were established through interim action records of decision (IARODs) based on existing knowledge of the waste sites (e.g., site types, processes, operating history, contaminants) and supplemented by limited amounts of characterization. In 1995, cleanup actions were initiated focusing on removal of contaminated soil and debris from waste sites with the highest potential to impact groundwater and the Columbia River. Actions to address existing plumes of groundwater contamination were also initiated.

Waste site and groundwater cleanup actions in the River Corridor have continued from 1995 to date. During that time, about 8 million tons of contaminated soil and debris have been removed from nearly 300 waste sites in the River Corridor and disposed of at authorized facilities. More than 2 billion gallons of contaminated groundwater have been processed through pump-and-treat systems. At each waste site where remediation has occurred, the goals and objectives of the IARODs have been met as demonstrated by evaluation of residual soil concentrations and verification documentation that has been completed and approved by the regulatory agencies.

COMPLETION OF CLEANUP ACTIONS

Cleanup actions in the River Corridor are not complete. Many waste sites and groundwater plumes that have been identified for cleanup actions have yet to be addressed. Consequently, waste site and groundwater cleanup actions in the River Corridor will continue for several years. In parallel with continuing the cleanup actions outlined in the existing IARODs, the Tri-Parties have established a strategy to develop final waste site and groundwater cleanup decisions for the River Corridor. Remedial investigation/feasibility study (RI/FS) processes have been initiated for the River Corridor to gather and evaluate information needed to make final cleanup decisions. A key element in the RI/FS decision-making process is performance of a baseline risk assessment.

Under CERCLA, a baseline risk assessment is needed to provide risk managers with an understanding of the current and potential future risks posed by a site. The RCBRA is being conducted part way through cleanup actions. As such, baseline conditions that are assessed include a mix of areas where cleanup has been completed in accordance with the IARODs, areas that are currently scheduled for cleanup, and areas where cleanup actions are not anticipated.

CURRENT CONDITIONS IN THE RIVER CORRIDOR

The RCBRA human health risk assessment addresses current conditions in the River Corridor based on historic information, results of site characterization, and cleanup actions completed in accordance with the IARODs. Under current conditions there are two general types of areas (see Figure ES-1):

- Operational areas where releases of hazardous substances posing a threat to human health and the environment are known to have occurred. These areas are identified as waste sites and include inactive ponds, trenches, burial grounds, landfills, and spill sites as well as inactive structures and facilities. Operational areas are typically found in the arid upland areas of the River Corridor and include a mix of remediated waste sites and yet-to-be-remediated waste sites.

- Nonoperational areas where historic information and other studies indicate there were no direct releases of hazardous substances posing a threat to human health or the environment. Nonoperational areas may be affected by transport of contaminants from operational areas through mechanisms such as windblown dust, groundwater migration, or biotic uptake, for example. Nonoperational areas include large portions of the River Corridor that are outside of the operational areas and are not anticipated to be adversely affected by Hanford Site releases.

Unacceptable risks are present in the River Corridor at waste sites that are identified in the IARODs but that have yet to be remediated. The determination of the presence of unacceptable risk and basis for action at yet-to-be remediated waste sites is supported by field investigation data as well as field experience and information gathered through implementation of the observational-approach soil cleanup actions in the River Corridor over the past 15 years. Waste site identification activities, including historical reviews, site walkdowns, remedial investigations, and discoveries during remedial actions, continue to be implemented in the River Corridor to ensure that all sites posing unacceptable risk are identified and addressed.

ASSESSMENT OF INTERIM ACTIONS

Screening-level cancer risks and noncancer chemical hazards were calculated for 156 individual waste sites that had been remediated under the interim actions. These waste sites that had been remediated from 1995 to 2005 included a representative selection of wastes site types from the various operating areas. Soil exposure concentrations were calculated using cleanup verification data collected to verify residual soil concentrations had met the remedial action goals following cleanup. For radionuclides, a residential scenario that included food pathways was used to calculate total cancer risk. For chemicals, risk calculations were based on the “Model Toxics

Control Act" (MTCA) Method B direct contact soil cleanup levels for unrestricted land use (WAC 173-340-740) with updated toxicity criteria current as of December 2009. MTCA specifies a total site risk threshold for all pathways of 10^{-5}.

> The EPA guidance memorandum "Role of the Baseline Risk Assessment in Superfund Remedy Selection Decisions" (OSWER Directive 9355.0-30) states that action is generally not warranted at sites where the cumulative carcinogenic site risk to an individual based on reasonable maximum exposure (RME) for both current and future land use is less than 1 x 10^{-4} (1 in 10,000), the noncarcinogenic hazard quotient is less than 1, and there are no adverse environmental impacts. In addition, the EPA guidance memorandum states that the upper boundary of the risk range is not a discrete line at 1 x 10^{-4}, although the EPA generally uses 1 x 10^{-4} in making risk management decisions. A specific risk estimate around 10^{-4} may be considered acceptable if justified based on site-specific conditions, and in certain cases, EPA may consider risk estimates slightly greater than 1 x 10^{-4} to be protective.

Total cumulative cancer risks for chemicals for 153 of the 156 remediated waste sites evaluated in RCBRA are less than 1×10^{-5} (1 in 100,000) using the MTCA Method B Unrestricted Use scenario. For three remediated waste sites, risks up to 3×10^{-5} were calculated, with the main risk driver for these sites being arsenic. Arsenic soil concentrations exceed the Hanford Site background value of 6.5 parts per million (DOE/RL-92-24) at a number of remediated waste sites with concentrations ranging from 1.1 to 17.3 parts per million. However, all of the reasonable maximum exposure (RME) values for arsenic are less than the IAROD cleanup value of 20 parts per million, which is based on the State of Washington Method A unrestricted cleanup level. When residual cancer risks for chemicals were calculated without the contribution from arsenic, all 156 remediated waste sites had cancer risk results less than 1×10^{-5} using the MTCA Unrestricted Use scenario with current toxicity criteria.

The noncancer chemical hazard indices for chemicals (including arsenic) are near or below the threshold of 1 at all of the 156 remediated waste sites evaluated in RCBRA using the MTCA Unrestricted Use scenario with current toxicity criteria. Two sites have hazard indices slightly greater than 1. The 316-1 site has a hazard index of 1.4 with arsenic and mercury as the main contributors. The 316-5 site has a hazard index of 1.2 with uranium as the main contributor.

Residual total cumulative cancer risks from radionuclides for 142 of the 156 remediated waste sites evaluated in the RCBRA are less than 1 x 10^{-4} (1 in 10,000) based on the IAROD residential scenario. Fourteen of the 156 remediated waste sites evaluated in the RCBRA have risks that are two to five times above the EPA upper risk management level (2 x 10^{-4} to 5 x 10^{-4}). For 11 of these 14 remediated waste sites, the risk drivers are residual levels of europium-152, cesium-137, and strontium-90. The radioactive half-lives of cesium-137 and strontium-90 are approximately 30 years, and the half-life of europium-152 is approximately 15 years. Three of the 14 remediated waste sites were cleaned up to industrial land-use assumptions and had higher cleanup levels. For these three remediated waste sites, the risk drivers are residual levels of cobalt-60, which has a half-life of approximately 5 years, and long-lived isotopes of uranium (uranium-235 and uranium-238).

The key uncertainty related to the risk calculations for the IAROD exposure scenarios is the applicability of using cleanup verification soil data for estimating future human exposure. Risks may be overestimated using cleanup verification data (from soil samples collected from the sidewalls and bottom of the excavations) to estimate exposure concentrations in surface soil. This is of particular significance because remediated waste sites with the highest levels of residual risk are sites that have been excavated and backfilled with clean fill material. Therefore, verification soil samples may not fully represent soil conditions on or near the surface of the remediated waste site.

> The basis of the IAROD cleanup levels for radionuclides is a radiation dose of 15 mrem/yr above background, and all of the 156 remediated waste sites evaluated meet the cleanup level. In general, a radiation dose of 15 mrem/yr equates to approximately a 3 × 10^{-4} lifetime cancer risk (OSWER Directive 9200.4-18, "Establishment of Cleanup Levels for CERCLA Sites with Radioactive Contamination"). Because of this, it is possible that some sites cleaned up to a 15 mrem/yr dose cleanup goal may show cancer risk greater than 1 x 10^{-4}.

ASSESSMENT OF GROUNDWATER

A screening-level groundwater risk assessment was completed to evaluate potential risks associated with groundwater exposures for each of the seven groundwater operable units (OUs), or impacted areas, within the River Corridor. The assessment provides an estimate of present-day exposure concentrations and risk in each OU using available data collected from 331 groundwater monitoring wells from 1998 to 2008.

The results of the groundwater screening-level risk assessment indicate potential risks above EPA thresholds within each groundwater OU. Reasonable maximum exposure (RME) cancer risks were above the upper end of the EPA target risk management level of 1×10^{-6} to 1×10^{-4} for five of the seven of the groundwater OUs. Noncancer chemical hazard results were above EPA's threshold value of 1 for six of the seven groundwater OUs.

> The RCBRA human health risk assessment includes an evaluation of risks under both reasonable maximum exposure (RME) and the central tendency exposure (CTE) conditions. Under RME conditions, risk is evaluated for individuals whose behavioral characteristics may result in higher potential exposure than seen in the average individual. The CTE conditions characterize potential risk to an average member of the target population. Including both the RME and CTE calculations provides a semi-quantitative measure of the range of risks that may occur under a particular exposure scenario. The CTE and RME provide risk managers with an estimate of the mean and upper percentile of estimates of exposure. Most CERCLA decisions are made based on RME.

A key uncertainty with the groundwater assessment is related to the ability of the groundwater data set collected from 1998 to 2008 to represent current baseline conditions and potential exposure within each groundwater OU. For this reason, the groundwater assessment is presented as a "screening-level" assessment. Additional groundwater data will be collected and evaluations will be presented in the RI/FS reports for the River Corridor ROD decision areas.

BROAD-AREA AND LOCAL-AREA RISK ASSESSMENTS

To assess potential risks related to a variety of potential land uses, a range of exposure scenarios are evaluated in the RCBRA. These range from the situation where recreational users occasionally visit the site to hypothetical scenarios where individuals live on site and consume food items that are predominantly grown or raised on site.

The exposure scenarios evaluated in the RCBRA are grouped according to general types of land use and associated exposure intensity. The exposure scenarios evaluated include the following:

- **Broad-Area Exposure Scenarios**
  - Recreational Use scenarios: Avid Hunter, Avid Angler, and Casual User
  - Nonresidential Tribal scenario.
- **Local-Area Exposure Scenarios - Occupational**
  - Industrial Worker scenario
  - Resident Monument Worker scenario.
- **Local-Area Exposure Scenarios - Residential**
  - Subsistence Farmer scenario
  - Confederated Tribes of the Umatilla Indian Reservation (CTUIR) Native American Resident scenario
  - Yakama Nation Native American Resident scenario.

Broad-Area Exposure Scenarios

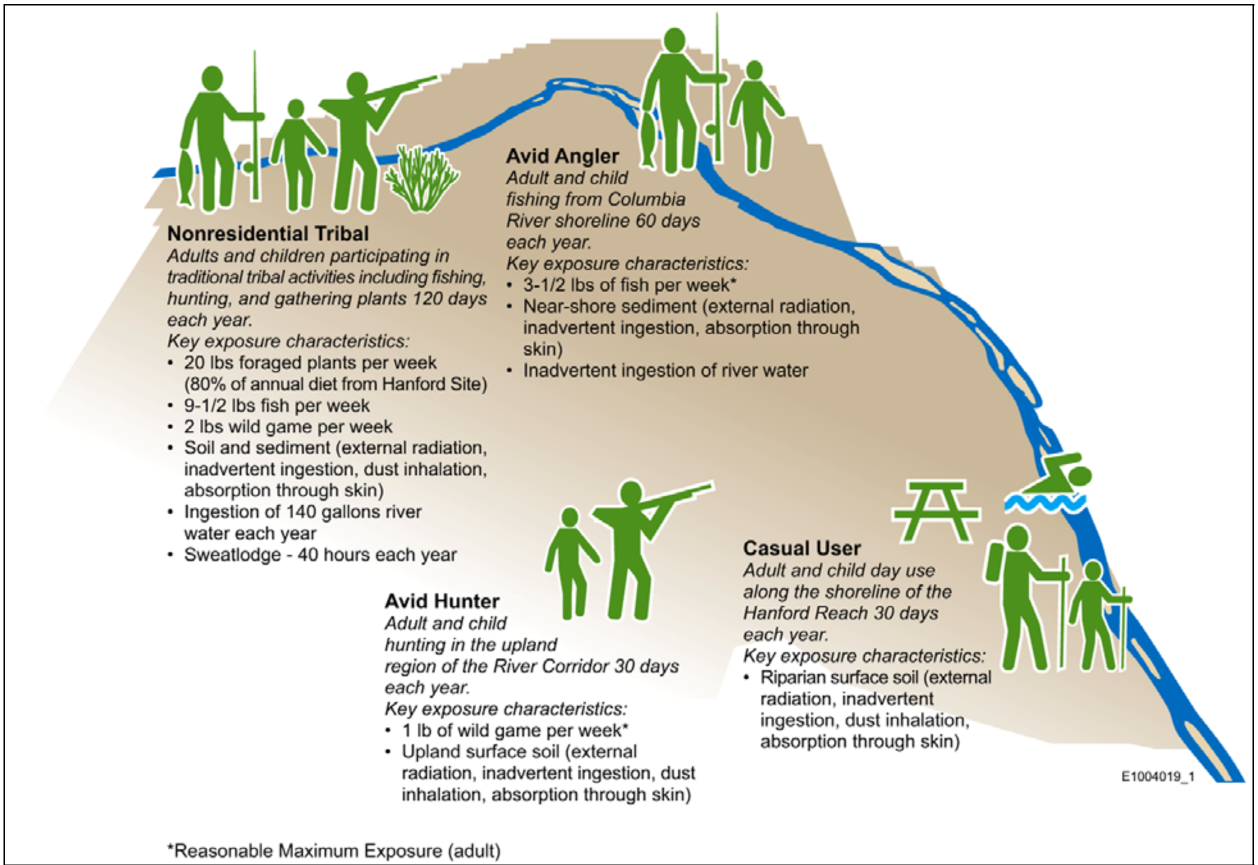
The broad-area risk assessment used surface soil, sediment, and surface water data from throughout the River Corridor to evaluate potential risks for receptors that may engage in activities where they are exposed to these media throughout the River Corridor, including the areas outside the waste sites. Potential exposures are based on soil samples collected from areas associated with 20 remediated waste sites that represent a cross section of waste site types, contaminants, and remediation methodologies in the River Corridor. Because these data were collected at or near remediated waste sites, they are considered to be a conservative representation of average contaminant concentrations over the entire River Corridor upland area.

The Recreational scenarios address child and adult exposures for different types of activities in the upland (Avid Hunter), riparian (Casual User), and shoreline (Avid Angler) environments. The Nonresident Tribal scenario is focused on adults and children engaged in a subsistence lifestyle who reside off site but who use the River Corridor for traditional tribal activities including fishing, hunting, gathering plants, and participating in sweat lodges using river water. For the Nonresident Tribal scenario, it is assumed that 80% of plant foods and 100% of fish and game consumed over an entire year by the children and adults comes from the Hanford Site. Figure ES-2 shows a brief description of the broad-area exposure scenarios and the key exposure pathways and assumptions.

Total cancer risks (i.e., sum of radionuclide and chemical risks) are near or below the EPA 1×10^{-6} de minimis threshold (1 in 1,000,000) for exposures to soil, sediment, and surface water in the Casual User and Avid Angler scenarios. Total cancer risks for the Avid Hunter scenario are within the EPA risk management range (1×10^{-6} to 1×10^{-4}). Noncancer chemical hazard is also below thresholds for each of these scenarios.

The results of the broad-area risk assessment for the Nonresident Tribal scenario indicate potential total cancer risks of approximately 1×10^{-2} (1 in 100) and chemical hazard index between about 75 and 95, related primarily to arsenic exposure by the plant ingestion pathway. This corresponds to risks approximately 100 times above the EPA guidelines. Arsenic in the plant ingestion pathway contributes over 90% of the total cancer risk. However, arsenic concentrations detected in the upland and riparian site soils are not significantly different from background levels in soil. The key contributors to total cancer risk other than arsenic include technetium-99, strontium-90, benzo(a)pyrene, and Aroclor-1254 by the plant and game ingestion pathways.

Figure ES-2. Broad-Area Exposure Scenarios.



The key uncertainty in the Nonresident Tribal results relates to exposure assumptions for consumption of native plants. The intake rates for various food items for Nonresident Tribal members are uncertain. Therefore, as a conservative estimate of exposure for this assessment, it was assumed that the majority of the food consumed by the receptors comes from the Hanford Site. This assumption results in potential overestimation of the exposure that might actually occur if only a portion of food items is obtained from the Hanford Site. There are also considerable uncertainties related to contaminant intake from these food pathways because concentrations in edible plant tissues are modeled from soil concentrations using literature uptake factor values, not site-specific data. Site-specific plant data were not available for edible species. However, collection of these materials is being considered as part of the RI/FS process.

Potential risks related to use of water from shoreline springs were evaluated by comparing the spring water concentrations of key contaminants to EPA short-term drinking water health advisories and Washington State and EPA maximum contaminant levels for long-term consumption of drinking water. The results of the risk assessment for groundwater contaminants released at shoreline springs along the Columbia River indicate that for the majority of the shoreline springs there is negligible risk related to exposure to key groundwater contaminants. At one spring (100-F Spring) concentrations of nitrate were elevated relative to short-term and long-term standards that apply to drinking water exposure. In addition, concentrations of uranium at Spring 42-2 in the 300 Area are routinely above drinking water standards. These contaminants are also found in groundwater in these areas. The groundwater plumes are part of ongoing remediation efforts.

A screening-level fish ingestion risk assessment was also completed to estimate fish ingestion risks for the Avid Angler, Nonresident Tribal, and residential exposure scenarios. The assessment is based on concentrations of contaminants detected in local populations of sculpin, shellfish, and crayfish. These species have a very limited home range and were sampled for the purpose of determining the extent that Hanford Site-related contaminants are accumulated by aquatic organisms in areas where contaminated groundwater plumes emerge at the Columbia River. Although these species are not plausible food sources for chronic human exposure, the results of the screening assessment help identify Hanford Site-specific contaminants and relative levels of risk. Carbon-14 was the only carcinogenic contaminant evaluated in the screening assessment, and it was detected only in sculpin in the 100-K Area. Nickel and selenium were the contaminants that contributed the most to the chemical hazard index. A more comprehensive risk assessment for food fish ingestion is being conducted for the remedial investigation of Hanford Site releases to the Columbia River.

Local-Area Exposure Scenarios – Occupational

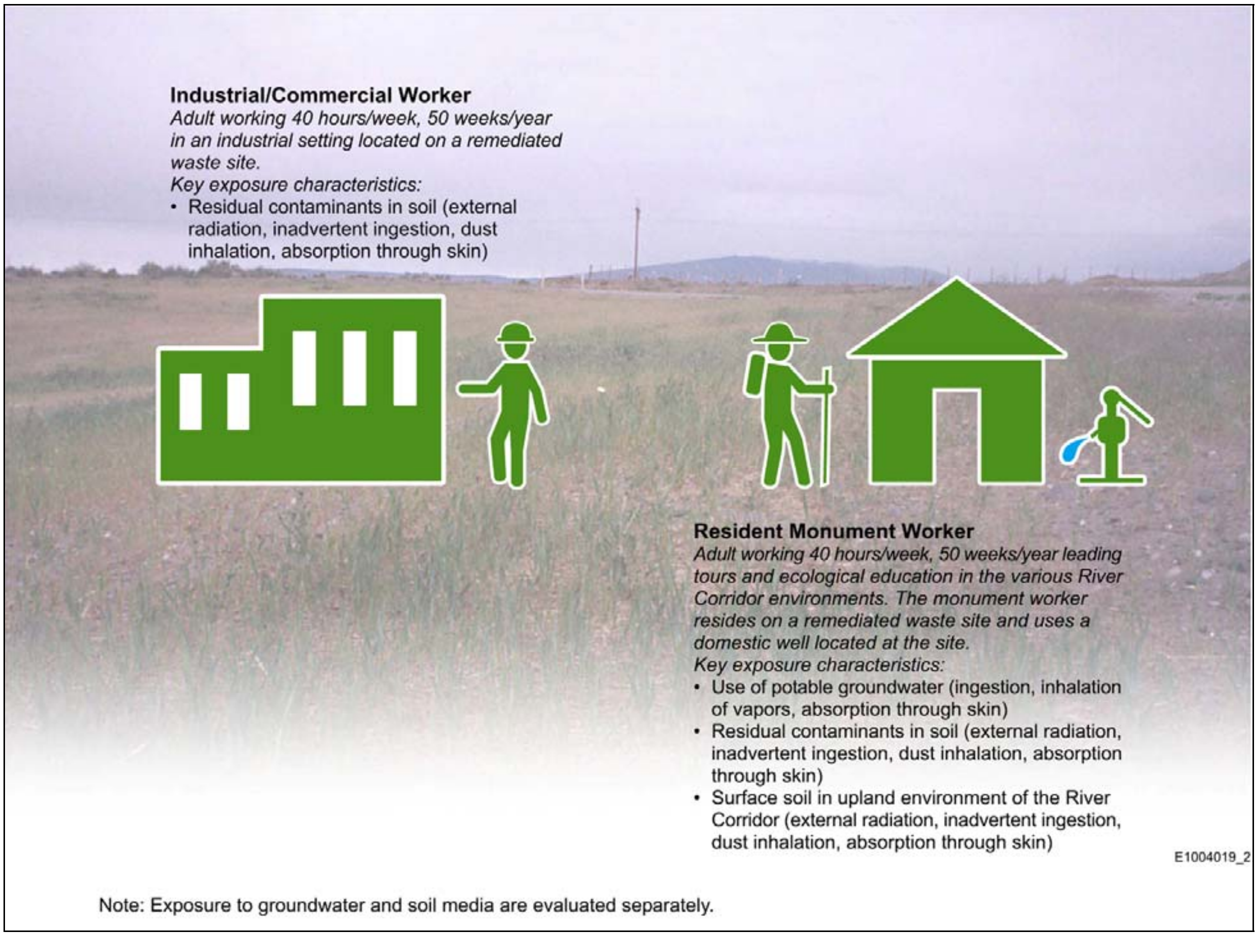
The local-area exposure scenarios used cleanup verification soil data from remediated waste sites to evaluate potential risks for receptors that may be exposed to soil from an individual waste site.

This exposure assumption is consistent with the way in which risks were assessed for the IARODs and provides a conservative evaluation of potential risks.

The Industrial/Commercial Worker scenario and the Resident Monument Worker scenario are the two occupational scenarios evaluated in the local-area risk assessment. Receptors for these scenarios are limited to adult workers. The Industrial/Commercial Worker receptor lives offsite and is assumed to work 40 hours per week at a building located on a remediated waste site. The receptor in the Resident Monument Worker receptor is assumed to live in a residence constructed on a remediated waste site and work outdoors in other portions of the River Corridor. The residential part of the exposure for the Resident Monument Worker scenario is based on remediated waste site exposure, and the occupational part of the exposure (40 hours per week) is based on broad-area exposure. Figure ES-3 shows a brief description of the occupational exposure scenarios and the key exposure pathways and assumptions.

Total cumulative cancer risks for 152 of the 156 remediated waste sites evaluated in the RCBRA are less than the EPA upper risk management level of 1×10^{-4} for the industrial/commercial worker scenario. For four remediated waste sites, risks up to two times the upper risk management level (2×10^{-4}) were calculated. This value is consistent with the 15 mrem/yr dose threshold of the IAROD cleanups; the 15 mrem/yr dose threshold is approximately equal to 3×10^{-4} risk. The main risk drivers are residual levels of cobalt-60, europium-152, and cesium-137. By the year 2075, all four of these waste sites are anticipated to have risks less than 1×10^{-4}. There are no remediated waste sites where the noncancer chemical hazard results exceed the threshold of 1 for the industrial/commercial worker scenario.

Figure ES-3. Local-Area Occupational Exposure Scenarios.



For the resident monument worker scenario, total cumulative cancer risks at 11 of the 156 remediated waste sites evaluated exceeded the EPA upper risk management level of 1×10^{-4}. Risks up to three times the upper risk management level (3×10^{-4}) were calculated. For 8 of the 11 sites, the main risk drivers are residual levels of cobalt-60, europium-152, and cesium-137 and correspond to the range of risk anticipated for sites cleaned up to a 15 mrem/yr dose. The main risk drivers for the remaining three sites are residual levels of uranium isotopes. Ten of eleven of these waste sites are expected to have risks less than 1 x 10^{-4} by 2075, while one of the sites will have similar risks well into the future. There are no remediated waste sites where the noncancer chemical hazard results exceed the threshold of 1 for this scenario.

A key uncertainty for both occupational scenarios is the applicability of using cleanup verification soil data at excavated sites to estimate future human exposure. Actual risks may be overestimated using cleanup verification data because present-day surface soil at these sites consists primarily of backfill material.

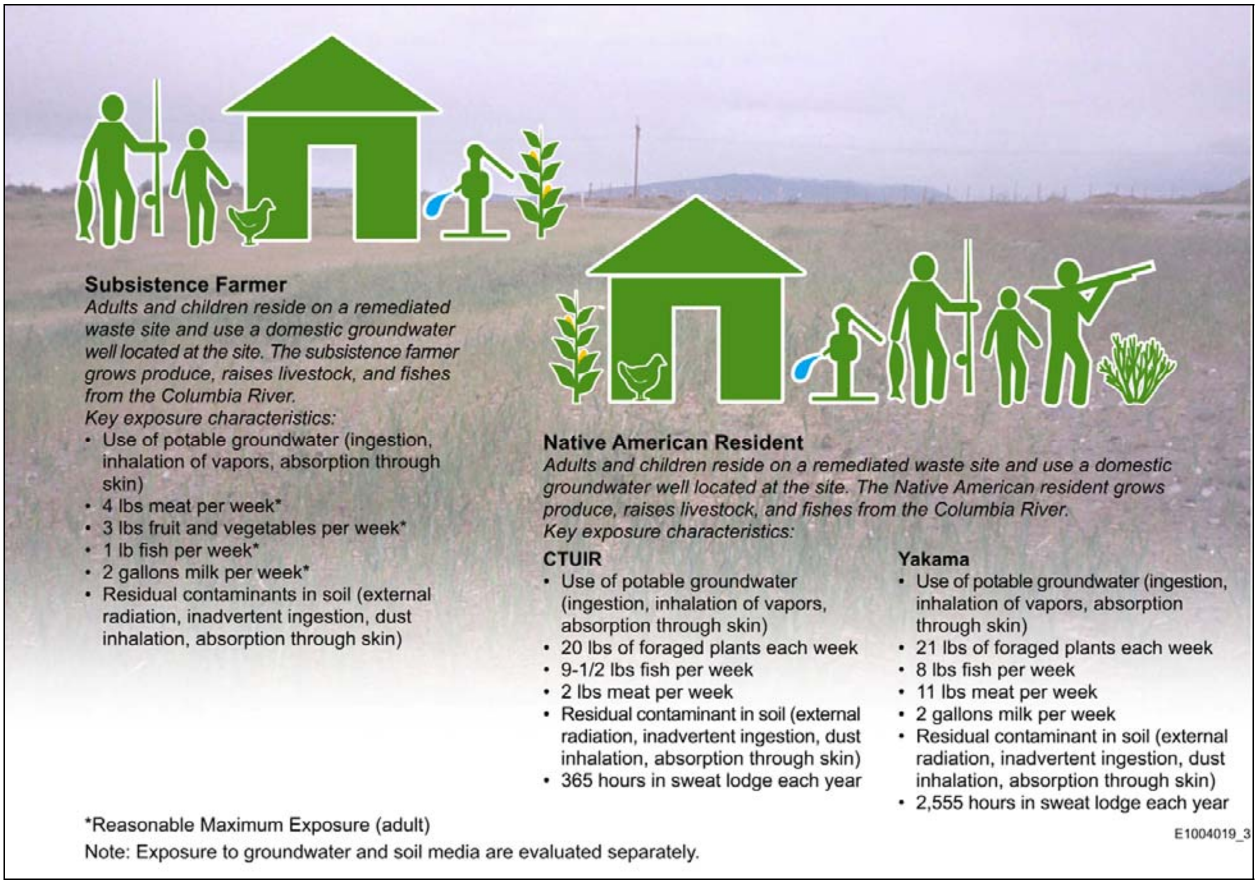
Local-Area Exposure Scenarios – Residential

The Subsistence Farmer and Native American Resident scenarios describe exposures related to residential land-use assumptions that include home-produced foods. The Native American Resident scenarios included separate scenarios submitted by the CTUIR and the Yakama Nation. The residential receptors are assumed to spend effectively all of their time in the area around a residence located on a remediated waste site. Figure ES-4 shows a brief description of the residential exposure scenarios and the key exposure pathways and assumptions.

For the Subsistence Farmer scenario, total cumulative cancer risks at 57 of the 156 remediated waste sites evaluated exceeded the EPA upper risk management level of 1×10^{-4}. Risks up to 30 times the upper risk management level (3×10^{-3}) were calculated. The noncancer chemical hazard results were greater than 1 at approximately 43 of the 156 remediated waste sites evaluated with results up to 220 times the threshold of 1. The primary exposure pathways related to risks above threshold criteria for the subsistence farmer are external radiation from residual levels of cobalt-60, cesium-137, and europium-152 and exposure to residual levels of strontium-90, arsenic, and other metals through ingestion of foodstuffs grown at the residence.

For the Native American scenarios, total cumulative cancer risks at 111 of the 156 remediated waste sites evaluated exceeded the EPA upper risk management level of 1×10^{-4}. The noncancer chemical hazard results exceeded the threshold of 1 at 62 of the 156 remediated waste sites evaluated. The primary exposure pathways for the Native American receptors are similar to the Subsistence Farmer scenario.

Figure ES-4. Local-Area Residential Exposure Scenarios.



Present-day total cancer risks exceeding thresholds for the residential exposure scenarios are almost entirely related to one of three factors:

- External irradiation from short-lived radionuclides including cobalt-60, cesium-137, and europium-152
- Exposure to arsenic from ingestion of garden produce
- Exposure to strontium-90 from ingestion of produce and livestock products.

By the year 2075, a major portion of the radionuclides will have decayed to a nonradioactive form and total cancer risks above the EPA risk management range (1×10^{-4}) for the Subsistence Farmer scenario are largely due to arsenic exposure from produce ingestion.

Because the Native American resident scenarios include very high food ingestion rates, strontium-90 continues to play a significant role in food-related exposures at year 2075. By year 2150, however, Native American resident cancer risks above 1×10^{-4} are also dominated by arsenic exposure from ingestion of garden produce.

Average arsenic concentrations at remediated waste sites range between 1.1 and 17.3 parts per million. Some of these arsenic concentrations exceed the Hanford Site background value of 6.5 parts per million (DOE/RL-92-24). However, all of the RME values for arsenic are less than the IAROD cleanup value of 20 parts per million, which is based on the State of Washington Method A unrestricted cleanup level.

As stated for the occupational scenarios, a key uncertainty in the local-area risk results pertains to the applicability of the cleanup verification soil data at excavated sites for estimating future human exposure at the waste sites that have been backfilled with clean fill material. Actual risks may be overestimated using cleanup verification data from these sites to estimate exposure concentrations in surface soil.

The second key uncertainty for the residential scenarios relates to the food pathways, particularly garden produce. Specific uncertainties related to these pathways include the likelihood that intensive home agriculture will exist in the future, whether any remediated waste site is large enough to reasonably support these activities, and the models used to estimate plant and animal tissue concentrations from the soil exposure concentrations. For example, the produce ingestion noncancer chemical hazard results for mercury, uranium, and copper may have a large conservative bias because soil concentrations of these metals at some sites are well above background levels and plant uptake factors used to model plant tissue concentrations are prone to overestimation in such circumstances. For arsenic, where the range of site soil concentrations is relatively small, uncertainty in produce concentrations is attributable to variability related to soil conditions, plant species, harvest time, and other factors. The arsenic plant uptake factor used for the residential risk calculations is on the upper end of the range of recommended published values.

PRELIMINARY REMEDIATION GOALS

Preliminary remediation goals (PRGs) were developed for soil for a range of exposure scenarios evaluated in the RCBRA. These include the Recreational, Industrial/Commercial Worker, and Resident Monument Worker scenarios. A variety of scenarios and exposure pathways were used to develop the PRGs presented in this section so that remedial alternatives under various land-use assumptions can be evaluated in the RI/FS reports for the River Corridor ROD decision areas. In addition, the PRGs may be used to help determine the final soil cleanup levels for the River Corridor.

The PRGs were developed in accordance with the methodology presented in Part B of the *Risk Assessment Guidance for Superfund* (EPA/540/R-92/003). The PRGs for radionuclides are based on a 1×10^{-4} target cancer risk. The PRGs for chemicals are based on a target cancer risk of 1×10^{-6} and/or a noncancer chemical hazard index of 1 depending on whether the chemical has published toxicity criteria for carcinogenic and/or noncarcinogenic health effects. The PRG values can be found in Section 7.0 of this volume.

> EPA risk assessment guidance (EPA/540/1-89/002) states that remedial actions should be based on an estimate of the reasonable maximum exposure (RME) expected to occur under both current and future land-use conditions. The RME is defined as the highest level of exposure that is reasonably expected to occur at a site. RMEs are estimated for individual pathways and if receptors may be exposed via more than one pathway, the combination of exposures across pathways represents an RME.

FINAL RECOMMENDATIONS

The RCBRA human health risk assessment evaluated total cumulative cancer risks and noncancer chemical hazards associated with conditions at remediated wastes sites and potentially affected areas within the River Corridor. A wide range of scenarios was included ranging from the hypothetical situation where recreational users may occasionally visit the site to residents living onsite and consuming food items that are predominantly raised or grown onsite.

Preliminary remediation goals were also developed for this wide range of scenarios to provide information for analysis of alternatives in the FS.

Uncertainties were identified that impact the estimates of cancer risk and noncancer chemical hazard results for the local-area risk assessment. These include the applicability of using cleanup verification soil data to represent current and future surface soil concentrations and the degree to which foods growing on the remediated waste sites may take up contamination from the soil.

Some data could be collected to help reduce key uncertainties related to the risk assessment results as follows:

- Additional soil sampling at the remediated waste sites could be helpful in establishing the lateral distribution of contaminants that remain after remedial actions are complete.
- Measuring actual contaminant concentrations in edible portions of native plants could help reduce uncertainties related to risks associated with the plant uptake pathways.
- Additional groundwater data collection efforts have also been identified for the RI to help reduce uncertainties related to spatial distribution, chemical composition, and temporal representativeness of the groundwater data.
- Additional sampling of fish in the Hanford Reach of the Columbia River has already been completed to support a comprehensive risk assessment for food fish ingestion.

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APPENDICES

FIGURES

ACRONYMS

AEA	*Atomic Energy Act of 1954*
ALARA	as low as reasonably achievable
ARAR	applicable or relevant and appropriate requirement
bgs	below ground surface
CDC	Centers for Disease Control
CERCLA	*Comprehensive Environmental Response, Compensation, and Liability Act of 1980*
CFR	*Code of Federal Regulations*
CLUP	*Hanford Site Comprehensive Land Use Plan Environmental Impact Statement*
COPC	contaminant of potential concern
COPEC	contaminant of potential ecological concern
CPERA	Central Plateau Ecological Risk Assessment
CRC	Columbia River Component
CRCIA	*Columbia River Comprehensive Impact Assessment*
CSF	cancer slope factor
CSM	conceptual site model
CTE	central tendency exposure
CTUIR	Confederated Tribes of the Umatilla Indian Reservation
CVP	cleanup verification package
DCF	dose conversion factor
DNA	deoxyribonucleic acid
DOE	U.S. Department of Energy
DQO	data quality objective
Ecology	Washington State Department of Ecology
ENRE	Environmental Restoration (database)
EPA	U.S. Environmental Protection Agency
EPC	exposure point concentration
ERA	ecological risk assessment
ERDF	Environmental Restoration Disposal Facility
FFS	focused feasibility study
FR	*Federal Register*
FS	feasibility study
FWS	U.S. Fish and Wildlife Service
FY	fiscal year
GiSdT	Guided Interactive Statistical Decision Tools
GRAS	generally recognized as safe
HEAST	Health Effects Assessment Summary Tables
HEIS	Hanford Environmental Information System
HGP	Hanford Generating Plant
HHRA	human health risk assessment
HI	hazard index

HPPS	Hanford Past-Practice Strategy
HQ	hazard quotient
HSRAM	*Hanford Site Risk Assessment Methodology*
IAROD	Interim Action Record of Decision
IEUBK	Integrated Exposure Uptake Biokinetic Model for Lead in Children
IRIS	Integrated Risk Information System
ISS	interim safe storage
K_d	distribution coefficient
LFI	limited field investigation
LOAEL	lowest observable adverse effect level
MCL	maximum contaminant level
MIS	*MULTI INCREMENT*® sampling
MTCA	*Model Toxics Control Act*
NCEA	National Center for Environmental Assessment
NCP	"National Oil and Hazardous Substances Pollution Contingency Plan"
NEPA	*National Environmental Policy Act of 1969*
NOAEL	no observable adverse effect level
NPL	National Priority List
ORNL	Oak Ridge National Laboratory
OU	operable unit
PAH	polycyclic aromatic hydrocarbon
PCB	polychlorinated biphenyl
PEF	particulate emission factor
PNNL	Pacific Northwest National Laboratory
PPRTV	provisional peer reviewed toxicity value
PQL	practical quantitation limit
QA	quality assurance
QAPP	Quality Assurance Project Plan
QC	quality control
QRA	qualitative risk assessment
RAG	remedial action goal
RAGS	Risk Assessment Guidance for Superfund
RAO	remedial action objective
RCBRA	River Corridor Baseline Risk Assessment
RCRA	*Resource Conservation and Recovery Act of 1976*
RCW	*Revised Code of Washington*
RESRAD	RESidual RADioactivity
RfC	reference concentration
RfD	reference dose
RI	remedial investigation
RL	U.S. Department of Energy, Richland Operations Office
RME	reasonable maximum exposure
ROD	record of decision
RSVP	remaining sites verification package
RTD	remove, treat, dispose

SAP	sampling and analysis plan
SESP	Surface Environmental Surveillance Program
SF	slope factor
SNF	spent nuclear fuel
SSE	safe storage enclosure
TEF	toxicity equivalency factor
Tri-Parties	U.S. Department of Energy, Richland Operations Office, U.S. Environmental Protection Agency, and Washington State Department of Ecology
Tri-Party Agreement	*Hanford Federal Facility Agreement and Consent Order*
UCL	upper confidence limit
UF	uncertainty factor
UR	unit risk
VOC	volatile organic compound
WAC	*Washington Administrative Code*
WCH	Washington Closure Hanford
WDOH	Washington State Department of Health
WIDS	Waste Information Data System
WRS	Wilcoxon rank sum

GLOSSARY

abiotic: Nonliving, chemical and physical factors in the environment.

baseline risk assessment: A baseline risk assessment is an analysis of the potential for adverse health effects (current or future) caused by hazardous substance releases from a site in the absence of any actions to control or mitigate these releases. The baseline risk assessment contributes to the site characterization and subsequent development, evaluation, and selection of appropriate remedial alternatives. The results of the baseline risk assessment are used to help determine whether additional remedial action is necessary at the site, modify preliminary remediation goals, and document the magnitude and the primary contributors to risk at a site.

biotic: Relating to, produced by, or caused by living organisms.

cancer slope factor: An upper bound, approximating a 95% confidence limit, on the increased cancer risk from a lifetime exposure to a potential carcinogen. This estimate, usually expressed in units of proportion (of a population) affected per milligram per kilogram per day, is generally reserved for use in the low-dose region of the dose-response relationship, that is, for exposures corresponding to risks less than 1 in 100.

carcinogen: A substance or agent that can cause cancer in humans or animals.

central tendency exposure: A calculation that assesses potential exposure for a member of the population with an average level of exposure intensity in the context of a particular exposure scenario, and uses "best estimate" values for average contaminant concentrations in environmental media.

conceptual site model: A description or model of the study area that identifies the sources of contamination, describes transport mechanisms through various environmental media, and identifies potential receptors that may be important in evaluating potential risks associated with exposures to human and ecological receptors.

contaminant of potential concern: A substance detected at a hazardous waste site that has the potential to affect a receptor adversely due to its concentration, distribution, and mode of toxicity.

correlation: An estimate of the degree to which two sets of variables vary together, with no distinction between dependent and independent variables.

dose conversion factor: A specific value that provides the dose equivalent for a unit intake of a radionuclide.

dose-response evaluation: A quantitative estimate of the relationship between exposure level and the incidence of effects.

exposure assessment: The evaluation of the magnitude, frequency, and duration of exposure of receptors to environmental contaminants.

exposure pathways: The manner in which an individual or components of a biological system are exposed to contaminants at a site. An exposure pathway includes a source or release, an exposure point, a receptor, and an exposure route.

exposure point concentration: The concentration of a contaminant at a location of potential contact with an organism.

exposure route: The way in which an individual is exposed to a contaminant in an environmental medium (e.g., ingestion, inhalation).

hazard index: The sum of the hazard quotients (see hazard quotient) for all chemicals.

hazard quotient: The ratio of an exposure level for a contaminant to a reference dose for that chemical. If the exposure level is higher than the reference dose (e.g., greater than 1) then there is the potential for an adverse health effect in the receptor.

human health risk assessment: An analysis of the potential for adverse health effects to an individual or a population caused by exposure to one or more hazardous substances.

intake: A specific measure of exposure expressed as the amount of contaminant taken into the body per unit body weight per time (e.g., milligram contaminant per kilogram body weight per day).

interim action record of decision: A legal document that selects, explains, and provides justification for the interim remedial actions that will be taken to clean up a CERCLA site.

interstitial water: The water filling the spaces between grains of sediment, also referred to as pore water.

limited field investigation: An investigation that consisted of historical data compilation; nonintrusive investigations (such as geophysics); intrusive investigations (such as boreholes); and the 100 Area aggregate studies, which included ecological, river water, and sediment sampling. The limited field investigations identified the sites that were candidates for interim remedial action, and provided a preliminary summary of site characterization studies and other information.

National Priorities List: A list of hazardous waste sites eligible for long-term remedial action under the federal CERCLA program.

natural attenuation: The natural dilution, dispersion, (bio)degradation, irreversible sorption, and/or radioactive decay of contaminants in soils and groundwater.

near-shore aquatic zone: The near-shore aquatic zone consists of a narrow band of the Columbia River adjacent to the shoreline to a depth of 1.8 m (6 ft).

nonparametric: Statistical methods that make no assumptions regarding the distribution of the data.

parametric: Statistical methods that assume a particular statistical distribution of the data.

qualitative risk assessment: A risk assessment that is based on qualitative data or that gives a qualitative result. The results are often stated as an estimated range of potential effects, such as low, moderate, or high risk.

radiation dose: A measure of the amount of ionizing energy from radioactive decay absorbed by a biological tissue.

reasonable maximum exposure: A calculation that assesses exposure to individuals whose behavioral characteristics may result in higher potential exposure than seen in the average individual. The reasonable maximum exposure calculations are based on a combination of upper-bound and average values for various exposure parameters.

receptor: The individual, species, population, community, or habitat that may be exposed to contaminants.

record of decision: A legal document that selects, explains, and provides justification for the remedial actions that will be taken to clean up a CERCLA site.

reference dose: An estimate (with uncertainty spanning perhaps an order of magnitude) of a daily exposure to the human population (including sensitive subgroups) that is likely to be without an appreciable risk of adverse effects during a lifetime.

reference site: A comparatively uncontaminated site used for comparison to contaminated sites in environmental monitoring studies. It can be the least impacted/unimpacted area of the site or a nearby site that is ecologically similar but not affected by the contaminants at the site under investigation.

representative concentration: The representative concentration is used to evaluate the potential for human or ecological exposure from sampled environmental media (soil, sediment, water, biota). The representative concentration is assumed to be representative of exposure to the sample medium over an area and time period consistent with the exposure assessment.

riparian zone: The riparian zone extends from the shoreline of the Columbia River to the point on the riverbank where upland vegetation becomes dominant.

risk: The probability of adverse effects resulting from known or estimated exposure to a contaminant.

risk characterization: A numerical expression of the potential for adverse health effects accompanied by text that interprets and qualifies the numerical results.

toxicity assessment: An evaluation of the potential of a contaminant to cause adverse effects. When possible, a toxicity assessment also includes a dose-response evaluation.

uncertainty analysis: A component of the risk characterization where the assumptions, biases, and uncertainties inherent in the risk assessment are explained in order to provide a context for interpreting the numerical risk estimates and support identification of potential data gaps.

upland zone: The upland zone consists of land that may be adjacent to the main channel of the Columbia River, but is situated at least 3 m (10 ft) above the river high-water mark.

1.0 INTRODUCTION

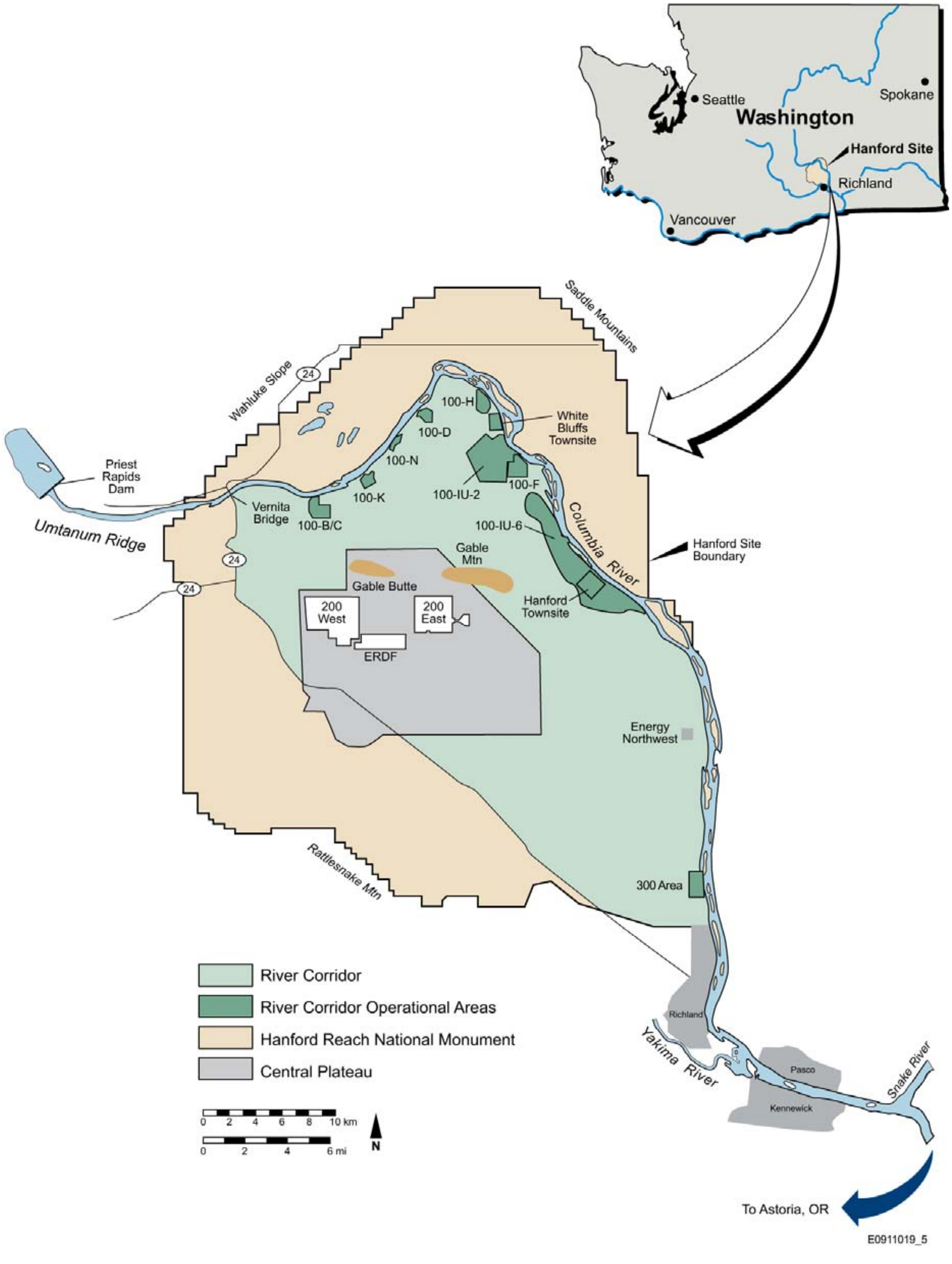
The U.S. Department of Energy's (DOE) Hanford Site is a 1,517-km^2 (586-mi^2) federal facility located within the semiarid shrub-steppe Pasco Basin of the Columbia Plateau in south-central Washington State. The site is situated north and west of the cities of Richland, Kennewick, and Pasco, an area commonly known as the Tri-Cities. The Columbia River flows through the northern part of the Hanford Site and, turning south, forms part of the site's eastern boundary. The general area and major features of the Hanford Site are shown in Figure 1-1.

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. The Hanford Site's production mission continued until the late 1980s, when the mission changed to cleaning up the radioactive and hazardous wastes that had been generated during production in the previous decades. In 1989, the 100 Area (encompassing the former plutonium production reactors along the Columbia River) and the 300 Area (which housed reactor fuel fabrication plants as well as many research and development projects) were two of four areas at the Hanford Site 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. Together, the 100 and 300 Areas comprise a 570-km^2 (220-mi^2) portion of the Hanford Site that is called the River Corridor (see Figure 1-1).

In order to allow cleanup to begin as soon as possible, the DOE, the U.S. Environmental Protection Agency (EPA), and the Washington State Department of Ecology (collectively called the Tri-Parties) developed an approach for expedited remediation of the River Corridor in 1991. Cleanup decisions were established through interim action records of decision (IARODs) based on existing knowledge of the waste sites (e.g., site types, processes, contaminants) and supplemented by limited amounts of characterization. In 1995, cleanup actions were initiated focusing on removal of contaminated soil and debris from waste sites with the highest potential to impact groundwater and the Columbia River. Actions to address existing plumes of groundwater contamination were also initiated.

Waste site and groundwater cleanup actions in the River Corridor have continued from 1995 to date. During that time, about 8 million tons of contaminated soil and debris has been removed from nearly 300 waste sites in the River Corridor and disposed of at authorized facilities. More than 2 billion gallons of contaminated groundwater has been processed through pump-and-treat systems. At each waste site where remediation has occurred, the goals and objectives of the IARODs have been met as demonstrated by verification documentation that has been completed and approved by the DOE and the regulatory agencies.

Figure 1-1. Geographical Boundaries and General Features of the Hanford Site.



Cleanup actions in the River Corridor are not complete. Many waste sites and groundwater plumes that have been identified for cleanup actions in the IARODs have yet to be addressed. Consequently, waste site and groundwater cleanup actions in the River Corridor will continue for several years. In parallel with continuing the cleanup actions outlined in the existing IARODs, the Tri-Parties have established a strategy to develop final waste site and groundwater cleanup decisions for the River Corridor. Remedial investigation/feasibility study (RI/FS) processes have been initiated for the River Corridor to gather and evaluate information needed to make final cleanup decisions. A key element in the RI/FS decision-making process is the performance of a baseline risk assessment.

Under CERCLA, a baseline risk assessment is needed to provide risk managers with an understanding of the current and potential future risks posed by a site. The River Corridor Baseline Risk Assessment (RCBRA) is being conducted part way through cleanup actions. As such, baseline conditions that are assessed include a mix of areas where cleanup has been completed in accordance with the IARODs, areas that are currently scheduled for cleanup, and areas that are currently not identified for cleanup actions.

This human health risk assessment (Volume II) provides the human health portion of the RCBRA and presents a comprehensive assessment of the River Corridor, considering all relevant sources of contamination, exposure pathways, and contaminants. Volume II will be used, with a complementary ecological risk assessment (Volume I), to support final cleanup 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 develop final cleanup decisions that will be protective of human health and the environment. Final cleanup decisions, applying to all portions of the River Corridor, will be identified in proposed plans that will undergo public review and documented in records of decision (RODs).

1.1 PURPOSE

The purpose of the RCBRA is to characterize current and potential future risks to human health and the environment that may be posed by releases of hazardous substances in the River Corridor of the Hanford Site. DOE is required to assess human and ecological risk under CERCLA, *Resource Conservation and Recovery Act of 1976* (RCRA), *National Environmental Policy Act of 1969*, and DOE orders. The "National Oil and Hazardous Substances Contingency Plan" (40 *Code of Federal Regulations* [CFR] 300), which implements CERCLA, specifically requires a site-specific baseline risk assessment to determine the need for action at sites, determine levels of contaminants that can remain onsite and still be protective, and provide a basis for comparing health impacts of various cleanup alternatives (40 CFR 300.430[d][4]).

Per the EPA (EPA/540/1-89/002, *Risk Assessment Guidance for Superfund Volume I Human Health Evaluation Manual [Part A], Interim Final*), a baseline risk assessment is an "analysis of the potential adverse health effects (current or future) caused by hazardous substance releases from a site in the absence of any actions to control or mitigate these releases (i.e., under an assumption of no action)."

The baseline risk assessment is part of the CERCLA RI/FS process. The RI/FS is the methodology the CERCLA program has established for characterizing the nature and extent of contamination associated with releases of hazardous substances to the environment, for assessing the potential risks posed by the environmental contamination to human and ecological receptors, and for developing and evaluating remedial options. Because the RI/FS is a process designed to support risk management decision making for CERCLA sites, the assessment of human health and environmental risk serves an essential role in the RI/FS process. The baseline risk assessment provides information to assist in the development, evaluation, and selection of appropriate response alternatives. The results of the baseline risk assessment are used to determine whether additional response action is necessary at the site, modify preliminary remediation goals, support selection of the ”no-action” remedial alternative where it is appropriate, and document the magnitude of risk and primary contributors (e.g., chemicals and exposure pathways) to risk at a site.

The primary human health risk assessment goal for CERCLA sites is to support remedial action decisions that reduce potential current and future risks to acceptable levels. The RCBRA evaluates contaminants potentially related to historical site releases of hazardous substances to the soil and groundwater in the River Corridor. The RCBRA addresses residual contamination at individual waste sites in the River Corridor and in addition, addresses residual contaminant concentrations related to the past transport of contaminants into Columbia River riparian and near-shore environments adjacent to and between the operational areas.

Results and conclusions from this human health risk assessment will be presented with the results from the ecological risk assessment (Volume I) in remedial investigation reports to support development of final cleanup actions for the River Corridor. The RCBRA will address the following questions that will provide information needed by risk managers to support final CERCLA decisions in the River Corridor that ensure protection of human health and the environment.

- Are cleanup levels currently established under the IARODs protective of human health and the environment?

- Are residual conditions for cleanup actions completed under the IARODs protective of human health and the environment?

- What are the uncertainties associated with the risk results and conclusions?

- What are recommended preliminary remediation goals that could be used to be protective of human health and the environment?

- Are there any recommendations for additional studies or monitoring that should be considered at this time to reduce uncertainties with specific risk results and conclusions?

This document is an important component of the overall cleanup mission at the Hanford Site. Site characterization and risk information from the River Corridor is part of the larger framework to provide risk characterization and guide risk management decisions for the principal Hanford cleanup components (i.e., River Corridor, Central Plateau, and Tank Waste). The overarching goals for the cleanup mission at Hanford include protecting the Columbia River, restoring groundwater to its intended beneficial use, remediating waste sites to protect human health and the environment while minimizing the cleanup footprint, and safely implementing long-term stewardship activities as necessary to monitor the performance of selected cleanup alternatives.

1.2 SCOPE

The scope of the RCBRA is a comprehensive assessment of current conditions for the River Corridor, considering relevant sources of contamination, exposure pathways, and contaminants. For this assessment, current conditions are represented through the end of 2005 in accordance with the work plan (DOE/RL-2004-37, *Risk Assessment Work Plan for the 100 Area and 300 Area Component of the RCBRA*).

The natural setting of the River Corridor includes three main types of environments: the upland, riparian, and near-shore environments, including soil, sediments, groundwater, and river water (the abiotic media) as well as ecological resources within them. Human receptors may be exposed to residual contaminants in one or more of the three environments depending on their activities. The human health risk assessment provides an evaluation of potential exposure to residual contamination at individual waste sites which are located within the upland environment (referred to as the “local-area” assessment in the document). In addition, potential human health risks are evaluated on a “broad-area” basis for receptors that may engage in activities that span large areas and include more than one environment.

This human health risk assessment addresses current conditions[1] in the River Corridor based on historic information, results of site characterization, and cleanup actions completed in accordance with the IARODs. Current conditions can be generally described as the following two types of areas:

- Operational areas where releases of hazardous substances posing a threat to human health and the environment are known to have occurred. These areas are identified as waste sites and include inactive ponds, trenches, burial grounds, landfills, and spill sites as well as inactive structures and facilities. Operational areas are typically found in the upland environment and are composed of a mix of remediated waste sites and yet-to-be-remediated waste sites.

- Nonoperational areas where historic information and other studies indicate there were no direct releases of hazardous substances posing a threat to human health or the environment. Nonoperational areas may be affected by transport of contaminants from operational areas through mechanisms such as fugitive dust, groundwater migration, or biotic uptake for example. Nonoperational areas include large portions of the River Corridor that are outside of the operational areas and are not anticipated to be impacted by Hanford Site releases.

As described above, operational areas include identified waste sites that are accepted in the Waste Information Data System and have been or are scheduled for cleanup. This risk assessment and the remedial investigation process may lead to identification of additional waste sites. Nonoperational areas may also contain yet unknown direct releases of hazardous substances that pose a threat to human health and the environment. These areas would be entered into the Waste Information Data System for disposition as accepted waste sites.

1.3 REQUIREMENTS AND GUIDANCE

This section summarizes requirements and guidance that establish the basis for a baseline risk assessment and provide the framework and methodology for human health risk assessment.

1.3.1 Tri-Party Agreement

The Tri-Parties signed a comprehensive cleanup and compliance agreement on May 15, 1989. The *Hanford Federal Facility Agreement and Consent Order* (Ecology et al. 1989), or Tri-Party Agreement, is a CERCLA federal facility agreement. It also is a framework for implementing the many environmental regulations that apply to the Hanford Site including the RCRA treatment, storage, and disposal unit regulations and corrective action provisions. The Tri-Party Agreement defines and ranks CERCLA and RCRA cleanup commitments, establishes

[1] For the purposes of this report, current conditions in the River Corridor are represented by the conditions present when the project work plan (DOE/RL-2004-37) was written. The work plan and ensuing planning documents defined the scope of the project based on the extent of remediation and characterization that was complete at the time. This corresponds to roughly the end of 2005.

responsibilities, provides a basis for budgeting, and reflects a concerted goal of achieving full regulatory compliance and remediation with enforceable milestones in an aggressive manner.

Additionally, an associated plan called the "Community Relations Plan" describes how the public will be informed and involved throughout the cleanup process. Throughout the risk assessment planning and implementation stages, a number of workshops and meetings were held to facilitate participation of interested parties including state and federal agencies, natural resource trustees, site contractors, and the public. These workshops served as important forums for soliciting input and feedback for project objectives, study design, and resource protection. Meeting notes from the data quality objective (DQO) and sampling and analysis plan workshops are provided on the Washington Closure Hanford Mission Completion project library web site at http://www.washingtonclosure.com/Projects/EndState/100-300_comp.html.

1.3.2 CERCLA

A site-specific baseline risk assessment is required by 40 CFR 300 as part of the CERCLA remedial action process. Therefore, human health and ecological risk assessments are required at all CERCLA sites. The baseline risk assessment evaluates current and future potential threats to human health and the environment. The scope of the human health and ecological risk assessment processes depend on site-specific factors such as reasonably anticipated future land use and anticipated beneficial uses of groundwater and surface water.

The baseline risk assessment provides a framework for developing risk information necessary to support decision making. The results of the risk assessment include development of a range of media-specific risk-based concentrations that would be protective of human and ecological receptors. These risk based concentrations will be considered for risk management decision making. Risk management decisions resulting from the risk assessment could include additional actions required to protect human health and the environment, development of revised remedial action objectives, or development of risk-based cleanup levels. However, target cleanup levels can also be derived from applicable or relevant and appropriate requirements rather than risk-based concentrations. Applicable or relevant and appropriate requirements typically used to establish risk-based concentrations include maximum contaminant levels for groundwater protection, ambient water quality criteria for surface water protection, and cleanup levels based on state cleanup laws and regulations.

1.3.3 Cultural Resources Management Plan

As a federal agency, the DOE has assumed a stewardship role on behalf of the American public with respect to managing the Hanford Site and the cultural resources contained therein. With the issuance of the *Hanford Cultural Resources Management Plan* (DOE/RL-98-10), the DOE Richland Operations Office has set forth its objective to ensure that the cultural resources entrusted to its care are managed with vision, leadership, care, and responsibility and are given full consideration in land-use planning and management decisions. Cultural and historic resource protection laws and regulations do not establish any specific standards pertaining to risk assessment.

1.3.4 Model Toxics Control Act

The “Model Toxics Control Act—Cleanup” (WAC 173-340) embodies the hazardous substance cleanup regulations for the State of Washington. Certain protectiveness standards for WAC 173-340 are pertinent to the baseline risk assessment effort.

1.3.5 Water Quality Criteria

Water quality criteria established under the *Clean Water Act of 1977* are pertinent to defining protection of the Columbia River from discharges from the Hanford Site (e.g., groundwater that discharges into the river). Ambient water quality criteria establish surface water standards for protection of human health (including ingestion of contaminated drinking water and contaminated fish) and for protection of aquatic life. Under *Clean Water Act of 1977* authority, the EPA periodically publishes recommended ambient water quality criteria in the Federal Register and promulgates criteria for certain toxic pollutants at 40 CFR 131, “Water Quality Standards.” State standards for surface water are based on the EPA criteria and are promulgated at WAC 173-201A-040, “Water Quality Standards for Surface Waters of the State of Washington.” WAC 173-340-730(6) establishes the point of compliance for groundwater discharges into surface water as close as technically possible to the actual point of groundwater flow into the surface water and prohibits the use of a mixing zone to achieve attainment of state water quality criteria.

1.3.6 Human Health Risk Assessment Guidance

The human health risk assessment evaluated the current and potential future risks to human receptors associated with releases of hazardous substances to the soil, sediment, and groundwater of the River Corridor. The human health risk assessment in this volume follows the evaluation process described in *Risk Assessment Guidance for Superfund, Volume 1, Human Health Evaluation Manual (Part A) (Interim Final)* (RAGS) (EPA/540/1-89/002). The goal of the Superfund human health evaluation process described in RAGS is to provide a framework for developing the risk information necessary to assist decision-making at remedial sites. Specific objectives of the process are to:

- Provide an analysis of baseline risks and help determine the need for action at sites
- Provide a basis for determining levels of chemicals that can remain onsite and still be adequately protective of public health
- Provide a basis for comparing potential health impacts of various remedial alternatives
- Provide a consistent process for evaluating and documenting public health threats at sites.

There are four steps in the baseline human health risk assessment process. These four steps are as follows:

1. Data collection and evaluation – site data are compiled and analyzed and contaminants of potential concern (COPCs) are identified.

2. Exposure assessment – potentially complete exposure pathways are identified including potential receptors and environmental media to which they may be exposed. Exposure parameters for the various receptors/exposure scenarios are identified.

3. Toxicity assessment – toxicity criteria used in the quantitative risk calculations are compiled.

4. Risk characterization – quantitative risk estimates are calculated and uncertainties associated with the risk estimates are identified and summarized.

1.4 INTEGRATION WITH HANFORD SITE RISK ASSESSMENTAND REMEDIAL INVESTIGATION/FEASIBILITY STUDY ACTIVITIES

The RCBRA is an important component of the overall cleanup mission at the Hanford Site. Site characterization and risk information from the River Corridor is part of the larger framework to provide risk characterization and guide risk management decisions for the principal Hanford cleanup components. This section summarizes the relationship of RCBRA with risk assessment and RI/FS activities that are being conducted in the River Corridor, Columbia River, and Central Plateau.

1.4.1 River Corridor Source and Groundwater Remedial Investigation/ Feasibility Study

Early cleanup actions in the River Corridor were authorized via a group of IARODs that were supported by qualitative risk assessments to establish a need for action. The Tri-Parties have established a strategy and begun the RI/FS process to develop final soil and groundwater cleanup decisions for the River Corridor. The RODs that are produced from this effort will establish the final remedial goals and objectives and any associated actions required to complete CERCLA cleanup for the River Corridor. The process to pursue final cleanup decisions has been organized into smaller pieces of work that are more manageable and aligned with Hanford Site operational functions. Six final remedy RODs will be developed associated with the operations areas.

The final remedy decision areas and the size of each are as follows:

- 100-B/C 4.45 mi^2
- 100-D/100-H 7.84 mi^2
- 100-K 3.47 mi^2
- 100-N 3.43 mi^2

- 100-F/100-IU-2/100-IU-6 145.23 mi^2 (100-F Reactor area = 2.10 mi^2)
- 300 Area 56.35 mi^2.

Figure 1-2 shows the location and boundaries of the decision areas. Each of the six final remedy RODs will be integrated to address both source and groundwater remedial actions for the decision area. Results and conclusions from the RCBRA are an integral component of the RI/FS reports for the River Corridor, providing essential information to enable risk managers to make decisions regarding final cleanup actions. If any additional characterization or monitoring activities are needed to help reduce uncertainties with RCBRA results and conclusions, they will be documented in the RI/FS reports. Some activities may be required before final cleanup decisions can be made, while others may be conducted as post-ROD activities as part of the implementation processes.

The schedule for preparing and submitting the RI/FS reports (Draft A) for the six River Corridor decision areas is governed by a series of Tri-Party Agreement targets and milestones as shown below. The RCBRA needs to be complete to support the schedules for the earliest RI/FS report.

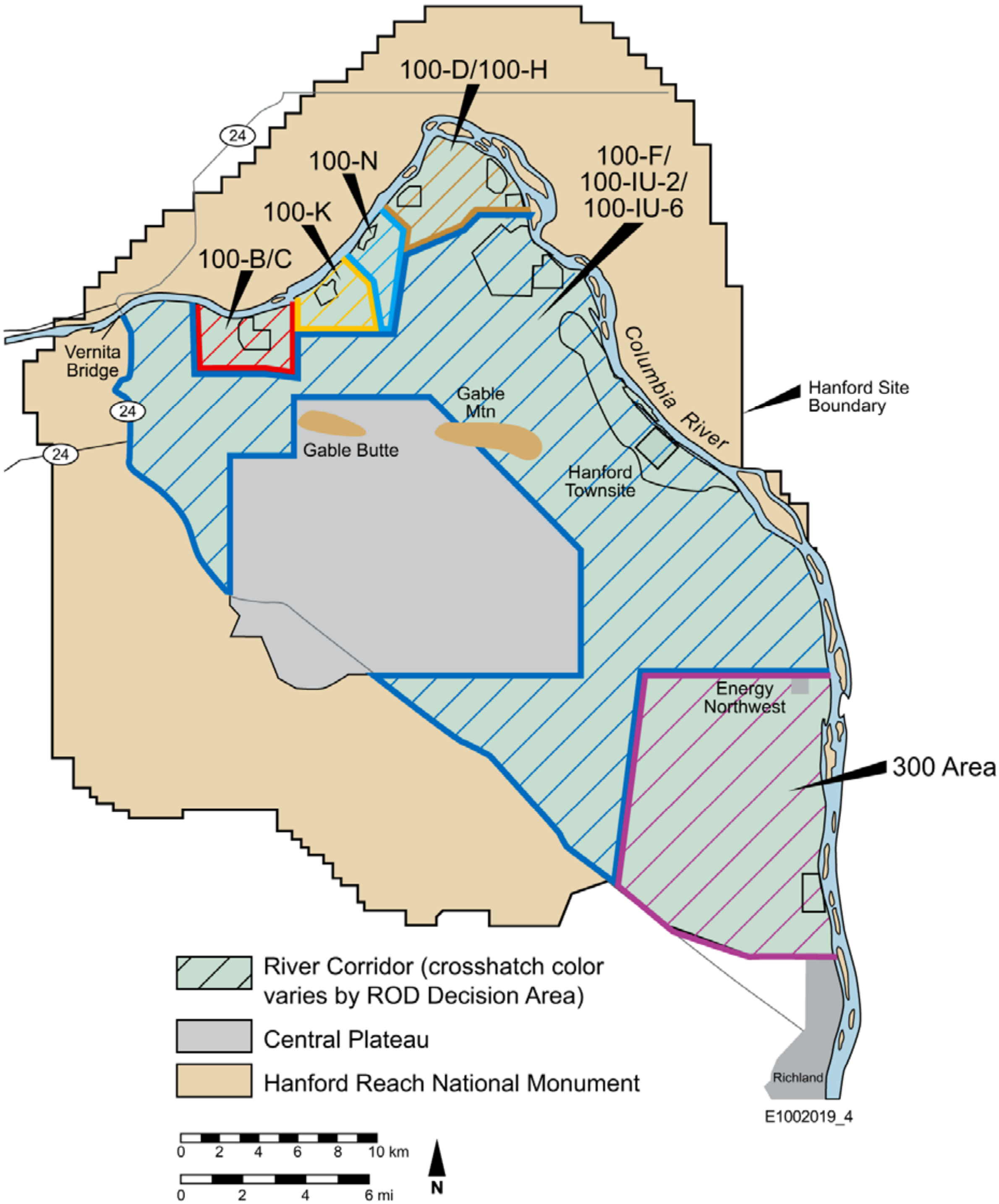
Decision Area	Draft A Submittal	Milestone
100-D/H	7/30/2011	M-015-70-T01
100-K	7/30/2011	M-015-66-T01
100-B/C	11/30/2011	M-015-68-T01
100-F/100-IU-2/100-IU-6	11/30/2011	M-015-64-T01
100-N	12/31/2011	M-015-62-T01
300	12/31/2011	M-015-72-T01

1.4.2 Other Hanford Site Risk Assessments and Studies

Information from previous risk assessment projects in the River Corridor is being integrated into the RCBRA. Results from the 100-B/C Pilot Study and the 100-NR-2 ecological risk assessment (DOE/RL-2005-22, *100-NR-2 Study Area Ecological Risk Assessment Sampling and Analysis Plan*) are part of the data set being used for the RCBRA and are integrated into the report to present a comprehensive picture of current and potential threats to human health and the environment from contaminants in the River Corridor.

Information gathered and lessons learned from a prior study called the Columbia River Comprehensive Impact Assessment (CRCIA) were also incorporated into the RCBRA. The purpose of CRCIA was to assess the effects of Hanford Site-derived materials and contaminants on the Columbia River environment, river-dependent life, and users of river resources for as long as those contaminants remain intrinsically hazardous. The CRCIA screening assessment scope included current conditions, the Columbia River and adjacent riparian zone between Priest Rapids Dam and McNary Dam, a limited number of contaminants, a limited amount of monitoring data, a limited number of species, and a limited number of scenarios. Several documents were published during the course of the CRCIA project, the most comprehensive of which is *Screening Assessment and Requirements for a Comprehensive Assessment*: *Columbia River Comprehensive Impact Assessment* (DOE/RL-96-16).

Figure 1-2. Decision Areas for the River Corridor.



The CRCIA study was instrumental in fostering an open process for assessing risk through the Tri-Party Agreement as well as other agreements and policies. The process resulted in an increased understanding of risk assessment among stakeholders and the formation of important working relationships. A significant technical contribution from CRCIA was the recognition of Native American exposure scenarios in risk assessment methodology for the Hanford Site.

1.4.3 Remedial Investigation of Hanford Site Releases to the Columbia River

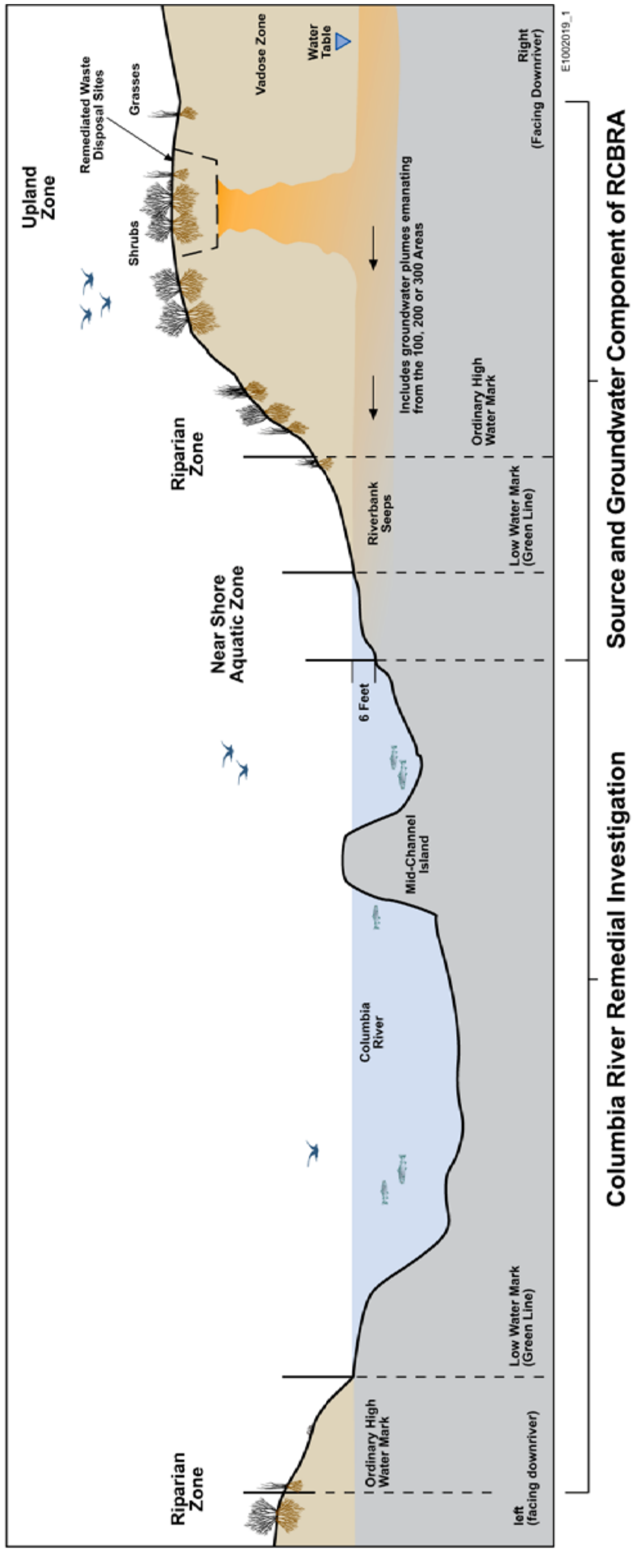
A primary objective of the Hanford Site cleanup mission is protection of the Columbia River through remediation of contaminated soil and groundwater that resulted from its weapons production mission. The impacts of Hanford Site hazardous substance releases to the Columbia River in areas upstream, within, and downstream of the Hanford Site boundary have been previously investigated as mandated by the DOE requirements under the *Atomic Energy Act of 1954*. The impacts are now being assessed under CERCLA via a remedial investigation, *Remedial Investigation Work Plan for Hanford Site Releases to the Columbia River* (DOE/RL-2008-11). The purpose of the remedial investigation is to:

- Characterize the nature and extent of Hanford Site-related contaminants that have come to be located within the Columbia River
- Assess the current risk to ecological and human receptors posed by Hanford Site-related contaminants
- Determine whether or not any cleanup actions are needed to lower the risk to ecological or humans receptors from being exposed to Hanford Site-related contaminants.

The *Remedial Investigation Work Plan for Hanford Site Releases to the Columbia River* (DOE/RL-2008-11) was developed and issued to initiate the remedial investigation. The work plan established a phased approach to characterize contaminants, assess current risks, and determine whether or not there is a need for any cleanup actions in the river. Field investigation activities over a 193-km (120-mi) stretch of the Columbia River began in October 2008 and are anticipated to be complete by mid-2010.

Beginning in early 2010, results from the new samples collected during the field investigation will be combined with existing data to conduct baseline human health and ecological risk assessments for the river component. The risk assessments to evaluate Hanford Site releases to the Columbia River will be performed in a manner that builds on, and is consistent with, the assessment conducted for the RCBRA. Hanford Site-related contaminants that are being evaluated by the RCBRA are being investigated in the river channel to the opposite shoreline and downstream where Hanford Site contaminants may have come to be located. Within the Hanford Reach area, the scope of the river investigation and assessments begins where the RCBRA left off at the near-shore zone, as depicted in Figure 1-3.

Figure 1-3. Remedial Investigation Area Within the Hanford Site.



The baseline risk assessments will help inform decision makers on whether or not there is a need for additional CERCLA investigation or response actions within the Columbia River. This scientific/management decision point is scheduled to occur in 2011 and is aligned to provide input to the RI/FS process being conducted to support each of the six source and groundwater RODs for the River Corridor. If any cleanup actions are needed to address Hanford Site contamination in the river, they may be included with the final decisions for one or more of the six areas. It is also possible that a separate cleanup decision could be made that is specific to the Columbia River. The objective for all of these decisions would be to protect human health and the environment.

1.4.4 Central Plateau Outer-Zone Remedial Investigation/Feasibility Study

The Central Plateau includes approximately 195 km^2 (75 mi^2) in the central portion of the Hanford Site and was primarily used for nuclear fuel processing and waste management/disposal activities. The DOE intends to minimize the area of the Central Plateau requiring long-term institutional controls to maintain protection of human health and the environment. Consequently, an “outer zone” strategy has been established for a large portion of the Central Plateau to guide near-term cleanup efforts. Outer-zone cleanup goals are envisioned to be similar to those that will be established for the River Corridor decision units.

The RI/FS process for the Central Plateau outer zone is being developed in a work plan that is scheduled for submittal in 2010. As with the River Corridor, a baseline risk assessment is a key element supporting the RI/FS process for the Central Plateau outer zone.

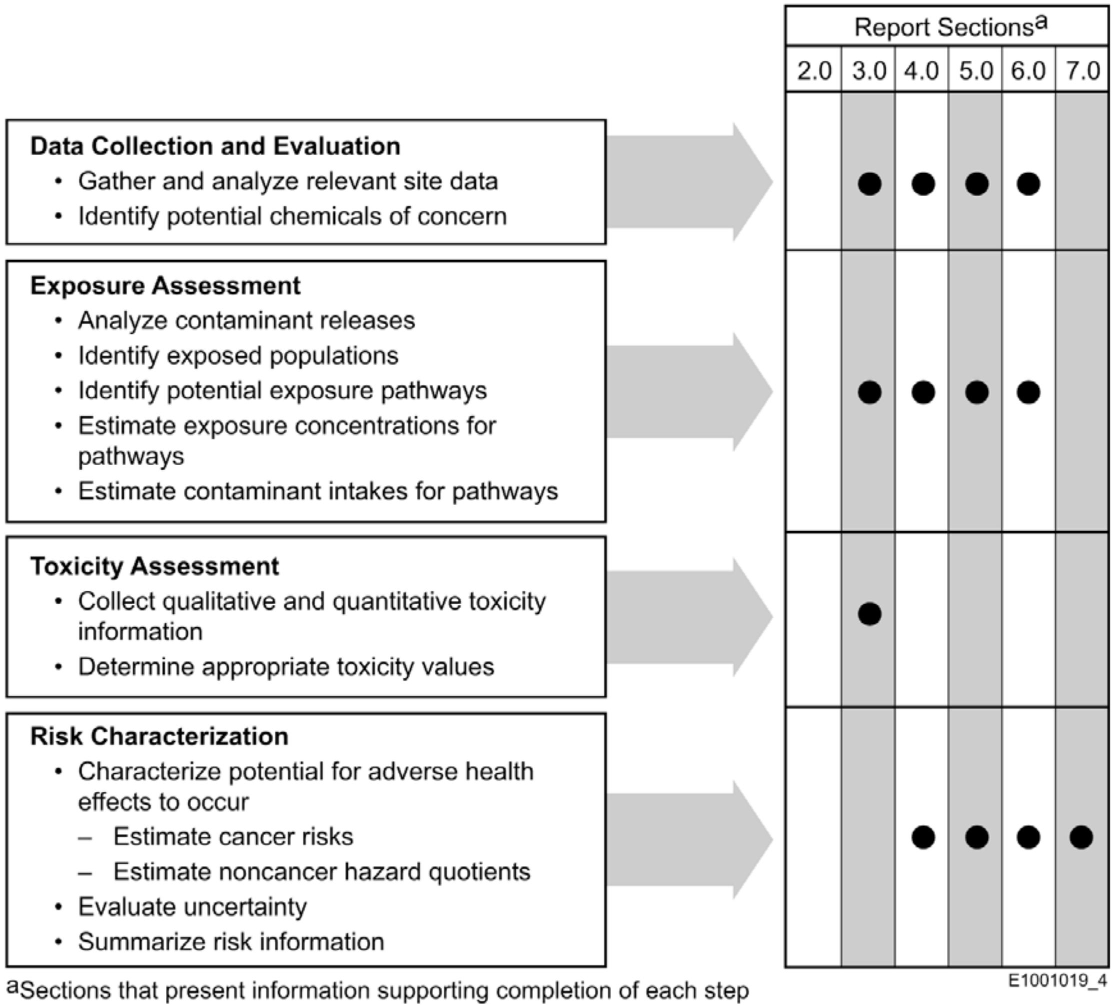
1.5 ORGANIZATION OF THE REPORT

The *River Corridor Baseline Risk Assessment* is presented in two volumes. Volume I contains the ecological risk assessment portion of the report, and Volume II contains the human health risk assessment portion of the report. The volumes are complementary but are written to stand alone with separate executive summaries, discussions, and conclusions. Figure 1-4 shows the linkage of the RAGS four-step process to the RCBRA Volume II report.

Volume II, *Human Health Risk Assessment* (this report), is composed of two parts. Volume II, Part 1 contains the text, figures, and references. Volume II, Part 2 contains the tables that support the text in Part 1. The tables correspond to the callouts in Part 1 and are organized by the applicable section. The tables were organized separately to facilitate side-by-side review of the information and the supporting data.

Technical appendices that provide additional supporting information are contained on a data compact disk (CD) that accompanies this report. The data CD also contains electronic files for the report sections and tables.

Figure 1-4. Linkage of the RCBRA Volume II Report Sections to the U.S. Environmental Protection Agency Four-Step Human Health Risk Assessment Process.



Volume II, *Human Health Risk Assessment*, is organized as follows.

- Section 1.0 provides a brief introduction to the Hanford Site and describes the purpose, scope, and regulatory framework for the risk assessment.

- Section 2.0 provides an overview of the environmental setting and historical background of the Hanford Site. This section also provides an overview of the cleanup actions conducted under CERCLA and summarizes the status of the cleanup as it relates to current levels of risk.

- Section 3.0 provides a summary of the conceptual site model (CSM) and the guidance, methods, assumptions, and data sources used in the human health risk assessment. This section describes how the four steps of RAGS are implemented. An overview of the data collection and evaluation step in provided in this section. The exposure assessment is included with a description of the various exposure scenarios that are quantitatively evaluated in the human health risk assessment. The toxicity assessment is also included in this section including a summary of toxicity criteria used in the risk calculations.

- Section 4.0 provides the broad-area risk assessment. The data sets used in the broad-area assessment are described along with the COPCs that are identified using those data sets. An analysis of baseline risks for the broad-area scenarios is presented and uncertainties related to the broad-area results are discussed.

- Section 5.0 provides the local-area risk assessment. The data sets used in the local-area assessment are described along with the COPCs that are identified using those data sets. An analysis of baseline risks for the local-area scenarios is presented and uncertainties related to the local-area results are discussed.

- Section 6.0 provides the groundwater risk assessment. The data sets used in the groundwater risk assessment are described along with the COPCs that are identified using those data sets. An analysis of baseline risks for groundwater is presented and uncertainties related to the groundwater results are discussed.

- Section 7.0 provides a summary of the broad-area, local-area, and groundwater risk assessments. In addition, human health preliminary remediation goals (PRGs) are presented for various scenarios and a comparison is made of these PRGs to the IAROD cleanup levels. This section provides the basis for determining the levels of contaminants that may remain onsite and still be adequately protective of human health. In addition, the PRGs may be used in comparing potential health impacts of various remedial alternatives.

Several technical appendices are provided in electronic format on a data CD that accompanies this report. The technical appendices are organized as follows.

A. Workshop Notes and Draft A Comment/Response Forms
B. Background Concentrations and Reference Sites
C. Data Analysis
D. Supporting Information for the Human Health Risk Assessment
E. Risk Assessment for the 100 Area River Effluent Pipelines.

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2.0 SITE BACKGROUND AND CLEANUP ACTIVITIES

This section provides a summary of the physical and ecological setting for the River Corridor of the Hanford Site. Consideration of the physical and environmental attributes of the site is an important element of a baseline risk assessment. Characteristics of the environment and natural resources present at the site may influence the nature and extent of potential contaminants.

Historical use of the site can impact the natural resources and release hazardous substances that may pose a threat to human health and the environment. This section provides an overview of early uses of the River Corridor prior to the Manhattan Project and government use. Hanford operations that have occurred in the River Corridor are discussed and the process for characterizing and cleaning up contaminated areas is described.

Finally, this section provides a summary of the current conditions[1] for the River Corridor. Information is provided for operational areas where releases of hazardous substances posing a threat to human health and the environment are known to have occurred and nonoperational areas where historic information and other studies indicate there were no direct releases of hazardous substances posing a threat to human health or the environment.

2.1 PHYSICAL SETTING

2.1.1 Climate

The Hanford Site is located within the semiarid shrub-steppe Pasco Basin of the Columbia Plateau in south-central Washington State. Average annual precipitation on the Hanford Site is 17 cm (6.8 in.). The climate is characterized by warm, dry, sunny summers and cool winters. Daytime temperatures in mid-summer can exceed 38 °C (100 °F) and winter temperatures can drop to below -18 °C (0 °F) (DOE/RL-2001-54, *Ecological Evaluation of the Hanford 200 Area – Phase 1: Compilation of Existing 200 Areas Ecological Data*). Precipitation that falls on the Hanford Site is generally lost through evapotranspiration and typically does not migrate through the soil column.

In some cases, natural precipitation infiltrates into the soil and may recharge groundwater flow systems. Recharge from natural precipitation is believed to be the most significant in the higher elevations of the site but has also been observed on a small scale in localized areas of disturbance that occur in and around waste disposal areas (PNL-10285, *Estimated Recharge Rates at the*

[1] For the purposes of this report, current conditions in the River Corridor are represented by the conditions present when the project work plan (DOE/RL-2004-37, *Risk Assessment Work Plan for the 100 Area and 300 Area Component of the RCBRA*) was written. The work plan and ensuing planning documents defined the scope of the project based on the extent of remediation and characterization that was complete at the time. This corresponds to roughly the end of 2005.

Hanford Site). Moisture movement through the vadose zone is important at the Hanford Site because it is a driving force for migration of most contaminants to the groundwater.

2.1.2 Geology

Hanford Site soils include strata consisting of unconsolidated sands and gravels, coarse-grained sand and gravel deposited by the Columbia River, and some fine-grained deposits. The *Soil Survey: Hanford Project in Benton County Washington* (BNWL-243) describes as many as 15 different surface soil types on the Hanford Site, including sand, sandy loam, and silt. The soil column, from the soil surface to the top of the groundwater (also called the vadose zone), underlying the River Corridor largely consists of materials belonging to the Hanford and Ringold Formations. The shallower Hanford formation consists predominantly of medium-to-dense sand and gravel, with varying amounts of silt and cobble. The underlying Ringold Formation consists primarily of dense, well-cemented gravels with sand and silt interbeds.

2.1.3 Groundwater

The hydrogeologic characteristics of the River Corridor are affected by proximity to the Columbia River. The unsaturated vadose zone ranges in depth from near 0 m (0 ft) at the edge of the Columbia River to more than 80 m (280 ft) at the interior of the Central Plateau. Groundwater beneath the Hanford Site is found in both an upper unconfined aquifer system and in deeper, basalt-confined aquifers. Portions of the upper suprabasalt aquifer system are locally confined, but because the entire suprabasalt aquifer system is interconnected on a site-wide scale, it is referred to as the Hanford unconfined aquifer system. The deeper, basalt-confined aquifer system is important because there is a potential for significant groundwater movement and, consequently, contamination movement between the two systems. Water can actually flow from the river into the aquifer at high river stage and then return to the river at low river stage, a phenomenon known as “bank storage.”

Groundwater beneath the Hanford Site originated as either natural recharge from rain and snowmelt or as artificial recharge during Hanford Site operations. Effective precipitation that can contribute to natural recharge occurs during the cold season and increases with elevation. Historically, the volume of artificial recharge from Hanford Site operations and wastewater disposal was significantly greater than the natural recharge from precipitation. There is no longer significant artificial recharge due to operations in the 100 and 300 Areas, as disposal of liquid wastes to ground has ceased. However, some localized artificial recharge may occur as a result of water line leakage, reservoirs, and application of water for dust suppression. Due to the reduction in discharges since 1984, groundwater levels are falling, particularly around the Hanford Site operational areas (PNNL-13080, *Hanford Site Groundwater Monitoring: Setting, Sources, and Methods*).

Groundwater in the upper, unconfined aquifer generally flows from west to east across the Hanford Site to discharge areas north and east along the Columbia River (PNNL-16346, *Summary of Hanford Site Groundwater Monitoring for Fiscal Year 200*). Total groundwater discharge from the unconfined aquifer on the Hanford Site to the Columbia River is estimated to range from 1.1 to 2.5 m^3/s (38.85 to 88.29 ft^3/s), with the average flow in the river being approximately 3,400 m^3/s (120,070 ft^3/s) (PNNL-16346). Although local groundwater flow in the River Corridor area is toward the Columbia River, the "bank storage" phenomenon described above may cause periodic reversals at locations adjacent to the river. The presence of pump-and-treat remediation systems in the 100-K, 100-D, and 100-H Areas may also affect local groundwater flow patterns (PNNL-16346).

2.1.4 Columbia River

The Columbia River is the second largest river in the contiguous United States in terms of total flow and is the dominant surface water body on the Hanford Site. The original selection of the Hanford Site for plutonium production and processing was based, in part, on the abundant clean, cold water provided by the Columbia River. Originating in the Canadian Rockies of southeastern British Columbia, Canada, the Columbia River drains a total area of approximately 680,000 km^2 (262,480 mi^2) en route to the Pacific Ocean. Most of the Columbia River is impounded by 11 dams within the United States, 7 upstream and 4 downstream of the Hanford Site. Priest Rapids is the nearest upstream dam and McNary is the nearest downstream dam. Lake Wallula, the impoundment created by McNary Dam, extends upstream past Richland, Washington, to the southern part of the Hanford Site. Except for the Columbia River estuary, the only unimpounded stretch of the river in the United States is the Hanford Reach, which extends from Priest Rapids Dam downstream approximately 82 km (51 mi) to the McNary Pool north of Richland, Washington. The existence of the Hanford Site has precluded development of this section of the river. The Hanford Reach of the Columbia River was recently incorporated into the land area established as the Hanford Reach National Monument (65 FR 37253, "Establishment of the Hanford Reach National Monument").

Flows through the Hanford Reach fluctuate significantly and are controlled primarily by releases from three upstream storage dams: the Grand Coulee Dam in Washington State and the Mica and Keenleyside Dams in British Columbia, Canada. Storage dams on tributaries of the Columbia River also affect flows. Flows in the Hanford Reach are directly affected by releases from Priest Rapids Dam. However, Priest Rapids operates as a run-of-the-river dam rather than a storage dam; its flows are controlled for power generation and to promote salmon migration. The Vernita Bar Agreement (signed June 16, 1988, by DOE, federal and state agencies, tribal governments, and public utility districts in Grant, Chelan, and Douglas Counties) was created to help prevent low-river flow periods from endangering Hanford Reach salmon spawning area deposits. Columbia River flow rates near Priest Rapids during the 83-year period from 1917 to 2000 averaged nearly 3,360 m^3/s (120,000 ft^3/s). Daily average flows during this period ranged from 570 to 19,500 m^3/s (20,000 to 690,000 ft^3/s). The lowest and highest flows occurred before the construction of upstream dams. During the 10-year period from 1991 through 2000, the average flow rate was also about 3,360 m^3/s (120,000 ft^3/s).

Columbia River flows typically peak from April through June during the spring snowmelt runoff and are lowest from September through October. As a result of daily fluctuations in discharges from Priest Rapids Dam, the depth of the river varies significantly over a short time period.

River stage changes of up to 3 m (10 ft) during a 24-hour period may occur along the Hanford Reach (PNNL-13230, *Hanford Site Environmental Report for Calendar Year 1999*). The width of the river along the Hanford Reach varies from approximately 300 m (1,000 ft) to 1,000 m (3,300 ft). The width also varies temporally as the flow rate changes, which causes repeated wetting and drying of an area along the shoreline.

2.2 ECOLOGICAL SETTING

A number of studies provide basic environmental information about the Hanford Site and specifically the 100 Area and 300 Area. The annual *Hanford Site National Environmental Policy Act (NEPA) Characterization Report* (PNNL-6415) provides a detailed summary of the ecology, biological resources, and hydrology for the entire Hanford Site, with selected information grouped by major operational areas. The document is updated annually and has been used extensively in the preparation of this ecological summary. The *Hanford Reach of the Columbia River: Comprehensive River Conservation Study and Environmental Impact Statement* (DOI 1996) also provides information on the riparian and aquatic environments found within the Hanford Reach. Other detailed characterization data for the 100 Area and 300 Area, including comprehensive lists of plant and wildlife species occurring in or near the study area, are presented and discussed in *Literature Review of Environmental Documents in Support of the 100 Area and 300 Area River Corridor Baseline Risk Assessment* (PNNL-SA-41467).

Natural habitat in the River Corridor can be divided into three general ecological zones. These three key ecological zones include the upland, riparian, and aquatic zones. This delineation is used to help to develop the conceptual site model which is used to determine exposure pathways and contaminants of potential concern.

2.2.1 Upland Zone

The upland environment of the River Corridor is the largest zone and consists of land that is adjacent to the main channel of the Columbia River, above the river high-water mark, and extends inland from the Columbia River. The upland environment is generally arid; it is not influenced by groundwater and river flow and depends on precipitation for its water supply. The animals and plants common to the upland environment are adapted to survival in arid conditions. Figure 2-1 is a photograph of an upland area in the 100-D Area with the Columbia River in the distance.

Historically, much of the upland habitat in the River Corridor was likely a community dominated by big sagebrush (*Artemisia tridentata*), with lesser amounts of rabbitbrush (*Chrysothamnus nauseosus*) and an understory of Sandberg's bluegrass (*Poa sandbergii*). During the Euro-American settlement of the area, a large portion of the areas near the Columbia River were disturbed by farming. Construction activities for the Manhattan Project further disturbed the vegetation and soils in the area. These two major impacts to the land resulted in changes to the native plant community, creating highly disturbed areas and other areas that have partially

Figure 2-1. Upland Ecological Zone Near the 100-D Area.

recovered and revegetated. In general, the affected areas do not recover to pre-disturbance conditions, due in part to the presence of invasive non-native species such as cheatgrass (*Bromus tectorum*).

Vegetation that occurs in highly disturbed areas, such as near old operating plants and waste sites, is typically sparse and consists of early successional species such as cheatgrass (*Bromus tectorum*), Russian thistle (*Salsola kali*), tumblemustard (*Sisymbrium altissimum*), and bur ragweed (*Ambrosia acanthicarpa*). Upland plant communities have been altered by the proliferation of non-native plant species, such as cheatgrass, Russian thistle, knapweed (*Centaurea* spp.), and yellow star thistle (*Centaurea solstitialis*). These invasive plants compete with native plants, affect the animals that inhabit an area, and can increase the magnitude of wildfires.

Some areas that are no longer disturbed have begun to revegetate naturally to communities dominated by gray rabbitbrush with an understory of Sandberg's bluegrass, bulbous bluegrass (*Poa bulbosa*), and cheatgrass. Sagebrush is present but infrequent. Each of these species is well adapted to the rocky soils characteristic of disturbed portions of the upland zone.

Large areas of cheatgrass and exotic annual species are present in the abandoned "old fields" associated the White Bluffs and Hanford townsite areas (PNNL-SA-41467). Upland vegetation at the Hanford townsite also differs from the other areas due to the presence of trees scattered along the remains of roadways and walkways associated with previous homesteads, the town itself, and the Hanford Construction Camp. The trees present along old streets of the town/construction camp provide important habitat for a number of mammals and birds. More detailed descriptions of these vegetation cover types are given in *Vascular Plants of the Hanford Site* (PNNL-13688).

The dominant ground-dwelling invertebrate species in the upland environment are harvester ants (*Pogonomyrmex owyheei*) and darkling beetles (family Tenebrionidae). Harvester ants can exist on vegetated and nonvegetated soils (PNL-2774, *Characterization of the Hanford 300 Area Burial Grounds, Task IV – Biological Transport*).

Soil type and depth, and vegetation types, densities, and stature influence the animal species that use the upland areas. The soils in the industrialized 100 Areas and 300 Area have been disturbed. This soil matrix limits the diversity of small mammals to species that live on the surface or in very shallow burrows. The dominant small-mammal species associated with remediated sites is the deer mouse (*Peromyscus maniculatus*). Other species that may be present, but likely in very small numbers, are the house mouse (*Mus musculus*) and the harvest mouse (*Reithrodontomys megalotis*). Burrowing species such as the Great Basin pocket mouse and the pocket gopher are limited to areas where fine-grained soils are at least 30 cm (12 in.) deep (PNL-4140, *Habitat Requirements and Burrowing Depths of Rodents in Relations to Shallow Waste Burial* Sites; RHO-SA-211, *Invasion of Radioactive Waste Burial Sites by the Great Basin Pocket Mouse [Perognathus parvus]*).

Mammals of the upland environment include the mule deer (*Odocoileus hemionus*), badger (*Taxidea taxus*), coyote (*Canis latrans*), Great Basin pocket mouse (*Perognathus parvus*), northern pocket gopher (*Thomomys talpoides*), black-tailed jackrabbit (*Lepus californicus*), and cottontail rabbit (*Sylvilagus nuttalii*) (WHC-EP-0620, *Areas CERCLA Ecological Investigations*). The abundance of these species and the occurrence of others vary according to the soil type and vegetative community. Large mammals, such as elk (*Cervus elaphus*), are also observed in the River Corridor upland areas. A complete list of mammals observed and expected in all habitats of the 100 Areas is provided in *100 Areas CERCLA Ecological Investigations* (WHC-EP-0620). PNNL-6415 presents a complete listing of Hanford Site wildlife species.

Several species of birds present in the upland zone rely on structures such as buildings, fences, and utility poles for some of their habitat needs. Raptors, such as the red-tailed hawk (*Buteo jamaicensis*), are present, and frequently nest on buildings, utility poles and towers, and trees along the river. Nonvegetated areas provide nesting habitat for nighthawks (*Chordeiles minor*) and killdeer (*Charadrius vociferus*). Canada geese (*Branta canadensis*) use open cheatgrass areas for winter grazing. Native shrub-steppe bird species include the horned lark (*Eremophila alpestris*), western meadowlark (*Sturnella neglecta*), savannah sparrow (*Passerculus sandwichensis*), loggerhead shrike (*Lanius ludovicianus*), and sage sparrow (*Amphispiza belli*). Raptors will continue to be present, but as the shrubs develop and the open grassy areas shrink in size, wintering geese will likely avoid the area, preferring the cheatgrass areas associated with nearby abandoned farm fields and orchards. A list of bird species observed in the 100 Areas is available in WHC-EP-0620. A catalogue of Hanford Site avian species is presented in PNL-6415, *Hanford Site National Environmental Policy Act (NEPA) Characterization*. Common reptiles found in upland environments at the Hanford Site include the rattlesnake (*Crotalus viridis*), gopher snake (*Pituophis melanoleucus*), yellow-bellied racer (*Coluber constrictor*), and side blotch lizard (*Uta stansburiana*) (PNL-8942, *Habitat Types of the Hanford Site: Wildlife and Plant Species of Concern*; WHC-EP-0601, *A Syntheses of Ecological Data from the 100 Areas of the Hanford Site*).

2.2.2 Riparian Zone

Riparian zones are areas of transition between aquatic and upland ecosystems. The riparian environment of the River Corridor extends from the point on the riverbank where upland vegetation is no longer dominant down to the shoreline of the Columbia River. The riparian zone is typically narrow, its width depending on the slope of the riverbank. The transition from the upland zone vegetation to riparian vegetation is generally abrupt. The vegetation that grows in the riparian zone along the river shoreline is thicker and taller than that in the upland area, attracting a broader range of wildlife species. The riparian zone is influenced by the emergence of groundwater originating from the upland areas and by fluctuations in the level of the Columbia River. The plants within this zone benefit from access to groundwater and water from the Columbia River via their root systems or periodic high-water conditions. The small mammals, birds, and reptiles common to the upland environment are also likely to inhabit the riparian environment. Figure 2-2 is a photograph of a riparian area near the 100-H Area.

The lateral extent of the shoreline vegetation varies along the Hanford Reach. The extent of physical disturbance from early settlement and Manhattan Project use is localized and has had relatively little impact on the riparian zone. The original native vegetation was likely dominated by grasses and sedges (*Carex* spp.), with a sparse distribution of willows (*Salix* spp.). Historically, there were very few trees along the Hanford Reach of the Columbia River before the construction of Priest Rapids Dam.

The riparian zone generally consists of a cobble shoreline with varying densities of vegetation. Dominant vegetation within the riparian zone includes mulberry (*Morus alba*), willow (*Salix* spp.), Siberian elm (*Ulmus pumila*), northern wormwood (*Artemisia campestris*), sweet clover (*Melilotus alba* or *M. officinalis*), and reed canarygrass (*Phalaris arundinacea*). WHC-EP-0620 lists plant and animal species that have been observed along the Columbia River shoreline.

Changes to the composition of shoreline vegetation over time have been influenced by a moderation in the river elevation changes, which are controlled by the operation of Priest Rapids Dam, approximately 18.5 km (10 mi) upstream of the Hanford Site. Because of the steepness of the shoreline, the transition from riparian to upland vegetation is abrupt.

Figure 2-2. Riparian Ecological Zone Near the 100-H Area.

Dominant plants in the riparian-upland transition area are bulbous bluegrass, Sandberg's bluegrass, cheatgrass, Russian thistle, and gray rabbitbrush. Detailed characterization information of the riparian environs associated with each reactor area is presented in PNNL-SA-41467.

There is considerable overlap of wildlife use between the riparian and upland zones. Some mammals common to the upland environment are also likely to use and inhabit the riparian environment, including the western harvest mouse, the Great Basin pocket mouse, and the deer mouse (PNNL-14516, *Synthesis of Ecological Data Collected in the Riparian and Riverine Environments of the Hanford Reach*). Wildlife use of the riparian zone is likely higher than that of the upland zone associated with the CERCLA waste sites due to its proximity to the Columbia River, which results in greater species diversity and the presence of higher density and higher stature vegetation that remains productive over a longer period of time.

A variety of snakes common to the upland areas may also use the riparian habitat. Other reptiles that may be found in the riparian zone include the western terrestrial garter snake (*Thamnophis sirtalis*) and the painted turtle (*Chrysemys picta*) (Hallock 1998, *Herpetofauna of the Hanford Nuclear Reservation, Grant, Franklin, and Benton Counties, Washington*; PNNL-14516).

Amphibians in the riparian and near-shore environments of the Hanford Reach include mostly Woodhouse's toads (*Bufo woodhousii*), but bullfrogs (*Rana catesbeiana*) and Great Basin spadefoot toads (*Scaphiopus intermontanus*) may also be present (PNNL-14516).

PNNL-14516 provides information on bird populations with respect to riparian vegetation. Location data are available in the electronic Environmental Monitoring and Compliance Project database managed by the Pacific Northwest National Laboratory. Research efforts have assessed winter bird populations in cottonwood/willow (*Populus/Salix*) communities of the Columbia River shoreline (Rickard 1964, "Bird Surveys in Cottonwood-Willow Communities in Winter"; Rickard and Rickard 1972, "A Comparison of Winter Bird Populations after a Decade"), quantified shorebird response to water fluctuations in the Columbia River near-shore environment (Books 1985, "Avian Interaction with Mid-Columbia River Water Level Fluctuations"), and evaluated habitat selection and use by spring migrant passerines (Duberstein 1997, "Riparian Stopover Habitat Selection by Spring Transient Landbirds of South-Central Washington"). The information gathered during these research efforts has been used to document the status and ecology of the Hanford Site's avian wildlife.

2.2.3 Near-Shore Aquatic Zone

The near-shore river zone consists of a narrow band of the Columbia River shoreline and aquatic environment adjacent to the riparian zone. The near-shore river zone includes the surface water of the Columbia River from the area that is permanently inundated by river water, extending from the low-water mark (i.e., a "green line" where the periphyton [sessile algae] remains green year round) into the river to a water depth of approximately 2 m (6 ft).

The near-shore aquatic zone consists of a narrow band of the Columbia River adjacent to the shoreline. This zone was selected to optimize the ability to measure the potential influence of emergent groundwater plumes and other Hanford contaminant sources within the Columbia River. The aquatic vegetation found in the near-shore zone supports aquatic insect populations, benthic organisms (organisms that live in or on the bottom of the river), birds, and fish. Many species of fish live in the Columbia River adjacent to the Hanford Site, and some use the river as a migration route to and from upstream spawning areas. Other species, such as sculpin, spend their entire lives in small sections of the river. The shoreline areas provide rearing habitat for many fish species, including spawning habitat for threatened and endangered fish species. This portion of the river environment immediately adjacent to the Hanford Site is considered to be habitat that could be impacted by potential releases. Figure 2-3 is a photograph of a near-shore aquatic area along the Hanford Site.

Near-shore river zone features include sloughs, rapids, and shoreline areas with varying water velocities. Sloughs are present in or near the Hanford townsite and White Bluffs shorelines and serve as important habitat for a number of aquatic species. Vegetation in the near-shore river zone consists of macrophytes and periphyton. Macrophytes are sparse in the Columbia River because of strong currents, rocky bottom, and frequently fluctuating water levels. Where macrophytes are found, they commonly include duckweed (*Lemna* spp.) and the native rooted

Figure 2-3. Near-Shore Aquatic Ecological Zone Along the Hanford Site.

pondweeds *(Potamogeton* spp. and *Elodea canadensis*). Macrophytes provide food and shelter for juvenile fish and spawning areas for some species of warm-water game fish. Eurasian milfoil (*Myriophyllum spicatum*), an introduced macrophyte, has increased to nuisance levels since the late 1980s and may encourage increased sedimentation of fine particulate matter. Periphyton communities develop on suitable solid substrate wherever there is sufficient light for photosynthesis and adequate currents to prevent sediment from covering the colonies.

More than 45 species of fish have been identified in the Hanford Reach of the Columbia River. Of these species, Chinook salmon (*Oncorhynchus tshawytscha)*, sockeye salmon (*Oncorhynchus nerka)*, coho salmon (*Oncorhynchus kisutch)*, and steelhead (*Oncorhynchus mykiss)* use the river as a migration route to and from upstream spawning areas and are of the greatest economic importance. 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)*, black crappie (*Pomoxis nigromaculatus*), channel catfish (*Ictalurus punctatus)*, and walleye (*Stizostedion vitreum)* are also popular. Other large fish populations include introduced common carp (*Cyprinus carpio)* and native species such as redside shiner (*Richardsonius balteatus*) and largescale suckers (*Catostomus macrocheilus)*. Smaller fish, such as sculpin (*Cottus* spp.), are associated with shoreline habitats and have small home ranges.

2.3 PRE-HANFORD HISTORY

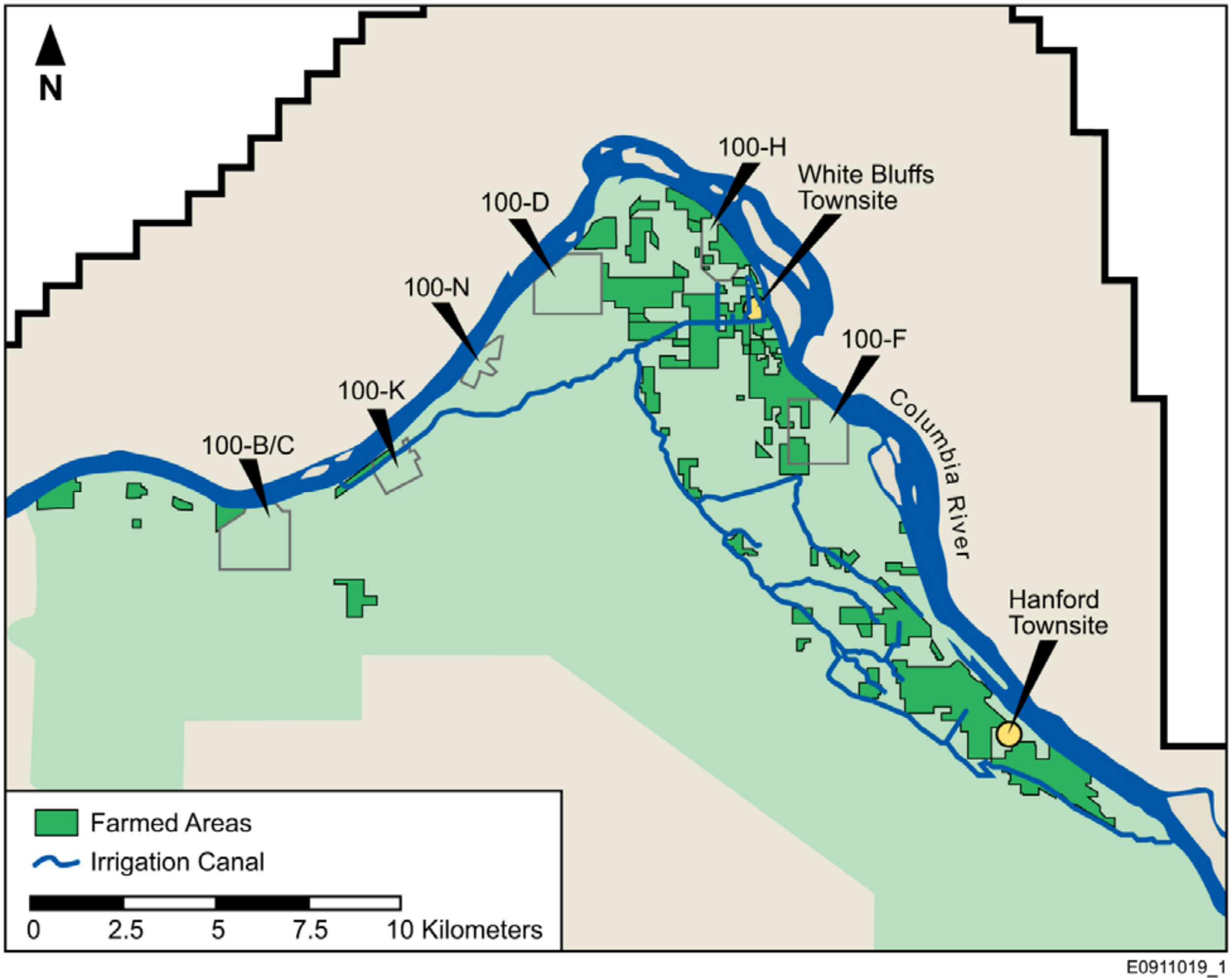
The Hanford Site is culturally rich, experiencing a history of multiple occupations by both Native and non-Native Americans. For thousands of years Native American peoples have inhabited the lands both within and around the Hanford Site (Relander 1956, *Drummers and Dreamers*; Spier 1936, "Tribal Distribution in Washington"; Walker 1998, *Handbook of North American Indians*). The Hanford Reach was a seasonal home to a large group of Native Americans prior to the arrival of Euro-Americans in the early 1800s. These groups included the Columbia, Nespelem, Sanpoil, Southern Okanogan, Umatilla, Wallula, Wanapum, Wauykma, and Yakama. Nearby groups such as the Cayuse, Chelan, Colville, Kittitas, Methow, Nez Perce, Palus, Spokane, Wayampum, Wenatchi, and Wishram also occasionally used the area to trade, gather resources, and conduct other activities (Andrefsky et al. 1996, *1995 WSU Archaeological Block Survey of the Hanford 600 Area*). Many descendants of these indigenous peoples retain traditional, cultural, and religious ties to the Hanford Site, with the Nez Perce, Umatilla, and Yakama holding treaty rights to the area. Native plant and animal foods, some of which can be found on the Hanford Site, are used in ceremonies performed by tribal members. Prominent landforms such as Rattlesnake Mountain, Gable Mountain, and Gable Butte, as well as various sites along and including the Columbia River, remain sacred to these peoples.

Non-Native American presence in the mid-Columbia began in 1805 with the arrival of the Lewis and Clark Expedition along the Columbia and Snake Rivers. Other visitors included fur trappers, military units, explorers, and miners who traveled through the Hanford Site on their way to lands up and down the Columbia River and across the Columbia Basin. In the late 19th and early 20th centuries, non-native people began intensive settlement on the Hanford Site, establishing an early settler and farming landscape.

Farmstead communities existed from 1880 to 1943, and their locations within the River Corridor are known from historic and current aerial photographs (1941, 1948, and 2002), real estate records, historic documents, personal interviews, and field walkdowns. The locations of pre-Manhattan Project farmsteads in the River Corridor are presented in Figure 2-4. As can be seen in the figure, the farmsteads are located primarily in the upland environment adjacent to the Columbia River and within the distribution pattern of early irrigation systems.

From 1880 to 1905 self-subsistence farming on small farms was the primary pursuit. The typical layout included a farm house, a well, an outhouse, outbuildings (barn, shop, storage), a vegetable garden, a root cellar, and areas for dumping refuse. Landscaping, particularly shade trees and windbreaks, also was common. Early farms along the Columbia and Yakima Rivers grew alfalfa and rye grasses. These farmers also experimented with vegetables, melons, berries, and fruit trees. Early irrigation systems included water elevators, which consisted of an endless chain of buckets powered by a horse. Other systems included windmills, waterwheels, and, particularly along the Yakima River, makeshift dams that diverted water from nearby creeks into wooden ditches or flumes and into nearby fields.

Figure 2-4. Location of Pre-Manhattan Project Farmsteads in the River Corridor.



In December 1905, the Hanford Irrigation and Development Company was organized in Seattle with the purpose of reclaiming 12,950 ha (32,000 ac) of arid land along the Columbia River near the White Bluffs. By 1909, the 29-km (18-mi)-long Hanford Irrigation Canal was carrying water from the Allard Pump House upstream near Coyote Rapids (near 100-K Area) on the Columbia River to the communities of White Bluffs and Hanford (see Figure 2-4). The Priest Rapids Valley soon became one of the premier orchard regions in the state. Farms were primarily family-operated and ranged in size from under 2 ha (5 ac) to over 16 ha (40 ac), averaging about 8 ha (20 ac). Hanford and White Bluffs farmers made large investments in their land, constructing irrigation systems and planting a variety of crops including apples, apricots, cherries, grapes, melons, peaches, pears, plums, strawberries, hops, alfalfa, asparagus, corn, and potatoes. Many farms had as many as eight different fields dedicated to different crops. Others, primarily orchardists, focused on a single crop. In 1913, settlement and agricultural development in the valley was bolstered by the construction of the Chicago, Milwaukee, St. Paul, and Pacific Railroad, which enabled the farmers to move from local to national-based markets by providing a means to ship fruit and other produce to either regional or more distant locations via transcontinental railroad.

In concert with the farms operating off the Hanford Irrigation Canal, a different type of farm appeared following World War I. "Soldier settlements" were offered to veterans for a small down payment and low interest rates. These 8-ha (20-ac) farms came complete with a house, outhouse, barn, chicken coop, and well. Wells were provided because they were considered more economical than paying for water from the local irrigation district.

Pest management practices designed to reduce crop loss were in place during this time. Spraying was undertaken in the orchards for codling moth and scale. Orchard sprays contained about 0.45 kg (1 lb) of arsenate or lead to 189 L (50 gal) of water.

Spraying 2.7 kg (6 lb) of lead arsenate paste or 1.4 kg (3 lb) of powder to 757 L (200 gal) of water in a mist often controlled the second brood of Codling moths. Codling moth control finally reached the point where it became too expensive for many of the producers and its discontinuance resulted in a decline in production in the early 1930s. Many of the apple orchards were replaced with soft fruit such as grapes or apricots (BHI-01326, *Pre-Hanford Agricultural History: 1900-1943*).

Scale, another orchard pest, was controlled with the use of lime sulphur (a mixture of calcium polysulfides). Red spiders were controlled with the use of lime sulfur, atomic sulphur, or flours of sulphur. The sprays were applied during the cool part of the day (BHI-01326).
Several other inorganic compounds have historically been used as insecticides including mercury, boron, thallium, arsenic, antimony, selenium, and fluoride. Arsenic-based insecticides included copper arsenate (Paris Green), lead arsenate, and calcium arsenate.

The small family-owned farms that dominated the economy of Hanford and White Bluffs struggled during the Great Depression. However, the farming life in Hanford and White Bluffs came to an abrupt halt in 1943 when the U.S. government took possession of the land and removed the people from their homes.

Recent archaeological work has been conducted on the farmsteads providing additional physical information. Based on information collected to date, the farmstead remains include small quantities of CERCLA hazardous materials (e.g., residual petroleum products, empty pesticide and paint cans) intermixed with larger amounts of debris (wood, fencing, concrete, glass, etc). The remains appear to present little risk to human health or the environment in their present configuration. Plant communities in the farmstead areas have re-established and habitat areas have naturally developed.

Most of the waste materials associated with the pre-Manhattan Project farmsteads are associated with individual farm refuse dumps and community disposal areas. Individual farm disposal areas were very common because no garbage service was available during that period. Often, refuse was disposed in unoccupied locations in the vicinity of the farm. Many of these locations contain a very limited number of items that could be classified as CERCLA waste materials, primarily lead-acid batteries and empty containers that once held oils, herbicides, pesticides, or paints.

2.4 HANFORD OPERATIONS IN THE RIVER CORRIDOR

In 1943, the Hanford Site became a federal facility when the U.S. Government took possession of the land to produce weapons-grade plutonium as a part of the Manhattan Project during World War II. Between 1943 and 1963, nine plutonium-production reactors were built along the Columbia River in six areas: the 100-B, 100-K, 100-N, 100-D, 100-H, and 100-F Areas. Together, these six areas comprise about 68 km^2 (26 mi^2) of the land adjacent to the Columbia River in the northern part of the Hanford Site. Large construction camps were established near the Hanford townsite in the central area of the River Corridor to support construction of the reactors. In the southern area of the River Corridor, the 300 Area was developed to support fuels fabrication and research and development activities. Figure 2-5 shows the geographical location and extent of each of the operating areas in the River Corridor.

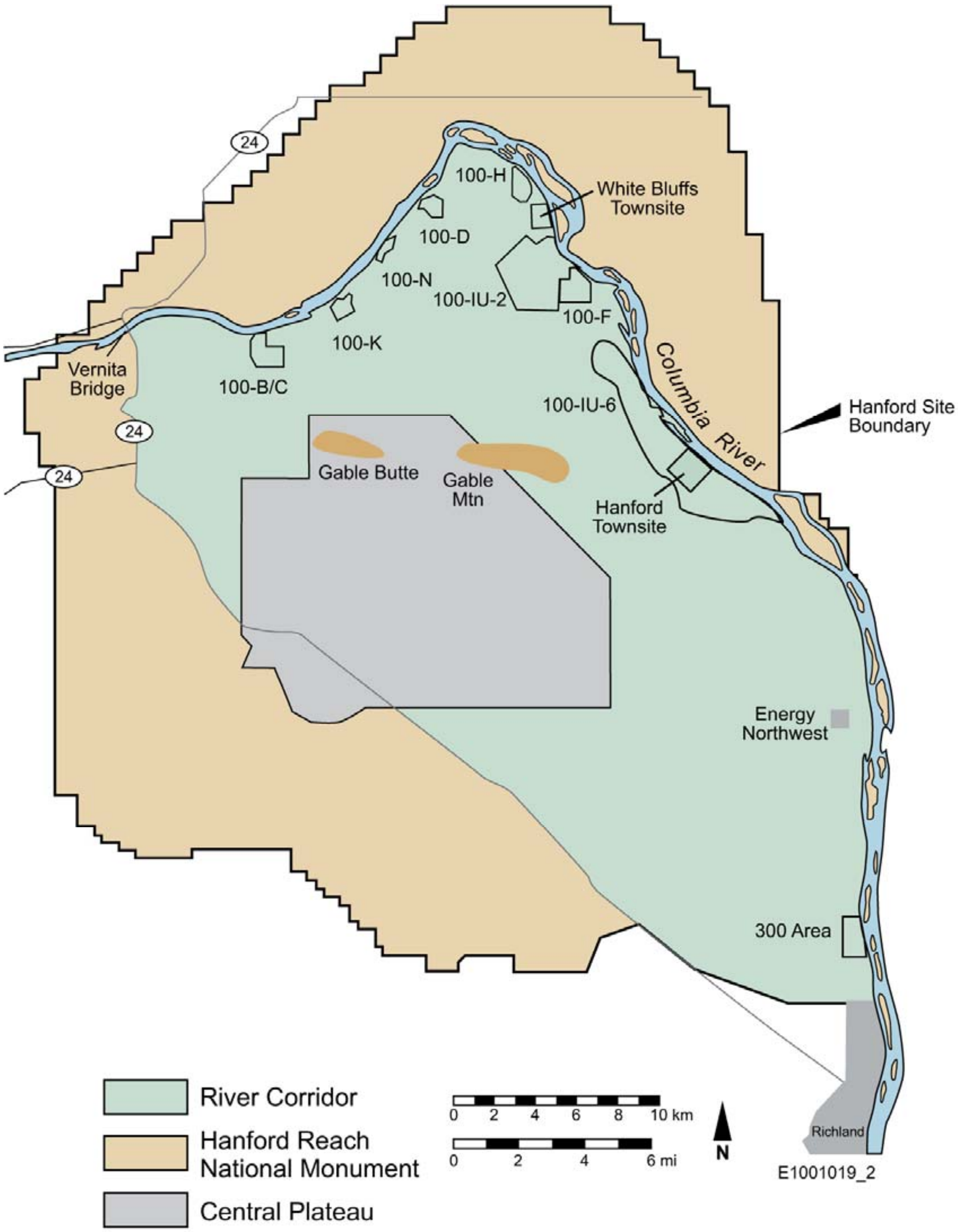
2.4.1 100 Area Reactor Operations

The main component of each reactor was a large stack of graphite blocks (called a pile) with steel process tubes containing the fuel elements and cooling water. Placing large numbers of uranium fuel elements into the reactor piles created an intense radiation field and a nuclear chain reaction that converted some uranium atoms to plutonium atoms. Along with uranium, other atoms in the pile structure, including nonradioactive constituents, were converted into radioactive fission products and activation products that were disposed of as waste. Each reactor area contained numerous support facilities, such as powerhouses, water treatment and pumping facilities, laboratories, railroad offloading facilities, office buildings, septic systems, and waste disposal facilities for reactor effluent that commonly became contaminated during plutonium production operations. The types of waste sites and expected contaminants in the reactor areas include disposal of liquid and solid radioactive waste and industrial chemicals. Nonradioactive wastes were also disposed in the 100 Areas. Reactor area-specific information is summarized in the following subsections.

2.4.1.1 100-B/C Area. The 100-B/C Area includes two water-cooled, graphite-moderated plutonium reactors, 105-B and 105-C, with their associated ancillary facilities for water treatment, air filtration, nuclear fuel handling, effluent disposal, as well as laboratories, administrative offices, and various other buildings.

Groundbreaking for the 105-B Reactor began in October 1943 by the U.S. Army Corps of Engineers as a part of the Manhattan Project. The reactor was fully constructed and operational by February 1945. The 105-B Facility was the world's first full-scale production reactor. The facility was operated until 1968.

Figure 2-5. Geographical Scope of the River Corridor and Location of the Operating Areas.



The design of the 105-C Reactor was started in March 1951, and construction was initiated on June 6, 1951. Initial startup of the 105-C Reactor was achieved in November 1952. The design of the facility was based on the earlier Hanford Site reactors. The 105-C Reactor was shut down in April 1969. Deactivation of the reactor was completed in early 1971.

The 105-B Reactor has been listed on the National Register of Historic Places and as a National Mechanical Engineering Landmark. Currently, the 105-B Reactor is planned to be preserved as a museum. A portion of the 100-B/C Area water treatment plant continues to operate to provide potable water to the 200 Area, and the 151-B Electrical Substation continues to provide electrical power.

2.4.1.2 100-K Area. The 100-K Area includes two water-cooled, graphite-moderated plutonium reactors, 105-KE and 105-KW, and their associated ancillary facilities. Construction on the 105-KE and 105-KW Reactors was initiated in 1948, with startup of the 105-KE Reactor on April 17, 1955, and the 105-KW Reactor on January 4, 1955. Both reactor complexes were permanently shut down by 1971; however, spent fuel and radioactive sludge remained in the fuel storage basins. Following the shutdown, leaks were discovered in these fuel basins, and activities were established to expeditiously remove spent fuel and sludge from the basins. In addition to this fuel basin leakage, various other operational processes, waste disposal practices, and unplanned releases generated many solid and liquid waste sites. As with other reactor sites, sodium dichromate, a chemical that was used in the cooling water treatment process, exists in the vadose zone and groundwater in the 100-K Area.

2.4.1.3 100-N Area. The 100-N Area contains the 105-N Reactor, which was designed to produce plutonium and use by-product steam to produce electric power. Rather than using water from the Columbia River as a once-through coolant, the 105-N Reactor design recycled the cooling water for the reactor core.

The reactor core is a graphite-moderated, light water-cooled, horizontal pressure-tube facility designed to produce plutonium. By-product steam was routed to a nearby privately operated facility (185-N Hanford Generating Plant) to produce approximately 860 megawatts of electricity. Construction of N Reactor began in December 1959 and was completed in October 1963. N Reactor was the last of the Hanford Site graphite-moderated reactors. On the south side of the building is the 109-N Heat Exchanger Building, which shares a common wall with 105-N. The reactor ceased operations in 1987 and preservation efforts were stopped in 1991. Facility deactivation was completed in July 1998, which placed the facilities in a safe and stable condition.

2.4.1.4 100-D Area. The 100-D Area contains two water-cooled, graphite-moderated plutonium reactors, 105-D and 105-DR, and their associated ancillary facilities. The 105-D Reactor was one of the original three Hanford Site reactors. Construction of the 105-D Reactor began in November 1943 and startup occurred in December 1944. During early operations, the reactor was thought to be failing because of the uncontrollable expansion of the graphite core. Consequently, the 105-DR Reactor was built as a replacement, with construction beginning in December 1947 and startup occurring in October 1950. Operational processes, waste disposal practices, and unplanned releases generated many solid and liquid waste sites. One of the most enduring and problematic waste issues has been the sodium dichromate contamination of the vadose zone and groundwater, caused by the cooling water treatment process. The 105-D Reactor shut down in June 1967. The 105-DR Reactor was shut down in December 1964.

2.4.1.5 100-H Area. The 100-H Area contains a single water-cooled, graphite-moderated plutonium reactor, 105-H and associated ancillary facilities. The 105-H Reactor was the first post-Manhattan Project reactor complex to begin operations on the Hanford Site. Construction of the 105-H Reactor was initiated in 1948; initial startup occurred in October 1949. The 105-H Reactor facility, including the fuel storage basin, was placed in final shutdown mode between 1965 and 1970. During shutdown activities, leaks in the basin were identified. In addition, various other operational processes, waste disposal practices, and unplanned releases generated many solid and liquid waste sites. Sodium dichromate, a chemical that was used in the cooling water treatment process, has been one of the most problematic waste issues encountered in the vadose zone and groundwater.

2.4.1.6 100-F Area. The 100-F Area contains the 105-F Reactor and associated ancillary facilities. Construction of the 105-F Reactor began in 1943 and was completed in February 1945. The 105-F Reactor was the third of the three original graphite-moderated plutonium production reactors built at the Hanford Site. The 100-F Area also contained an experimental animal farm that began operation in 1945 and continued until 1976. Early studies at the experimental animal farm were conducted to measure the effects of reactor effluents on fish. Later research included the studies of swine, sheep, dogs, and rats. The experimental animal farm facilities included numerous laboratories, barns, pens, pastures, kennels, and waste facilities. At one time nearly 40% of the 100-F Area buildings were devoted to biological research. The reactor and support facilities operated until 1965 when the reactor facility was deactivated and permanently retired from service. Many of the reactor support buildings were decommissioned with the reactor in 1965. Biological research to study the effects of ionizing radiation on plants and animals continued in several buildings until 1976. The facilities related to the experimental animal farm were decommissioned between 1978 and 1979.

2.4.2 Support Areas

During the Manhattan Project, several facilities were built to support construction of the reactor and fuel fabrication complexes. In some cases, buildings remaining from before the Manhattan Project were modified to serve a support function. These temporary support areas are described in the following sections.

2.4.2.1 Central Shops. The White Bluffs area was selected as the location for the central shops to support the Manhattan Project. Construction began for new buildings or modification of existing facilities in 1943. Central Shops contained a wide variety of support facilities and a large equipment surplus area. The Central Shops area was operated from 1943 until 1952. During its operational use, the area underwent several modifications including relocation of warehouses from North Richland. In 1951, the Central Shops area was enlarged with the addition of general offices, craft shops, automotive and machine shops, a paint shop, a service station, steel fabrication sheds, oxygen and acetylene storage, lumber storage, electrical distribution, a sewer system, and a water system. In 1955, the White Bluffs Bank was used as an office equipment warehouse and storage area for the Hanford Fire Department. All facilities associated with the Central Shops were later demolished.

2.4.2.2 Construction Camps. Following removal of the local residents, the U.S. Atomic Energy Commission created a construction camp at the Hanford townsite that housed approximately 50,000 people between 1943 and 1945. During the life of the camp, 1,175 buildings and 9 service facilities were constructed. In addition, seven trailer camps were provided. For that period of time, the Hanford construction camp was the largest voting precinct in the United States and had the largest general delivery post office in the world.

Construction of housing for the camp began in March 1943. Some of the pre-Manhattan Project facilities remained and were used to support the construction camp. Existing facilities included several residences, the passenger and freight station, the grange hall, the Masonic Hall, the high school and church, and the branch line for the Chicago Milwaukee Railroad (HAN-10970, Vol. 2, *Hanford Engineer Works, History of the Project*). The construction camp covered an area of 607 ha (1,500 ac) approximately 4.2 km (2.6 mi) long and 1.6 km (1 mi) wide. It was designed to house, feed, and provide for recreational, religious, and other needs of the work force (WHC-MR-0425, *Manhattan Project Buildings and Facilities at the Hanford Site: A Construction History*).

The riverbank area north of Avenue A was used for storage of construction material and equipment and was served by the Chicago Milwaukee Railroad. This location contained most of the 65 craft shops, warehouses, offices, and 6 facilities used for construction of the camp. Twelve of the 65 buildings were pre-Manhattan facilities such as residences, fruit packing warehouses, and commercial store buildings. The area between First and Fourth Streets was used primarily for lumber storage and fabrication (HAN-10970, Vol. 2).

According to report issued in October 1944, the Hanford area contained administrative buildings, main construction office, hospital and employment buildings, construction guard headquarters, 131 men’s barracks with a housing capacity of 24,319; 912 men’s hutments with a 9,834 capacity and 64 women’s barracks with a housing capacity of 4,897. Trailer space for 3,639 units with 139 bathhouses was also constructed. Running water was supplied to each trailer lot. Service water faucets were located midway between adjacent lots. Individual sewer drains were available for each lot for trailer sink and toilet connections.

The Hanford trailer camps numbered 7 in all and were designed to provide accommodations for 3,639 trailer houses. Trailer camps were constructed with canopies, bathhouses, ice houses, coal storage buildings, playgrounds, clothes drying lots, trailer office, trailer camp warehouse, and dog pound. Trailer services included water and electrical services, sewer and septic tanks, telephones, general grading and landscaping, steam, roads, and walks.

At its peak, the number of people living in the camp reached nearly 50,000 between 1943 and 1945. Operation of the camp was terminated on February 17, 1945 (HAN-2914, *Monthly Field*

Progress Report HEW Project 9536 Period Ending 03/31/1945; IN-3023, *Report HEW 01/01/1945 Through 03/31/1945*). All portable hutments were dismantled and shipped. Trailers left in the camp were disposed of by the Benton County Sheriff. When the construction camp was dismantled, about 80% of the camp was sold to a Chicago salvage company with the remaining 20% kept as a residual camp for possible use.

2.4.3 300 Area Fuels Fabrication, and Research and Development

Development of industrial operations in the 300 Area began in 1943 and encompassed the fuel fabrication complex for the Hanford Site. Fuel for the reactors was fabricated from uranium shipped in from offsite production facilities. Completed fuel elements were then transported north to the 100 Areas for irradiation in the reactors. In addition to housing the Hanford Site fuel fabrication plants, the 300 Area was the center of many research and development facilities, chemical process laboratories, test reactors, and numerous ancillary support structures. As the Hanford Site production reactors were shut down, fuel fabrication in the 300 Area ceased. Portions of the 300 Area continue to be an active industrial complex, housing research and development facilities and analytical laboratories.

2.4.4 Contaminant Sources and Waste Streams

Operation of the Hanford facilities produced a variety of wastes and by-product effluents. The types of wastes produced were dependent on the specific process being conducted and included liquid wastes, solid wastes, and air emissions. Based on the process, some of these waste streams contained chemical contaminants, radioactive contaminants, or both. The following sections provide an overview of wastes and effluents that were generated in the River Corridor during operation of the Hanford facilities.

2.4.4.1 100 Area Waste Streams. The first eight plutonium production reactors at the Hanford Site (105-B, 105-C, 105-KE, 105-KW, 105-D, 105-DR, 105-H, and 105-F) used large quantities of water from the Columbia River for direct cooling of the reactor "piles." Reactor piles were made of stacks of carbon blocks containing uranium fuel rods where the irradiation of the fuel took place, yielding plutonium. The reaction produced a great deal of heat, so water was used to cool the piles and was discharged through large pipes into retention basins for short periods of time to cool. The cooling water was later discharged into the Columbia River or to disposal cribs and trenches. This process was referred to as "single pass" cooling. The discharged cooling water contained activation products primarily from impurities in the river water made radioactive by neutron activation and radioactive materials (fission products) that escaped from the fuel elements or tube walls during the irradiation process (DOE/RL-97-1047, *The Hanford Site* Historic *District*; PNNL-13230, *Hanford Site Environmental Report for Calendar Year 1999*). The ninth reactor, 105-N Reactor, had a modified design that recirculated purified water through a secondary, closed-loop cooling system. Water in the secondary loop cooled the reactor core primary closed-loop system. Secondary cooling loop water was used to produce steam (for electricity production) or discharged directly to the Columbia River.

The waste disposal practices associated with these operations resulted in releases of radionuclides and chemicals to soil and groundwater near the reactors. The primary source of these contaminants was the cooling water that flowed through the reactor core. Leaks in the reactor cooling water transfer systems, as well as the intentional effluent disposal into cribs and trenches, resulted in extensive contamination. Liquid waste sites were considered the first priority for remediation in the River Corridor. Some of these high-priority waste sites are included in this ecological risk assessment. In addition, solid wastes containing radionuclides and chemicals were buried in unlined burial grounds to isolate those wastes from ongoing operational activities.

Solid waste burial grounds were used for near-surface disposal of solid waste containing hazardous substances (radioactive and nonradioactive). Radioactive solid wastes in the 100 Area were segregated as soft waste (combustibles) or hard waste (greater than 99% metallic). The radioactive hard waste included process tubes, fuel element spacers, equipment, tools, and control rods. Most of the combustible waste from reactor operations was burned in open pits or in a natural draft incinerator located in a 100-K Area solid waste burial ground. Biological studies generated low-level soft waste containing radioactive tracers and low-level fission and activation products, which were buried in the 100-F Area.

In addition to these liquid waste sites and solid waste burial grounds that were documented and characteristic of the reactor operations, there are many other waste sites. These sites are located in both the reactor areas and the old townsites of White Bluffs and Hanford (100-IU-2 and 100-IU-6 Operable Units [OUs], respectively), and they include a wide variety of sites, including septic systems from contaminated facilities, burn pits, french drains, pre-Hanford Site and Hanford-era waste dumps in the town sites, small oil spills, and nonreactor-effluent pipelines.

Figure 2-6 is an aerial photograph of the 100-F Area during operation. Liquid waste retention basins can be seen in the lower right. Solid waste disposal trenches can be seen in the lower center of the photograph.

Reactor buildings present in each of the 100 Area reactor areas contain residual chemical and radionuclide contamination. As part of the remedy selected in *Record of Decision: Decommissioning of Eight Surplus Production Reactors at the Hanford Site, Richland, Washington* (58 FR 48509), the contamination will be sealed in safe storage enclosures and remain in interim safe storage (ISS). The safe storage enclosures encapsulate the reactor cores while radionuclide activities decay and are engineered to prevent contaminant releases to the environment, as well as biological intrusion. While the 105-B Reactor was initially identified for ISS (58 FR 48509), it has since been listed on the National Register of Historic Places and as a National Mechanical Engineering Landmark and is planned to be preserved as a museum.

Figure 2-6. Aerial Photograph of the 100-F Area During Operation.

2.4.4.2 300 Area Wastes Streams. Operations in the 300 Area created both liquid and solid waste. Prior to 1973, a series of solid waste burial grounds were used for waste and debris generated by 300 Area operations. These burial grounds were located just north and west of the 300 Area complex. Prior to 1994, liquid waste was discharged to a series of unlined ponds and process trenches in the 300 Area. The primary contaminant in the 300 Area is uranium from the fuel fabrication processes. However, numerous other potential radioactive and chemical contaminants exist for individual waste sites.

The 300 Area contains solid waste disposal sites, burn pits, ash pits, catch tanks, cribs, drain fields, dumping areas, foundations, french drains, injection wells, laboratories, process sewers, ponds, process facilities, radioactive process sewers, storage areas, storage tanks, surface impoundments, trenches, and unplanned releases. The 300 Area liquid and process waste sites routinely received discharges of millions of gallons of contaminated waste water from operations between 1943 and 1994. These sites and the burial grounds are suspected to be the primary source of uranium contamination in 300 Area groundwater.

Contaminant plumes from Central Plateau waste sites have migrated southeast toward the 300 Area. These plumes were driven east and southeast by the natural groundwater gradient across the Hanford Site and the large-volume discharges of cooling water to ponds and ditches in the Central Plateau.

Figure 2-7 shows an aerial photograph of the 300 Area looking northwest.

Figure 2-7. Aerial Photograph of the 300 Area, October 2007.

2.4.4.3 Air Emissions. Air emissions were generated at all of the main production facilities on the Hanford Site during operations. The type and quantity of air emission was dependent on the processes being conducted. Fuel separations facilities in the 200 Area produced the largest amounts of both radioactive and nonradioactive air emissions. The reactor facilities in the 100 Areas produced a smaller amount of air emissions. Fuel fabrication facilities in the 300 Area produced relatively low amounts of air emissions.

The following sections provide an overview of radioactive and nonradioactive air emissions that had the potential to impact the River Corridor by air deposition.

Radioactive Air Emissions. Most of the Hanford Site radioactive air emissions occurred between 1944 and 1972, primarily from the facilities in the 200 Areas that separated plutonium and uranium from irradiated reactor fuel (WDOH 2004, "The Release of Radioactive Materials from Hanford: 1944-1972"). The major radioactive air releases occurred between 1944 and 1957. The largest releases from these facilities occurred in 1945, before effective collection devices were installed upstream of the stacks to prevent the discharge of volatile and particulate radionuclides.

More than half of the entire amount of iodine-131 released during the 1944 to 1972 period was emitted in 5 months during 1945 (PNWD-2222 HEDR, *Radionuclide Releases to the Atmosphere from Hanford Operations, 1944 - 1972*). Water scrubbers and sand filters were installed upstream of the stacks at the 200 Area separation plants in 1948, greatly reducing the air emissions. More advanced technology was installed in 1950 (silver nitrate reactors) specifically to remove iodine-131, further reducing the stack releases.

Radioactive stack emissions from the 100 Area reactor facilities represented a minor contribution to the total releases. High-efficiency particulate air (HEPA) filters were used upstream of the reactor stacks to reduce the discharge of particulate contaminants. The 105-N Reactor also used charcoal filtration on its air discharges. As stated previously, the majority of air emissions from 100 Area facilities occurred from 1944 until the last of the single-pass reactors was shut down in 1972. The remaining reactor, 105-N Reactor, was shut down in 1989.

No radioactive stack releases from 300 Area operations were reported in *Summary of Environmental Contamination Incidents at Hanford 1952-1957* (HW-54636) or in PNWD-2222 HEDR. These reports reviewed the historical releases to the atmosphere from Hanford Site operations between 1944 and 1972. The 300 Area primarily supported fuel fabrication processes and research activities and did not provide a major contribution to radioactive airborne emissions on the Hanford Site.

Nonradioactive Air Emissions. Each of the operating areas had coal-fired and/or oil-fired powerhouses to provide heat and steam that produced emissions typical of these types of facilities. Chemical contaminant stack emissions from Hanford Site operations primarily consisted of volatile organic compounds associated with solvents used in the 200 Area fuel separation processes. There was no significant source of chemical emissions from the 100 Area processes. Inorganic compounds such as metals were an insignificant component of stack emissions. Organic solvents and degreasers were used in the 300 Area fuel manufacturing processes and produced a minor contribution of volatile organic compound emissions compared to the 200 Area processes. Water scrubbers were also used to reduce chemical emissions from the 300 Area facilities.

2.5 TRANSITION TO CLEANUP IN THE RIVER CORRIDOR UNDER CERCLA

This section summarizes the transition from plutonium production and fuels fabrication operations to a cleanup mission within the River Corridor. This information provides the rationale and basis for the cleanup process currently implemented in the River Corridor portion of the Hanford Site.

2.5.1 Addition of Hanford Site to National Priorities List

Past nuclear production and processing at the Hanford Site released hazardous substances to the environment and resulted in areas of contaminated soil and groundwater that pose a risk to human health and the environment. Between 1985 and 1988, preliminary assessment/site inspection activities were completed to identify waste sites and prioritize the relative hazards. Waste disposal information was collected through exhaustive reviews of literature and maps, employee interviews, and visual inspection of all sites and unplanned releases. Results were organized and sites were ranked with respect to potential environmental impacts in accordance with a slightly modified version of the CERCLA hazard ranking system. The results from this process provided information to support addition of the 100 and 300 Areas to the National Priorities List (NPL).

2.5.2 Tri-Party Agreement

In anticipation of the NPL listing, the U.S. Environmental Protection Agency (EPA), Washington State Department of Ecology (Ecology), and U.S. Department of Energy (together referred to as the Tri-Parties) entered into the *Hanford Federal Facility Agreement and Consent Order* (Tri-Party Agreement) (Ecology et al. 1989). The Tri-Party Agreement uses both CERCLA and *Resource Conservation and Recovery Act of 1976* (RCRA) requirements to define a framework for the characterization, evaluation, and remediation of waste sites and groundwater at the Hanford Site. A listing of waste sites was included in Appendix C of the Tri-Party Agreement.

2.5.3 Hanford Past-Practice Strategy

The conventional CERCLA process for establishing remedial action decisions required conducting comprehensive characterization via remedial investigations (RIs), and preparing baseline risk assessments, feasibility studies, and final records of decision (RODs). In 1991, the Tri-Parties adopted a “bias-for-action” approach to expedite the decision-making process and allow cleanup actions in the River Corridor to begin as soon as possible. Known as the *Hanford Past-Practice Strategy* (HPPS) (DOE/RL-91-40), this “bias-for–action” approach streamlined the RI/FS process to begin remediation of contaminated waste sites earlier than typically performed under the traditional CERCLA process. The HPPS incorporated limited field investigations (LFIs), focused feasibility studies (FFSs), and qualitative risk assessments (QRAs), and allowed further site characterization to proceed in tandem with waste site remediation. The waste sites with the highest potential to contribute to contamination of groundwater and the Columbia River were prioritized for the first remediation efforts. As opposed to a full baseline risk assessment, the QRAs were performed using limited data and included an abbreviated quantitative risk analyses. The approach outlined in the HPPS is consistent with later EPA initiatives developed to support expedited cleanups, such as the *Superfund Accelerated Cleanup Model* (EPA/540/R-98/025) and the *RCRA Facility Stabilization Initiative* (DOE/EH-231-076/0295r 2003).

2.5.4 Investigation and Assessment Activities

Prior to the start of remedial actions, an adequate understanding of the contaminated areas was necessary to support evaluation and selection of sound cleanup decisions and subsequent remedial designs. This was accomplished through a series of investigation and characterization activities that included the following:

- Technical baseline reports summarizing existing process and contamination information
- Limited field investigations conducted to collect additional characterization data and support qualitative risk assessments
- Focused feasibility studies prepared to select interim remedial actions.

The early characterization, field investigations, and assessments established a basis for action and supported remedy evaluation and selection for waste sites included in the River Corridor. Each of these activities is described in the following sections.

2.5.4.1 Technical Baseline Reports. Technical baseline reports were prepared for each operating area and provided to DOE, regulatory agencies, and contractors with a “baseline” of technical information related to operational processes and resulting contaminated waste sites. Table 2-1 is a list of the technical baseline report prepared for each River Corridor operating area. The information in the reports was based on evaluation of numerous Hanford Site reports, drawings, and photographs supplemented with on-site inspections and interviews with employees and other sources to provide first-hand knowledge of sites, facilities, and processes. Intrusive field investigation or sampling was not a part of developing the technical baseline reports.

Each technical baseline describes the industrial process history. Information includes years of operation and intensity of use, as well as containment failure events, process improvements, or research activities unique to a given area. The reports also contain descriptions of the types of waste streams that resulted from the operations, with estimated volumes and suspected contaminants. The reports contain maps and photographs of the facilities cited in the reports and information on the sampling for environmental monitoring conducted for each area. A detailed description is provided for each waste site within each area, describing known contamination and condition as of the time the report was written.

2.5.4.2 Investigation Work Plans. The technical baseline reports were used to prepare work plan documents. The work plans provided the direction to conduct follow-on field investigations to provide supplemental data. Table 2-2 lists the work plan documents for the River Corridor source OUs.

2.5.4.3 Limited Field Investigations. The LFI reports completed for the 100 and 300 Area OUs contained a compilation of site characterization data including information from historical data compilation, nonintrusive investigations (e.g., geophysics), intrusive investigations (e.g., boreholes), and the 100 Area aggregate studies (i.e., ecological, river water, and sediment sampling) (DOE/RL-88-36, *RCRA Facility Investigation/Corrective Measures Study Work Plan for the 100-HR-3 Operable Unit, Hanford Site, Richland, Washington*). The LFI reports completed for River Corridor waste sites are listed in Table 2-3.

The LFIs recommended sites for interim remedial action and categorized them as high or low priority. High-priority sites were considered to have the greatest potential to contribute to contamination of groundwater and the Columbia River. The reports also provided a preliminary summary of site characterization studies and identified contaminant-specific and location-specific applicable or relevant and appropriate requirements (ARARs). The data collection activities associated with the LFIs supplemented existing information to support formulation of conceptual models, as well as performance of QRAs for each area.

2.5.4.4 Qualitative Risk Assessments. Site-specific information from the technical baseline reports and the LFIs were used to prepare QRAs for specific waste sites in each of the operating areas. The QRAs were combined into summary reports for each of the applicable OUs. The QRA reports are listed in Table 2-4.

The QRAs were performed for the high-priority sites in each OU. Conservative assumptions, such as highest reported contaminant levels from either the LFI or historical data from *Radiological Characterization of the Retired Areas* (UNI-946), were used in the QRAs. The QRAs provided estimates of human health risks, assuming frequent use and occasional use, and included considerations such as the attenuation of external dose provided by layers of clean gravel fill that overlie many sites. The QRAs identified the human health risk to be primarily from external exposure to the radionuclides cobalt-60, cesium-137, europium-152, and europium-154. Ecological risks were also estimated for a single receptor, the great basin pocket mouse.

The high-priority sites were evaluated using the following criteria to help identify those recommended for remedial actions:

- Magnitude of risk identified in the QRA
- Exceedance of a chemical-specific ARAR
- Potential to contaminate groundwater
- Insufficient information for conceptual model

- Multiple exposure pathways
- Expected natural attenuation and radioactive decay.

The QRAs were used to determine whether contaminant concentrations pose an unacceptable risk that warrants remedial action and have been used to establish the basis for action for all waste sites identified in the River Corridor interim records of decision (IARODs) and RODs.

2.5.4.5 Focused Feasibility Studies. The purpose of the FFSs performed in the 100 and 300 Areas was to support selection of interim remedial actions for sites within the OUs. The *100 Area Source Operable Unit Focused Feasibility Study* (DOE/RL-94-61), *Focused Feasibility Study for the 300-FF-2 Operable Unit* (DOE/RL-99-40), and *Phase III Feasibility Study Report for the 300-FF-1 Operable Unit* (DOE/RL-94-49) reports provided the decision makers with the information they require from the investigation activities for selection of remedial actions. The FFSs developed site profiles for the high-priority waste sites (as identified in the LFI reports) and made comparative analyses of the remedial action alternatives. Because a final land-use decision had not been made at the time the remedies were being evaluated, the FFSs proposed baseline assumptions to be used during analysis of remedial alternatives.

2.5.5 Cleanup Action Decisions

Under the HPPS, the Tri-Parties established cleanup actions through IARODs as summarized in Table 2-5. The cleanup decisions in the IARODs were based on pre-existing knowledge of the waste sites and similarities in types of waste sites and contaminants of potential concern (COPCs) among the operational areas, supplemented by additional characterization from the LFIs. The IARODs identify waste sites to be evaluated and specify the requirements for achieving protectiveness of potentially exposed receptors, groundwater, and the Columbia River.

The selected remedy established for cleanup of waste sites in the River Corridor is remove, treat, dispose (RTD). A fundamental assumption that supported the regulatory decision to select this remedy was investigation information indicating that contaminant concentrations decrease with depth, thereby making “source removal” the desired choice when measured against the CERCLA feasibility study criteria. If contaminant concentrations did not decrease with depth, it is likely that another remedial alternative (e.g., capping) would have been selected in place of RTD.

The remedial action objectives (RAOs) stated in the IARODs are narrative statements that define the extent to which the waste sites require cleanup. Remedial action goals (RAGs) are contaminant-specific numerical cleanup criteria developed to guide the remedial actions to meet the RAOs. The RAGs used in the IAROD process employed residential and industrial exposure scenarios to evaluate risks from contaminants in soil. Cleanup levels in the 100 Area of the River Corridor are based on a residential exposure scenario. Cleanup levels for the 300 Area are based on a mix of residential and industrial exposure scenarios. Remedial action goals related to radiation dose were calculated in the IAROD process using the RESidual RADioactivity (RESRAD) computer code. RAGs related to chemical cancer risk and hazards were based on screening models of the *Model Toxics Control Act* (MTCA) in the 1996 *Washington Administrative Code* (WAC) Part 173-340.

The IAROD RAGs were based on two different risk models. The RAGs for chemical constituents were based on the MTCA chemical soil cleanup levels based solely on inadvertent soil ingestion. The RAGs for radionuclide constituents were calculated using RESRAD and included additional exposure pathways. The differences between the risk models for the chemical and radionuclide RAGs were greatest for the residential scenario, where the RESRAD calculation addressed exposure via home-grown foodstuffs (e.g., produce, beef, and milk).

2.6 IMPLEMENTATION OF SOURCE CLEANUP ACTIONS

This section summarizes the process for implementing and verifying source cleanup actions at waste sites in the river corridor. The RTD remedy selected in the IARODs is implemented using the "observational approach." The observational approach allows remediation to proceed under a set of assumptions called a "conceptual model," which describes operational and contamination characteristics of a given waste site based on known characteristics of analogous, or similar, sites. During remediation, the assumptions of the conceptual model are continually evaluated, and the approach for implementing the remedy is reassessed or refined as appropriate. Processes for identifying new waste sites and adding them to existing cleanup actions are also described.

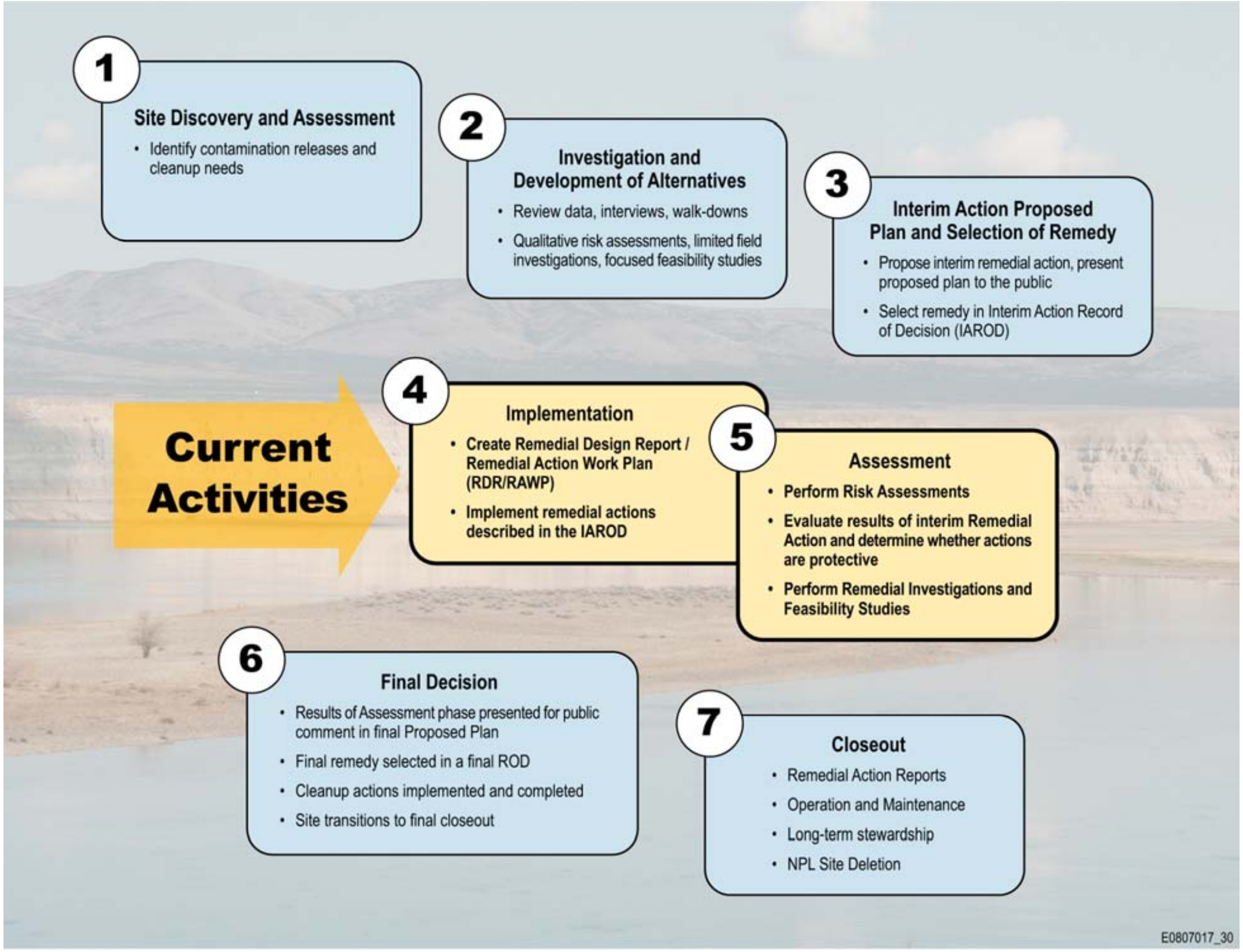
Figure 2-8 shows a summary of the CERCLA process as it is being implemented for the River Corridor. Current activities are shown as steps 4 and 5.

2.6.1 Excavation and Disposal

The RTD remedy presented in source OU interim action RODs requires removal of wastes, treatment (as necessary), and subsequent disposal at the Environmental Restoration Disposal Facility (ERDF) or other approved facilities. The RTD remedy requires excavation of contaminated soil and debris to the extent required to meet established cleanup levels. For engineered structures such as a burial ground or landfill, the entire structure and its contents must be excavated. Excavation following removal of the engineered structure must continue to the extent required to meet established cleanup levels. At waste sites that contain a heterogeneous mixture of soil and debris (such as a solid waste burial ground), it is possible to segregate and set aside material that meets established cleanup levels for potential use as backfill material.

If information gathered during the course of excavation suggests source removal to meet established cleanup levels is not practical or another remedy is more appropriate, an explanation of significant differences (ESD) or amendment to the IAROD must be issued before implementing a modification to the established cleanup approach.

Figure 2-8. CERCLA Process as Implemented for the River Corridor.



Engineering designs for implementation of the RTD remedy at individual waste sites are intended to reflect all potential contaminants that may have been carried through engineered structures, direct buried, or released to the soil. In addition, source unit actions are designed to protect groundwater as a potential drinking water source regardless of the presence of existing groundwater contamination or controls preventing use that may be prescribed by the IAROD or ROD. Based on observation of waste and debris removed from the waste site during implementation of the remedial design, additional contaminants may be identified that must be considered for excavation and site closeout (including groundwater impacts). Observation, monitoring surveys, field screening, and laboratory analytical data demonstrate that the interim cleanup goals have been met.

A majority of the contaminated soil and debris that is removed during the excavation process is transported to the ERDF for disposal. Some of the waste must be treated (e.g., macroencapsulated) prior to disposal at the ERDF. Waste streams that cannot meet the

ERDF acceptance criteria are packaged and sent to other approved onsite or offsite disposal facilities.

2.6.2 Confirmatory Process for Selected Waste Sites

Some waste sites in the River Corridor lack the process knowledge or data to make a decision on whether or not cleanup actions are required in accordance with the existing RTD remedy. These waste sites are identified as "remaining sites" or "candidate sites" in the 100 Area and 300 Area decision documents, respectively. Remaining sites and candidate sites undergo a confirmatory sampling process as an intermediate step to support making cleanup decisions.

The confirmatory process involves development of a site-specific sample design to obtain additional characterization information. Each sample design is submitted by DOE for approval by the lead regulatory agency. Once samples are collected and associated results are evaluated, cleanup decisions are made. Sites that exceed the IAROD cleanup standards are "plugged in" to the existing RTD remedy and addressed as described in Section 2.6.1. Sites that meet the IAROD cleanup standards without further action are typically reclassified as "no action" in accordance with the verification and site reclassification process described in the following section.

2.6.3 Verification of Cleanup Actions and Site Reclassification

Once a remedial action at a waste site is complete, cleanup verification packages (CVPs) or remaining sites verification packages (RSVPs) are prepared to document completion of cleanup actions in accordance with the applicable decision document and support waste site reclassification as defined in RL-TPA-90-0001, *Tri-Party Agreement Handbook Management Procedures*, Guideline Number TPA-MP-14, "Maintenance of the Waste Information Data System (WIDS)." Definitions for the different reclassification categories identified by TPA-MP-14 are presented in Table 2-6.

In order to verify that an individual waste site meets the cleanup levels in the ROD, samples are collected in accordance with regulator approved sample designs and analyzed in a laboratory. Residual concentrations for waste site contaminants of concern (COCs) are then evaluated to verify that the site does not pose an unacceptable risk to human health, the environment, groundwater, and Columbia River.

Part of the closeout documentation includes the results from modeling to predict the future migration of COCs through the soil column and into groundwater. To meet the criteria for reclassification, all source waste sites must be shown through predictive modeling to be protective of groundwater unless an alternative endpoint is agreed to with appropriate documentation (e.g., ESD or amendment to the applicable ROD). Consequently, any contamination remaining at source waste sites following remediation will not present an adverse impact on underlying groundwater and protectiveness of source units can be demonstrated independent of the remediation status for existing groundwater plumes.

Site-specific data evaluations and conclusions are presented in CVPs and RSVPs. A protectiveness statement is included in the CVP or RSVP that supports reclassification of the site from "Accepted" to "Closed Out," "Interim Closed Out," or "No Action." All reclassification actions are approved by DOE and the lead regulatory agency. In some instances where waste will remain at the site after cleanup actions are completed there is also a requirement to maintain some level of institutional control.

2.6.4 Backfill and Revegetation

Excavated waste sites are backfilled to match the surrounding contours with clean fill from an approved borrow pit or approved overburden that meets cleanup levels. To promote the reintroduction and colonization of native species, remediated upland waste sites were revegetated with the goal of reestablishing sagebrush/Sandberg's bluegrass communities with a mixture of other native grasses and forbs adapted to rocky soils and arid conditions. Following vegetation, the initial plant varieties, density, and species diversity may not be the same as an undisturbed area. However, as the restored plant communities mature, improvements in plant coverage provide important habitat for native wildlife species.

2.6.5 Ongoing Waste Site Identification

A fundamental assumption to the River Corridor cleanup program is that site risks are associated predominantly with locations where waste management/disposal activities or other environmental releases have occurred. These locations are identified as waste sites. An inventory of known and potential waste sites has been maintained in the Waste Information Data System (WIDS) database since the early 1980s. The WIDS list for the Hanford Site has grown to more than 2,800 sites, both within the areas where plutonium production and research operations occurred and in areas of lower intensity use. Cleanup activities conducted to date provide real-time information and data to support the process of identifying waste site locations in the River Corridor. In addition, processes have been established to address new discoveries when identified.

Waste site identification activities in the River Corridor fall into two categories: systematic and observational. Various systematic programs have been conducted at different times, while observation-based identification activities can happen at any time and will continue into the future. The following sections describe systematic and observational methodologies for identifying waste sites.

2.6.5.1 Orphan Site Evaluations. The most current systematic program in the River Corridor consists of a series of ongoing investigations that were initiated in 2004 to identify potential new waste sites that may have been overlooked. These investigations, called "orphan site evaluations," supplement past systematic efforts that identified source waste sites and are a systematic approach to evaluate land parcels in the River Corridor to ensure that all waste sites or releases requiring characterization and cleanup have been identified. Information collected through these evaluations also supports elements of the CERCLA Section 120(h)(4), Federal

Property Real Disposal Process, requirements for review and identification of uncontaminated property at federal facilities.

Two of the key elements that make up an orphan sites evaluation include a historical review and field investigation.

- **Historical review**: Review historical information (documents, photographs, drawings, geophysical surveys, etc.) associated with facilities, piping systems, operational processes, and waste sites to identify potential orphan sites and target areas for field investigation.

- **Field investigation (operational areas)**: Conduct a systematic walking survey of operational areas to document potential orphan sites (field-based observation) and to follow up on potential orphan sites identified from historical review. Geophysical surveys also may be conducted in target areas as part of the field investigation. Walking surveys are conducted on a 30- by 30-m (98- by 98-ft) reference grid system. Hand-held global positioning system units and digital cameras are used to record locations and information for observed items.

- **Field investigation (nonoperational areas)**: A graded approach is used for the regions between the operational areas. Digital high-resolution aerial photographs and light detection and ranging imagery of the River Corridor were collected in 2008 and are used to conduct "virtual walkdowns" of the areas. Based on results of these "virtual walkdowns," areas are selected to conduct walking surveys consistent with the approach for operational areas (e.g., 30- by 30-m reference grid system). Vehicle surveys along accessible roads and utility easements are also part of the field investigation.

Results of the evaluations are reviewed with participation from the regulatory agencies and are summarized in an orphan site evaluation report. Following completion of the orphan sites evaluation for a given area, new waste sites identified by the process are "plugged in" to an appropriate interim action ROD for subsequent characterization and remediation. If one or more of the new waste sites does not meet the criteria for "plug in" under the provisions of an existing ROD, the Tri-Parties will determine the regulatory strategy for selecting cleanup actions under an appropriate decision document.

2.6.5.2 Observation-Based Waste Site Discoveries. In addition to the systematic processes that have been conducted in the River Corridor to identify waste sites, observation-based discoveries can lead to identification of new waste sites (often referred to as discovery sites). These discoveries are a planned and expected element of the observational approach based cleanup actions. Demolition and removal of retired facilities, cleanup of existing waste sites, and routine monitoring or area management activities provide new opportunities for discovery of potential waste sites. Discoveries can occur at any time and may be identified by any individual. Observation-based discoveries that become waste sites typically are added to the scope of existing interim action RODs in the same way as sites identified through the systematic processes. The opportunities for these discoveries will continue throughout cleanup of the river corridor, including activities conducted after final RODs are issued.

2.6.6 Groundwater Cleanup Actions

The goal of groundwater cleanup actions is to restore groundwater to drinking water standards and to ensure that aquatic life in the Columbia River is protected by achieving aquatic water quality standards at groundwater discharge points to the river. It is intended that these objectives be achieved, unless technically impracticable, within a reasonable time frame. In those instances where RAGs are not achievable in a reasonable time frame or are determined to be technically impracticable, programs will be implemented to prevent further migration of the plume, prevent exposure to contaminated groundwater, and evaluate further risk reduction opportunities as new technologies become available.

Unlike the source units, where interim cleanup actions are designed to address all potential contaminants for each waste site, the interim actions for groundwater address contaminants that were considered to be principal threats to public health and the environment. In the 100 Area, chromium and strontium-90 were identified as principal threats to the Columbia River and are the subject of ongoing response actions. Principal groundwater contaminants in the 300 Area include tritium and uranium.

Multiple groundwater treatment systems are in place to remediate groundwater plumes that threaten the Columbia River. The principal cleanup remedy for groundwater in the River Corridor is pump and treat. Contaminated groundwater is extracted from the plume using a series of wells and processed through an external treatment system designed to remove target contaminants. The resulting “clean” groundwater is then pumped back into the aquifer using a series of “injection” wells. A network of monitoring wells is sampled periodically and results are used to track plume migration and progress of the cleanup actions.

The DOE is also conducting various new technology treatment tests to explore the application and effectiveness of the using the following:

- Adding organic nutrients to the aquifer to stimulate native bacteria, which act to remove contaminants from the groundwater
- Adding nontoxic chemicals (e.g., phosphate) to the aquifer to trap contaminants, rendering them immobile
- Injecting micron-size or smaller iron particles to the aquifer to immobilize contaminants
- Adding apatite, a stable mineral found in teeth and bones, to adsorb and hold contaminants, preventing further migration
- Introducing a strong reducing chemical, such as calcium polysulfide, into the aquifer to change contaminants to a less mobile and less toxic form
- Plants to extract and/or sequester soil contaminants.

2.7 OVERVIEW OF RIVER CORRIDOR CONDITIONS

The following sections provide a summary of the current conditions of the River Corridor. As previously stated, the current condition of the River Corridor is composed of a combination of remediated waste sites, yet-to-be remediated waste sites, and nonoperational areas where historic information and other studies indicate there were no direct releases of hazardous substances posing a threat to human health or the environment. Unless specified, current conditions in the River Corridor are represented by the conditions present when the RCBRA work plan (DOE/RL-2004-37, *Risk Assessment Work Plan for the 100 Area and 300 Area Component of the RCBRA*) was written. The work plan and ensuing planning documents defined the scope of the project based on the extent of remediation and characterization that was complete at the time. This corresponds to roughly the end of 2005.

2.7.1 Facilities and Reactors

Final remediation of the River Corridor includes decommissioning and demolition of the structures and facilities at the operational areas. Following demolition and disposition of any below-grade structures, the facility footprint is verified to meet the cleanup requirements of the IARODs and closed out. In some cases, contaminated structures and soil beneath a structure are established as discrete waste sites and remediated according to the process previously described.

Table 2-7 presents a summary of the current conditions for the reactor facilities in the River Corridor. Five of the nine production reactors have been put in ISS awaiting final disposition. The ISS includes decontamination of reactor structures, reduction of the reactor footprint through decommissioning, and construction of a safe storage enclosure (SSE) over the reactor core to prevent deterioration and release of contamination for up to 75 years. The DOE currently plans to implement interim safe storage followed by deferred one-piece removal as the final disposition alternative for the surplus reactors. Three production reactors (105-N, 105-KW, 105-KE) are currently being put in ISS condition. One reactor, 105-B, is currently designated as a national historic landmark and is planned for extended preservation.

To date, approximately 300 support facilities and structures in the River Corridor have been decommissioned and demolished. The demolition debris has been disposed in approved disposal facilities. At the current time, about 300 facilities and structures in the River Corridor remain to be decommissioned and demolished.

2.7.2 Waste Sites

Since the establishment of the 100 and 300 Areas on the NPL, waste sites have been identified and evaluated following the processes that were previously described in Sections 2.5 and 2.6. An inventory of waste sites has been developed and is maintained in WIDS.

Under the observational approach, large-scale cleanup began in the River Corridor in the mid-1990s. The soil and waste sites with the highest potential to contribute to contamination of groundwater and the Columbia River were prioritized for the first remediation efforts. After nearly 13 years of waste site cleanup in the River Corridor, DOE is part way through the work that is planned under the interim action RODs and the Tri-Party Agreement. Cleanup actions at hundreds of waste sites have been completed. Much has been learned from the work that has been conducted, which has included excavation of large liquid waste disposal sites, burial grounds and landfills, and unplanned releases (spill sites). More than 8 million tons of contaminated soil and debris have been excavated and disposed at the ERDF. Hundreds of wastes sites remain that are scheduled to be addressed in the next several years, with a majority of the cleanup work in the River Corridor anticipated to be complete by 2015.

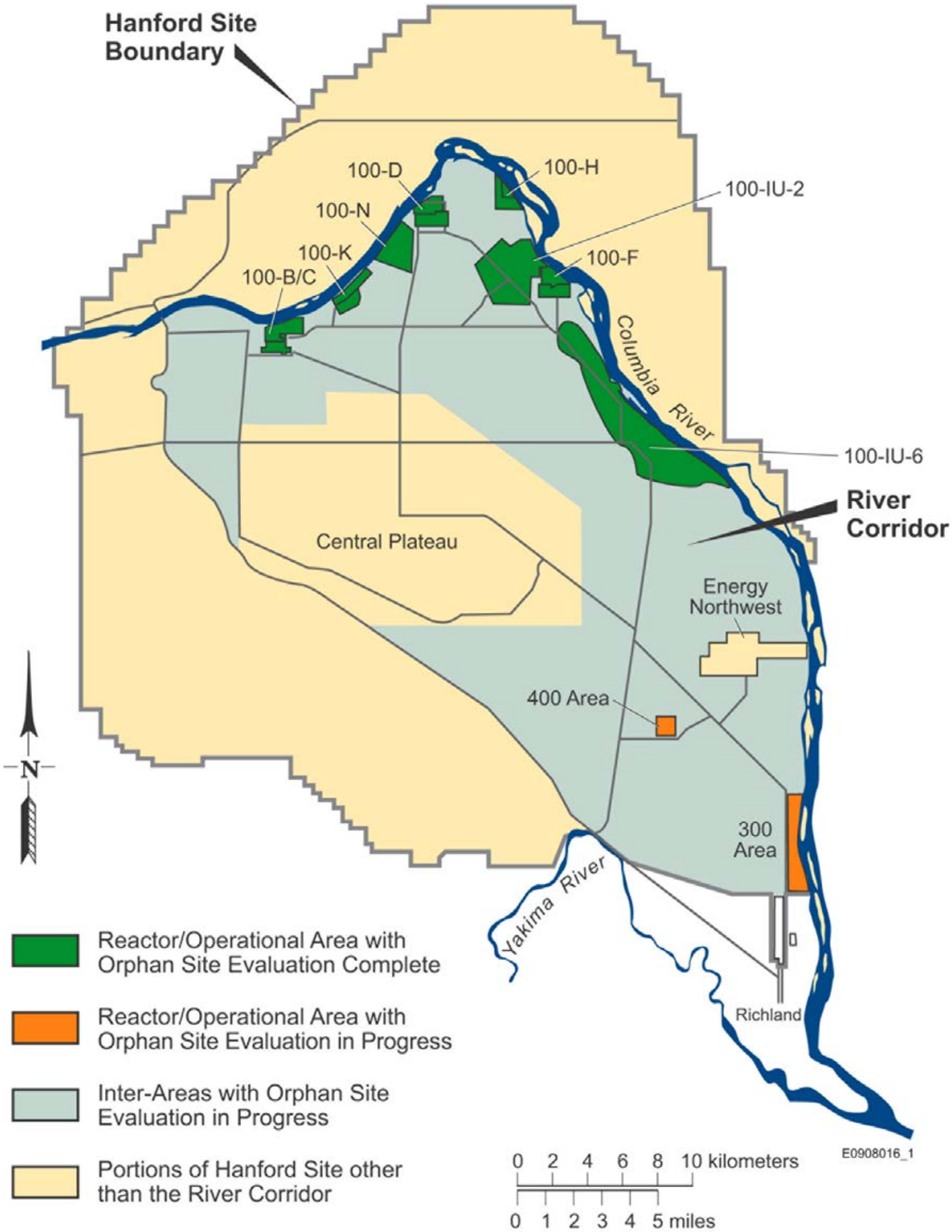
Early cleanup actions have helped sharpen the focus of data-collection efforts in recent years to fine tune remedial actions. Efforts to understand the nature and extent of contamination beyond the areas adjacent to reactors have been extensive and have demonstrated that the focus of early actions on waste sites associated with reactor areas has been instrumental in addressing the highest priority environmental risks. The Tri-Parties are at a point in the River Corridor cleanup process to begin the transition from interim actions to final cleanup actions, using the RI/FS process.

Table 2-8 provides a summary of the waste sites that were identified in the RCBRA work plan (DOE/RL-2004-37) for evaluation in the RCBRA. The table shows the waste site name, type of site, estimated cancer risk prior to remediation, and the major contaminants for each site. The estimated risk prior to remediation was obtained from the QRA evaluations using the residential exposure scenario. The table also shows the status of each waste site following remediation including excavation depth, surface area, waste removed, estimated dose and cancer risk following remediation, current status, and date reclassified. The estimated radiation dose and cancer risk following remediation is obtained from the closeout verification sampling results evaluated using the exposure scenarios specified by the IARODs. Radiation dose (mrem/yr) is calculated from direct exposure to the radionuclide contaminants of concern evaluated for each waste site. Excess cancer risk is calculated from nonradionuclide carcinogenic contaminants of concern for each waste site. If a waste site did not have radionuclide contaminants or nonradionuclide carcinogenic contaminants, then no radiation dose or cancer risk is shown for that site. Hazard index (HI) is also calculated for all nonradionuclide noncancerous contaminants of concern for each site. Hazard index values must be less than one in order to meet the IAROD RAGs and reclassify the waste site. Since all HI values are less than one for the sites evaluated, they are not shown in Table 2-8. This table represents the present-day current condition of each waste site in the scope of the RCBRA.

2.7.3 Identification of New Waste Sites

The progress of orphan site evaluations that began in the River Corridor in 2004 is shown in Figure 2-9. Through this systematic process, 155 new waste sites have been identified to date. These new waste sites have been added to the WIDS inventory using the TPA-MP-14 process and have been “plugged in” to the applicable RODs based on their location for future characterization and cleanup. The orphan site evaluation process will continue to cover the entire geographical area of the River Corridor. In addition, observation based discovery of new waste sites will continue as demolition and cleanup actions are conducted in the River Corridor.

Figure 2-9. Orphan Site Evaluations Completed or in Progress in the River Corridor.



2.7.4 Groundwater Cleanup

Table 2-9 identifies the groundwater OUs with remedial actions, the type of remedy, and status. These actions are expected to continue until RAOs are achieved or subsequent decisions are made that set final groundwater restoration requirements. Four large pump-and-treat systems are operated at the 100-K, 100-D, and 100-H Areas to remove hexavalent chromium from the underlying groundwater. To date, over 7.6 billion L (2 billion gal) of groundwater have been treated, removing nearly one ton of hexavalent chromium from the aquifer. Other technologies have included tests of biostimulation, electrocoagulation, calcium polysulfide injection, and in situ Redox manipulation treatment.

Treatment technologies at 100-N Area are being tested to slow migration of strontium-90 contamination to the Columbia River. Technologies include developing an apatite mineral barrier between the contamination source and the Columbia River to absorb and retain the strontium-90. Also, stands of coyote willow (*Salix exigua*) are being tested as a method for localized removal of strontium-90 near the river.

Current actions in the 300 Area include monitoring the conditions while natural processes act on the levels of tritium and uranium contamination in the groundwater. Other potential treatment technologies are also being investigated.

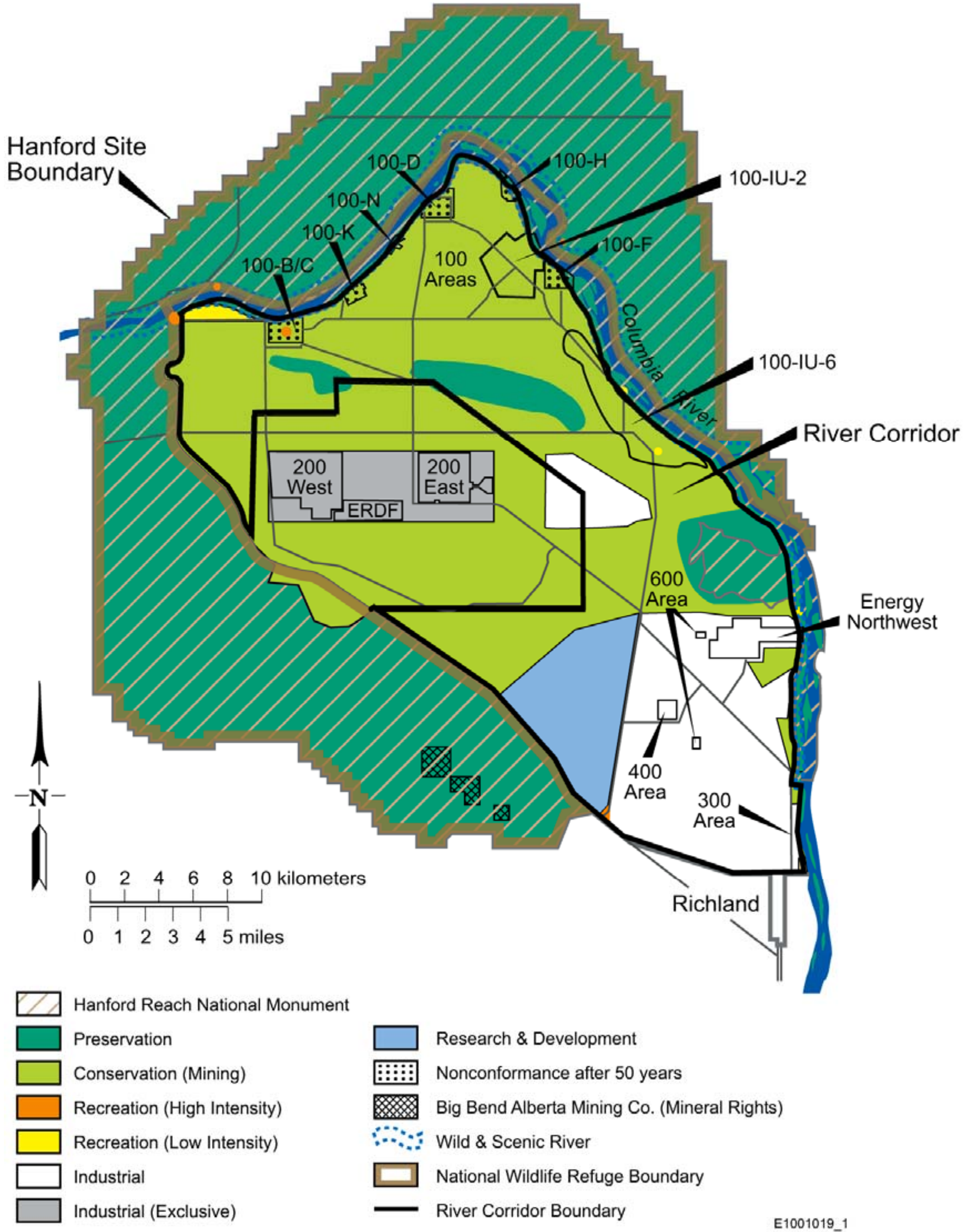
2.7.5 Land Use

Current land use in the River Corridor consists of waste management, environmental monitoring, soil and groundwater remediation, and conservation and restoration activities. Present-day exposure is limited primarily to Hanford Site employees and contractors and is controlled by access restrictions.

DOE’s reasonably anticipated future land use is predominantly conservation/preservation in the River Corridor’s 100 and 600 Areas. The likely human receptors in these areas are part time users of the land and could include recreational users, tribal users, and monument workers. In the 300 Area, DOE’s reasonably anticipated future land use is industrial and the likely human receptors could include recreational users, tribal users, and industrial workers.

The Tri-Parties have participated in multiple discussions with the many affected parties regarding reasonably anticipated future land use. That input, including advice from the Hanford Advisory Board, and other considerations presented in EPA’s OSWER Directive 9355.7-04, *Land Use in the CERCLA Remedy Selection Process*, for land use planning, will be used by the Tri-Parties to support final remedy selections. The Hanford Comprehensive Land-Use Plan (CLUP) (DOE/EIS-0222-F, *Hanford Comprehensive Land-Use Plan Environmental Impact Statement*) also provides information on reasonably anticipated future land use. Figure 2-10 shows anticipated future land use as designated by the CLUP.

Figure 2-10. National Monument Boundaries and Future Land Use Designated by the Comprehensive Land-Use Plan.



Disclaimer: This figure must be viewed in color for total content and meaning. Please contact WCH if color copy is needed.

2.7.5.1 National Monument Designation. The stretch of the Columbia River flowing through the Hanford Site is referred to as the Hanford Reach. It is a major nontidal, free-flowing stretch of the Columbia River. The river, islands, gravel bars, sloughs, riparian areas, and dune field of the Hanford Reach provide a variety of habitats that are now rare along the Columbia River due to the extensive reservoir system, development, and agriculture. Since 1943, DOE (or its predecessor federal entities) has held title to the lands that make up the Hanford Reach. Administration of this unit is multi-jurisdictional and complex, with the U.S. Fish and Wildlife Service (FSW), Bureau of Land Management, DOE, and various state and local agencies each playing specific roles.

In 1994, the National Park Service completed the *Hanford Reach of the Columbia, Comprehensive River Conservation Study and Environmental Impact Statement* (NPS 1994). This study evaluated outstanding features of the Hanford Reach and provided recommendations on alternatives for protecting the features. The associated ROD recommended designating the Hanford Reach and approximately 41,279 ha (102,000 ac) of adjacent lands as a National Wild and Scenic River and a National Wildlife Refuge, respectively (DOI 1996). In 2000, Presidential Proclamation 7319 was signed, creating the Hanford Reach National Monument to be managed by the FWS and DOE (65 FR 37253). The Monument was established for the purpose of protecting the biological, historic, and scientific objects contained within.

The Hanford Reach National Monument consists of an 82.1-km (51-m)-long unimpounded stretch of the Columbia River and federally owned land on either side of the river with an average width of 402 m (1,320 ft). The boundaries of the Hanford Reach National Monument, as presented in the proclamation, are shown in Figure 2-10. They encompass approximately 78,917 ha (195,000 ac). The Monument encompasses approximately 793 km^2 (306 mi^2) of lands already owned by the federal government that had previously been designated for preservation or conservation under the CLUP (DOE/EIS-0222-F). To support continued protection of natural and cultural resources, the proclamation stated that the Monument would not be developed for residential or commercial use in the future (65 FR 37253).

The majority of the Monument is managed by the FWS through a Permit and Memorandum of Understanding granted by DOE in 2001. The portion of Monument lands that are managed by the FWS are included in the *Hanford Reach National Monument Comprehensive Conservation Plan and Environmental Impact Statement* (FWS 2008). The remaining Monument lands that are managed by DOE are undergoing or supporting environmental cleanup.

2.8 EVALUATION OF RESIDUAL CANCER RISK AND NONCANCER HAZARD

Final remedy selection (development of final RODs) must be completed in order for the NPL CERCLA sites in the River Corridor to reach final closeout. Although interim remedial measures are intended to achieve remedies that are likely to lead to a final ROD, final remedy decisions could potentially be different than those identified by the IARODs. This RCBRA human health risk assessment and the companion ecological risk assessment provide an evaluation of ecological and human health risk from residual contamination at waste sites remediated under the IARODs and from potentially affected environmental media under various exposure scenarios. The site-specific risk information provided by the RCBRA will be used to support final RODs for the River Corridor.

This section presents a screening-level assessment of residual cancer risks and noncancer hazards for the remediated wastes sites using the exposure scenarios that were the basis of the residential RAGs for the IAROD cleanups. This assessment was done to provide information about the residual risks and noncancer hazards associated with post-interim action conditions at the remediated waste sites and help assess whether residual conditions are protective of human health. Results of this screening-level assessment can be compared to risk assessment results presented in later sections of the RCBRA where a range of exposure scenarios were evaluated.

As discussed previously, waste sites evaluated in the River Corridor were Interim Closed using RAGs related to soil exposure by human receptors and protection of groundwater from contaminants leaching from soil. Remedial action goals in the 100 Area of the River Corridor were based on a residential exposure scenario. Remedial action goals for the 300 Area were based on a mix of residential and industrial exposure scenarios. The IAROD residential scenario for radionuclides is a Rural Residential scenario that includes food exposure pathways (e.g., ingestion of homegrown produce, beef, and milk). The IAROD residential scenario for chemicals is based on the MTCA Method B Soil Cleanup Levels for Unrestricted Land Use (WAC 173-340-740, "Model Toxics Control Act – Cleanup"). The MTCA Method B levels are based solely on incidental soil ingestion and do not address the food exposure pathways that were included for the radionuclide Rural Residential scenario.

As discussed in Section 2.6.3, CVPs or RSVPs were prepared to document completion of IAROD cleanup actions in accordance with the applicable decision document and support waste site reclassification. The screening-level calculations presented in this section use the IAROD risk assessment models but differ from the calculations used in the CVPs and RSVPs to document the IAROD cleanups in four ways. These differences are summarized below. More detailed information on these differences is provided in Appendix D-4.

1. The analytes included in this calculation of risk for individual waste sites are those COPCs identified for each ROD area in Section 5.2 according to the protocol described in Section 3.2. In the IAROD cleanups, RAGs were only calculated for target analytes defined by Tri-Party concurrence based on process knowledge and analytical results from the LFIs and earlier interim actions.

2. Residual soil concentrations are calculated using cleanup verification sample data and the RCBRA protocols for calculating representative concentrations discussed in Section 3.4. The RCBRA protocols differ from those used in the IAROD cleanups. For chemicals, guidance on calculating the 95% upper confidence limit (UCL) under MTCA was followed in the IAROD cleanups. However, if 50% or more of a data set was nondetect, then the maximum chemical analytical results were used in lieu of a 95% UCL for a chemical.

For radionuclides, a 95% UCL was always calculated for the IAROD cleanups using a nonparametric method based on the "z" statistic.

3. Current chemical toxicity criteria are used in the RCBRA. Toxicity criteria from EPA and other federal or state sources (as cited in the EPA Regional Screening Level Table [EPA 2009]) were used to calculate current MTCA Method B values. Chemical toxicity criteria that were current as of the 2006 revision of MTCA were used for the IAROD cleanups.

4. Radionuclide cancer risk is evaluated relative to a cancer risk range of 1 x 10^{-6} to 1 x 10^{-4} in the RCBRA, and a threshold of 1 x 10^{-4} was used to calculate radionuclide risk-based screening levels in this section. Radionuclide RAGs were calculated based on a radiation dose threshold of 15 mrem/yr for the IAROD cleanups.

A simplified "sum of ratios" approach was used for the calculations in this section as described in the EPA Regional Screening Level User's Guide (http://www.epa.gov/reg3hwmd/risk/human/rb-concentration_table/usersguide.htm). This approach uses the ratio of soil representative concentrations and risk-based soil concentrations to calculate risks for individual chemicals or radionuclides. The risks from the individual analytes are summed to provide estimates of cumulative risks for chemicals and cumulative risks for radionuclides. The same approach is used for calculating the hazard quotients (HQs) for individual chemicals and cumulative HIs for noncancer effects.

The calculation of the risk-based concentrations used in this screening-level assessment is explained in detail in Section 7.5. The risk-based concentrations for radionuclides are based on the IAROD Rural Residential scenario. The radionuclide risk-based concentrations are calculated for present-day direct soil exposures using the RESRAD computer code and a target cancer risk of 1 x 10^{-4}.

The risk-based concentrations for chemicals are the current MTCA Method B values. As noted above, toxicity criteria from EPA sources (as cited in the EPA Regional Screening Level Table [EPA 2009]) were used to calculate current MTCA Method B values. The chemical risk-based concentrations are based on the MTCA Method B Unrestricted Use scenario with a target cancer risk of 1 x 10^{-6} or target HQ of 1. MTCA uses a target risk of 1 x 10^{-6} for individual chemicals and a target cumulative cancer risk of 1 x 10^{-5}.

The soil concentrations used in these calculations were the reasonable maximum exposure (RME) values and central tendency exposure (CTE) values described in Section 3.4 and provided in electronic format in Appendix C, Section C-3, "Representative Concentrations." The lists of COPCs for each ROD area that may contribute to the screening assessment results are shown in Table 5-26. Table 2-10 presents a summary of the RME cumulative cancer risk and HI calculations for the remediated waste sites grouped by ROD areas. Calculations showing RME and CTE risks and hazard quotients for individual chemicals and radionuclides for each waste site are included in Appendix D.

2.8.1 Residual Cancer Risks and Noncancer Hazards for Chemicals

As shown on Table 2-10, residual cumulative cancer risks from chemicals for most of the 156 remediated waste sites evaluated in the RCBRA are less than 1 x 10^{-5} using the IAROD residential scenario (i.e., MTCA Method B Unrestricted Use scenario). For some remediated waste sites, risks up to 3 x 10^{-5} were calculated with the main risk driver for these sites being arsenic. Although site RME arsenic concentrations exceed reference area concentrations and sometimes Hanford Site background 90th percentile values (DOE/RL-92-24, *Hanford Site Background: Part 1, Soil Background for Nonradioactive Analytes*), and consequently were included in the risk calculations, RME concentrations at all remediated waste sites are less than the residential IAROD cleanup level of 20 mg/kg for arsenic (i.e., RME concentrations range from 1.1 to 17.3 mg/kg). The 20 mg/kg cleanup level was established by Ecology as an unrestricted land use cleanup value; this value is adjusted for natural background in soil (http://apps.leg.wa.gov/WAC/default.aspx?cite=173-340-900). When residual cancer risks for chemicals were calculated without the contribution from arsenic, all 156 remediated waste sites have results less than 1 x 10^{-5}.

The noncancer HIs for chemicals do not exceed a threshold of 1.0 at remediated waste sites except for two sites in the 300 Area (316-1 had an HI of 1.4 and 316-5 had an HI of 1.2). For the 316-1 waste site, arsenic and mercury are the main contributors to the HI with HQs of 0.5 and 0.6, respectively. Without the contribution from arsenic, the HI at 316-1 is less than 1 (0.7). At the 316-5 waste site, total uranium is the main contributor to the HI with an HQ of 1.0. It is important to note that the 316-5 waste site was remediated to an industrial IAROD cleanup level. The RME for calculated total uranium at 316-5 is 242 mg/kg, which is essentially equal to the industrial IAROD cleanup level of 240 mg/kg.

2.8.2 Residual Cancer Risks for Radionuclides

As shown in Table 2-10, residual cumulative cancer risks from radionuclides for most remediated waste sites are less than 1 x 10^{-4} based on the IAROD Rural Residential scenario. There are 14 of the 156 remediated waste sites that were evaluated in the RCBRA that have risks greater than 1 x 10^{-4} (between 2 x 10^{-4} and 5 x 10^{-4}) using the IAROD Rural Residential scenario.

Eleven of the fourteen sites with risks greater than 1 x10^{-4} are in the 100 Areas: six sites are in the 100-B/C Area, two sites are in the 100-D/100-H Area, one site is in the 100-F/100-IU-2/100-IU-6 Area, and two sites are in the 100-K Area. For the 100 Area sites with risks greater than 1×10^{-4}, the risk drivers are europium-152, cesium-137, and strontium-90. The basis of the IAROD cleanup levels for radionuclides was a radiation dose of 15 mrem/yr. A radiation dose of 15 mrem/yr equates to approximately a 3×10^{-4} lifetime cancer risk (OSWER Directive 9200.4-18, *Establishment of Cleanup Levels for CERCLA Sites with Radioactive Contamination*). Therefore, although all of the sites were cleaned up to 15 mrem/yr, the residual levels of particular radionuclides at some sites may exceed 1×10^{-4} cancer risk.

For one of the three sites in the 300 Area with risks greater than 1×10^{-4} based on the IAROD Rural Residential scenario, the risk driver is cobalt-60 (the 316-1 waste site). For the other two sites in the 300 Area with risks greater than 1×10^{-4} based on the IAROD Rural Residential scenario, the risk drivers are uranium-235 and uranium-238 (the 316-2 and 316-5 waste sites). It is important to note that the IAROD cleanup levels for the 316-1, 316-2, and 316-5 waste sites in the 300 Area were based on future industrial land use. For the remediated waste site with cobalt-60 as a risk driver (316-1), the RME concentration of cobalt-60 is 1.67 pCi/g, which is well below the industrial IAROD cleanup level of 5.2 pCi/g. In addition, for the two sites with uranium isotopes as the risk drivers the RME concentrations are less than the industrial IAROD cleanup levels. The RME concentration for uranium-238 at the 316-2 waste site is 55 pCi/g, and the industrial IAROD cleanup level is 167 pCi/g. The RME concentrations for uranium-235 (12 pCi/g) and uranium-238 (80 pCi/g) at the 316-5 waste site are also well below the industrial IAROD cleanup levels (16 pCi/g for uranium-235 and 167 pCi/g for uranium-238). Therefore, although residual cumulative risks based on the IAROD Rural Residential scenario for three sites in the 300 Area are greater than 1×10^{-4}, the industrial IAROD cleanup levels have been achieved at these sites.

2.8.3 Uncertainties Associated with Screening-Level Cancer Risk and Noncancer Hazard Calculations

A key uncertainty related to these risk calculations for the IAROD exposure scenarios is how applicable the cleanup verification soil data at excavated and backfilled sites are for future human exposure. Residual contamination at backfilled sites is characterized by cleanup verification soil samples collected on the sidewalls and (if the excavation depth is <4.6 m [15 ft]) bottom of an excavation. After the verification data are accepted and the site is shown to meet the IAROD requirements, the site is reclassified and backfilled with clean fill material. Risks may be overestimated using cleanup verification data from excavated sites to estimate exposure concentrations in surface soil. This is of particular significance because the great majority of remediated waste sites with the highest levels of residual risk are sites that have been excavated and backfilled, so confirmation soil samples are not necessarily representative of soil on or near the surface of the waste site.

2.8.4 Summary of Residual Cancer Risks and Noncancer Hazards Using IAROD Scenarios

Based on these cancer risk and noncancer hazard calculations using the IAROD residential exposure scenarios for radionuclides and chemicals, the conditions at 128 of 139 remediated waste sites in the 100-B/C, 100-K, 100-D/100-H, 100-N, and 100-F/100-IU-2/100-IU-6 ROD areas following interim cleanup actions are protective of human health based on the following results:

- Cumulative cancer risks for chemicals (not including arsenic) are less than1×10^{-5}
- Hazard indices for chemicals (not including arsenic) are less than 1
- Cumulative cancer risks for radionuclides are less than 1×10^{-4}.

There are 11 of 139 remediated waste sites in the 100 Area that have cumulative cancer risks for radionuclides that exceed 1×10^{-4}. The cumulative cancer risks for radionuclides at these sites range from 2×10^{-4} to 5×10^{-4}. For these 11 sites, the risk drivers are europium-152, cesium-137, and strontium-90.

Residual conditions in the 300 Area meet the industrial IAROD cleanup levels. In addition, the 300 Area remediated waste sites that were evaluated in the RCBRA have cumulative cancer risks for chemicals (not including arsenic) that are less than 1×10^{-5}. The HIs are less than 1 based on the IAROD residential scenario for all 300 Area sites except for 316-5. At this site, the HI is slightly greater than 1 (1.2) and the main contributor is total uranium. The calculated total uranium RME soil concentration at this site is 242 mg/kg and the industrial IAROD cleanup level is 240 mg/kg. For radionuclides, there are three remediated waste sites that have cumulative cancer risks that exceed 1×10^{-4} based on the IAROD Rural Residential scenario. The risk drivers for these three sites are cobalt-60, uranium-235, and uranium-238. The concentrations of these three risk drivers meet the industrial IAROD cleanup levels.

2.8.5 Evaluation of Residual Cancer Risks and Hazards for Additional Exposure Scenarios

In the following sections of this volume, risks due to residual levels of radionuclides and chemicals at remediated waste sites and in multiple environmental media in the River Corridor are evaluated using recreational, occupational, and residential human health exposure scenarios. The risk evaluation is based on a range of exposure assumptions and data sets (Section 4.0, Broad Area Exposures to COPCs in Soil; Section 5.0, Local Area Exposures to COPCs in Soil; and Section 6.0, Exposures to COPCs in Groundwater) to determine if residual COPC concentrations pose a potential risk to human health under various exposure scenarios. In Section 7.0, the results of the evaluations in Sections 4.0, 5.0, and 6.0 are summarized, and a range of soil preliminary remediation goals are presented that are protective of human health based on the exposure scenarios evaluated in this volume.

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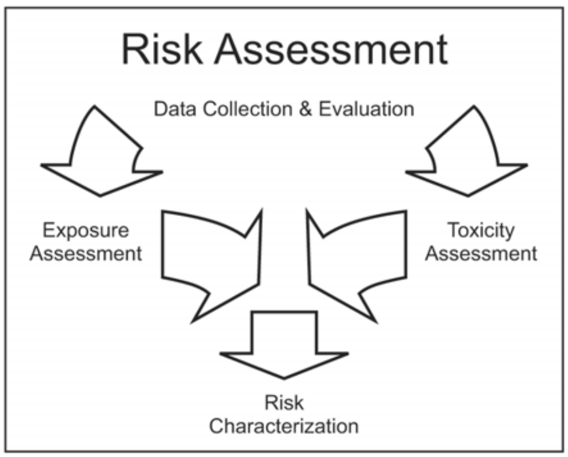
3.0 HUMAN HEALTH RISK ASSESSMENT APPROACH

This section provides a description of the risk assessment methods used in the RCBRA, with additional detailed aspects of the risk assessment methodology provided in technical appendices to this report. The overall framework for performing risk assessment in the River Corridor is discussed in Section 3.1. Subsequent sections address specific components of a baseline human health risk assessment (HHRA).

The methodology of a baseline HHRA is generally structured in four steps according to *Risk Assessment Guidance for Superfund, Volume I: Human Health Evaluation Manual, Part A (RAGS)* (EPA/540/1-89/002). These steps are (1) data collection and evaluation, (2) contaminant exposure assessment, (3) toxicity assessment, and (4) risk characterization. The components of each of the four steps, and where in the HHRA they are addressed, are briefly described below.



Data Collection and Evaluation. Data collection and evaluation includes a summary of the relevant historical operations and releases, a discussion of the sampling and analysis conducted, presentation of data processing protocols, and methods for background comparisons and identification of contaminants of potential concern (COPCs). Historical operations and releases were discussed in Section 2.0. A summary of sampling and analysis relevant to the data used in the HHRA, and methods relating to data evaluation and identification of COPCs is provided in Section 3.3. Additional details specific to the broad-area (Section 4.0), local-area (Section 5.0), and groundwater (Section 6.0) risk assessments are provided in subsequent sections of the HHRA.

Exposure Assessment. The exposure assessment focuses on identifying how much of a chemical is present in an environmental medium, determining who might be exposed and how, and quantifying the rate of exposure. The conceptual site model (CSM) describing the land-use scenarios evaluated in the HHRA and potentially complete exposure pathways for receptors within each scenario provides the context for the quantitative aspects of the exposure assessment and is described in Section 3.3. Methods for calculating representative concentrations in sampled environmental media are discussed in Sections 3.4.1 and 3.4.2. Methods for estimating exposure point concentrations in other media and for estimating concentrations in environmental media at future dates are discussed in Section 3.4.3. Then, methods for calculating the rate of chemical and radionuclide intake into the body via each potentially complete exposure pathway are presented in Section 3.4.4. Additional details on exposure assessment specific to the broad-area (Section 4.0), local-area (Section 5.0), and groundwater (Section 6.0) risk assessments are provided in subsequent sections of the HHRA.

Toxicity Assessment. The different types of potential adverse health effects associated with chemical and radionuclide exposure are discussed in Section 3.5. For most chemicals, the higher the level of exposure, the more likely it is that the chemical will cause a toxic effect. Because every chemical has the potential to cause a toxic effect at *some* level, the toxicity assessment is used together with the exposure assessment in order to assess the potential for human health risks.

Risk Characterization. The risk characterization is the part of an HHRA in which estimates of potential health effects and radiation dose for each exposure scenario are presented. The risk characterization also clarifies which COPCs and exposure pathways are associated with these risks. Uncertainty related to the various assumptions and inputs used in the HHRA is also assessed in the risk characterization step to qualify the risk assessment results. Potential human health risks for nonradiological COPCs are characterized for two health endpoints: (1) the risk of potential carcinogenic (cancer-related) health effects, and (2) the potential for noncancer health effects (systemic hazards). For radiological COPCs, radiation dose and carcinogenic risk are calculated. Based on EPA guidance, cancer risk is summed across chemicals and radionuclides. The methods employed for the risk characterization are discussed in Section 3.6. Presentation of the risk assessment results, the uncertainty analysis, identification of data gaps, and conclusions and recommendations are the subjects of Sections 4.0 through 7.0 of the HHRA.

Files of the representative concentrations in sampled media that were used to calculate exposure point concentrations in the various exposure media are provided in electronic format as an attachment to Appendix C-3. The exposure point concentrations, and the results of the risk assessment calculations, are provided in electronic attachments to Appendix D-5. The representative concentrations, exposure point concentrations, and risk assessment results are also accessible via a web-based interface. To access data through the RCBRA Guided Interactive Statistical Decision Tools (GiSdT) interface, you must have a username and password. These are available by sending an email request to rcbra@gisdt.org.

3.1 RIVER CORRIDOR ASSESSMENT FRAMEWORK

This section presents an overview of the assessment framework for the River Corridor. The framework reflects consideration of historical information and current conditions within the upland, riparian, and near-shore environments described in Section 2.0.

3.1.1 Assessment of Risk for Operational Areas in the Upland Environment

Part way through the cleanup process for the River Corridor, the current setting for the operational areas in the upland environment includes a mix of waste sites that have been remediated in accordance with interim action record of decisions (IARODs) and yet-to-be remediated waste sites that have been identified for action under the IARODs. These waste sites were characterized through qualitative risk assessments (QRAs) and limited field investigations (LFIs) as previously described in Section 2.6. The QRAs presented pre-remediation risk levels and established the basis for action at sites scheduled for remedial action.

3.1.1.1 Yet-To-Be Remediated Waste Sites. Unacceptable risks are present in the River Corridor at waste sites that are identified in the IARODs but have yet to be remediated. Table 3-1 shows the human health risks calculated in the QRA reports for several yet-to-be remediated waste sites based on pre-remediation conditions. The QRA risks were based on a Frequent-Use scenario that models exposure to soil based on residential use. The EPA uses 1×10^{-6} to 1×10^{-4} cancer risks as a target risk range within which they strive to manage risks as a part of a *Comprehensive Environmental Response, Compensation, and Liability Act of 1980* (CERCLA) cleanup. Generally, remedial action under CERCLA is considered warranted when cancer risks are greater than 1×10^{-4} (OSWER 9355.0-30, "Role of the Baseline Risk Assessment in Superfund Remedy Selection Decisions"). The table also shows the human health cancer risk calculated from the same data set using a suite of exposure scenarios evaluated in the RCBRA HHRA.

The exposure scenarios shown in Table 3-1 range from low-intensity use (casual recreational user) to higher intensities of exposure (farmer and residential scenarios). The QRA Frequent-Use scenario is most similar to the RCBRA Subsistence Farmer scenario, except that exposure pathways related to home-raised foods were not evaluated in the QRAs. Cancer risks calculated for the QRA Frequent-Use scenario are very similar to risks calculated for the RCBRA Subsistence Farmer scenario, which is likely because risk from external radiation dominates the calculations when using pre-remediation site soil concentrations at these waste sites.

The original determination of the presence of unacceptable risk and basis for action at yet-to-be remediated waste sites is supported by the field experiences and information gathered through implementation of the observation approach based soil cleanup actions in the River Corridor over the past 13 years. The risk associated with the quantity and type of waste that has been excavated confirms unacceptable risks that appropriately drive CERCLA cleanup actions. Examples of types of waste and contaminants removed from several waste sites are shown in Table 3-2. This information supports the qualitative risk conclusions and basis for action for several waste site types.

Waste streams summarized in Table 3-2 represent sources of both chronic and acute risk to human health and the environment. This information supports the conclusion that there is currently unacceptable risk at yet-to-be remediated waste sites in the River Corridor. In parallel with establishing final cleanup actions for the River Corridor through the remedial investigation/feasibility study (RI/FS) process, U.S. Department of Energy (DOE) is committed to continuing cleanup actions at these sites according to goals and objectives of the IARODs. Consequently, the risks associated with the yet-to-be remediated waste sites are not a focus of the remainder of this report, but will be discussed again in the context of recommendations and conclusions (Section 7.0).

3.1.1.2 Remediated Waste Sites. Evaluation of human health risks from residual contamination in the upland environment where cleanup actions had been completed in accordance with the IARODs is a major focus area of this risk assessment report. In Section 4.0 of this report, surface soil samples from within an area of approximately 1 ha (2.47 ac) associated with selected remediated waste sites are used to assess human health risks from exposure activities that occur over a large region. Human health risk calculations and conclusions for exposure scenarios that take place on smaller spatial scales are presented in Section 5.0 of this report for 156 individual waste sites remediated through the year 2005. The assessments in Sections 4.0 and 5.0 are intended to evaluate whether protectiveness was achieved for various exposure scenarios where soil cleanup had been performed under the IARODs. Risk results and conclusions for these waste sites will provide decision makers with information on the protectiveness of the current IAROD cleanup goals and implementation process. Evaluation of the current IAROD cleanup goals in regard to protection of groundwater from soil leaching will be addressed in RI reports.

3.1.2 Assessment of Risk for Nonoperational Areas in the Upland Environment

Open undisturbed areas comprise more than 492 km^2 (190 mi^2), or roughly 87%, of the 570-km^2 (220-mi^2) area of the River Corridor. Evaluating these large amounts of open undisturbed spaces in the upland environment where no Hanford Site operations occurred presents unique challenges. Nonoperational areas are evaluated through a combination of historical information, results from environmental monitoring programs and studies, and the waste site identification processes (including the ongoing orphan sites evaluation process).

The historical and current waste site identification processes provide confidence that the areas of greatest risk within the River Corridor are identified and addressed in accordance with established cleanup objectives. The open and often undisturbed areas between the waste sites are generally considered to be undifferentiated from the present day conditions at other similar shrub-steppe environments in the region. Assumptions about the risk associated with these areas are supported by the results from various monitoring programs, best management practice work control activities, and reporting requirements performed by DOE as part of its site management responsibilities and will be discussed in RI reports. When observations or results from various monitoring activities indicate that this is not the case, there are processes in place as described previously to document these discoveries as new waste sites and address them in accordance with existing cleanup objectives.

Various surveillance and maintenance activities and monitoring programs conducted at the Hanford Site provide information that may be useful for evaluation of the nonoperational areas. Examples of some of these activities and programs included the following:

- **Environmental Monitoring.** DOE has completed routine radiological surveys of the river shore (PNL-3127, *Radiological Survey of Exposed Shorelines and Islands of the Columbia River Between Vernita and the Snake River Confluence*), as well as sampling of the riverbank springs and sediment (DOE/RL-92-12, *Sampling and Analysis of 100 Areas Springs*;

PNNL-13230, *Hanford Site Environmental Report for Calendar Year 1999 [Including Some Historical and Early 2000 Information]*; WHC-SD-EN-TI-198, *100 Area Columbia River Sediment Sampling*). The Hanford Site annual environmental monitoring reports also document and evaluate surveillance sampling of many media on and off the Hanford Site (e.g., vegetation, terrestrial and aquatic wildlife, air, soil, and water) to quantify potential contaminant levels.

- **Aerial Surveys.** Aerial radiological surveys were completed (EGG-10617-1062, *An Aerial Radiological Survey of the Hanford Site and Surrounding Area*) to define areas of manmade radioactive contamination. The EGG-10617-1062 survey covered the Hanford Site and the banks of the Columbia River downriver to McNary Dam. The radiation levels over more than 95% of the site were reported to be due to normal levels of background radiation. Areas of elevated radionuclide activity outside of operational areas have been investigated and are identified in Waste Information Data System (WIDS). Several slough areas along the Columbia River also showed elevated radioactivity; these areas were sampled and the radionuclide content shown to be only slightly above background (WHC-SD-EN-TI-198).

- **Air Emissions Evaluations.** In 2005, an evaluation of the releases on the Hanford Site from air emissions stacks located in the 100 and 300 Areas was made (DOE/RL-2005-49, *RCBRA Stack Air Emissions Deposition Scoping Document*) using previous background soil sampling results, radiological surveys, and an evaluation of the emissions (radionuclides and metals) emitted. The report concluded there were no areas of elevated radioactivity or metals in the 100, 300, or associated 600 Areas due to aerial deposition, other than those discrete areas already identified as waste sites in WIDS. This information was considered, along with soil sampling results, to evaluate the sites selected as reference sites for the RCBRA.

- **Environmental Radiation Monitoring and Assessment Program.** The Washington State Department of Health (WDOH), Division of Environmental Health (DOH) has an oversight program that independently verifies the quality of the DOE monitoring programs at the Hanford Site. The DOH performs this oversight by conducting split, collocated, and independent sampling at locations having the potential to release radionuclides to the environment or any location which may be impacted by such releases. The DOH uses the oversight data to assess impacts to the public and to address public concerns related to radiation at the Hanford Site. The DOH publishes an annual Hanford Site environmental oversight program summary report.

Assessment of risks in the nonoperational areas of the upland environment is not a focus of the remainder of this report, but will be discussed again in the context of recommendations and conclusions (Section 7.0). In addition, a more in-depth evaluation of these areas that are not anticipated to have been affected by Hanford Site operations will be conducted as part of the remedial investigation activities for the River Corridor. The evaluation of the nonoperational areas will include a data compilation effort followed by evaluation of the compiled data to determine what conclusions may be drawn and what potential data gaps exist. The results of the data compilation activities and analysis will be summarized in the remedial investigation reports for the six River Corridor record of decision (ROD) decision units. If there are additional data needs to address identified data gaps, a data quality objectives (DQO) process will be conducted to develop a sample design and a sampling and analysis plan (SAP) will be written to guide field collection and analysis.

3.1.3 Assessment of Risk in the Riparian Environment

The riparian environment is a nonoperational portion of the Hanford Site that is potentially affected by Hanford contaminants through transport of releases from the operational areas. Evaluation of human health risks in the riparian environment where releases from upland operational areas have potentially affected the adjacent environmental media along the river shoreline is a component of the broad-area risk assessment presented in Section 4.0 of this report.

Human health risk is evaluated based on sample results from nonoperational areas that may be affected by transport of contaminants from operational areas in order to study areas with high potential for residual contamination. Risk results and conclusions for these potentially affected riparian areas provide decision makers with information on the type of risks that would be anticipated following continued cleanup of the River Corridor.

One of the river islands, 100-D-Island, or D-Island, represents a special area within the riparian environment. The 100-D Island is located in the Columbia River approximately 250 m (820 ft) offshore of the 100-D Area of the Hanford Site. The 100-D Island is a low-lying island that becomes partially submerged due to daily fluctuations in the river level. Most of the 10-ha (25-ac) island is covered by a layer of sediments predominated by cobbles 2.5 to 15 cm (1 to 6 in.) in diameter, and coarse sandy gravel. A coarse sandy beach is located on the downstream end of the island. Vegetation on the higher elevations of the island consists of grass, small bushes, and a few small trees. The following section provides a summary of available information on 100-D Island as well as an evaluation of risk.

3.1.3.1 100-D Island Investigation. This section summarizes two radiological surveys at the 100-D Island. The first investigation was conducted in 1992 by Westinghouse Hanford Company. The second investigation was conducted in 1995 by the WDOH. In addition, descriptions of other investigations of the 100-D Island can be found in the *Screening Assessments and Requirements for a Comprehensive Assessment, Columbia River Comprehensive Impact Assessment* (DOE/RL-96-16).

Purpose and Scope. The purpose of the 100-D Island radiological surveys was to determine the extent of detectable radiological contamination on the island and to attempt to determine the source of the contamination. The surveys conducted by Westinghouse Hanford Company involved approximately 5 ha (12.5 ac) of reconnaissance across the upstream portion of 100-D Island. The WDOH evaluation looked at particle density of speck contamination on downstream portions of the island.

Summary of Investigation Data. From April 12 to 18, 1992, a series of radiological surveys were performed at the upstream half of 100-D Island. Radiological surveys were completed using the Ultrasonic Ranging and Data System and conducted using both a digital count rate meter with a sodium iodide detector reporting in counts per minute and a dose rate meter reporting in microroentgens per hour (µR/h). Five ha (12.5 ac) of 100-D Island were surveyed with Ultrasonic Ranging and Data System equipment. The radiological survey indicated low levels of cobalt-60 contamination on the island's surface. Results of the radiological survey are published in *100-D Island USRADS Radiological Surveys Preliminary Report – Phase II* (BHI-00134). The results of the upstream survey indicate that cobalt-60 contamination on the upper end of 100-D Island does not pose a significant human health risk.

An additional radiological survey was conducted by the WDOH on a sandy downstream section of 100-D Island in 1995. A walking survey of exposed island shoreline was conducted when river levels were at their lowest. All surveys were performed using µR meters suspended approximately 2 to 4 cm (approximately 0.78 to 1.5 in.) above the ground. Background exposure rates along the river shore ranging from 7 to 8 µR/hr were measured. A pressurized ion chamber was used to measure variations in background at three island locations. Ambient background measurements recorded by the pressurized ion chamber varied from 8.8 to 9.5 µR/hr.

Conclusions reached by the WDOH were consistent with findings of previous studies. The ambient gamma radiation level measured at several island locations was near background. Burial depth and contact radiation levels of excavated discrete particles were within the range of values previously reported. No particles were found on the sandy downstream section of the island. The number of particles per unit volume was 1.3×10^{-1} particles per cubic meter. Three discrete particles were detected and contact measurements ranged from 85 to 2,000 µR/hr, or about 10 to 200 times greater than background levels.

The WDOH survey (WDOH/ERS-96-1101, *100-D Island Radiological Survey*) concluded that radiological hazards and potential health effects from exposure to cobalt-60 particles on the downstream section of 100-D Island are consistent with evaluations documented in previous correspondence and reports. The net results from the survey support a conclusion that cobalt-60-contaminated particles in downstream 100-D Island sediments do not pose significant human health risks. However, the WDOH recommended removal of such particles if found during the course of cleanup actions. Radiological postings are maintained in a manner consistent with Hanford Site contractor protocols because the island is owned by DOE.

3.1.4 Assessment of Risk in the Near-Shore Environment

The near-shore environment is also a nonoperational portion of the Hanford Site that is potentially affected by Hanford contaminants through transport of releases from the operational areas. Similar to the riparian environment, evaluation of human health risks in the near-shore environment will focus primarily on locations where releases from upland operational areas have potentially affected the adjacent environmental media along the river shoreline. Many of the near-shore sampling locations are paired with sampling locations in the adjacent riparian zone.

Near-shore sampling sites are located where contaminated groundwater plumes enter the Columbia River along the shoreline.

Human health risk is evaluated at nonoperational areas that may be affected by transport of contaminants from operational areas in order to study areas with high potential for residual contamination. Risk results and conclusions for these potentially affected areas provide decision makers with information on the type of risks that would be anticipated following continued cleanup of the River Corridor.

Effluent pipelines from the historical reactors are a specific source of potential risk within the near-shore environment. The following section will provide a summary of available information on the effluent pipelines as well as an evaluation of risk.

3.1.4.1 River Effluent Pipelines. Between 1943 and 1988, water that was used during fuel production to cool the reactors was discharged to pipelines that extended into the Columbia River. The liquid, called effluent, entered the river through "outfall" structures located in each of the 100 Areas of the Hanford Site. Release of cooling water through the river effluent pipelines ended when the associated reactors and facilities were shut down beginning in the late 1960s until the late 1980s. Today, the inactive effluent pipelines remain in their original location on or beneath the Columbia River channel bottom. The effluent pipelines comprise 7 waste sites that include 15 separate pipelines. The river effluent pipeline waste sites in the 100-B/C, 100-D, 100-H, 100-F, 100-K, and 100-N Areas are listed in Table 3-3.

Purpose and Scope. Most of the river effluent pipelines are known or suspected to contain low levels of residual contamination from past reactor operations. The river pipelines are described in Table 3-4. Two past characterization efforts obtained samples of the river effluent pipelines from the 105-B, 105-C, 105-D, 105-DR, and 105-F Reactors. Characterization data collected during the river pipelines evaluations were used to evaluate risks from contaminants within the pipelines and to propose remedial action alternatives, such as pipeline removal.

Effluent pipelines from the 105-H Reactor could not be sampled because of high river flow velocities and cold water temperatures. Effluent pipelines at the 100-KE Reactor, 100-KW Reactor, 105-N Reactor, and the Hanford Generating Plant (HGP) also were not sampled. These effluent pipelines were used for discharge of nonradioactive effluent in accordance with a National Pollutant Discharge Elimination System (NPDES) permit.

Summary of Investigation Data. In 1984, the *River Discharge Lines Characterization Report* (UNI-3262) discussed samples of scale (flakes of mostly rust) from the interior surfaces and enclosed sediment of the effluent pipelines from the 105-C, 105-DR, and 105-F Reactors. The pipelines were also visually inspected underwater by a diver, and their positions and physical conditions were assessed. Samples of scale and sediment were analyzed for radionuclides. The major radionuclides detected included cobalt-60, cesium-137, europium-152, europium-154, and europium-155. Radionuclide concentrations were greater in the scale than in the sediment. Direct beta-gamma radiation measurements were also obtained for interior and exterior pipe surfaces. The dose rates measured for direct contact with the interior of the pipe surfaces were low at less than 1 mrem/hr, and readings on the exterior were lower than the instrument's detection capability.

In 1994, a comprehensive geophysical survey (WHC-SD-EN-TI-278, *Columbia River Effluent Pipeline Survey*) located and mapped the reactor effluent pipelines. The study relied mainly on remote sensing geophysical techniques, including navigation and echo sounding, side-scanning radar, sub-bottom profiling, seismic reflection profiling, and ground-penetrating radar. The results indicated that the pipelines have neither broken loose nor moved from their original locations. However, portions of some pipelines are no longer buried and have been exposed. Exposed pipe sections were believed to be associated with areas of turbulent flow conditions at the river bottom.

In 1995, pipe scale and sediment from the interior of the effluent pipelines from the 100-B and 100-D Areas were sampled and physically characterized using a robotic transporter (BHI-00538, *100 Area River Effluent Pipelines Characterization Report*). Analytical data from these two effluent pipelines were intended to complement the 1984 radionuclide data (UNI-3262) and were expected to represent "worst-case" conditions with respect to radiological contamination. This assumption was based on the long years of pipeline service and the volume of effluent known to have been discharged from the 105-B and 105-D/DR Reactors. The samples taken in 1995 were analyzed for a larger number of radionuclides than in the 1984 study and were also analyzed for metals and total organic carbon.

In most cases, when the results of all radionuclide analyses are decayed to 2005, the concentrations of the samples taken in 1995 are lower than 1984 concentrations. Most metals were at concentrations below the analytical detection limits. However, the concentrations of total chromium and mercury were above detection limits; total chromium detections were over 1,000 ppm in the scale of some samples.

The analytical results from the 1984 and 1995 effluent pipeline characterization studies at the 105-B, 105-C, 105-D/DR, and 105-F Reactors may reasonably be extrapolated to effluent pipelines at the 100-H and 100-K Areas because operations among these reactors were similar. However, operating histories for effluent pipelines 100-N-77 or 100-N-80 suggest that contamination would only be found at negligible levels. Reactor cooling operations at the 100-N Area differed significantly from the other 100 Area reactor areas in that the 105-N Reactor used a secondary system intended to keep cooling water from becoming contaminated. As a result, the 100-N-77 effluent pipeline primarily discharged raw river water that was used to remove heat from the secondary cooling system at the 105-N Reactor. It also provided a disposal method, although only on an emergency basis, for primary cooling water and fuel storage basin water, that were more likely to be contaminated. Effluent in the 100-N-77 effluent pipeline would have normally contained zero to very low levels of radioactive fission products (DOE/RL-95-111, *Corrective Measures Study for the 100-NR-1 and 100-NR-2 Operable Units*).

The 100-N-80 effluent pipeline served the same purpose as 100-N-77, but serviced only the HGP, a power generating facility at the 100-N Reactor area. Effluent in the 100-N-80 HGP pipeline first passed through the 1908-NE HGP outfall structure. Analytical results for water and sediment samples, combined with radiological survey data and process knowledge, show that the 1908-NE HGP outfall is an uncontaminated structure that currently meets required cleanup standards. As a result, the Tri-Parties selected "continued institutional control" as the interim action for this outfall structure under the *Record of Decision for the 100-NR-1 and 100-NR-2 Operable Units, Hanford Site, Benton County, Washington* (EPA 1999), and no removal action is required (Energy Northwest 2004, *Cleanup Verification Package for the Hanford Generating Plant 100-N-4 Tile Field (SWMU #5); 100-N-1 Settling Pond (SWMU#6); 1908-NE Outfall (SWMU #7); 1716-NE Maintenance Garage (SWMU #8) and 100-N-52 Underground Storage Tank; 100-N-3 Maintenance Garage French Drain, 100-N-41 Gate House Septic Tank, and 100-N-45 Septic Tank (SWMU #9); 100-N-5 Bone Yard (SWMU #10); and 100-N-46 Underground Storage Tank, HGP-CVP-SWMUs 5, 6, 7, 8, 9, & 10*). The analytical results from the 1908-NE HGP outfall may reasonably be extrapolated to the 100-N-80 HGP effluent pipeline because both sites were exposed to the same effluent during operations.

Risk Evaluation for River Effluent Pipelines. Evaluations of human health and ecological risk have been performed for the river effluent pipelines as they are today, located on or beneath the river channel bottom, and for a scenario in which a pipeline section breaks away from the main pipeline and is washed onto the shore of the river. Both the 1996 risk assessment effort (BHI-00538) and the 1998 risk assessment effort (BHI-01141, *100 Area River Effluent Pipelines Risk Assessment*) relied on data collected from the 1984 and 1995 characterization work. The evaluation of human health and ecological risk performed in 1998 (BHI-01141), concluded that the concentrations of chromium and mercury in the scale and sediment within the pipelines pose minimal ecological risk because they have been in contact with river water without dissolving since the reactors were shut down in 1971. Based on the results of the 1998 risk evaluation of the pipelines under current conditions (in the river), there were no unacceptable risks, and therefore no requirement under CERCLA to remediate the river effluent pipelines. However, the risk evaluation did determine that should portions of the river pipelines become dislodged and wash ashore, there may be elevated human health risk.

Based on the conclusions from this evaluation, the river effluent pipelines are not a focus of the remainder of this risk assessment report. However, they will be discussed again in the context of recommendations and conclusions (Section 7.0).

3.2 METHODOLOGY FOR DATA EVALUATION

Data evaluation for analytical results used in the broad-area, local-area, and groundwater risk assessments are presented in the results sections for each assessment. There are components of data analysis that are common to all HHRAs and are summarized below. A summary of the analytical data used in the HHRA is provided in Section 3.2.1. The contaminant refinement process identifies COPCs based on background and reference data statistical evaluations, as described in Chapter 5 of EPA/540/1-89/002. The methodology for identifying COPCs is presented in Section 3.2.2.

3.2.1 Summary of Sampling and Analysis

For the purposes of completing an HHRA, it is important to collect and use the appropriate types of data, including pertinent existing data. This section summarizes the various datasets used in the assessment and presents the following discussions:

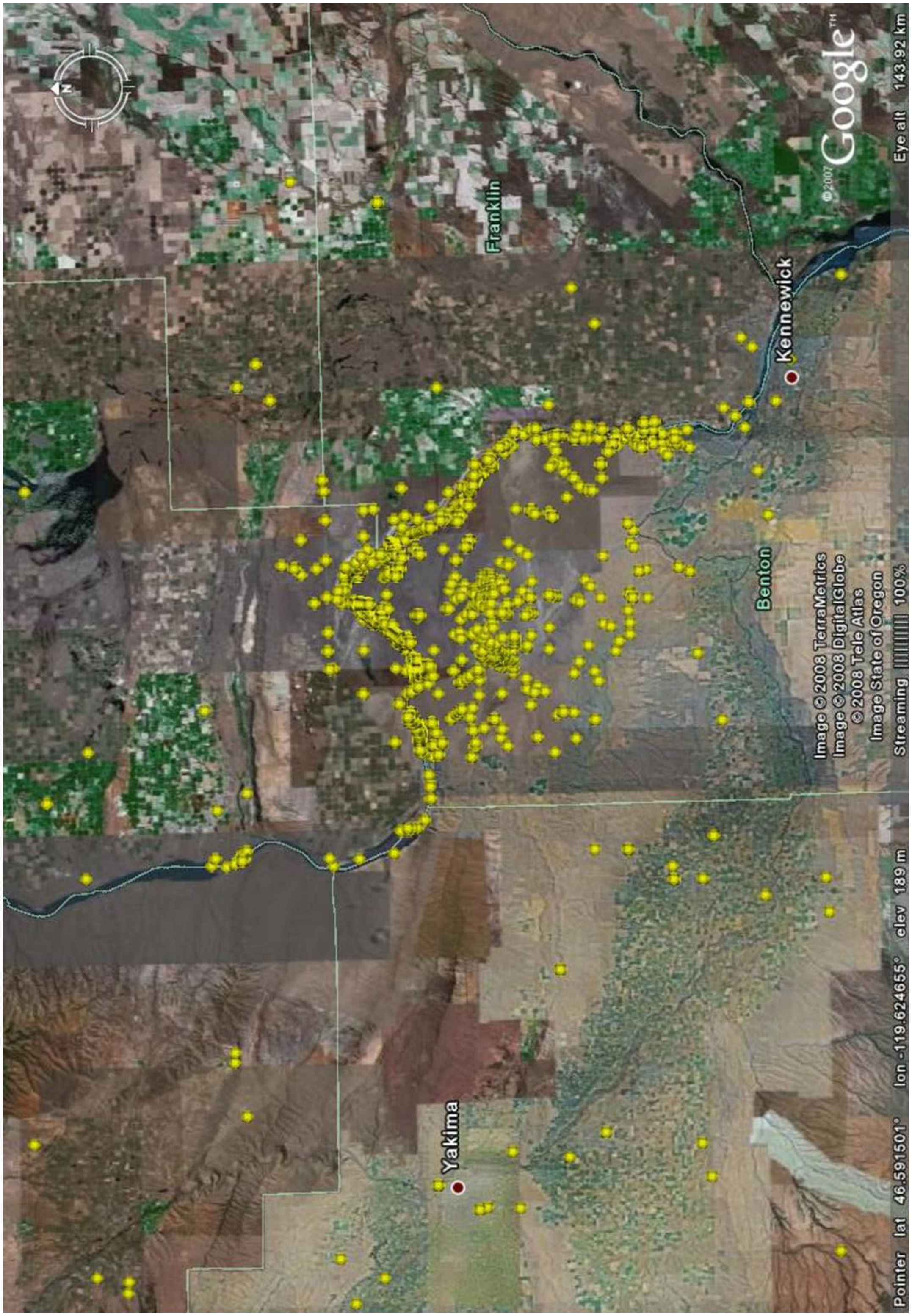
- A discussion of the background and reference site data used in the HHRA
- A discussion of the data collected by the RCBRA project
- A discussion of the waste site cleanup verification data used in the HHRA
- A discussion of the groundwater data used in the HHRA
- A discussion of other data used in the HHRA, including data related to the Surface Environmental Surveillance Program (SESP) and special studies.

The data evaluated in this report are available electronically through a web-based interface. The GiSdT interface for the RCBRA was developed as a repository and interface for analytical data used in this report. The data repository is located on the Internet at http://rcbra.gisdt.org/. The sample collection locations of the RCBRA data used in the assessment can also be viewed on an interactive map accessed at the web site. To access data through the RCBRA GiSdT interface, one must have a username and password. These are available by sending an email request to rcbra@gisdt.org. Figure 3-1 provides an example of the sample location information available on the interactive map, including background sample locations outside the Hanford Site.

Table 3-5 presents a summary of the types of analytical results for different environmental media for each of the data sources in the project database. More detailed information on the data used in the broad-area, local-area, and groundwater risk assessments is tabulated in Sections 4.0, 5.0, and 6.0, respectively.

3.2.1.1 Background and Reference Site Data. Fundamental to analysis of site-related risks in the RCBRA is establishing whether analyte concentrations in environmental media have been impacted by site operations. Background data and data from reference sites were used for this purpose. Calculated health risks may be significant even at naturally occurring levels. However, it is the incremental risk above background levels that is of primary concern in a risk assessment to determine the effects of contaminated media and to support remedial action decisions. A summary of the reference site and background data used in the RCBRA is provided in this section. Detailed information on reference sites and background is also provided in Appendix B.

Figure 3-1. Example of Interactive Map Showing Sample Locations. (http://rcbra.gisdt.org/)



Background locations are defined by the EPA as those not influenced by the releases from a contaminated site (EPA/540/R-01/003, *Guidance for Comparing Background and Chemical Concentrations in Soil for CERCLA Sites*). The concentrations of substances at background locations are relevant to this assessment and to remedial action decisions because CERCLA does not typically require cleanup to concentrations below natural levels or those caused by unrelated, widespread human activity (anthropogenic).

In addition, *Natural Background Soil Metals Concentrations in Washington State* (Ecology 1994) gives the following examples of background:

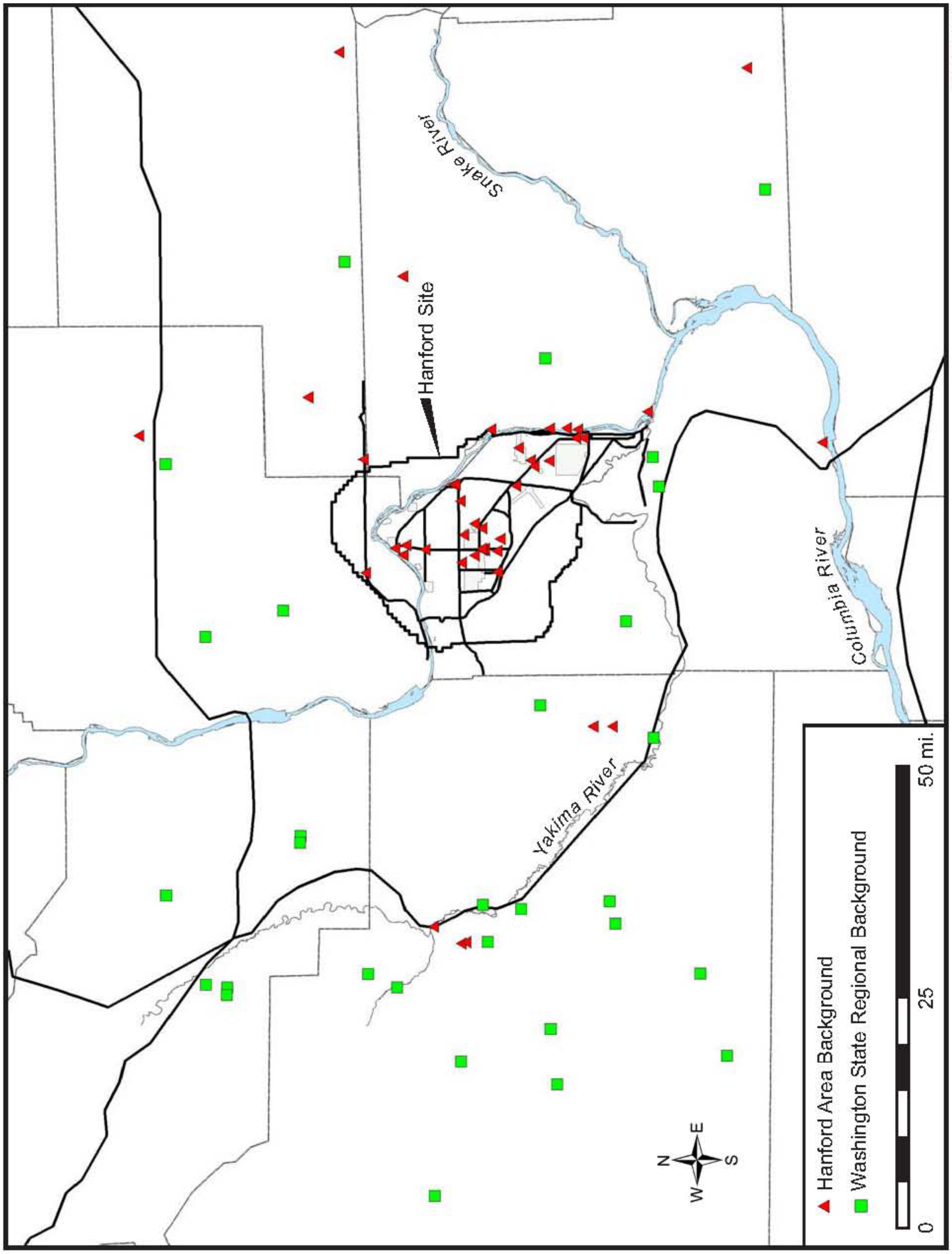
- Metals and radionuclides that occur in the bedrock, sediment, and soils due to the geologic processes that formed them (naturally occurring levels)

- Low concentrations of some organic compounds, such as polychlorinated biphenyls (PCBs) and radionuclides, present due to global distribution of these substances (anthropogenic background). These concentrations of hazardous substances are consistently present in the environment in the vicinity of a site as the result of human activities unrelated to releases from that site.

The outcome of the Hanford Site background characterization studies for metals, other inorganic analytes, and radionuclides at the Hanford Site were documented in three reports (DOE/RL-92-24, *Hanford Site Background: Part 1, Soil Background for Nonradioactive Analytes*; DOE/RL-95-55, *Hanford Site Background: Evaluation of Existing Soil Radionuclide Data*; DOE/RL-96-12, *Hanford Site Background: Part 2, Soil Background for Radionuclides*). The background studies included a DQO process following EPA guidance and approved SAPs (DOE/RL-92-24). The results of the background studies are used by DOE and its contractors in Hanford Site cleanup, as well as by federal and state regulatory agencies.

Washington State background samples cover the entire state and provide concentrations for 13 inorganic chemicals (Ecology 1994) (i.e., aluminum, arsenic, beryllium, cadmium, chromium, copper, iron, lead, manganese, mercury, nickel, selenium, and zinc); samples collected from locations in the area around the Hanford Site were termed "regional background." Figure 3-2 shows the locations of the Hanford Area and Washington State Regional background samples.

Figure 3-2. Hanford Area and Washington State Regional Background Sample Locations.



To help facilitate characterization of risks due to site contamination, data from the site are normally compared to data from similar areas not impacted by contamination. The EPA (EPA/540/F-94/012, *Using Toxicity Tests in Ecological Risk Assessment*) describes a reference site as a location that has similar physical, chemical, geological, and biological characteristics as the Superfund site, preferably nearby, that is either least affected or altogether unaffected by the site contamination. The EPA further states (EPA/540/F-94/012):

> The investigator should try to locate the reference site as close as possible to the Superfund site so that the reference site will accurately reflect the site's conditions. Yet the reference site should lie at a great enough distance from the Superfund site to be unaffected by site contamination.

The proximity of a reference site to the Superfund site must be close enough so that characteristics such as geology, climate, and habitat remain comparable to the site conditions. Differences other than the presence or absence of contamination between the Superfund site and reference sampling locations, such as different soil chemistry or vegetation, is likely to confound the interpretation of results. Reference sites were selected for the RCBRA based on both similarity of site characteristics and absence of contamination.

The ecological risk assessment (ERA) evaluates the potential for ecological risk in three environments: upland, riparian, and near-shore aquatic. Multiple reference locations were selected in the upland, riparian, and near-shore areas to cover the range of substrate types and habitat conditions. Reference site locations in the upland environment are shown in Figure 3-3. Reference site locations in the riparian and near-shore aquatic environments are shown in Figure 3-4.

Although the reference sites were primarily selected to support the ERA, these sites are equally applicable for differentiating site and background contaminant concentrations for the HHRA. Unlike the Hanford Site and Washington State Regional background samples, which have limited data for radionuclides and essentially no information for organic chemicals, reference area samples were analyzed for all constituents evaluated in this report. The EPA guidance related to human health risk assessment discusses reference sites in the context of collecting samples to represent background conditions (EPA/540/R-01/003). Several of the attributes of a reference site selected for an ERA (e.g., commonality with site physical, chemical, and geological characteristics) also render reference sites appropriate for establishing background concentrations for an HHRA. A summary of the reference sites selected is provided below. Additional information relating to the selection of reference sites for each environment is provided in Volume I of the RCBRA.

Figure 3-3. Location of Upland Reference Sites.

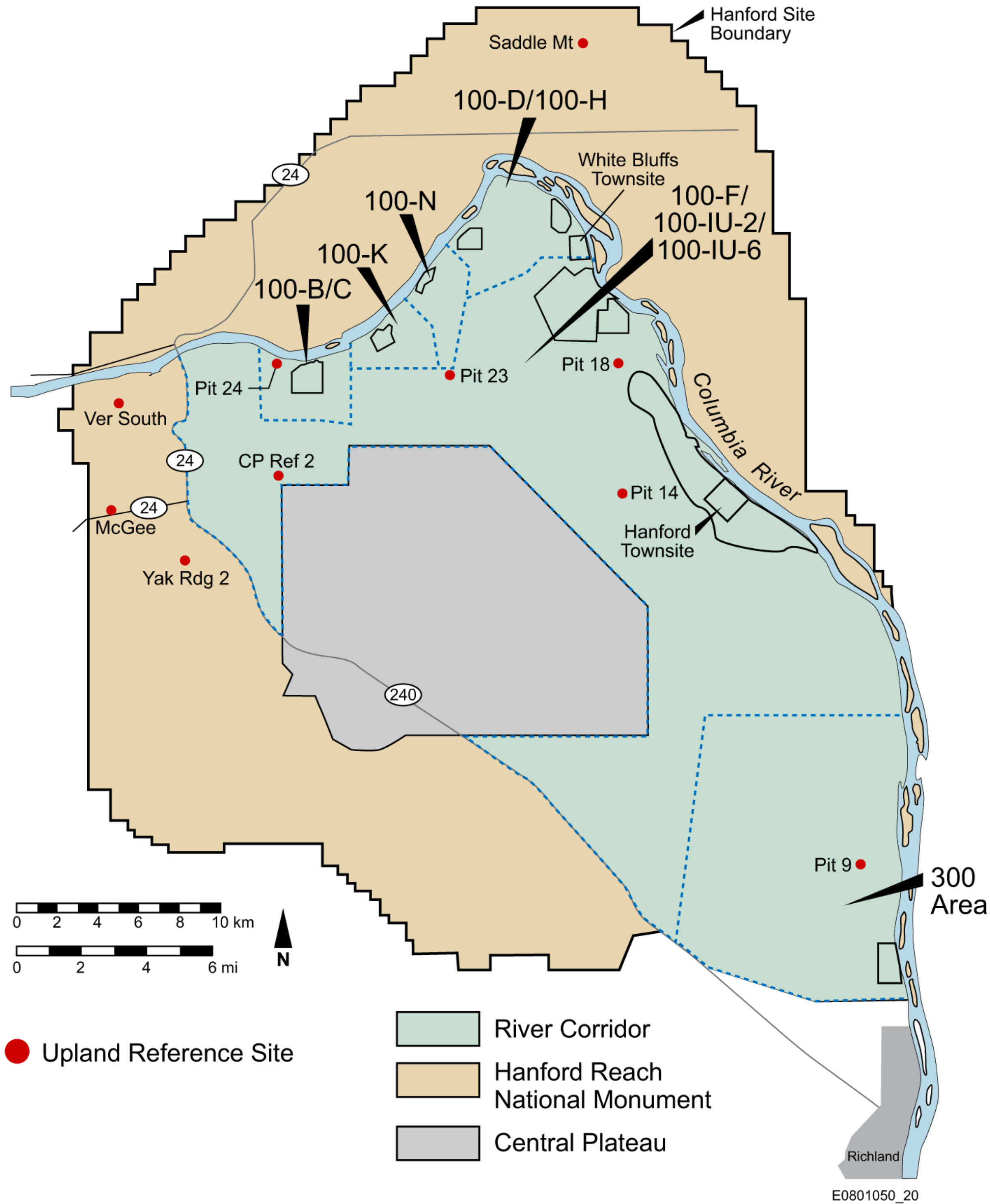
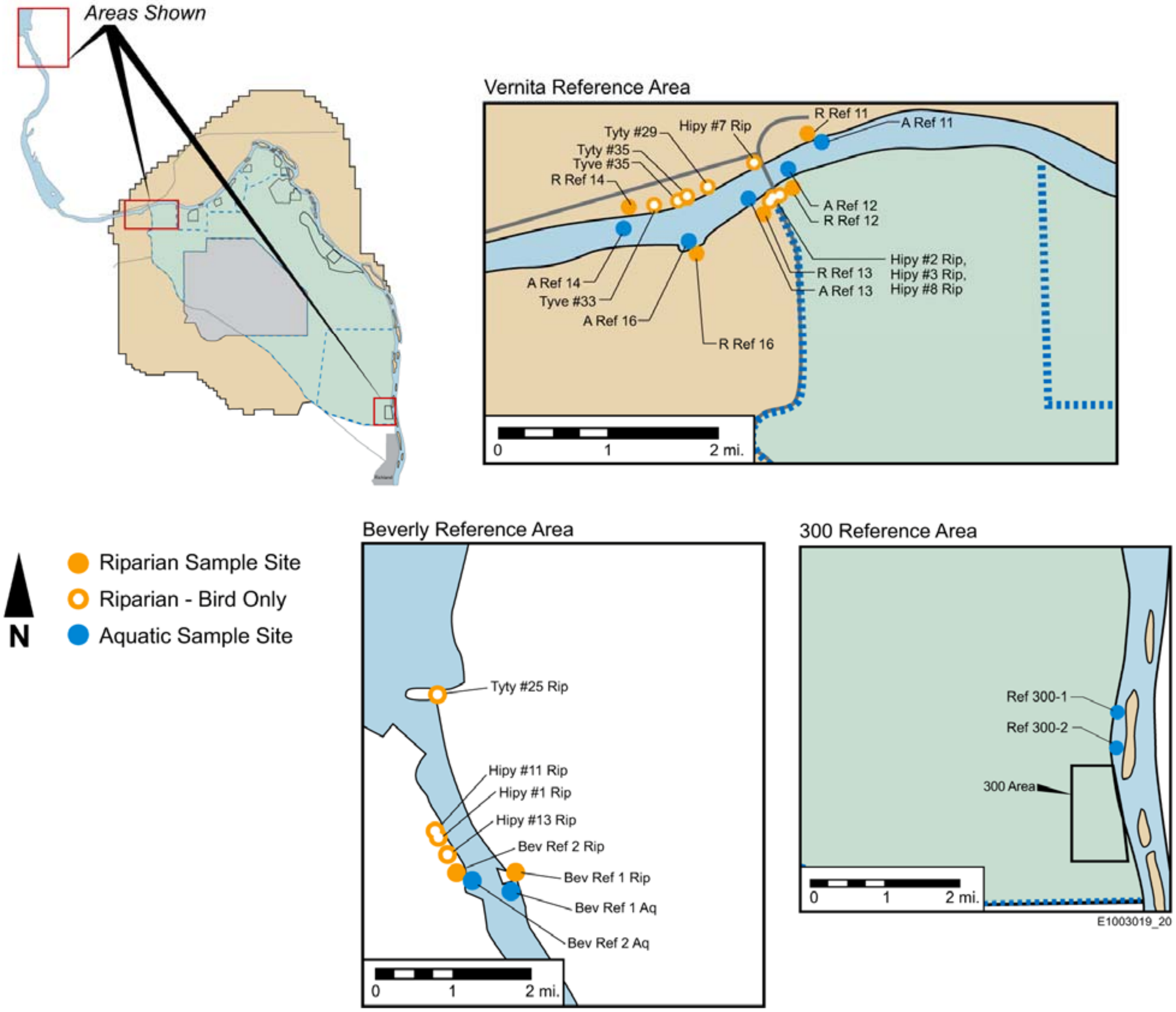


Figure 3-4. Location of Riparian and Near-Shore Aquatic Reference Sites.



Upland Reference Sites. Ten upland reference sites were selected. Five of the upland reference sites were located within "native" (mostly undisturbed) habitat areas. These are shown in Figure 3-3 and include Saddle Mountain, Vernita South, McGee Ranch, Central Plateau Reference 2, and Yakima Ridge 2. These sites are ecologically similar to remediated waste sites that required little or no excavation; tend to have heterogeneous, or patchy, contamination; and have little or no imported backfill. The soils are less coarse than backfilled areas and are vegetated with a mix of both native and nonnative species, such as rabbitbrush (*Chrysothamnus nauseousus*), Sandberg's bluegrass (*Poa sandbergii*), and cheatgrass (*Bromus tectorum*). Figure 3-5 shows the Vernita South reference site.

Figure 3-5. Vernita South Upland Reference Site.

The other five upland reference areas were located within uncontaminated borrow pits, which are disturbed areas used to acquire backfill material for remediated waste sites. These reference sites are shown in Figure 3-3 and include Pit 9, Pit 14, Pit 18, Pit 23, and Pit 24. They are ecologically similar to remediated waste sites that undergo significant amounts of excavation, contaminated soil removal, and application of imported backfill. The borrow pits used as reference areas usually contain coarse materials, such as sandy cobble, and have naturally revegetated. Figure 3-6 is a photograph of the Pit 9 upland reference site.

Figure 3-6. Pit 9 Upland Reference Site.

Riparian Reference Sites. Seven riparian reference sites were selected on the Columbia River upstream of the Hanford Site. These reference sites are shown in Figure 3-4 and include Bev Ref 1 Rip, Bev Ref 2 Rip, R Ref 11, R Ref 12, R Ref 13, R Ref 14, and R Ref 16. These reference sites are ecologically similar to the riparian study areas downstream on the Hanford Site. The slope of the bank from the upland boundary to the river varies from site to site and influences the width of the riparian zone. Flatter sections of the shoreline result in larger, more frequently flooded riparian areas. For example, riparian areas adjacent to the 100-F Area and 100-H Area sloughs are much flatter and wider than those below the 100-D Area, which are much steeper and narrower. The vegetation within the riparian study sites are characterized by a mix of reed canary grass (*Phalaris arundinacea*), mulberry trees (*Morus alba*), and willows (*Salix* spp). Figure 3-7 is a photograph of the Rip 16 riparian reference site.

Figure 3-7. Rip 16 Riparian Reference Site.

Near-Shore Aquatic Reference Sites. Seven near-shore aquatic reference sites were selected on the Columbia River upstream of the Hanford Site. These reference sites are shown in Figure 3-4 and include A Ref 11, A Ref 12, A Ref 13, A Ref 14, A Ref 16, Bev Ref 1 Aq, and Bev Ref 2 Aq. In addition, two near-shore aquatic reference sites were selected upstream of the 300 Area (Ref 300-1, Ref 300-2). Each location was characterized with a single sample of sediment and surface water. Biotic tissue samples and pore water were also collected from each location. However, while most of the near-shore riverbed adjacent to the reactor areas consists of sand and embedded cobbles, there are also slough and backwater areas where finer grained sediments have accumulated. Vegetation in the near-shore zone areas is generally sparse because of the prevalence of coarse substrate, along with daily and seasonal river level fluctuation. Figure 3-8 is a photograph of the Rip 14 near-shore aquatic reference sites.

Comparisons of Reference Site Concentrations to Background Concentrations. Although reference site data and background data are expected to be similar for most analytes, these data were prepared and tabulated to verify their comparability. In addition to evaluating RCBRA reference site data against background concentrations, the data were also compared to reference site data collected for other DOE projects. These included the Central Plateau Environmental Risk Assessment (CP ERA) and the SESP.

Figure 3-8. Rip 14 Near-Shore Aquatic Reference Site.

To understand the nature and variation in contaminant concentrations and comparability of background and reference data, reference data pertaining to the RCBRA investigation were compared to similar data from a variety of sources. These data sources are summarized in Table 3-6. Data sources were categorized as "Hanford Area background," "Washington State background," "reference," or "reference comparison." Washington State background data consists of published values for natural background of soil metals in the State of Washington (Ecology 1994). Hanford Area background values for metals and radionuclides in soil at the Hanford Site are described in three reports: DOE/RL-92-24, DOE/RL-95-55, and DOE/RL-96-12. Reference site data are available for the RCBRA project and the CP ERA. Reference comparison data consisted of data collected as part of SESP or special studies. Each of the reference comparison data sets was categorized by its data source.

Appendix B presents statistical and graphical comparisons of the background and reference site data sources listed in Table 3-6. The general conclusion from these comparisons is that, although the concentration ranges of many analytes for various background and reference sites overlap, there are many statistically significant differences among data sources. In general, there are more significant differences in the results for inorganic chemicals (metals) compared to the radionuclide results.

Below is a brief summary of the comparisons for frequently detected radionuclides in reference area soil and inorganic chemicals measured in Washington State background soils.

- Radionuclides
  - Seven radionuclides were frequently detected in upland or riparian reference soil: cesium-137, plutonium-238, plutonium-239/240, strontium-90, uranium-233/234, uranium-235, and uranium-238.
  - Lower concentrations of the uranium isotopes were measured in upland reference soils compared to Hanford Site background. Concentrations of other radionuclides were basically the same.
- Inorganic chemicals (metals and metalloids) measured in Washington State background soils
  - Thirteen inorganic chemicals were measured in Washington State background soils: aluminum, arsenic, beryllium, cadmium, chromium, copper, iron, lead, manganese, mercury, nickel, selenium, and zinc.
  - Upland reference soils were less than Washington State regional background for five of these inorganic chemicals (aluminum, beryllium, cadmium, manganese, and selenium). The concentrations of eight inorganic chemicals (arsenic, chromium, copper, iron, lead, mercury, nickel, and zinc) in upland reference soil were not different from Washington State regional background levels.
  - Riparian reference soils had lower concentrations of five of these inorganic chemicals (aluminum, beryllium, iron, manganese, and selenium) compared to Washington State regional background. The concentrations of chromium and copper in riparian reference soil were not different from Washington State regional background levels. Riparian reference soils had greater concentrations than Washington State regional background for the remaining six inorganic chemicals (arsenic, cadmium, lead, mercury, nickel, and zinc).

The comparisons presented in Appendix B show that background data source is important. These results indicate that given the difference in concentrations among various data sources evaluated in Appendix B, sample results from these data sources should not be combined for the purposes of identifying COPCs (see Section 3.2.2). An exception is that reference site samples collected for the Central Plateau Project were comparable in concentrations and in sample collection and laboratory analytical methods. These data were combined with the RCBRA reference site data for the purpose of identifying COPCs. In addition, these results show that concentrations of inorganic chemicals measured in upland reference sites are not different or less than those measured in Washington State background soils. However, these results also show that concentrations of inorganic chemicals measured in riparian reference sites generally differ from those measured in Washington State background soils.

3.2.1.2 Data Collected for the RCBRA. The data used in the HHRA to characterize exposure at locations other than an individual remediated waste site were collected under the *100 Area and 300 Area Component of RCBRA Sampling and Analysis Plan* (DOE/RL-2005-42). The RCBRA SAP originally identified sampling locations within the 100 Area and 300 Area. The SAP was amended during the first year of field sampling to include sampling of the shoreline regions between the operating areas, referred to as the "Inter-Areas." This assessment uses data from both the operating and the "Inter-Areas" to characterize contaminant concentrations in the River Corridor and does not differentiate between the original sampling distinctions. Together, these data allow evaluation of exposures within the upland, riparian and near-shore environments of the River Corridor.

Particular media and locations were selected for sampling as a result of a DQO process conducted specifically for the RCBRA (BHI-01757, *DQO Summary Report for the 100 Area and 300 Area Component of the River Corridor Baseline Risk Assessment*). Through this process, gaps in existing data were identified and conceptual models were developed that considered the relationship between contaminant sources and potentially exposed receptors.

Biotic and abiotic samples were collected from onsite investigation areas and reference sites between October 2005 and October 2007. Site descriptive information and species/habitat metrics were also recorded for use in Volume 1. Appendix C-1 provides a summary of the data sources used in this report. Additional details on specific analyses performed for the RCBRA sample media are presented in Appendix C-1.

Additional information relating to data collected for the RCBRA, including discussions of data quality, data completeness, and data representativeness, is provided in the broad-area risk assessment in Section 4.0 where these data are applied. A figure showing the locations where RCBRA samples were collected is also provided in Section 4.0.

3.2.1.3 Waste Site Cleanup Verification Data. Cleanup verification soil samples were collected at individual CERCLA waste sites within the River Corridor to document completion of remedial actions according to (primarily) IARODs. The data from cleanup verification samples were used to evaluate attainment of remedial action objectives (RAOs) identified in the IARODs. Two types of cleanup verification soil data are distinguished according to the sample collection methodology. Most of the shallow zone verification samples were collected by combining numerous individual subsamples into a single composite sample that was submitted for laboratory analysis. These are referred to as "statistical" samples because they were intended to be used to produce statistical estimates of average soil concentrations at the waste site. The other type of verification sample is referred to as a "focused" sample. The focused samples are individual samples collected from a single location. Sometimes these locations were selected based on where contamination was either evident or expected. Focused samples were also commonly collected subsequent to a targeted soil removal to confirm that residual concentrations were below IAROD cleanup levels.

The cleanup verification soil data are distinguished by whether they were collected in shallow zone soil (0 to 4.6 m [0 to 15 ft] below ground surface [bgs]) or deep zone soil (greater than 4.6 m [15 ft] bgs). Figure 3-9 shows the shallow zone and deep zone in a completed waste site excavation prior to final verification sampling. As discussed in Section 3.3, only shallow zone sample results were used to assess human health risks from residual soil contamination at remediated waste sites. Additional information relating to the waste site cleanup verification data, including data quality and the influence of variability between statistical and focused sampling results, is provided in the local-area risk assessment in Section 5.0, where these data are applied. Figures showing the locations of the remediated waste sites evaluated in the RCBRA in each of the six ROD decision areas are also provided in Section 5.0.

Figure 3-9. Excavated Waste Site Showing Shallow Zone and Deep Zone Soils.

3.2.1.4 Groundwater Data. The primary data used in the HHRA to characterize groundwater exposure over the separate ROD decision areas were obtained from the Hanford Environmental Information System (HEIS) database. For the purpose of characterizing groundwater exposure to human receptors, decision areas are assigned on the basis of the boundaries of each groundwater operable unit (OU). The following groundwater OUs are evaluated in this risk assessment: 100-BC-5, 100-KR-4, 100-NR-2, 100-HR-3, 100-FR-3, 300-FF-5, and the groundwater underlying the 100-IU-2 and 100-IU-6 decision area. The groundwater underlying the portion of the 100-IU-2 and 100-IU-6 source area OU is assigned to the 200-PO-1 OU. Groundwater data from unfiltered samples collected during the past 5 to 10 years were used in the HHRA. The data were obtained from a selection of wells representative of present-day groundwater conditions.

Work plans for the purpose of characterizing groundwater exposures and performing HHRA are currently under development. The groundwater risk assessment presented in this report is of a preliminary nature and will be refined in the RI reports prepared for each ROD decision area. At this time, DOE monitors groundwater at the Hanford Site to fulfill a variety of state and federal regulations, including the *Atomic Energy Act of 1954*; the *Resource Conservation and Recovery Act of 1976* (RCRA); CERCLA; and Section 173 of the *Washington Administrative Code* (WAC). Data collected to fulfill these monitoring requirements were used for the HHRA. While the monitoring data can be used for risk assessment purposes, there are uncertainties associated with its use. Specifically, target analytes, sampling frequencies, and method detection limits (or reporting limits) are different between programs because the information is used to meet different requirements.

Additional information relating to the programs under which the groundwater data used in the HHRA were collected is provided in the groundwater risk assessment in Section 6.0, where these data are applied. Figures showing the locations of the monitoring wells evaluated in the HHRA in each of the decision areas and concentration contours for the key groundwater plume contaminants in each area are also shown in Section 6.0.

3.2.1.5 Other Data Used in the HHRA. This section presents an overview of other Hanford Site data not collected specifically for the RCBRA but identified as potentially relevant to the HHRA. Sources of non-RCBRA analytical data include the following:

- 100-B/C Pilot Project Risk Assessment
- 100-NR-2 Shoreline Evaluation
- Central Plateau Environmental Risk Assessment
- Surface Environmental Surveillance Program and Special Studies.

These data were provided electronically from various Hanford Site data repositories, including the HEIS database, Environmental Restoration (ENRE) database, and other sources. Integrating laboratory data from multiple sources to support a single study presents a number of general challenges, which are discussed below.

The project-specific analytical data sources listed above were developed independently by multiple contractors. Because Hanford Site data may be formatted or configured for electronic storage with slight differences from one contractor to another, but are stored in common Hanford Site repositories (e.g., HEIS), it is possible for duplicate entries of a given sample to occur. Likewise, it is also possible for the database to appear to contain duplicate results for a specific analyte in a given sample. For example, laboratory analysis sometimes produces two results for a particular analyte, because two different methods are available and were performed. In either of these cases, the most appropriate result based on the source of the information or on the preferred analytical method, was retained for use in risk characterization, the other was flagged and not used in the risk calculations.

It was important to the RCBRA project for the data to be carefully reviewed and a data set be constructed that contained only a single entry of each pertinent sample result, and to ensure that it contained all of the relevant analyses associated with that sample. In cases where data were identified as truly duplicative and not appropriate for inclusion in risk characterization, the data were flagged in the database as “GiSdT not usable.” Results such as these were flagged during development of the risk assessment data set, rather than removed, in order to maintain transparency. Appendix C-1 of this report provides additional detail on the data sources and the data importing process used to support this HHRA.

Along with laboratory analytical data, databases contain other information, sometimes referred to as “metadata.” It can include information such as the location where a sample was collected (e.g., waste site name) or the depth at which the collection occurred, among other things. Integration of the data from the sources listed above required “normalization” of the metadata, where possible, in order to create consistency and allow the data to be used. The units used to report analytical results (e.g., mg/kg, pCi/g) often need to be normalized when multiple data sources are integrated. Occasionally, a sample result in the databases lacked critical information such as geographic location (coordinates), in which case the data were flagged and not used for risk calculations. Accurate sample location information is important for supporting final remedial action decisions within a given ROD decision area. Where normalization is possible, the changes aid greatly in the ease of data presentation and interpretation.

The quality assurance (QA) and quality control (QC) measures performed in sample collection and laboratory analysis are typically identified on a project-specific basis. Consequently, the QA/QC varies among datasets and is reviewed when results from a variety of databases are integrated. Generally, data used to support cleanup on the Hanford Site are collected according to approved SAPs that include a quality assurance project plan (QAPP). However, a review was conducted for each data set used in the RCBRA to determine its usability according to six different criteria. The criteria used and the review of the data sets is provided in detail in Appendix C-2.

100-B/C Pilot Project Risk Assessment. The 100-B/C Pilot Project Risk Assessment (100-B/C Pilot Project) (DOE/RL-2005-40) purpose and scope is described in Section 2.0 of this report. The investigation yielded analytical data for abiotic and biotic media, biological health metrics information for selected receptors, and numerous records of contaminants in abiotic media from a compilation of cleanup verification results and Hanford Site monitoring data. Historical and recent analytical data evaluated for the 100-B/C Pilot Project included shallow zone soil (0 to 4.6 m [0 to 15 ft] bgs), deep zone soil (greater than 4.6 m [15 ft] bgs), sediment, surface water from the Columbia River, and biotic tissues including sculpin, clams, and macroinvertebrates. Sampling and analytical data collected between June 1995 and January 2004 were evaluated in the 100-B/C Pilot Project. Sources of data for the 100-B/C Pilot Project investigation included the following:

- All 100-B/C Area soil data residing in the ENRE database maintained for remedial action projects

- All 100-B/C Area soil and water analytical data residing in the HEIS database

- Analytical results for data contained in the *100-B/C Pilot Project Data Summary for 2003 and 2004* (BHI-01724).

These data were used to identify COPCs in this report. As discussed in Section 4.1.3, upland surface soil data related to the 100-B/C Pilot Project are not used in the HHRA to calculate representative concentrations because only a single soil grab sample was collected at each 100-B/C waste site sampled in the 100-B/C Pilot Project. A single grab sample is a poor indicator of average soil concentrations across an entire site.

Some overlap in 100-B/C Pilot Project data with the other data sources was encountered. Specifically, cleanup verification data for individual remediated waste sites are evaluated in the local-area risk assessment (Section 5.0) and are not assessed in any other context. Information on the criteria and process used for making determinations of data usability and for ranking of data sources is provided in Appendix C-1.

100-NR-2 Shoreline Evaluation. The 100-NR-2 shoreline evaluation was conducted during 2005. The purpose and scope of the evaluation is described in Section 2.0 of this document. A report titled *Aquatic and Riparian Receptor Impact Information for the 100-NR-2 Groundwater Operable Unit* (DOE/RL-2006-26) describes the objectives, methods, and findings of the investigation.

Data collected during the 100-NR-2 investigation included analytical data for surface water, pore water, and sediment. Data were also collected for several species of aquatic and riparian biota, including clams, clam shells, sculpin, milfoil, and periphyton in the Columbia River, and plants, small mammals, and terrestrial invertebrates in the riparian environment. Abiotic and biotic media were analyzed for radionuclides, metals, PCBs, and total petroleum hydrocarbons (TPH). Sediment, surface water, sculpin, and clam tissue data were used to identify COPCs and calculate representative concentrations in the 100-N Area.

Some overlap in 100-NR-2 investigation data with the other data sources was encountered. Information on the criteria and process used for making determinations of data usability and for ranking of data sources is provided in Appendix C-1.

Central Plateau Ecological Risk Assessment Data. The CP ERA evaluated the potential for ecological risk from waste sites located in the 200 Area. The purpose and scope of the CP ERA is described in Section 2.0 of this document. Sampling methods, analytical suites, and other ecological measures used in the Central Plateau were consistent with those used for the RCBRA project. Media collected and analyzed for the CP ERA included terrestrial soil, vegetation, invertebrates, and small mammals. These Central Plateau data were used in the HHRA to supplement RCBRA information on reference site conditions and to identify COPCs. Additional information on the types of samples collected and the analyses performed using Central Plateau data is provided in Appendix C-1.

Surface Environmental Surveillance Program and Special Studies. A variety of environmental media are routinely monitored for the SESP. These include air, surface water, sediments, soil, natural vegetation, agricultural products, fish, and wildlife. Radionuclides and nonradiological chemicals (including metals, organic compounds, pesticides, and PCBs) are measured in various media. Ambient external radiation levels in the environment are also routinely monitored. Information obtained from these surveillance efforts are provided to federal, state, county, and city agencies; regional Native American tribes; other stakeholders; and the general public. The collected data are used to document Hanford Site compliance with environmental regulations, to provide information to the public about environmental conditions on the Hanford Site and adjoining properties, and to satisfy monitoring requirements of some contractors engaged in site cleanup activities.

Biotic tissue data are published by the SESP in the Hanford Site annual monitoring reports. Tissue data published between 1990 and 2005 that are evaluated in the HHRA include contaminant tissue concentrations in certain upland and riparian biota. A limited amount of SESP tissue data were also published for aquatic species including salmon, steelhead, trout, bass, carp, sculpin, sucker, squawfish, and whitefish. As discussed in Section 3.3.2, these aquatic tissue data are not used in the HHRA.

Surface water and sediment have historically been characterized under the SESP at the Hanford Site shoreline and within the Columbia River for special characterization purposes. These data were maintained in the HEIS database or in project-specific documentation (reports and databases). Surface water data used in the HHRA include samples from seeps and springs associated with the near-shore environment. Columbia River surface water and sediment data collected by SESP are not used in the HHRA, as described in Section 3.3.2.

Some overlap in the SESP data with the other data sources was encountered. Information on the criteria and process used for making determinations of data usability and for ranking of data sources is provided in Appendix C-1.

3.2.2 Contaminant of Potential Concern Refinement

Selection of the appropriate COPCs is critical to preparing an assessment that is representative of risks resulting from Hanford Site operations and useful for making remedial action decisions. COPC selection should occur through a process that is deliberate, systematic, and based on established selection criteria. The risk assessment must be able to differentiate between background materials, nonsite-related materials, and contaminants directly related to site processes or materials. A consequence of not establishing an appropriately focused list of Hanford Site-related contaminants for this type of assessment can be that the calculated risks from non-Hanford Site constituents are high enough to mask the impacts from contaminants that are potentially related to Hanford operations. Examples of such constituents include naturally occurring metals and radionuclides, and ubiquitous manmade contaminants, such as certain persistent organic pollutants and radionuclides related to global fallout from atmospheric testing. This section describes the approach developed to identify and focus the COPCs identified for the risk assessment evaluation. Additional details on the implementation of this process are also described in Sections 4.0, 5.0, and 6.0.

The approach used for COPC refinement builds upon approaches and methods for COPC selection presented in the Tri-Parties approved RCBRA SAP (DOE/RL-2005-42). The SAP outlined a process for focusing contaminants based on comparing mean concentrations at study sites to background or reference sites using conclusions and data summaries from limited field investigation, cleanup verification packages (CVPs), Hanford Site monitoring, and related projects. This process is consistent with guidance pertaining to selection of COPCs for risk assessment (EPA/540/1-89/002, *Risk Assessment Guidance for Superfund* [*RAGS]*) Part A, Chapter 5, "Data Evaluation"). The focus on risks related to site releases is also consistent with the National Contingency Plan (40 CFR 300.430[d][(2]), which states, "…gather data necessary to assess the extent to which the release poses a threat to human health or the environment or to support the analysis and design of potential response actions." EPA guidance for conducting background comparisons (EPA/540/R-01/003, Appendix B) suggests comparing site concentrations with risk-based screening levels to identify COPCs and carrying chemicals through the risk assessment process as COPCs if they exceed risk-based screening levels, even though the measured concentrations may represent background conditions. This EPA guidance further recommends addressing the contribution to risk from background levels of COPCs in the risk characterization of the risk assessment. Such an approach is problematic for this HHRA because numerous exposure scenarios are evaluated, and there are no published risk-based screening levels that correspond to these scenarios. In addition, the goal of the COPC selection process developed for Hanford through agency workshops was to clearly focus the risk assessment efforts on site-related chemicals. Therefore, a risk-based screening was not included as part of the COPC selection process for the HHRA, and the COPC selection process relied on comparisons to background levels and reference areas.

The COPC refinement process includes a number of complementary steps and criteria, including a pre-selected list of contaminants that will be excluded and a list that will be included, as determined and agreed upon among the Tri-Parties. 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.

Additional selection steps included evaluation of all of the data according to detection status, statistical comparisons of Hanford Site data to background and reference site data, and an analyte-specific evaluation. The analyte-specific evaluation integrated a variety of information (such as the magnitude and significance of statistical comparisons results, sample results in other media, and sample results for similar analytes) to support a conclusion on COPC identification when the results of statistical comparisons were inconclusive. These evaluations were conducted using quantitative methods and divided the analytes into workable groups for these individual analyses. The quantitative methods provided valuable information for the included analytes and also provided a sound technical basis for eliminating less relevant analytes from the quantitative risk assessment.

3.2.2.1 Exclusions. Some analytes have been excluded from consideration as COPCs by agreement among the Tri-Parties based on relevant Hanford Site data. Separate exclusion lists have been developed for soil and groundwater contaminant plumes.

The exclusion lists are based on the following types of analytes:

- Radionuclides with a half-life of less than 3 years: Radionuclides with half-lives less than 3 years would not be present as a result of historical Hanford Site operations due to radioactive decay that would have occurred since operations ceased.

- Essential nutrients: Essential nutrients that are present at relatively low concentrations and are toxic only at high concentrations need not be considered in a quantitative risk assessment (EPA/540/1-89/002).

- Water quality or soil physical property measurements: These analytes were measured to obtain information on water quality or soil properties to understand potential confounding factors for bioassays conducted for soil, sediment, or water, or to interpret their influence on the toxicity of COPCs (e.g., grain size for soils, water hardness for metals effects).

- Background radionuclides (potassium-40, radium-226, radium-228, thorium-228, thorium-230, and thorium-232): These background radionuclides were identified by consensus of Tri-Party managers as not directly related to Hanford operations or processes.

The list of excluded analytes is provided in Table 3-7.

3.2.2.2 Inclusions. Certain analytes were included as COPCs based on evaluation of the commonly reported analytes in waste site CVPs or based on the most prevalent contaminants in the groundwater plumes. The inclusion list reflects those contaminants that the Tri-Parties agree need to be addressed in this risk assessment in order to prepare a meaningful and effective regulatory document.

- Waste site (soil and related media): The list of contaminants evaluated in the CVP/remaining sites verification package (RSVP) reports was compiled for waste sites in the 100 Area and the 300 Area. Some contaminants were only reported in a single report, and others were reported in nearly all CVP/RSVP documents. To develop the inclusion list for the 100 Area and 300 Area, the frequency of reporting for analytes was ranked, and those analytes reported at one-third or more of the waste sites were placed on the 100 Area inclusion list. The 300 Area list was developed by removing those 100 Area analytes that were not reported in at least one of the 300 Area waste sites.

- Key plume (groundwater and related media): The list of key plume contaminants in the 2006 Hanford Site annual groundwater monitoring report (PNNL-16346, *Hanford Site Groundwater Monitoring for Fiscal Year 2006*) was compiled for the 100 Area and the 300 Area. Some contaminants were reported only in a single area, and others were reported in nearly all areas.

The list of included analytes is provided in Table 3-8. Additional information on the key groundwater plume analytes is provided in Section 6.0.

3.2.2.3 Nondetected Analytes. Analytical results for soil, sediment, water, and biota collected for the RCBRA investigation are evaluated against the quality criteria specified in the QAPP of the RCBRA SAP (DOE/RL-2005-42). As a measure of data quality, analytical results identified as nondetects in the RCBRA data set are compared to the laboratory-required detection limits prescribed in the QAPP. Nondetect results reported at values higher than the prescribed detection limit are identified for additional consideration in the uncertainty analysis of the risk assessment.

The evaluation of nondetect values above a practical quantitation level (PQL) focuses on whether reporting limits that exceed a PQL also exceed one or more of the human health reference values defined in the QAPP. Human health reference values were provided for soil, groundwater, and surface water in the QAPP (DOE/RL-2005-42; Tables 2-2 and 2-3). If an analyte that exceeds a PQL has no associated reference value, the potential significance of exceeding the PQL is discussed in relation to risk assessment results in other media and/or results for analytes with similar properties (e.g., PCB Aroclors, organochlorine pesticides). A comparison of sample-specific detection limits versus PQLs for nondetect results in the RCBRA data set is provided for soil and surface water in a spreadsheet attachment to Appendix C-3. The results are summarized in Section 4.7 of the HHRA.

3.2.2.4 Detected Analytes (Soil and Other Media Except for Groundwater). Detected analytes are the dominant focus of COPC refinement. Statistical and analyte-specific analysis of detected analytes are used to determine the COPC list. The COPC list is determined by sorting the detected analytes into the following groups:

- Detected analytes present at concentrations that are clearly different from the reference area and/or background data based on one or more of the statistical tests. These analytes are COPCs. Demonstration of statistical difference was based on p-values less than or equal to 0.05 for one or more of the following statistical tests: Gehan, quantile, slippage, and chi-square detection frequency. These tests are described below.

- Detected analytes present at concentrations that are clearly not different from the reference area and/or background data. These analytes are not COPCs. Lack of statistical difference was based on p-values greater than to 0.05 for all of the following statistical tests that are applicable: Gehan, quantile, slippage, and chi-square detection frequency. These tests are described below.

- Detected analytes that remain after the above steps are subject to an analyte-specific evaluation (described in the following section). The analyte-specific evaluation integrates a variety of information, such as the magnitude and significance of statistical comparisons results, process knowledge, results in other media, and results for similar analytes, to support a conclusion on COPC identification. Based on the results of the evaluation, these analytes are sorted as either COPCs or not COPCs.

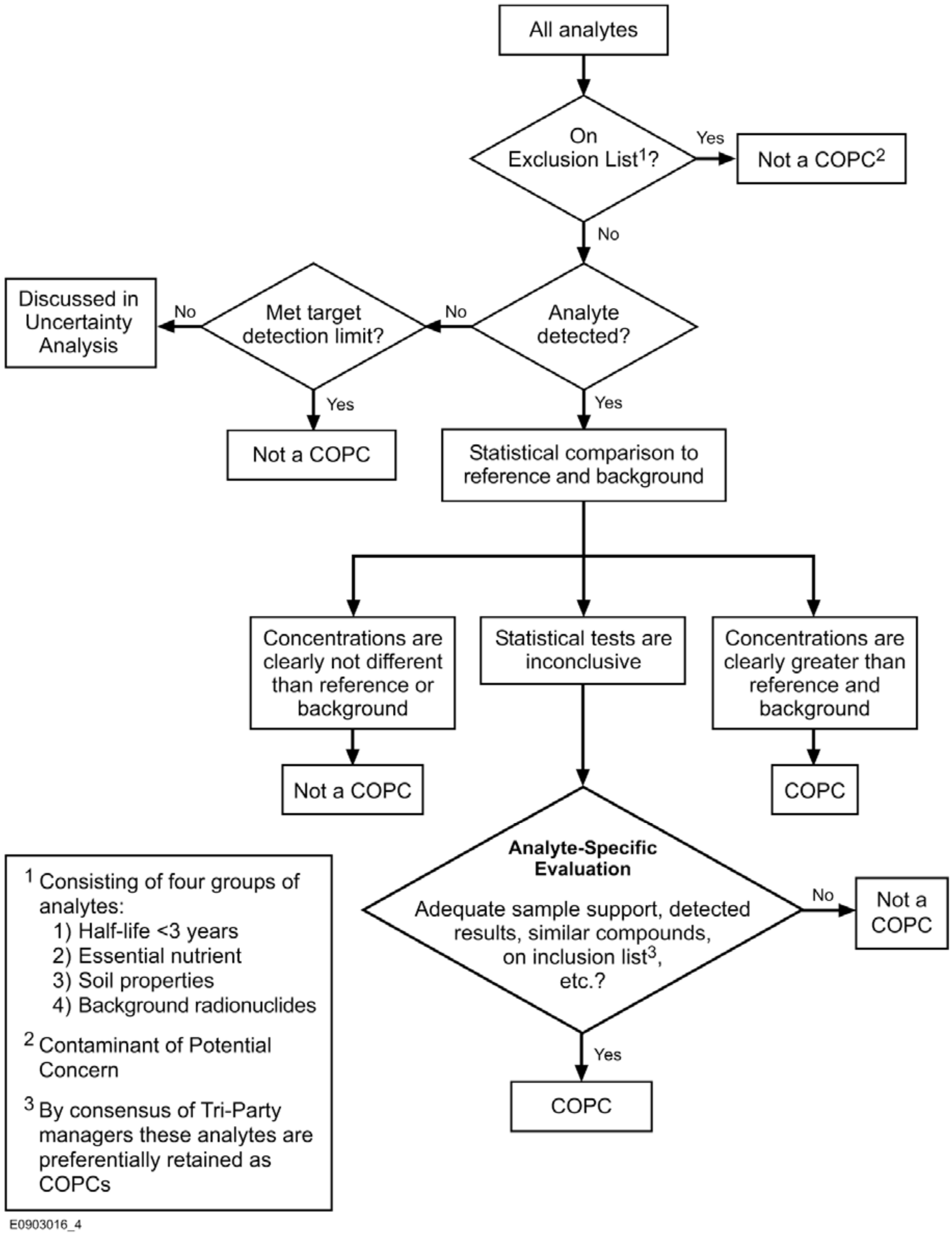
Figure 3-10 provides an overview of the COPC refinement process for soil and other media with the exception of groundwater.

Statistical Tests. Background comparisons are possible based on the availability of appropriate data. In the case of soil data, available background data include natural background for Washington State (Yakima Basin region) for selected inorganic chemicals and Hanford Site background for selected radionuclides and inorganic chemicals. For analytes with background data from multiple sources, the Hanford Site background data are used in preference to Washington State (Yakima Basin region) background data.

Reference site comparisons are used to evaluate detected analytes in soil, sediment, surface water, and biotic tissues. Although reference site soil data and background soil data are similar for most analytes, both comparisons are performed. If site concentrations are clearly different in one or more statistical tests for either data set, that analyte is identified as a COPC. Reference sites are used for constituents and media not included in the various sources for background levels. Additional information relating to background and reference site data is provided in Appendix B of this report.

The study site and reference site soil data were composite samples, and thus represented the average concentration in the sampled area. Cleanup verification soil data include both composite and discrete (grab) samples, and background soil data sets are based on discrete samples. Some potential bias was introduced by comparing discrete sample results to composite sample results. However, running a suite of statistical tests is a rigorous way of comparing contaminated site analyte concentrations to background, and this approach has been recommended in several papers (e.g., EPA/540/1-89/002; Gilbert and Simpson 1990, "Statistical Sampling and Analysis

Figure 3-10. Contaminant of Potential Concern Refinement Process Flow Diagram for Soil.



Issues and Needs for Testing Attainment of Background-Based Cleanup Standards at Superfund Sites"; EPA/230/R-94/004, *Statistical Methods for Evaluating the Attainment of Cleanup Standards, Volume 3: Reference-Based Standards for Soils and Solid Media*). This approach represents an enhancement over the comparison of site data to a single statistic of background (e.g., the 90th percentile), as all fixed percentiles will be exceeded with some frequency.

Four statistical tests are used because the kind of differences detected by each test differs, and the overall results of the tests are complementary. The statistical tests employed for the background and reference site comparisons, including data adequacy requirements, are discussed in greater detail below.

Overall Concentration Shift. The Wilcoxon rank sum (WRS) test detects a shift in the central tendency of the data. The Gehan test is used in this report and is a variation on the WRS that uses a statistically robust method for ranking nondetect values. These tests are used to detect differences in the interquartile range of the data (25th percentile to the 75th percentile) or the range represented by the box in box and whisker plots. These tests detect an overall shift in the site data compared to background or reference. The Gehan test is used when nondetects are relatively frequent (greater than 10% and less than 50%). It handles data sets with nondetects reported at multiple detection limits in a statistically robust manner (Gehan 1965, "A Generalized Wilcoxon Test for Comparing Arbitrarily Singly-Censored Samples;" Millard and Deverel 1988, "Nonparametric Statistical Methods for Comparing Two Sites Based on Data with Multiple Nondetect Limits"). The Gehan test was not performed if either of the two data sets had more than 50% nondetects. The WRS test performs a test for a difference between two populations of data (Gilbert 1987, *Statistical Methods for Environmental Pollution Monitoring*). This is a nonparametric method that relies on the relative rankings of data values. Knowledge of the precise form of the population distributions is not necessary. The WRS test has less power than the two-sample t-test when the data originate from a normal distribution, but the assumptions are not as restrictive. This version of the WRS test uses the Gehan approach for ranking (Gehan 1965, Millard and Deverel 1988). When some of the data are "censored" or reported as below a detection limit, the Gehan approach assigns ranks to the combined set of detects and nondetects in a statistically robust manner. The Gehan approach defaults to standard ranks when the dataset contains no censored data (nondetects). The Gehan ranking approach is recommended in EPA-sponsored workshops and publications due to its broad applicability to environmental data sets (Gilbert and Simpson 1990, 1992).

Shift in Upper Range of Concentrations. The quantile test (tests for difference in the upper percentiles versus background) evaluates the upper range of data sets and is sensitive to a small number of observations from one or more sites being greater than background. Thus, the quantile test complements the WRS test and provides a greater chance to detect differences among a small set of sample results. The quantile test is applied at a pre-specified quantile or threshold. The 80th percentile was chosen for this project. The test cannot be performed if more than 80% (or, in general, more than the chosen percentile) of the combined data are nondetected values. The quantile test determines whether more of the observations in the top 20% (chosen percentile) of the combined data set come from the site data set than would be expected by chance, given the relative sizes of the site and background or reference data sets. If the relative proportion of the two populations being tested is different in the top 20% of the data than in the remainder of the data, the distributions may be partially shifted due to a subset of site data. This test is capable of detecting a statistical difference when only a small number of site concentrations are elevated (EPA/230/R-94/004). The quantile test is the most useful distribution shift test for sites at which samples from a release represent a small fraction of the overall data collected. Thus, the quantile test is sensitive to concentration shifts based on a fraction of the remediated waste sites or study sites that are pooled for statistical analyses. The quantile test is applied at a pre-specified quantile or threshold. The quantile test is more powerful than the WRS (Gehan) test for detecting differences when only a small percentage of the concentrations are elevated.

Differences in Range of Concentrations. The slippage test (tests for difference in the upper range of data compared to background) evaluates the highest values and could detect as few as one elevated measurement in the site data (dependent on the number of sample results in the data sets). Thus, the slippage test complements the WRS test and the quantile test and will detect a difference in concentration from a single sample or a single site. The slippage test was not performed if there were no detected concentrations in the background or reference data set, but was performed if the data set contained at least a single detected sample result. The slippage test (Gilbert and Simpson 1990) is a nonparametric test appropriate for comparing between data sets with low detection rates. This test is based on the maximum observed concentration in the background data set and the number (“n”) of site concentrations that exceed the maximum concentration in the background set (Gilbert and Simpson 1990). The result (p-value or significance level) of the slippage test is the probability that “n” site samples (or more) exceed the maximum background concentration by chance alone. The test accounts for the number of sample results in each data set (number of sample results from the site and number of sample results from background) and determines the probability of “n” (or more) exceedances if the two data sets came from identical distributions. This test is similar to a “hot measurement” test, a comparison of the maximum site value to an upper percentile or threshold calculated from the background data, in that it evaluates the largest site measurements. It is more useful than a “hot measurement” comparison because it is based on a statistical hypothesis test, not simply on a statistic calculated from the background distribution. Because the slippage test evaluates the maximum concentration, it is sensitive to detecting an elevated concentration at a single remediated waste site or study site, even if they are pooled for statistical analyses.

Difference in Detection Frequency. The chi-square test is an assessment of detection frequency. It is used to determine if site data have a greater frequency of detections compared to background and/or reference site data sets. The chi-square assessment is useful for comparing differences between data sets for analytes that are not frequently detected. It was used in concert with the other tests to help determine differences between site and background and/or reference site data sets when detection frequency was low. The chi-square test (Zar 1984, *Biostatistical Analysis*) is used in conjunction with 2 x 2 contingency tables to test for differences in categorical data. It represents the measure of association between two dichotomous variables and so indicates the strength of correlation, or lack thereof, indicating independence. In the case of detection rate, it tests whether the frequency of occurrence of detects in a dataset is independent of the category of that dataset, (e.g., whether the dataset originates from remediated waste sites or background locations). The chi-square test does not consider the magnitude of the analyte, only the frequency of detection. The box plots were reviewed to determine if the difference in detection frequency was also associated with a difference in the range of remediated waste site versus background or reference site concentrations.

Analyte-Specific Evaluation. Contaminant of potential concern refinement stresses detected sample results in the primary environmental media. The results of statistical tests using data from remediated waste sites and study site sampling locations and media are supplemented by review of results in the primary media for individual sampling locations and decision areas in order to identify COPCs. The primary media are abiotic media that include soil (upland and riparian), sediment, and pore water (near-shore). River water was not identified as a primary medium because of the influence of dilution. Factors such as mobility of biota, interspecies variability, uncertainty in uptake and depuration rates, and other temporal influences are reasons that biotic tissues were not considered primary media. The analyte-specific evaluation for each environment included the following components.

- Status on the inclusion list was considered for retaining an analyte as a COPC.

- Emphasis was placed on reference site concentrations as these are used as comparison values for the exposure evaluation. A difference from reference concentrations demonstrates that there is a biologically meaningful range of exposure levels for gradient analyses. The slippage test is the most definitive test for showing a difference in the range of concentrations at operational areas compared to reference sites.

- The results of primary contaminated media and biotic tissue statistical tests are considered, but emphasis was placed on differences in the primary contaminated media for the reasons stated above.

- Box plots showing sample results for the primary media are reviewed. The plots are reviewed to identify concentration shifts for a particular location and to identify outliers. Concentration shifts are indicated by differences in the interquartile range (the position of the box) or by the range of concentrations in the upper quartile (points that plot above the box). Analytes that show a concentration shift are identified as COPCs. Outlier points are indicated on the plots by data that lie outside of the whiskers on the plots. A significant outlier is a factor used to decide if an analyte should be retained as a COPC. The plots also help to determine if the detected concentrations fall with the range of the nondetects; these analytes were not retained as COPCs.

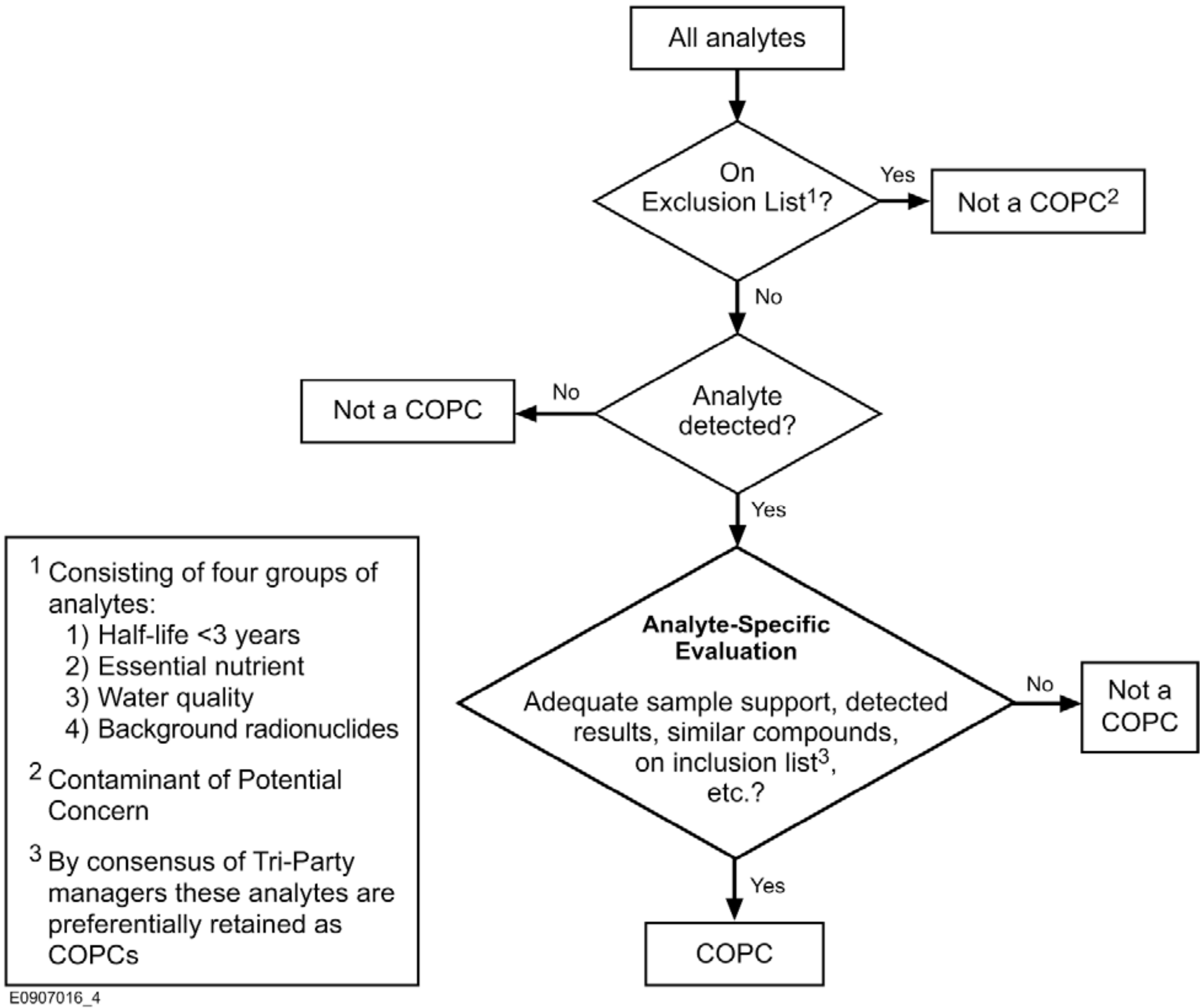
- The box plots showing biotic tissue sample results are reviewed. The plots are reviewed to identify outliers as described above. A significant outlier is a factor used to decide if an analyte should be retained as a COPC. The plots also help to determine if the detected concentrations fall within the range of the nondetects; these analytes are not retained as COPCs.

Detected Analytes (Groundwater). Detected analytes are the primary focus of COPC refinement in groundwater. Neither background nor reference site data are available for groundwater. For this reason, the COPC identification process for groundwater does not employ such comparisons. Graphical analyses, comparisons of the range of concentrations among the groundwater OUs, and other information described below are used to differentiate COPCs from analytes not identified as COPCs. The COPC list for groundwater is determined by sorting the detected analytes into the following three groups.

- Analytes that were determined not to be COPCs based on the analyte-specific evaluation. This evaluation includes an assessment of the range of the detected concentrations relative to detection limits, consistency in groundwater concentrations among different groundwater OUs, detection frequency, status as a COPC in the cleanup verification soil data in the ROD area, and status as a preferred inclusion list analyte.
- Analytes that were determined to be COPCs based on the analyte-specific evaluation. This evaluation includes an assessment of the range of the detected concentrations relative to detection limits, consistency in groundwater concentrations among different groundwater OUs, detection frequency, status as a COPC in the cleanup verification soil data in the ROD area, and status as a preferred inclusion list analyte.
- Analytes for which COPC status is uncertain based on the analyte-specific evaluation.

Figure 3-11 provides an overview of the COPC refinement process for groundwater. The degree of difference in analyte concentrations among the seven groundwater OUs is the primary tool to assign COPC status for metals and radionuclides. Other information summarized in the above bullets is more frequently employed for organic chemicals and when detection frequencies of naturally occurring metals and radionuclides are too low to assign an analyte to a decision logic endpoint with confidence. The rationale for assigning a detected groundwater analyte to one of the three groups defined in these bullets is documented for each of the seven groundwater OUs in Section 6.2. A more systematic evaluation of groundwater COPCs, in support of the identification of groundwater data gaps and recommendations for additional sampling, will be implemented in the groundwater remedial investigation work plans currently in preparation.

Figure 3-11. Contaminant of Potential Concern Refinement Process Flow Diagram for Groundwater.



3.3 CONCEPTUAL SITE MODEL

The CSM relates what is known about site conditions, such as the nature and extent of contamination, to the spatial and temporal distribution of potential receptors. It also describes transport and exposure pathways through various environmental media that may be important in evaluating potential exposures to human receptors. Information on the physical and environmental setting of the Hanford Site and the present-day nature and extent of contamination in the Columbia River Corridor is provided in Section 2.0 of this report. This CSM will, therefore, focus on the relation of residual contamination, as reflected in the environmental data sets described in Section 3.2, to potential human exposures.

For the purposes of remedial action decisions in the 100 and 300 Areas, the River Corridor has been divided into six decision areas corresponding to the final RODs that will be developed to define the final requirements for closing the waste sites. The decision areas are as follows: 100-B/C Area, 100-K Area, 100-N Area, 100-D and 100-H Areas, 100-F/100-IU-2/100-IU-6 Area, and 300 Area. Figure 3-12 shows the boundaries of the decision areas.

Figure 3-13 presents the general features, waste streams, and sources of contamination for the River Corridor and illustrates the conditions and transport mechanisms that exist during operations. At the time this assessment was conducted, many waste sites had been remediated. The potential residual contamination still present in soils associated with former waste sites constitutes an exposure medium for receptors and a potential source of groundwater contamination at these locations. A simplified representation of current site conditions is provided in Figure 3-14, which shows an example of a remediated waste site that has been backfilled and revegetated.

This section provides a description of the human health conceptual exposure models for the RCBRA. The exposure models are used to identify potentially complete human exposure pathways for a variety of hypothetical exposure scenarios. As described in the EPA risk assessment guidance (EPA/540/1-89/002), an exposure pathway describes the course a contaminant takes from a source to a receptor. Assuming that exposure occurs at a location other than the source, every complete exposure pathway contains the following elements (EPA/540/1-89/002):

- Known or potential sources and/or releases of contamination
- Contaminant transport pathways
- Potential exposure media
- A point of potential receptor contact with the impacted medium
- An exposure route (such as ingestion or inhalation).

Figure 3-12. Record of Decision Areas for the River Corridor.

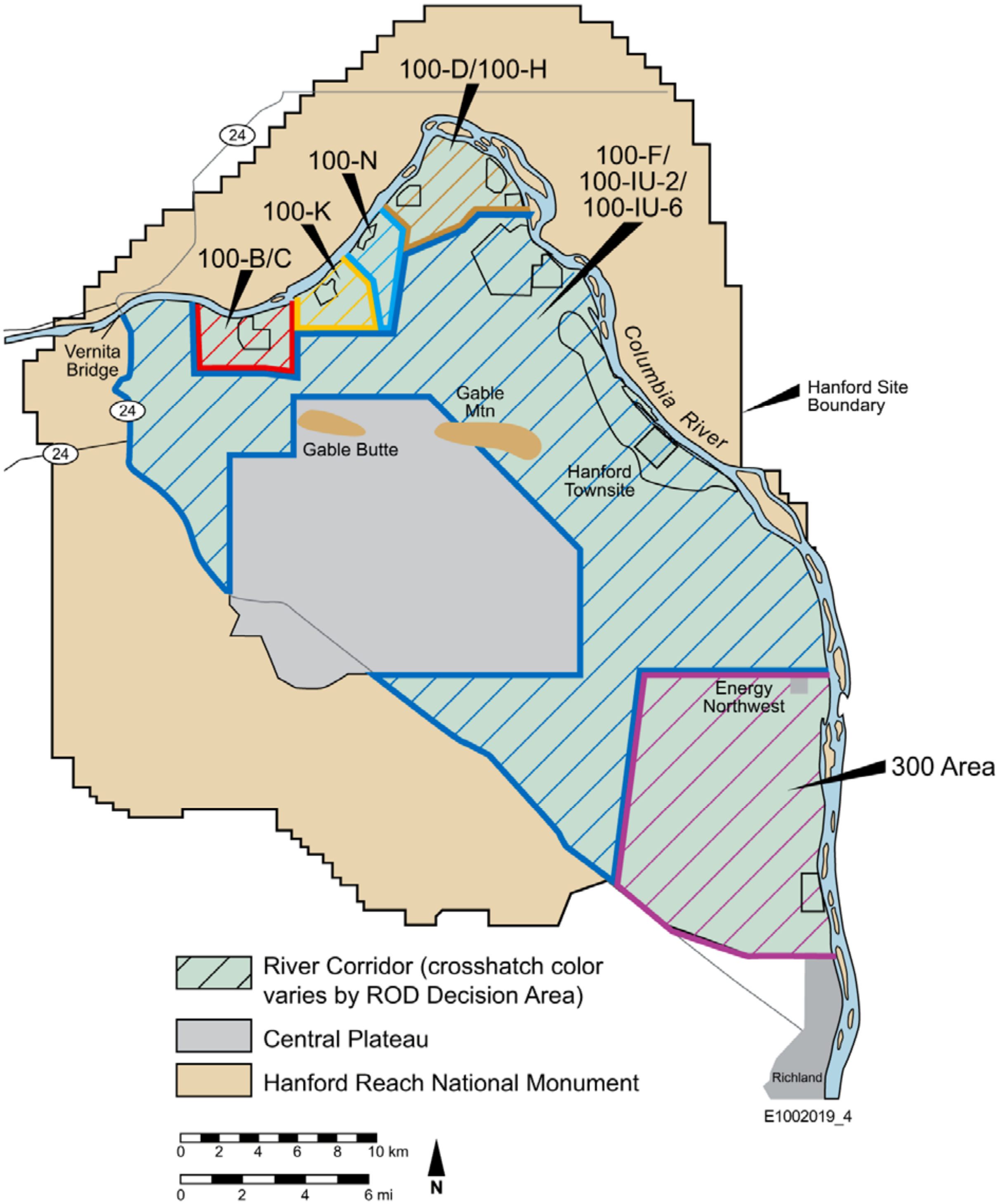


Figure 3-13. Conceptual Model of Contaminant Sources Potentially Affecting the River Corridor.

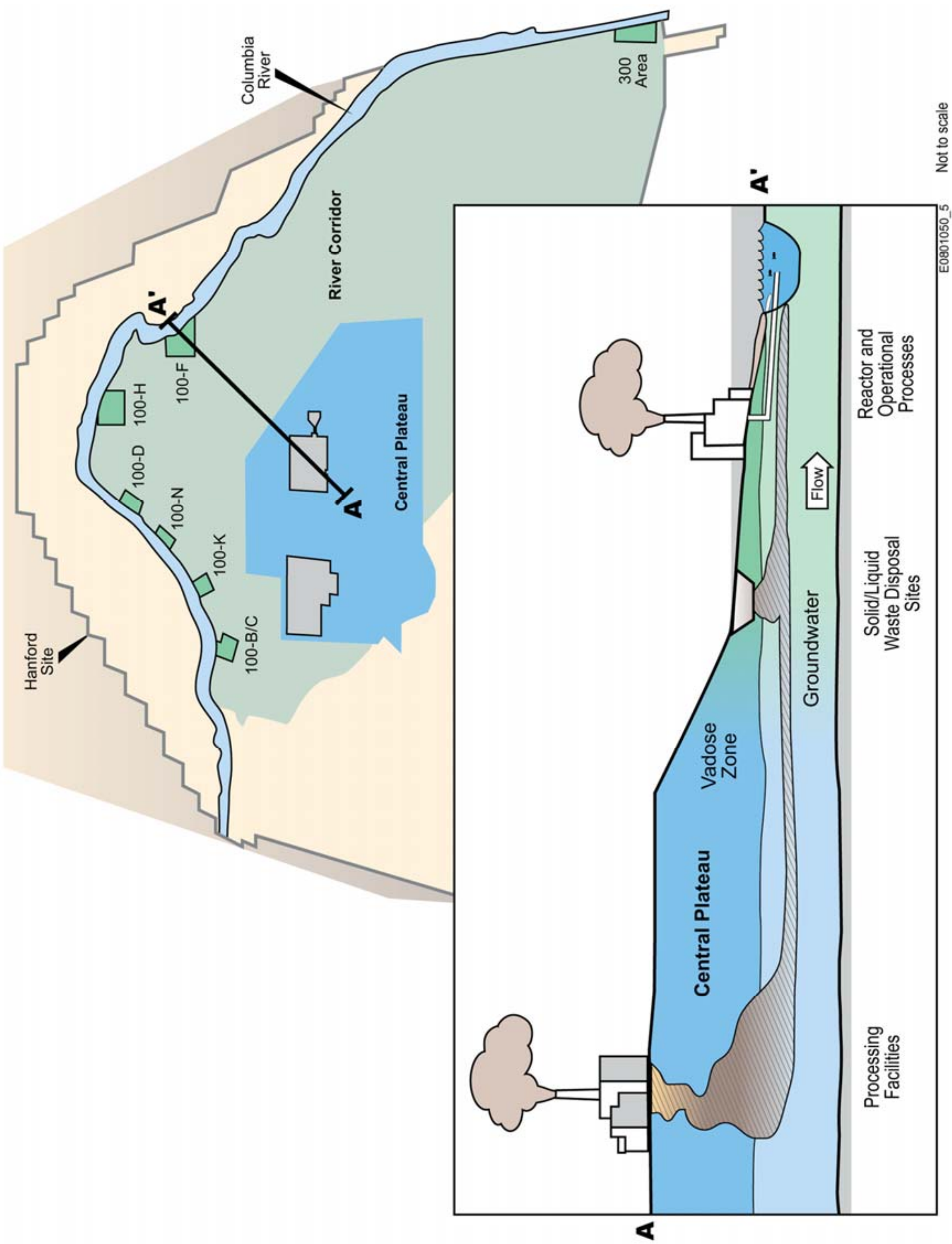
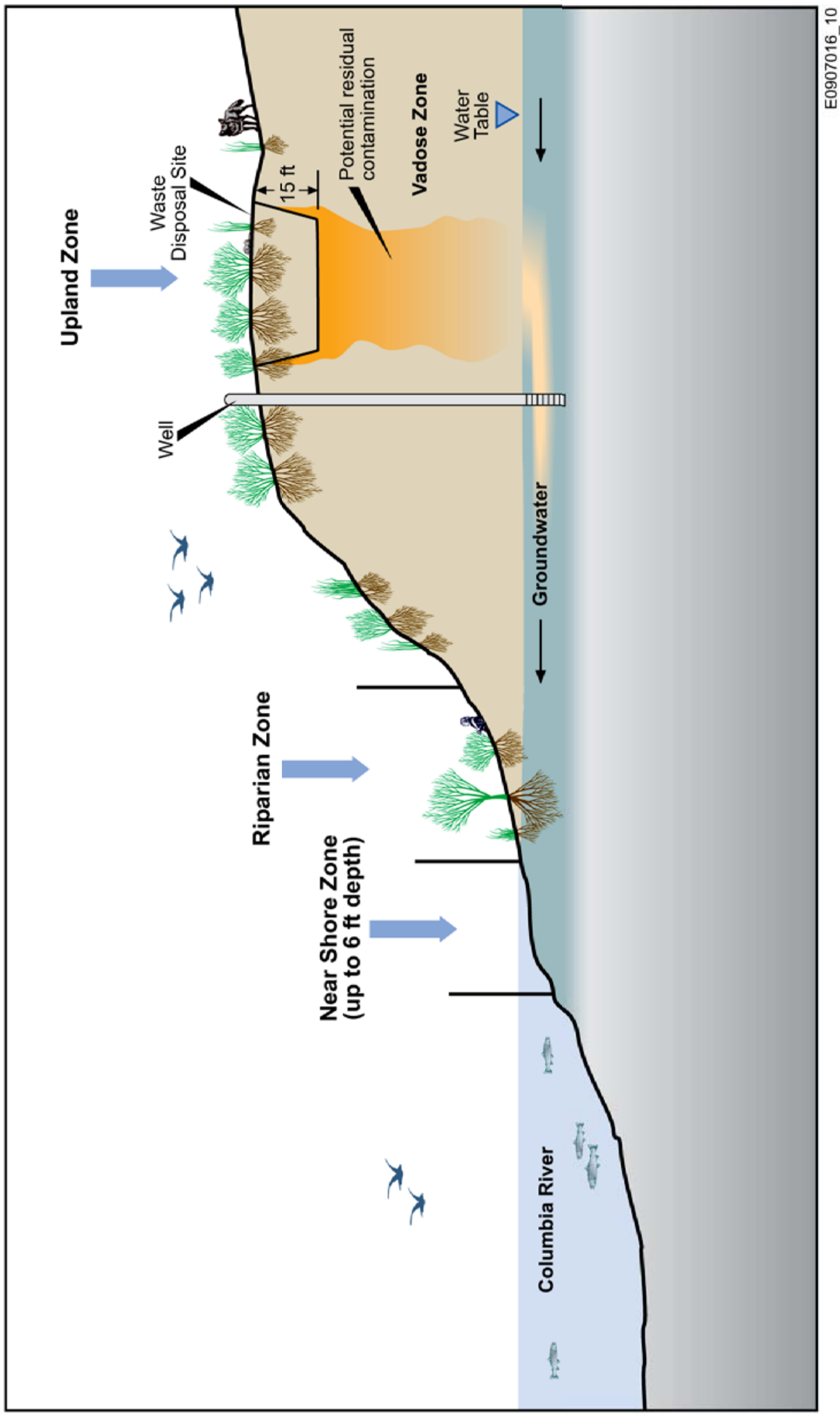


Figure 3-14. Conceptual Model of the Upland, Riparian, and Near-Shore Aquatic Zones of the River Corridor.



In the absence of any one of these components, an exposure pathway is considered incomplete, and by definition there is no risk or hazard (EPA/540/1-89/002; EPA/540/1-89/001, *Risk Assessment Guidance for Superfund, Volume 2: Environmental Evaluation Manual*). With the exception of external irradiation from radionuclides, environmental contaminants must cross a cellular barrier and enter the body of a receptor for an exposure to occur.

Contaminant sources in the River Corridor historically consisted of solid and liquid wastes generated by Hanford Site reactor operations, as described in Section 2.0. In addition to the waste site exposures in the upland environment, the potential for human exposure to contaminants in the riparian and near-shore aquatic environments of the Hanford Site is also evaluated in the HHRA. The transport mechanisms by which media and receptors outside the immediate location of waste sites can be impacted by contamination will be discussed in this section. This section also identifies potentially complete exposure pathways for the receptors and scenarios evaluated in the upland, riparian, and near-shore aquatic zones. The pathways have been identified in accordance with risk assessment guidance, and using stakeholder involvement and site-specific information as allowed by the guidance.

The exposure scenarios used to evaluate potential risks in the HHRA are described in Section 3.3.1. Details on the identification of potentially complete exposure pathways for each scenario are provided in Sections 3.3.2 and 3.3.3.

3.3.1 Present-Day and Potential Future Human Health Exposure Scenarios

Present-day exposures in the six decision areas are controlled by access restrictions. Present-day activities in these areas are limited to security surveillance and remedial action and sampling activities conducted by remedial action workers. Activities conducted by Hanford Site workers are managed under a site health and safety plan. This plan addresses worker training and protective measures to minimize potential exposures and requires monitoring of potential radiological exposure, where necessary. Because potential exposures and associated risks are monitored for these workers, they are not considered potential receptors for the HHRA.

The nature of the potential exposure scenarios to be used in the RCBRA has been the subject of numerous discussions among the Tri-Parties and various affected parties. One outcome of the early discussions was a decision to implement a pilot human health and ecological risk assessment for the 100-B/C Area. The draft pilot assessment is documented in *100-B/C Pilot Project Risk Assessment Report* (DOE/RL-2005-40). The 100-B/C Pilot Project identified five exposure scenarios: a Rural Residential scenario (called Subsistence Farmer in this report), Resident Monument Worker scenario, Industrial/Commercial Worker scenario, Recreational Use scenarios (Avid Hunter, Avid Angler, and Casual User applications), and a Native American User scenario. Among these exposure scenarios, contaminant exposure and potential health effects were quantified in the 100-B/C Pilot Project for all except a Native American User scenario.

With the exception of the Native American User scenario, the scenarios identified in the 100-B/C Pilot Project are also employed in the RCBRA. Native American User scenarios based on Tribal exposure models published subsequent to the 100-B/C Pilot Project are used in the RCBRA. The exposure scenarios evaluated in the RCBRA have been grouped according to general types of potential exposure and land use. The types of exposure scenarios evaluated in the RCBRA include the following:

3.3.1.1 Recreational and Nonresident Tribal Scenarios.

- Recreational Use scenarios: Avid Hunter, Avid Angler, and Casual User Applications (Recreational)
- Nonresidential Tribal scenario.

3.3.1.2 Occupational Scenarios.

- Industrial Worker scenario (Industrial/Commercial)
- Resident National Monument Worker scenario (Resident National Monument/Refuge).

3.3.1.3 Residential Scenarios.

- Subsistence Farmer scenario (Subsistence Farmer)
- Confederated Tribes of the Umatilla Indian Reservation (CTUIR) Native American Resident scenario (CTUIR Resident)
- Yakama Nation Native American Resident scenario (Yakama Resident).

Potential exposure media, and the spatial scale for which exposure is evaluated, are described in Section 3.3.2. The specific exposure pathways and receptors associated with each of these exposure scenarios are described in Section 3.3.3.

In order to account for the changes in contaminant concentrations over time due to radionuclide decay, risk assessment results for each exposure scenario will be presented for a number of points in time. As discussed in Section 3.4.3, cancer risk and radiation dose will be calculated using present-day radionuclide activities in soil, and with radionuclide activities in soil decayed to the years 2075 and 2150. An interval of 75 years was selected based on the approximate length of time reactors may be kept in a "safe storage" condition and the corresponding period of institutional control.

3.3.2 Potentially Complete Human Exposure Pathways: Exposure Areas, Media, and Data Sets

Environmental media relevant to the HHRA are grouped into source media and exposure media in this CSM. The source media, including surface soil, shallow zone soil, deep zone soil, sediment, and groundwater, are the principal reservoirs of residual environmental contamination in the RCBRA. A second group of media, including biota, air, and surface water, may be affected secondarily due to migration of contaminants from the source media over time. The two groups together compose the potential exposure media evaluated in the HHRA.

The spatial scale over which exposure occurs is related to the presumed activities described for the receptors for each exposure scenario listed in Section 3.3.1. For the three residential scenarios and the Industrial/Commercial scenario, most contact with soil may occur primarily within a limited area surrounding a home or workplace. By contrast, soil contact for the Recreational and Nonresident Tribal scenarios may take place over hundreds of acres. Therefore, although soil concentrations at an individual remediated waste site may represent an exposure point for Residential and Industrial/Commercial receptors, soil concentrations at any one site would have minimal impact on Recreational and Nonresident Tribal receptors who are likely to be exposed over much broader areas. The Resident National Monument/Refuge scenario is a composite scenario with respect to spatial scale, with an occupational component that pertains to broad areas and a residential component that pertains to localized exposure.

The issue of defining the appropriate spatial scale of exposure is also important for evaluating individual exposure routes involving foodstuffs. Some foodstuffs, such as garden produce or poultry products, may be raised in a relatively small area near a residence. In these cases, soil exposure point concentrations appropriate for modeling concentrations in foodstuffs may be analogous to concentrations used to calculate exposure via incidental ingestion, dermal absorption, and external irradiation around the home. However, the home range of certain game species (such as elk or game birds) can be very large, and contaminant tissue concentrations may be expected to reflect exposure across the entire home range.

Spatial scale is discussed in a general manner in this CSM. A “local-area” scale refers to exposure in a region commensurate with the area at and around an individual remediated waste site. A “broad-area” scale refers to exposure that occurs in a region as large as an individual ROD decision area and potentially as large as the entire River Corridor.

The sizes of the ROD decision areas for the RCBRA are as follows:

100-B/C ROD Area	11.53 km^2 (4.45 mi^2)
100-D/100-H ROD Area	20.31 km^2 (7.84 $mi^{2)}$
100-K ROD Area	8.99 km^2 (3.47 mi^2)
100-N ROD Area	8.88 km^2 (3.43 mi^2)
100-F/100-IU-2/100-IU-6 ROD Area	376.15 km^2 (145.23 mi^2)
	(100-F Area = 5.44 km^2 [2.10 mi^2])
300 ROD Area	145.95 km^2 (56.35 mi^2).

Waste sites in the 100-F Area operational area are aggregated with sites in the 100-IU-2 and 100-IU-6 operational areas to form a single ROD decision area. The 100-F/100-IU-2/100-IU-6 ROD decision area contains various waste sites that are not spatially contiguous. Although all decision areas include regions between the waste sites, the size of the 100-F/100-IU-2/100-IU-6 ROD decision area is such that the area between the waste sites is comparatively large.

Within the HHRA, environmental data are processed to support two activities: identification of COPCs and calculation of representative concentrations. For each activity, data must be aggregated into appropriate groups. A summary of the application of these data in the HHRA is provided below.

3.3.2.1 Upland and Riparian Soil and Biota. A crosswalk of the application of the environmental data for upland and riparian soil and biota described in Section 3.2 to spatial scale and environment within the River Corridor is shown in Table 3-9. Specifically, this table indicates which data sets are used to identify COPCs and calculate representative concentrations in these environments. More detailed information relating to use of these data to identify COPCs, calculate representative concentrations, and model exposure concentrations in unsampled media is provided in Sections 4.0 and 5.0 of the HHRA for the broad-area and local-area assessments, respectively. In each of the exposure scenarios, direct and indirect human exposures to environmental contaminants present in soil are likely to be a key aspect of the risk assessment. Direct soil exposure includes exposure routes such as incidental ingestion, inhalation of dust (i.e., suspended soil), dermal absorption, and external irradiation. Indirect exposure refers to human exposure that is mediated by transport from soil to a secondary exposure medium. For example, indirect exposure to soil contaminants may occur via plants and/or animals whose tissues contain contaminants that have been taken up from soil.

The depths at which residential human activities (e.g., gardening and lawn care) and small animal activity occur impacts potential exposure concentrations for direct receptor exposure to residual contaminants in soils. These activities are expected to mix soils primarily to a depth of approximately 0.2 m (6 in.) from the ground surface (i.e., surface soil). Soil excavation and activities such as trenching or constructing a basement during building construction would be expected to have the effect of mixing and averaging residual soil contamination to a greater depth. The depth to which excavation may cause soil to be available for subsequent surface exposure has historically been assumed during the cleanup verification process to be the 0- to 4.6-m (0- to15-ft) "shallow zone."

Soil data from 0 to 4.6 m (0 to 15 ft) are also identified in Ecology regulations as important to human exposure assessment. The 2001 version of "Model Toxics Control Act – Cleanup" (WAC 173-340-740[6][d]) states,

> For soil cleanup levels based on human exposure via direct contact or other exposure pathways where contact with the soil is required to complete the pathway, the point of compliance shall be established in the soils throughout the site from the ground surface to fifteen feet below the ground surface. This represents a reasonable estimate of the depth of soil that could be excavated and distributed at the soil surface as a result of site development activities.

At remediated waste sites in the upland environment where contaminated soil was removed to meet the IAROD cleanup levels, clean backfill was used to return the site to the original elevation. Therefore, at these sites, residual soil contamination may exist below the bottom and sides of the excavation and below a layer of clean backfill. Backfill layers vary with the depth of excavation, but generally range from 3 to 6 m (10 to 20 ft) thick. The excavation depths for the excavated sites evaluated in the HHRA are provided in Table 2-8 in Section 2.0 of this HHRA.

Use of the shallow zone soil data from sites where clean fill has been placed following remediation implies that humans have excavated the potentially contaminated residual soil and relocated it to the ground surface where exposure may take place. This excavation in turn implies that subsurface soils with residual contamination have been mixed with the overlying clean backfill. However, based on the Tri-Party Agreement, this HHRA does not apply a model of soil mixing to account for the presence of clean backfill when assessing potential exposure to residual contaminants in shallow zone soil. The degree of protective bias from assuming that subsurface contamination at excavated waste sites exists on the ground surface is bounded by comparisons of representative concentrations and risk assessment results for remediated waste sites where both RCBRA surface soil and CVP/RSVP data are available (see footnote "d" to Table 3-9). These comparisons are summarized in Section 5.0.

If volatile organic compounds (VOCs) are identified as COPCs, soil data from below 4.6 m (15 ft) may be relevant for assessing indoor exposure to contaminants that can migrate to the ground surface via gas-phase diffusion. Another potential exposure mechanism for contaminants in deep zone soil that could exist involves mixing material from water-well drill cuttings into surface soil in the vicinity of a residence. Quantification of such exposures would require development of spatially explicit soil exposure models, discussed in relation to shallow zone soil in the preceding paragraphs. Because the volume of drill cuttings will be very small relative to the volume of soil encountered by receptors when averaging exposure over many years, the potential contribution of drill cuttings to chronic health risks from soil exposures is likewise small.

Exposure to contaminants in both shallow and deep zone soil may also occur due to leaching of COPCs from soil to groundwater, followed by domestic uses of groundwater. An accurate evaluation of potential risks related to leaching of soluble contaminants from soil to groundwater over time requires soil data showing the vertical profile of concentrations of soluble COPCs between the waste site and the aquifer. Data gaps for modeling leaching to groundwater will be identified in the RI work plans for each ROD decision area and an evaluation of potential groundwater impacts presented in the RI reports.

Human exposure to upland and riparian contaminants through plants and animals is evaluated in the Residential, Avid Hunter, and Nonresidential Tribal exposure scenarios. Home-grown fruits and vegetables may have elevated levels of contaminants if they are growing in contaminated soils. Livestock, such as milk cows, beef cattle, or poultry, and wild game, such as deer or elk, may also uptake contaminants from soil and accumulate them in their tissues. Game exposure may occur both through the ingestion of plants growing on contaminated soils as well as via direct ingestion of soil while grazing.

Potential contaminant concentrations in agricultural foodstuffs, and in wild plants and game, are modeled from contaminant concentrations in soil. Cultivated crops and livestock that pertain to the Residential exposure scenarios include garden produce, fruits and berries, poultry, beef cattle, and milk cows. A potentially important consideration for food chain modeling with larger domestic animals such as cattle and milk cows is the size of the area needed to raise fodder relative to the area of contamination. As discussed above, shallow zone soil data from remediated waste site cleanup verification samples are being used to assess soil exposure at each individual waste site in this risk assessment without consideration of the presence of overlying backfill. Given this condition, it is illogical to develop a spatial model to combine local-area cleanup verification data with broad-area surface soil data in order to estimate more realistic contaminant concentrations for livestock exposure at smaller remediated waste sites.

A variety of biota data have been collected specifically for the RCBRA, providing contaminant concentrations in upland and riparian vegetation, ground-dwelling invertebrates, fledgling birds, and small mammals. The upland and riparian vegetation data are used for identifying COPCs in these environments. As discussed in Section 3.4.3 and in the uncertainty analyses of the HHRA in Sections 4.0 and 5.0, representative concentrations for wild plants were modeled from soil rather than calculated from plant tissue samples due to elevated detection limits for many organic chemicals that resulted from sample dilutions performed in the analytical laboratory.

3.3.2.2 Near-Shore Biotic and Abiotic Media. Environmental media in the near-shore environment evaluated in the HHRA include sediment, pore water, surface water from seeps and the Columbia River, and biota such as sculpin, clams, and crayfish. The application of the environmental data for near-shore media for identifying COPCs and calculating representative concentrations is shown in Table 3-10. These data are used only in the broad-area risk assessment. More detailed information relating to use of these data to identify COPCs and calculate representative concentrations and associated health risks for the broad-area assessment is provided in Section 4.0 of the HHRA.

Exposure to contaminants in Columbia River sediments via inadvertent ingestion, dermal absorption, and external irradiation is evaluated in the Recreational and Nonresident Tribal scenarios. Exposure to sediments is also a possibility for receptors in the Resident National Monument/Refuge exposure scenario. However, broad-area occupational exposure under this scenario is protectively evaluated using upland environment soil data because residual contaminant concentrations are generally higher than in the sediment data.

Sediment sampling sites in the RCBRA investigation were selected for sampling based on locations of known groundwater plumes, the results of a 2005 conductivity survey, and past biota sampling results (DOE/RL-2005-42). Many of the sampling locations are within the 300 Area (uranium plume), 100-D and 100-K Areas (chromium plumes), and 100-N Area (strontium plume). Because of uncertainties related to sampling location and sample extraction methods, and because adequate RCBRA data exist to characterize sediments in the near-shore environment, SESP sediment data are not used in the HHRA.

Human exposure to contaminants in near-shore biota is evaluated in the Residential, Avid Angler, and Nonresidential Tribal exposure scenarios. Biota data that have been collected specifically for the RCBRA include aquatic macroinvertebrates and bivalves, and fish with a very small home range. Fish data collected under the SESP program are limited, and tissue data for many of these larger-ranging species are being supplemented by additional data obtained for the Columbia River risk assessment. Therefore, the SESP data will not be used in the HHRA. The fish ingestion risk calculations in this report focus on COPCs identified in sculpin, clams, and crayfish.

Use of surface water from the Columbia River is a potentially complete exposure pathway associated with the Recreational and Nonresident Tribal exposure scenarios. Surface water samples from the Columbia River were collected as part of the RCBRA investigation at the locations where sediment samples were acquired. Because of uncertainties related to sampling location, and because adequate RCBRA data exist to characterize Columbia River surface water in the near-shore environment, SESP river water data are not used in the HHRA.

Chronic exposure to contaminants in water from seeps at the banks of the Columbia River is unlikely because flows are generally seasonal and of low volume. Risks from exposure to seep water using samples acquired under the SESP are evaluated in a semiquantitative manner in the HHRA. Pore-water data obtained in the RCBRA investigation from submerged sediments in the near-shore environment are used to support COPC identification, but pore water is not a potential human health exposure medium.

3.3.2.3 Groundwater. Groundwater in the River Corridor areas is relatively shallow and flows in the direction of the Columbia River. In addition to local sources of contamination from the individual waste sites, groundwater at some of the monitoring wells in the River Corridor may at some time harbor contamination from upgradient releases in the 200 Area.

Present-day residual contamination in groundwater is characterized in the risk assessment by the use of unfiltered groundwater monitoring well data, as described in Section 3.2.1. Monitoring well data from several years of data collection were obtained from a selection of wells identified as representative of present-day groundwater conditions within each of the ROD decision areas. Contaminant concentrations are calculated on the scale of a ROD decision area using the groundwater data to represent present-day concentrations. Evaluation of future groundwater concentrations due to leaching of soil contaminants and migration of contamination from outside of the boundaries of the River Corridor is not performed in this risk assessment.

Exposure to groundwater is evaluated for the three residential scenarios and the residential component of the Resident National Monument/Refuge Exposure scenario. Direct exposure to contaminants in groundwater is evaluated for household uses of groundwater in each of these scenarios, such as drinking and cooking (ingestion) and bathing (dermal absorption). If VOCs are measured in groundwater, indirect exposure by inhalation of VOCs in air may occur while bathing or when using groundwater in the home for other purposes. The inhalation pathway for VOCs associated with household use of groundwater is evaluated for VOCs that are identified as COPCs in groundwater. Additionally, ingestion, inhalation, and dermal exposures to COPCs in groundwater used in a sweat lodge is evaluated in the CTUIR Resident and Yakama Resident scenarios.

Indirect exposures to contaminants in groundwater may also occur if contaminated groundwater is used to irrigate a garden or provide water for livestock or by migration of vapors from groundwater upward through the vadose zone and into a building. The vapor intrusion pathway and irrigation pathway are not evaluated in the risk assessment calculations, although the potential significance of these pathways is discussed in the uncertainty analysis of the local-area risk assessment in Section 5.0 based on modeling presented in Appendix D-1.

3.3.2.4 Air. Exposure to contaminants on suspended soil (dust) or in the gas phase (VOCs and some radioisotopes) through inhalation may occur in indoor or outdoor environments for all exposure scenarios. Air concentrations used in the RCBRA are modeled from concentrations in soil due to the absence of applicable air data, particularly on the scale of individual waste sites. Concentrations of contaminants on dust in outdoor air are calculated from soil data using air dispersion modeling. Indoor concentrations of dust-borne contaminants are assumed to be equivalent to those in outdoor air.

Volatile organic compounds were not identified as COPCs in shallow zone or deep zone soil (see Section 5.0). Therefore, indoor concentrations of gas-phase contaminants emanating from the vadose zone were not evaluated quantitatively. Volatile organic compounds were identified as COPCs in groundwater; consequently, they may contribute to contamination in indoor air via volatilization from domestic uses of water in the bathroom and kitchen. The EPA models for estimating such indoor air concentrations are used to calculate exposure via this pathway.

3.3.3 Potentially Complete Human Exposure Pathways: Exposure Routes and Receptors

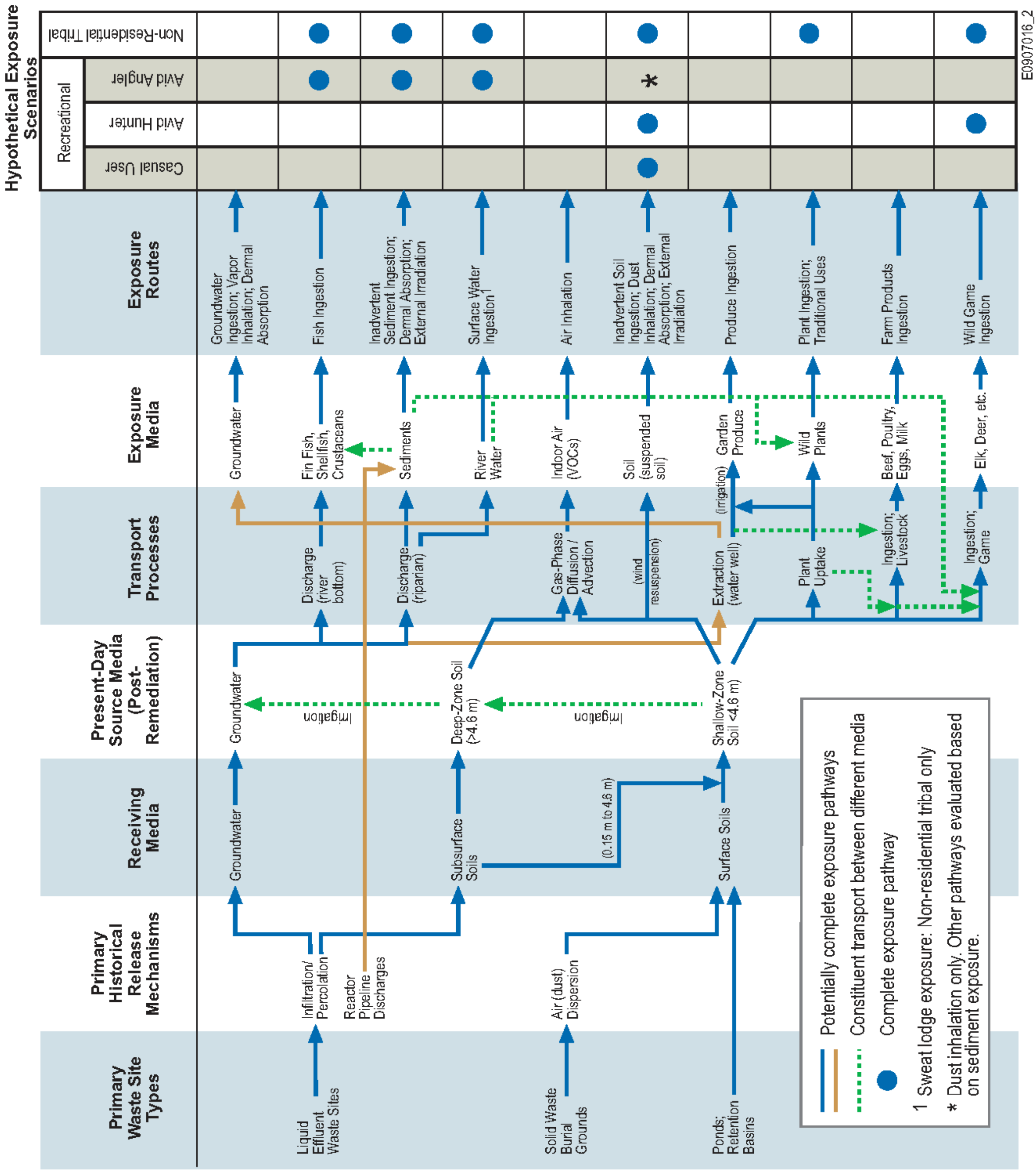
Exposure routes and receptors for each of the exposure scenarios evaluated in the RCBRA are described in the following sections.

3.3.3.1 Recreational and Nonresident Tribal Use Scenarios. Figure 3-15 shows the conceptual model of potentially complete exposure pathways for the Recreational and Nonresidential Tribal scenarios. Table 3-11 shows the exposure routes, exposure media, and exposure areas applicable to the Recreational and Nonresident Tribal exposure scenarios evaluated in the RCBRA. There are three distinct applications of the recreational exposure scenario that relate to different types of activities. Each recreational scenario has been assigned to one of the three River Corridor environments (upland, riparian, and near-shore). The Casual User application addresses occasional recreational use and is focused on activities such as walking and picnicking in riparian areas near the river. The Avid Hunter is focused on individuals who are recreational hunters, as opposed to those engaged in subsistence hunting of game species such as deer and elk. This application has been associated with upland regions of the River Corridor. The Avid Angler, like the Avid Hunter, is focused on individuals who are not engaged in a subsistence lifestyle. The Avid Angler application is associated with exposure in the near-shore region of the River Corridor.

The Nonresidential Tribal scenario includes adults and children and evaluates potential risks related to traditional tribal activities including fishing, hunting, gathering plants, and participating in sweat lodge ceremonies and other religious and cultural uses of the environment. Hunting is assumed to occur within upland regions of the River Corridor, while fishing exposures relate to riparian and near-shore regions. The Nonresidential Tribal scenario is focused on individuals engaged in a subsistence lifestyle who reside offsite but who use the River Corridor for the aforementioned purposes.

Casual User. The potentially exposed population for this exposure scenario includes adults and children. Adults and children may be exposed to surface soil contaminants via external irradiation, incidental ingestion, dermal absorption, and inhalation of dust. This scenario is focused primarily on activities such as walking and picnicking near the river where paths and benches are likely to exist. Therefore, riparian soils are employed as the key exposure medium. Although these receptors may also have some exposure to contaminants in river sediments and water, exposure to these media is a focus of the Avid Angler scenario and, therefore, is not assessed in this scenario.

Figure 3-15. Human Health Conceptual Exposure Model for the Recreational and Nonresidential Tribal Scenarios.



Avid Hunter. The potentially exposed population engaged in hunting for this exposure scenario includes adults and older children.[1] While hunting, these receptors may be exposed to surface soil contaminants via external irradiation, incidental ingestion, dermal absorption, and inhalation of dust. The potentially exposed population for ingestion of wild game is not restricted to older children and adults who actively hunt, but may also include younger children at home.

The signature exposure pathway associated with this scenario is exposure to site contaminants through the consumption of wild game. Wild game are assumed to be primarily exposed to contaminants in the upland environment through direct ingestion of soil as well as ingestion of plants containing contaminants absorbed from soil. Contaminants in riparian soil, sediments, and surface water may also contribute to contaminant concentrations in game tissues. However, the area of the upland portion of the River Corridor is much larger than the riparian area, and residual concentrations of Hanford Site-related contaminants are also likely to be higher in the vicinity of the remediated waste sites where upland sampling was focused. For these reasons, exposures to humans while hunting and the grazing area of game animals are protectively assumed to be limited to the upland environment.

Avid Angler. The potentially exposed population engaged in this exposure scenario includes adults and older children engaged in sport (rather than subsistence) fishing. Avid Anglers in this scenario may be exposed to surface sediment in the Columbia River through external irradiation, incidental ingestion, and dermal absorption while fishing or swimming in the Columbia River. Because some incidental ingestion of river water may occur during swimming, contaminant exposure via river water is also assessed. Exposure via inhalation of dust will be evaluated using soil data from the adjacent riparian environment. Although these receptors may also have some direct exposure to contaminants in riparian soil, exposure to these media is a focus of the Casual User scenario and is not assessed in this scenario. As with the Avid Hunter scenario, the potentially exposed population for ingestion of fish is not restricted to older children and adults who actively fish, but may also include younger children at home.

Nonresidential Tribal Scenario. The potentially exposed individuals addressed in the Nonresidential Tribal exposure scenario include adults and children who reside off the Hanford Site, but use portions of the site to hunt, fish, gather resources, and participate in other traditional tribal activities. Subsistence-level consumption of plant, animal, and fish resources gathered from the site is a key component of the scenario. Exposure assumptions for this scenario are based on information provided in exposure scenario documents developed by the CTUIR (Harris and Harper 2004, *Exposure Scenario for CTUIR Traditional Subsistence Lifeways*) and the Yakama Nation (Ridolfi 2007, *Yakama Nation Exposure Scenario For Hanford Site Risk Assessment, Richland, Washington*). The Nonresidential Tribal scenario is applied on a broad-area spatial scale, consistent with the nature of exposure activities such as hunting and gathering activities. In lieu of information regarding specific locations of potential fishing camps, a broad-area scale is also proposed to quantify exposure to river sediments, riparian soil, and river water during fishing activities.

[1] The EPA traditionally defines child receptors as between the ages of 1 and 6. In the Avid Hunter and Avid Angler scenarios, child receptors engaged in frequent hunting or fishing are envisioned to be older than age 6.

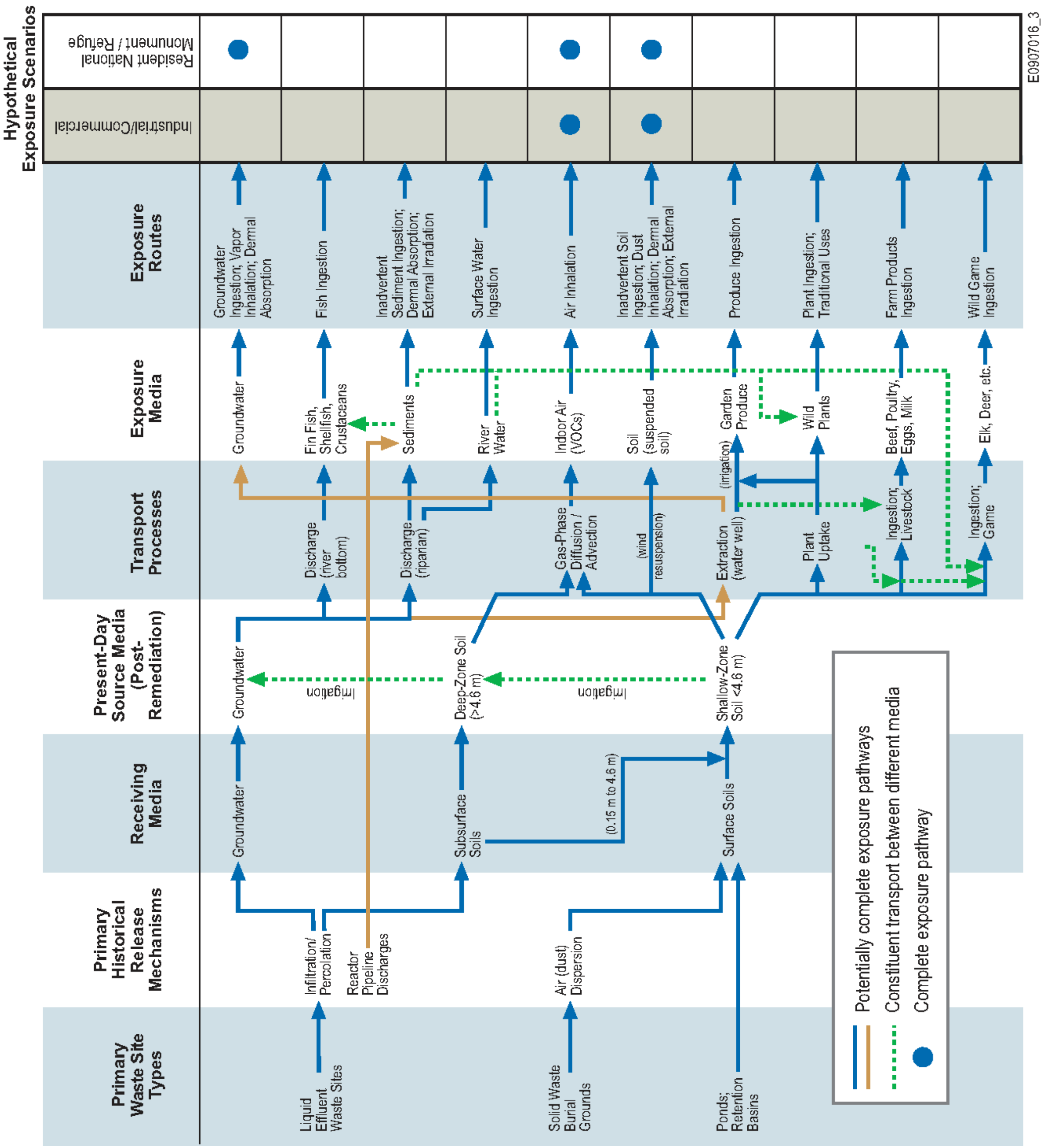
Children (beginning at an age of 2 years) and adults are assumed to be exposed to surface soil via external irradiation, incidental ingestion, inhalation, and dermal absorption. With the exception of dust inhalation, these exposure pathways also pertain to river sediments. Adults and children are also assumed to be exposed by use of river water for drinking, and adults are assumed to participate in sweat lodges with associated exposure to river water by ingestion, dermal absorption, and inhalation of volatile and semivolatile chemicals. Although children as young as age 2 may participate in the sweat lodge (Harris and Harper 2004; Section 2.2.3), the frequency of daily use cited in Section 2.3.1 of Harris and Harper (2004) would appear to be applicable to older individuals. Exposure pathways related to use of natural resources include ingestion of wild plants and game meat obtained from the site and ingestion of fish from the Columbia River.

Onsite exposure for the Nonresidential Tribal scenario is evaluated using environmental data collected from within upland, riparian, and near-shore environments. The primary exposure activities related to the upland environment are hunting and gathering wild plants. For these activities, individuals are assumed to travel to the River Corridor and spend daylight hours there but return to an offsite residence in the evening. The primary exposure activity in the riparian and near-shore environments is fishing, although individuals are also assumed to gather wild plants and participate in the sweat lodge. It is assumed that individuals reside in temporary dwellings along the river for many consecutive days while catching and drying fish. A single individual is assumed to engage in all these activities, so onsite exposures in the upland, riparian, and near-shore environments are additive.

Game tissue concentrations will be modeled based on contaminant concentrations in upland surface soil, where residual contaminant concentrations related to the remediated waste sites are likely to be highest. For plant ingestion, 50% of wild plants will be assumed to be gathered in the upland environment and 50% in the riparian environment. COPC concentrations in wild plants, as well as forage plants eaten by game animals, are modeled from the upland surface soil data.

3.3.3.2 Occupational Scenarios. Figure 3-16 shows the conceptual model of potentially complete exposure pathways for the occupational scenarios. Table 3-12 shows the exposure routes, exposure media, and exposure areas applicable to the occupational exposure scenarios evaluated in the RCBRA. These include the Industrial/Commercial and Resident National Monument/Refuge scenarios. The Industrial/Commercial scenario is a traditional scenario that has been described in various EPA documents (EPA/540/R-95/128, *Soil Screening Guidance: Technical Background Document*; OSWER 9355.0-30, “Role of the Baseline Risk Assessment in Superfund Remedy Selection Decisions”; OSWER 9355.4-24, *Supplemental Guidance for Developing Soil Screening Levels for Superfund Sites*). It is associated with upland regions of the River Corridor where permanent buildings may be erected. The Resident National Monument/Refuge scenario is a site-specific scenario that envisions a resident employee of the Hanford Reach National Monument. It pertains primarily to the upland region of the River Corridor. This scenario is described as an occupational scenario because it is defined in relation to employment.

Figure 3-16. Human Health Conceptual Exposure Model for the Occupational Scenarios.



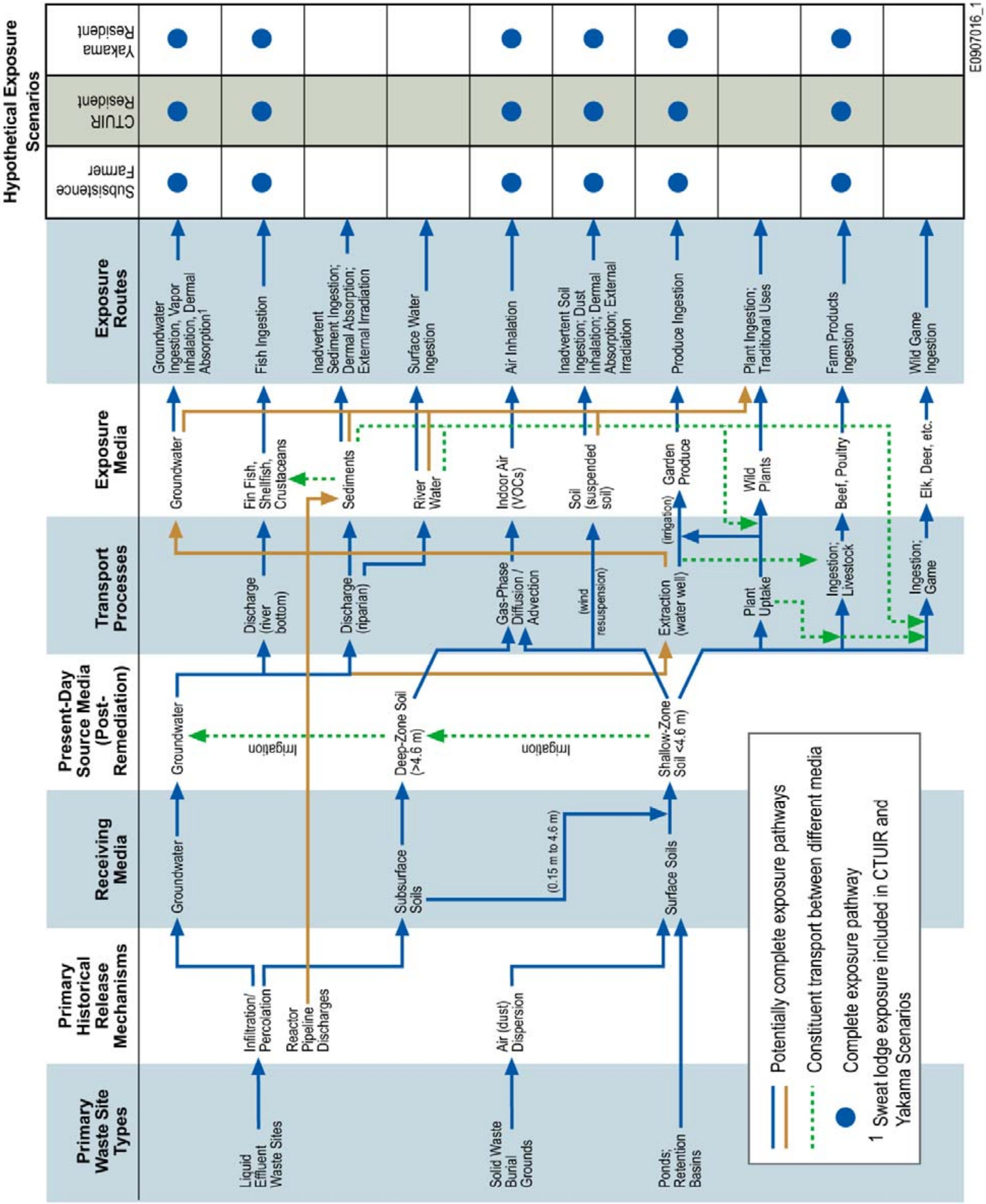
Industrial/Commercial Scenario. The potentially exposed population for this exposure scenario includes adult workers. Exposure routes for contaminants in shallow zone soil evaluated in this exposure scenario include external irradiation, incidental ingestion, dermal absorption, and inhalation. Possible exposure to contaminants in deep zone soil is limited to inhalation of gas-phase constituents in indoor air. The Industrial/Commercial scenario is specific to land-use designation for a portion of the 300 Area, but has been applied in the HHRA to all decision areas to provide risk management information.

Resident National Monument/Refuge Scenario. The potentially exposed population for this exposure scenario includes adult workers. The National Monument workers are assumed to be exposed primarily in an outdoor environment as they lead tours, conduct ecological education, or similar activities. When not working, these receptors are envisioned to live in an onsite residence associated with the Monument. Adult workers could potentially be exposed to site contaminants in shallow zone soil at their residence through external irradiation, incidental ingestion, dermal absorption, and inhalation. Possible exposure to contaminants in deep zone soil at the residence is limited to inhalation of gas-phase constituents in indoor air, but no VOCs were identified as COPCs in deep zone soil (see Section 4.0). By use of a domestic well at their residence, these receptors may also be exposed to groundwater contaminants through ingestion, dermal absorption, and inhalation of volatiles released to indoor air during showering, dishwasher use, or other household activities.

During working activities, these receptors may be exposed to contaminants in surface soil by external irradiation, incidental ingestion, dermal absorption, and inhalation. Although workers in this exposure scenario could also be exposed to contaminants in river sediments, these exposures would probably be less frequent than exposure to soil, and contaminant concentrations in upland soil are generally higher than in sediments. Exposure to sediments is considered in the Avid Angler and Nonresidential Tribal exposure scenarios.

3.3.3.3 Residential Scenarios. Figure 3-17 shows the conceptual model of potentially complete exposure pathways for the Residential scenarios. Table 3-13 shows the exposure routes, exposure media, and exposure areas applicable to the residential exposure scenarios evaluated in the RCBRA. The residential scenarios describe exposures related to a rural land-use pattern that involves home-produced foods. The Subsistence Farmer scenario envisions a diet that includes a substantial quantity of home-produced foods. The Native American Resident scenarios envision a complete subsistence lifestyle where all foods are grown at the home or (in the case of fish) caught in the Columbia River. These residential scenarios provide upper-bound estimates of human health risk for exposure activities more intensive than the conservation/preservation land uses envisioned in the recreational and occupational scenarios.

Figure 3-17. Human Health Conceptual Exposure Model for the Residential Scenarios.



Subsistence Farmer Scenario. The potentially exposed population for this exposure scenario includes adults and young children. Residents could potentially be directly exposed to site contaminants in shallow zone soil (0 to 4.6 m [15 ft] depth) through external irradiation, incidental ingestion, dermal absorption, and inhalation. Possible exposure to contaminants in deep zone soil (soils greater than 4.6 m [15 ft] depth) is limited to inhalation of VOCs diffusing through the soil column into indoor air, but no VOCs were identified as COPCs in deep zone soil (see Section 5.0). Residents may potentially be exposed to groundwater contaminants through ingestion, dermal absorption, and inhalation of volatiles released to indoor air during showering, dishwasher use, or other household activities. A major distinction between the Subsistence Farmer and Resident National Monument/Refuge scenarios is that the latter does not include agricultural activities such as gardening and raising livestock or fishing in the Columbia River.

Adults and children may also be indirectly exposed to soil contaminants from consumption of produce (fruits and vegetables) raised in a backyard garden. Exposure from consumption of products from domestic livestock is also feasible for a rural homestead and is assessed in the Subsistence Farmer scenario. Such products may include beef, poultry, eggs, and milk. Nationally, about 60% of rural households have home gardens (National Gardening Association 1987, *National Gardening Survey: 1986-1987*) and probably fewer households raise livestock for domestic consumption. Therefore, not all potential residential receptors are likely to be exposed via these biotically mediated exposure pathways. Individuals in households engaged in raising livestock and/or gardening represent a subpopulation of individuals whose activities may cause them to have a higher rate of exposure to residual contaminants in environmental media. Similarly, individuals in some households may fish in the Columbia River and supplement their diet in this manner. Exposure via ingestion of fish will also be considered in the Subsistence Farmer scenario.

As described in Section 3.3.2, COPC concentrations in shallow zone soil at excavated remediated waste sites are used in the local-area risk assessment calculations as if they represented present-day surface soil. Assessing risk at these sites under the condition that the backfill present at these sites does not exist may result in a substantial overestimate of reasonable exposure concentrations. Comparisons of local-area risk assessment results for the 10 excavated remediated waste sites where both RCBRA surface soil and CVP/RSVP shallow zone data are available (see Table 3-9) are presented in Appendix D-1. Modeling of the lateral dispersion of water during operations at liquid waste disposal sites to evaluate the potential for migration beyond the region of excavation (and hence whether surficial contamination at concentrations similar to the excavation sidewalls may exist adjacent to the site) is also described in Appendix D-1.

Native American Resident Scenarios. As discussed in Section 2.3, several local and regional tribes have ancestral ties to the Hanford Reach of the Columbia River and surrounding lands. Each tribe has been requested by the DOE to provide an exposure scenario(s) that reflects its traditional activities. The CTUIR has submitted an exposure scenario report to DOE (Harris and Harper 2004). The Yakama Nation has also submitted an exposure scenario report to DOE (Ridolfi 2007). Both of these exposure scenarios are evaluated in this risk assessment.

Exposure assumptions for these scenarios are consistent with information provided in Harris and Harper (2004) and Ridolfi (2007). Like the Subsistence Farmer scenario, the CTUIR Resident and Yakama Resident scenarios address potential exposures to COPCs in shallow zone soil (0 to 4.6 m [15 ft] depth), groundwater, and fish.

In applying the CTUIR and Yakama exposure scenarios on the scale of individual remediated waste sites, it was recognized that upper-bound exposure to site contaminants would involve continuous exposure at a residence located near a remediated waste site. Therefore, the complexity of the exposure models described in Harris and Harper (2004) and Ridolfi (2007) have been simplified and receptors have been assumed to spend effectively all of their time in the area around a residence. However, such receptors are still assumed to have access to fish harvested by other tribal members and to raise food for their consumption. Although the scenario documents discuss a number of specific and unique exposure activities, the exposure media and exposure routes for child and adult receptors in the Native American Resident scenarios in the area of a residence are analogous to those described for the Subsistence Farmer. The only additional exposure routes relate to groundwater exposures by ingestion, dermal absorption, and inhalation during the sweat lodge. As discussed in relation to the Nonresidential Tribal scenario, sweat lodge exposure is evaluated for adult receptors.

Exposure to COPCs from ingestion of plants and meat could be evaluated in an agricultural manner or by consideration of hunting and gathering. Because evaluation of risks related to gathering wild plants and hunting game animals is addressed in the Nonresidential Tribal scenario, plant and animal foods (except fish) in the Native American Resident scenarios are assumed to be domestically raised. This assumption also maximizes potential exposures related to residual contamination at an individual remediated waste site. Exposure concentrations in agricultural products are calculated from COPC concentrations in soil at the individual remediated waste sites using models that relate tissue concentrations in plants and animals to concentrations in the soil where foods are raised.

3.3.4 Summing of Risks from Potentially Complete Exposure Pathways

The HHRA is organized according to spatial scale and media. For all exposure media except groundwater, the risk assessment results are grouped into either the broad-area risk assessment (Section 4.0) or the local-area risk assessment (Section 5.0). A screening-level groundwater risk assessment is presented for each groundwater OU in Section 6.0.

Although it is desirable in principle to calculate cumulative risks for all exposure pathways that apply to one scenario, there are practical reasons why this is not always appropriate. As discussed in Section 3.3.2, exposure concentrations in groundwater within a groundwater OU are not spatially defined. Groundwater COPC concentrations are calculated as percentiles of the concentrations in all wells in a groundwater OU. However, the risk calculations for the exposure scenarios that include groundwater use (residential scenarios and Resident National Monument/Refuge) are conducted on the local-area spatial scale of a single remediated waste site. In other words, the exposure concentrations for soil are defined for a particular location in a

ROD decision area, whereas those for groundwater are not. Also, as summarized in Section 3.4.2, the groundwater assessment in this HHRA is preliminary in nature pending additional groundwater data collection to fill data gaps identified in the RI work plans for each ROD decision area. For these reasons, groundwater risks are not directly summed with risks related to other exposure media.

Although tissue contaminant concentrations for sculpin, clams, and crayfish pertain to individual ROD decision areas, there was little differentiation in tissue concentrations for near-shore COPCs among the various decision areas or between River Corridor and reference area sampling locations. Also, these species have high site fidelity and were used to identify possible Hanford-related releases, but they are not credible food fish for which to calculate chronic health risks. Therefore, the fish ingestion risk assessment results are not directly summed with risks from other exposure pathways.

The Executive Summary and Section 7.0 include a discussion of the potentially additive risks for exposure scenarios that include exposure to two or more of the exposure media described above (soil or sediment, groundwater, and fish). This discussion of additive risk pertains to the Avid Angler scenario (fish ingestion summed with exposure to sediment and surface water), the Nonresident Tribal scenario (fish ingestion summed with exposure to surface soil, sediment, and surface water), the Resident National Monument/Refuge scenario (groundwater and waste site shallow zone soil exposures are summed), and the three residential scenarios (groundwater, fish, and waste site shallow zone soil exposures are summed).

3.4 METHODOLOGY FOR EXPOSURE ASSESSMENT

3.4.1 Methods for Computation of Representative Concentrations for Soil, Sediment, Surface Water, and Biota Data

Thousands of laboratory results are used in this assessment to estimate exposure of human receptors to Hanford Site contaminants and to estimate the potential health risks to those receptors. From the large body of individual analytical results from the many samples collected, it is necessary to calculate a concentration of each COPC that represents the body of relevant data. Calculation of those representative concentrations takes into consideration the spatial and temporal scale upon which receptors may be exposed, as well as the expected exposure pathways. Field sampling is designed to support exposure estimates of this kind, and the relevance and appropriateness of existing data from other sources is evaluated in these terms as well. Representative concentrations are provided in electronic format in Appendix C, Section C-3, “Representative Concentrations.”

In the HHRA, an exposure point concentration (EPC) is an estimation of the COPC concentration in a given medium likely to be contacted by a receptor over time within the exposure area. Exposure point concentrations are calculated based on representative concentrations. The distinction between EPCs and representative concentrations for the HHRA is that representative concentrations pertain to concentrations in a sampled medium, whereas EPCs also include modeled concentrations in other exposure media.

The HHRA includes an evaluation of the reasonable maximum exposure (RME) and the central tendency exposure (CTE). An RME scenario assesses risk to individuals whose behavioral characteristics may result in higher potential exposure than seen in the average individual. A CTE exposure characterizes potential risk to an average member of the target population. The inclusion of both RME and CTE calculations provides a semiquantitative measure of the range of expected risks that may occur under a particular exposure scenario. The CTE and RME provide risk managers with an estimate of the mean and upper percentile of estimates of exposure. Even though they are intended to represent a conservative estimate of risk, RME risk calculations should not be composed of upper-bound estimates of every parameter included (EPA/230/02-89/042, *Methods for Evaluating the Attainment of Cleanup Standards, Volume I: Soils and Media*); otherwise, conservatism will compound into an unrealistic upper-bound risk estimate.

The statistical methods for calculating the upper confidence limit (UCL) values for use in risk assessment have evolved over time. Earlier EPA guidance (EPA/540/R-92/003, *Guidance for Data Usability in Risk Assessment*) provided methods for calculating UCLs assuming that the data being used had been derived from a normal or lognormal statistical distribution. This guidance also allowed for use of the maximum concentration in cases where the UCL was larger than maximum value. This early approach was deficient in several ways. Since 2002, EPA has released versions of the ProUCL software to address technical and computational issues associated with calculating UCLs from environmental data. Many of these techniques have been possible due to advances in computer hardware and software since the original EPA guidance was released in 1992. An accepted process for calculating analyte concentrations in environmental media is needed so that RME risks can be calculated.

In general, the process described in the following sections follows EPA guidance as provided in the *ProUCL Version 4.0 User Guide* (EPA/600/R-07/038).

Several issues need to be considered for determining the most appropriate methods for estimating representative concentrations for CTE and RME risk calculations.

- What was the intended use of the sample results (what were the DQOs)?
- How many sample results are available for the exposure unit?
- Are the data censored (are there nondetect sample results)?
- What estimation methods are mathematically stable for the data being evaluated, and, therefore, provide reasonable estimates of the mean and upper bound on the mean?

The following sections provide additional information on the factors and issues that are important for estimating representative concentrations for CTE and RME scenarios.

3.4.1.1 Data Sets and Exposure Units. Every representative concentration used in the risk assessment has an associated spatial scale to which the underlying data apply. Risk assessment results derived from the representative concentration pertain to that spatial scale. Spatial scales are designated as exposure units, or exposure areas, in a risk assessment. These are commonly defined on a volumetric basis for abiotic media. For example, RCBRA multi-increment soil samples represent the top 15.2 cm (6 in.) of soil within an area of approximately 1 ha (2.47 ac). Sediment samples are collected from the biologically active 0- to 10.2-cm (0- to 4-in.) zone and, in the case of samples collected from operational areas, represent conditions within the area where groundwater contaminant plumes enter the Columbia River. For biotic media, an exposure unit is a function of the home range of the sampled species. A crosswalk of spatial scale and the data sets used to calculate representative concentrations for the HHRA are provided in Tables 3-9 and 3-10.

In the case of the RCBRA data, the spatial scale used for calculation of representative concentrations depends upon the outcome of data analysis that is performed to determine if there are significantly different concentrations among sampling locations. Grouping of data for these analyses is related to the conceptual site model. Ultimately, how data are aggregated influences the method used to estimate representative concentrations largely through the number of samples, the number of detected samples, and the skewness of the sample dataset at a particular scale.

For some of the data groups and sample media combinations shown in Tables 3-9 and 3-10, sampling was not always performed with the intention of calculating a UCL. In these cases, there may be too few samples available to develop a statistical estimate of the RME representative concentration. Sometimes there are few samples because they were collected to represent “worst-case” conditions, such as biased cleanup verification focused samples based on the results of field screening measurements.

3.4.1.2 Treatment of Censored Values. A nondetect is an analytical sample result where the concentration is deemed to be lower than could be reliably identified and quantified using the method employed by the analytical laboratory. A value is reported, and a qualifier is assigned, indicating that the sample concentration was smaller than that value. The data are essentially censored at this value. Thus, nondetect results are referred to as censored data.

Many radionuclide sample results are not censored, meaning that the value reported is the actual measured concentration. These noncensored data are treated in the same manner as detected sample results.

Nondetects may correspond to concentrations that are actually or virtually zero, or they may correspond to values that are larger than zero but are below the laboratory’s ability to provide a reliable measurement. All approaches to working with nondetects use substitution values to estimate the sample results that might have been obtained with a more sensitive analytical method. Some of the most widely used methods are substitutions of the nondetects with the detection limit, half the detection limit (DL/2), or zero. The EPA's recently released *ProUCL Version 4 Technical Guide* (EPA/600/R-07/041) "strongly recommends avoiding the use of the DL/2 method even when the percentage (%) of nondetects is as low as 5%-10%." Two approaches for handling nondetects discussed in the ProUCL guidance include Kaplan-Meier estimation and regression on order statistics (ROS).

Kaplan-Meier is a nonparametric approach for estimating the mean and standard deviation of censored data that is commonly used in survival analysis. The Kaplan-Meier method provides an estimate of the sample distribution functions, adjusting for censored data. The Kaplan-Meier estimate of the sample distribution function is calculated as follows (EPA/600/R-07/041):

$$
\begin{aligned}
&\widetilde{F}(x) = 1, && x \le x'_{n'} \\
&\widetilde{F}(x) = \prod_{j = x'_j > x} \frac{n_j - m_j}{n_j} && x'_1 \le x \le x'_{n'} \\
&\widetilde{F}(x) = \widetilde{F}(x'_1) && x_{\min} \le x \le x'_1 \\
&\widetilde{F}(x) = 0 && 0 \le x \le x_{\min}
\end{aligned}
$$

where:

x = a vector of n samples
x' = a vector of n' distinct detected samples
m_j = the number of detects at x'_j
n_j = the number of samples ≤ x'_j
x_{min} = the minimum value of x.

The Kaplan-Meier estimate of the population mean is given by:

$$\hat{\mu} = \sum_{i=1}^{n'} x'_i \left[\widetilde{F}(x'_i) - \widetilde{F}(x'_{i-1})\right], \text{ with } x_o = 0 .$$

The Kaplan-Meier estimate of the standard error of the mean is given by:

$$\hat{\sigma}^2_{SE} = \frac{n-k}{n-k-1} \sum_{i=1}^{n'-1} a_i^2 \frac{m_{i+1}}{n_{i+1}(n_{i+1} - m_{i+1})}$$

where k is the number of nondetects and:

$$a_i = \sum_{j=1}^{i} \left(x'_{j+1} - x'_j\right) \widetilde{F}(x'_j),\ i = 1,2,...,n'-1 .$$

The Kaplan-Meier parameter estimates can then be used to estimate the UCL parametrically or the mean estimator can be used in a bootstrap resampling algorithm.

Alternatively, ROS is a parametric approach for imputing nondetect concentrations. Regression on order statistics estimates a linear model of the detected sample values versus the quantiles from a hypothesized probability distribution and then uses this linear model to assign values for the nondetects. The quantiles can be based on an assumed distribution such as a normal, lognormal, or gamma. The first step in computing the quantiles is to compute the plotting positions or percentiles. For example, in the case of one detection limit for the gamma distribution, ProUCL computes the plotting position for the i^{th} ordered sample value as $(i-½)/n$. The gamma quantiles are then computed using the probability statement $P(X \leq q(i)) = (i - ½)/n$, $i = 1, 2, \ldots, n$, where X represents a gamma random variable. Details for computing plotting positions for multiple detection limits can be found in *Nondetects and Data Analysis: Statistics for Censored Environmental Data* (Helsel 2005). Once the model is fit, values for the nondetects can be imputed concentrations and combined with the detected concentrations, and a mean and UCL can be estimated using a parametric UCL method.

Given the use of a linear model, at least two detected observations are needed to estimate the model. However, the reliability of the assignment is highly dependent on the number of detected values. ProUCL recommends using at least 8 to 10 detected values for a reliable model.

Both Kaplan-Meier and ROS can handle multiple detection limits. With multiple detection limits, ROS can assign values for nondetects that are higher than some detected values, which is not the case with Kaplan-Meier. Although this may appear to be a disadvantage, it can also be seen as an advantage given that there is a nonzero probability that a nondetect value is actually above the detection limit. Other trade-offs between Kaplan-Meier and ROS are typical of the general trade-offs between parametric and nonparametric approaches. When parametric assumptions are met, parametric methods are considered to be more powerful; if the assumptions are not met, then nonparametric methods are considered more powerful. Both the Kaplan-Meier and ROS approaches for handling censored values were employed in estimating means and confidence and levels and, subsequently, the values used as the CTE and RME.

3.4.1.3 Methods. This section provides an overview of the technical approach for calculating means and UCLs on the mean for the RCBRA. The decision logic for choosing an appropriate statistical method is largely based on the number of detected samples and the statistical distribution of the available samples for the spatial scale of interest. The process is based on EPA guidance as presented in ProUCL (EPA/600/R-07/038).

In *Calculating Upper Confidence Limits for Exposure Point Concentrations at Hazardous Waste Sites* (OSWER 9285.6-10), “EPA recommends using the average concentration to represent ‘a reasonable estimate of the concentration likely to be contacted over time’ (OSWER 9285.701A, *Risk Assessment Guidance for Superfund, Human Health Evaluation Manual, Part A, Interim Final*) and ‘because of the uncertainty associated with estimating the true average concentration at a site’” recommends that the 95% UCL on the mean be used for assessing a reasonable maximum exposure. There are many parametric and nonparametric methodologies available for calculating UCLs. Representative concentrations are based on the mean and UCL using both parametric and nonparametric methodologies in cases in which the number of detected samples, n', is greater than 4. To provide robustness to distributional assumptions, outliers, and methodology instabilities, the median of the calculated means and the median of the calculated UCLs are used as the CTE and RME, respectively. Methods differ based on the number of detected sample results as described below. The methods for identifying CTE and RME representative concentrations based on the number of detections was developed with input from the Tri-Parties.

Small Number of Detected Sample Results. The following is a summary of the approaches for calculating representative concentrations for detect samples sizes, n', from 0 to 4:

- $n' = 0$; then no CTE, no RME
- $n' = 1$; then the detected result is used for the CTE and RME value
- $n' = 2$; then maximum detect is used for the CTE and RME value
- $n' = 3$ or 4; then the average is the CTE, and the maximum detect is used as RME.

In some cases, noncensored radionuclide sample results may be less than zero. There were several occasions where a radionuclide representative concentration was below zero when the number of samples was four or less. These value representative concentrations were not used in the HHRA.

Large Number of Detected Sample Results – Parametric Approach. For parametric approaches, the particular mean and UCL estimation methodology used depends on the distributional assumptions made for the population that generated the data. ProUCL recommends avoiding "the use of a lognormal model even when the data appear to be lognormally distributed" because "its use often results in incorrect and unrealistic statistics of no practical merit" (EPA/600/R-07/038). ProUCL further notes that "the gamma distribution is better suited to model positively skewed environmental data sets" based on the following:

- The conclusion that use of a gamma distribution results in reliable and stable UCL values for datasets without nondetects

- The fact that there is no need to transform the data and back-transform the resulting statistics.

The gamma distribution also has the advantage that it can also be used to model normally distributed data. Given the above, the assumption is made that concentration data are gamma distributed, and the gamma shape and scale parameters can be used to estimate the UCL based on the following methods (EPA/600/R-07/041):

- Student's t-test 95% UCL
- Approximate gamma 95% UCL
- Adjusted gamma 95% UCL.

Table 3-14 provides the parameter estimation decision logic based on the gamma shape parameter, *k*, and the number of observations, *n* (EPA/600/R-07/041). For situations that include nondetects, a gamma ROS approach is used to impute concentrations prior to mean and UCL estimation.

Large Number of Detected Sample Results – Nonparametric Approach. In combination with the previously discussed parametric approaches based on the gamma distribution, the mean and the UCL is also estimated nonparametrically following the decision logic in Table 3-15. In the case of nondetects, the mean (and standard deviation for the Chebyshev approaches) is estimated using the Kaplan-Meier estimator.

Supplemental Representative Concentrations for Uncertainty Analysis. The protocol described above has been supplemented with calculations for censored COPCs based on all sample results. These calculations were used in the uncertainty analysis. Nondetects were used "as is" and were not be replaced. This implies the use of a sample-specific reporting limit for chemicals and a method-specific minimum detectable activity for radionuclides. The following is a synopsis of the supplemental representative concentrations calculations based on the number of sample results (detects and nondetects).

- If no sample results are detected, then no representative concentrations can be calculated.
- If there are from one to four sample results (at least one detect), all values are used to estimate representative concentrations as either an average or maximum of the values.
- If the data set contains five or more sample results, then three statistical methods are used to calculate the mean and the UCL of the mean based on all data (detects and nondetects). In this case, the nondetect sample results are used "as-is" (no replacement values).

The approaches described above provide both a parametric and nonparametric estimate of the mean and 95% UCL. The representative concentrations decision logic is largely based on simulation studies presented in the ProUCL technical guidance document (EPA/600/R-07/038), while the strongest theoretical basis for calculating a UCL on the mean is the central limit theorem that points to a Student's t-estimation approach. ProUCL, in some cases, recommends estimating the UCL by two approaches and taking the maximum of the estimates. Here a similar type of approach was followed in which the median of the three UCLs estimated by using the methods presented above was used as the RME.

Additional representative concentrations for different cleanup verification sample types were also calculated and are discussed in Section 5.0 of this report.

3.4.2 Methods for Computation of Representative Concentrations for Groundwater

Underlying much of the River Corridor are widely dispersed and overlapping plumes of various contaminants in groundwater (i.e., all the highest concentrations or the lowest concentrations do not occur at the same location). For the major groundwater OUs, using the typical procedure of estimating representative concentrations on a well-by-well basis would generate a large number of representative concentrations and risk estimates for the hundreds of individual wells evaluated in the HHRA. The purpose of this groundwater risk assessment is to provide risk managers with preliminary information for establishing the range of potential risks across large OUs. Therefore, contaminants in groundwater were evaluated for a range of concentrations for each COPC, with the high end of the range sufficient to cover the RME to groundwater, rather than on a well-by-well basis. The groundwater remedial investigations will focus on subareas within each of the groundwater OUs.

The 50th and 90th percentile values for each COPC from the groundwater data set in an OU were selected to represent a reasonable range of potential exposure concentrations. These representative concentrations can be used to evaluate central tendency and reasonable maximum groundwater concentrations for potential groundwater exposures. At the Hanford Site, the use of 50th and 90th percentile values for estimating groundwater risks was initially performed for the baseline risk assessment for the 200-ZP-1 OU (DOE/RL-2007-28, *Feasibility Study Report the 200-ZP-1 Groundwater Operable Unit*). That baseline risk assessment was used to support a ROD for the 200-ZP-1 OU (DOE/RL-2007-33, *Proposed Plan for Remediation of the 200-ZP-1 Groundwater Operable Unit*). This methodology does not provide risks at a specific location but, instead, results in information on the range of possible risks for each COPC across an OU. When multiple COPCs contribute to calculated risk in an OU, the cumulative risks may be overestimated unless these COPCs are collocated. Representative concentrations are provided in electronic format in Appendix C, Section C-3, "Representative Concentrations."

The 90th percentile values are intended to provide a protective estimate of groundwater COPC concentrations at almost any location within an OU, while the 50th percentiles are intended to provide a central tendency estimate of such groundwater concentrations. However, some COPC concentrations vary greatly among wells, and the number of samples also varies. For these reasons, there are instances when all or most COPC concentrations above the 90th percentile value for an OU come from just one or two individual wells. This is not necessarily problematic unless the COPC is a major risk-driver at the OU, and/or the concentrations of the COPC in a well are much higher than the 90th percentile in the OU. If exposure at the location of any individual well could result in calculated health risks greater than those associated with the 90th percentile, then those wells were identified and the groundwater risk assessment for the OU supplemented with an additional well-specific risk assessment. Identification of monitoring wells for the supplemental well-specific risk assessment calculations is described in Section 6.0.

The methods for the calculation of these percentiles, the mean, and 95% UCL in an individual monitoring well are described below. In the case of the 90th percentiles for an entire OU, the intent is to capture a concentration that is near the upper bound of results across a large and heterogeneous area, rather than an average exposure concentration at any particular location. When calculating a representative concentration for an individual monitoring well, where the results pertain to a specific location, it is appropriate to employ the 95% UCL for the RME concentration, as discussed in Section 3.4.1.

Operable unit CTE and RME values were computed as the 50th and 90th percentiles, respectively, using all groundwater data from the OU, with the exception of data from the individual well-analyte combinations mentioned above. The sample-specific detection limit was substituted for nondetects. The p^{th} percentile is a value such that p percent of the data lies below the value and 100 – p percent of the data lies above the value. For example, the 50th percentile (the median) of a data set with sample size 8 can be defined as the average of the fourth and fifth values because four data points lie below it and four lie above. However, any value between the fourth and fifth values shares this trait, so they, too, can be said to be the median. This lack of uniqueness has led to numerous percentile calculation methods. The method used in this report is the method described in EPA's *Data Quality Assessment: Statistical Tools for Practitioners* (EPA/240/B-06/003). First, the minimum value is assigned to the 0th percentile, and the maximum value is set to the 100th percentile. Then, linear interpolation between order statistics is used to determine the 90th percentile. As sample size increases, the difference between the results of the different methods for calculating percentiles becomes negligible. A summary of the number of groundwater percentiles calculated with fewer than 10 samples, and with 10 or more samples, is provided in Table 3-16.

The CTE and RME values for the individual well-analyte combinations were calculated as the arithmetic mean and a 95% UCL, respectively. The method of UCL computation was selected using the decision logic described in Section 3.4.1. Across the six individual well-analyte combinations, there was only one nondetect value. For simplicity, the detection limit was substituted for this nondetect rather than imputation of a value as described in Section 3.4.1.

3.4.3 Methods for Modeling Exposure Point Concentrations from Representative Concentrations in Sampled Media

An EPC is the concentration of a contaminant in an exposure medium at the time and location where a receptor may contact that medium. Exposure point concentrations are calculated based on representative concentrations in sampled media, as described in Sections 3.4.1 and 3.4.2. The difference between EPCs and representative concentrations is that representative concentrations are calculated from analytical results for a sample medium, whereas EPCs also include modeled concentrations in other exposure media and/or at future times.

The availability of data, the size of an area related to potential human exposure, data comparability issues, and concentration trends in the environmental data all contribute to decisions on the spatial scale over which data are aggregated to calculate EPCs. The spatial scale for which EPCs were calculated for different combinations of environment and media is summarized in Tables 3-9 and 3-10.

Risk calculations for the soil source term are conducted to evaluate potential exposures at different times in the future. The point in time when the hypothetical future exposure scenarios are applied may be an important consideration impacting the calculated risks. As shown in Table 2-7, all but one of the 100 Area reactors (i.e., 105-B) are being placed in interim safe storage (ISS) to allow radiation levels to decrease prior to final disposition. This ISS status indicates additional decommissioning operations may be conducted at these reactors, which in turn suggests that unrestricted development is particularly unlikely in the vicinity of the reactors during this interim period. In order to provide risk managers with information on potential risks for exposures that may not occur until some future date, risk assessment calculations are conducted using present-day radionuclide activities that have been decayed to the years 2075 and 2150. These calculations supplement the assessment of risks using present-day COPC concentrations in environmental media.

The RESidual RADioactivity (RESRAD) computer code was used to calculate the ratio of radionuclide soil EPCs for the years 2075 and 2150 to the measured activities used to assess present-day risks. Although radionuclide soil data used in the HHRA were collected at times ranging between 1990 and 2007, the majority of the data were collected between 1995 and 2005. The ratio of present-day to future radionuclide activity was calculated assuming that all measured radionuclide activities pertain to the year 2005. The ratio of present-day to future soil activities for radionuclide soil COPCs is presented in Table 3-17. Additional details on the use of RESRAD to calculate decayed activities in soil are provided in Appendix D-1.

A discussion of the methods for modeling EPCs in unsampled media is provided below. Additional details on the models used to calculate EPCs for ambient dust, VOCs in indoor air, produce, and various animal products are provided in Appendix D-1. The EPCs are provided in electronic format through a web-based interface at http://rcbra.gisdt.org and are also tabulated as an electronic attachment to Appendix D-5 of this report. The web site interface for the RCBRA was developed as a repository and interface for analytical data and risk assessment results for this report.

3.4.3.1 EPCs for Dust in Ambient Air. Exposure point concentrations related to wind-borne soil particulates in ambient air (e.g., dust) are estimated using a screening-level model that relates soil concentrations to the concentration of respirable particles in air. The specific model that is used for these calculations is EPA's particulate emission factor (PEF) model (OSWER 9355.4-24, *Supplemental Guidance for Developing Soil Screening Levels for Superfund Sites*), which combines an atmospheric dispersion term with a particulate emission model related to wind erosion. Details of the PEF model, and input values used for calculating dust concentrations in ambient air, are provided in Appendix D-1.2.0.

Dust concentrations in air are calculated for both individual remediated waste sites (local-area risk assessment) and for broader areas (the River Corridor for the upland environment; each ROD decision area for the riparian environment). For local-area receptors, dust exposures to the waste site source and the upland environment source are additive because the remediated waste sites lie within the River Corridor.

Inhalation risks based solely on broad-area soil EPCs are used in the recreational and nonresident Tribal scenarios and the occupational portion of the Resident National Monument/Refuge scenario. Because airborne dust surrounding a residence or workplace may include contributions from surrounding areas, the local-area risk calculations for residential scenarios, the

Industrial/Commercial scenario, and the residential component of the Resident National Monument/Refuge Exposure scenario include a sum of exposure to both local-area and broad-area airborne dusts. A default size of 0.4 ha (1 ac) is assigned to the potential source term for the local-area dispersion model calculations. An area of 202.5 ha (500 ac), the largest area for which EPA dispersion model calculations are documented (OSWER 9355.4-24), is used for the broad-area dispersion model calculations. Exposure point concentrations for dust in ambient air are provided in electronic format through a web-based interface at http://rcbra.gisdt.org and are also tabulated as an electronic attachment to this report.

3.4.3.2 Indoor Air EPCs. Indoor air concentrations of VOCs related to domestic uses of groundwater contaminated with VOCs are estimated using models described in *Volatilization Rates from Groundwater to Indoor Air, Phase II* (EPA/600/R-00/096). This guidance describes the development of models for estimating chemical emissions from washing machines, dishwashers, showers, and bathtubs. The models make use of source- and chemical-specific mass transfer coefficients and air exchange rates for the shower and appliances to estimate VOC releases from water to indoor air.

Models described in EPA/600/R-00/096 were used to estimate exposure to VOCs in domestic water while bathing. Potential exposures related to other water uses (such as dishwashers and washing machines) are subject to high degrees of uncertainty and were not quantified. Uncertainties and biases related to these models are discussed in the uncertainty analysis of the local-area risk assessment in Section 5.0. Details of the volatilization model and input values used for calculating VOC concentrations in indoor air are provided in Appendix D-1.3.0. Another potential exposure route to VOCs in groundwater is migration of gas-phase VOCs upward through vadose zone soil and into a residential or commercial building. Risks related to this exposure pathway were not quantified in the HHRA, but an evaluation of the potential magnitude of exposure via this pathway using EPA's Johnson and Ettinger vapor intrusion model is provided in Appendix D-1 and discussed in the uncertainty analysis of the local-area risk assessment in Section 5.0. Because VOCs were not identified as COPCs in remediated waste site soils, evaluation of vapor intrusion from soils to indoor air is not performed in the HHRA.

3.4.3.3 Sweat Lodge Air EPCs. Exposure to COPCs in sweat lodge air is evaluated for the Nonresidential Tribal, CTUIR Resident, and Yakama Resident scenarios. Appendix 4 of Harris and Harper (2004) provides an exposure assessment methodology for calculating contaminant air concentrations associated with the use of contaminated water in a sweat lodge. Contaminants are assumed to be introduced into the sweat lodge predominantly through water poured over heated rocks that is used to create steam. Equations are provided for both volatile and nonvolatile contaminants. For volatile and semivolatile organic compounds, Equation 7 of Harris and Harper (2004) provides the following method for calculating air-phase concentrations in the sweat lodge:

$$EPC_{air,v} = C_w \cdot \left(\frac{V_{w,total}}{2} \right) \cdot \left(\frac{1}{\frac{2}{3} \cdot \pi \cdot r^3} \right) \quad \text{Equation 3-1}$$

where:

$EPC_{air,v}$ = exposure point concentration of volatile or semivolatile compound in air (mg/m^3)

C_w = concentration of volatile or semivolatile compound in water (mg/L)

$V_{w,total}$ = total volume of water used to create steam during the lodge (L)

R = radius of a hemispherical lodge (m).

The last term in the equation represents the internal volume of the sweat lodge. Values for $V_{w,total}$ (4 L) and r (1 m) were obtained from Appendix 4 of Harris and Harper (2004). The equation for volatile and semivolatile chemicals will be used for all organic chemicals.

Appendix 4 of Harris and Harper (2004) also provides equations for estimating air-phase contaminant concentrations for nonvolatile chemicals, including metals and most radionuclides.

For nonvolatile organic compounds, Equation 14 of Harris and Harper (2004) provides the following method for calculating air-phase concentrations in the sweat lodge:

$$EPC_{air,nv} = C_w \cdot \left(\frac{MW_w}{R \cdot T \cdot \rho_w} \right) \cdot EXP\left(18.3036 - \frac{3816.44}{T - 46.13} \right) \quad \text{Equation 3-2}$$

where:

$EPC_{air,nv}$ = exposure point concentration of nonvolatile compound in air (mg/m^3)
C_w = concentration of volatile or semivolatile compound in water (mg/L)
MW_w = molecular weight of water (AMU)
R = ideal gas law constant (0.06237 mm Hg – m^3 / gmole - K)
T = temperature of the sweat lodge (K)
P_w = density of liquid water (g/L).

The model presented in Harris and Harper (2004) and reproduced in Equation 3-2 calculates the concentration of nonvolatile contaminant in the air as a function of the amount of water needed to create a water vapor-saturated atmosphere in the lodge. Appendix D of Harris and Harper (2004) states, "Nonvolatile COPCs become airborne as an aerosol as the water they were carried in vaporizes." This implies that water is added to the heated rocks to constantly maintain saturated steam in the lodge, and nonvolatile solute presumably becomes a respirable aerosol in equilibrium with the amount of water vapor. There is no physical basis provided in Harris and Harper (2004) to correlate respirable aerosol concentrations with saturated water vapor, and exposure concentrations of nonvolatiles in sweat lodge air calculated in this manner are considered physically implausible. Nevertheless, this model was used in the CTUIR Resident exposure scenario for consistency with Harris and Harper (2004). Based on correspondence between DOE and the Yakama Nation discussing these technical concerns, inhalation risks for nonvolatile COPCs in the sweat lodge are not calculated for the Yakama Resident scenario. Sweat lodge inhalation risks for nonvolatile COPCs are not calculated for the Nonresident Tribal exposure scenario. A comparison of nonvolatile exposure concentrations in sweat lodge air modeled using Equation 14 of Harris and Harper (2004) with aerosol measurements from boiling and flashing of aqueous solutions is presented in Appendix D-1 and discussed in the uncertainty analysis in Sections 4.7 and 6.9 of the HHRA.

Exposure point concentrations for sweat lodge air are provided in electronic format through a web-based interface at http://rcbra.gisdt.org and are also tabulated as an electronic attachment to this report.

3.4.3.4 Garden Produce Exposure Point Concentrations. Exposure to COPCs in garden produce is evaluated for the three residential scenarios. Contaminant concentrations in garden produce (e.g., fruits and vegetables) are modeled from the local-area soil representative concentrations. Exposure point concentrations for metals and radionuclides in plant tissues are calculated based on root uptake using plant-soil concentration ratios ($K_{p\text{-}s}$) obtained from the RESRAD computer code or other sources as described in Appendix D-1.4.0. For organic chemicals, $K_{p\text{-}s}$ values were calculated as a function of a chemical's octanol-water partition coefficient based on EPA methodology (EPA/530/R-05/006, *Reference Guide to Non-Combustion Technologies for Remediation of Persistent Organic Pollutants in Stockpiles and Soil*). The $K_{p\text{-}s}$ values are expressed as the linear ratio of the concentration of a COPC in plant tissue to the concentration in soil. Details of the modeling of COPC concentrations in produce from the soil data are provided in Appendix D-1.4.0. The potential for protective bias when applying linear ratios to high soil concentrations of COPCs is discussed in the uncertainty analysis of the local-area risk assessment in Section 5.0.

As discussed in the CSM (Section 3.3.4), risks related to groundwater contamination are not summed with risks related to the soil source term. Therefore, EPCs for garden produce do not account for potential contribution from groundwater contaminants due to the use of groundwater for irrigation. The potential for exposure to groundwater contaminants via agricultural pathways is discussed in the Uncertainty Analysis of the local-area risk assessment in Section 5.0.

Exposure point concentrations for garden produce are provided in electronic format through a web-based interface at http://rcbra.gisdt.org and are also tabulated as an electronic attachment to this report.

3.4.3.5 Poultry and Egg Exposure Point Concentrations. Exposure to COPCs in poultry and eggs is evaluated for the Subsistence Farmer exposure scenario. Evaluation of human exposure to soil contaminants via animal products has not traditionally been common in Superfund risk assessments. Relatively recent EPA guidance for conducting HHRA for hazardous waste incinerators addresses transfer factors for chicken tissue and eggs for organic chemicals, but for only a limited number of metals, including mercury compounds, cadmium, selenium, and zinc (EPA/530/R-05/006). By contrast, evaluation of exposure via animal products (particularly beef and milk) has long been routine in radiation dose assessment.

Exposure point concentrations for metals and radionuclides in chicken tissue and eggs are calculated based on transfer factors from EPA publications or other sources as described in Appendix D-1.4.2. For organic chemicals, transfer factors were calculated as a function of a chemical's octanol-water partition coefficient based on EPA methodology (EPA/530/R-05/006). Transfer factors describe the relationship between intake of a contaminant in feed and the associated tissue concentration in an animal. In the HHRA, the poultry transfer factors will be applied to uptake of contaminants in soil rather than feed. It is assumed that chicken feed is store-bought, rather than produced from grain grown onsite, and that exposure to soil contaminants for free-range chickens is a result solely of their foraging habits. Details of the modeling of COPC concentrations in poultry and eggs from the soil data are provided in Appendix D-1.4.

Exposure point concentrations for poultry and eggs are provided in electronic format through a web-based interface at http://rcbra.gisdt.org and are also tabulated as an electronic attachment to this report.

3.4.3.6 Beef and Milk Exposure Point Concentrations. Exposure to COPCs in beef and milk is evaluated for the three residential scenarios, with the exception of the CTUIR Resident for which only beef ingestion is specified in Harris and Harper (2004). The modeling of exposure point concentrations in beef and milk is conceptually similar to that described for poultry and eggs. In both cases, transfer factors are used to describe the relationship between intake of a contaminant in feed and the associated tissue concentration in an animal. In the case of poultry, all contaminant intake was assumed to occur due to ingestion of soil particulates. Penned cattle are assumed to be exposed via ingestion of fodder plants grown on contaminated soils.

Beef cattle or milk cows are assumed to be penned near a homestead and exposed to soil contaminants via ingestion of home-grown fodder. Contaminant of potential concern concentrations in fodder are modeled from representative concentrations in remediated waste site soil. For some remediated waste sites, the area of a site may not be sufficiently sized to grow enough hay to support beef and milk cattle. For example, in the RESRAD computer code, if the size of the contaminated zone is less than 2 ha (4.94 ac), risk results for the beef and milk ingestion pathways are reduced proportionally (ANL/EAD-4, *User's Manual for RESRAD Version 6.0*, Section D.2.1.2). This will result in a protective bias when exposure for these pathways is assessed for smaller sites. Details of the modeling of COPC concentrations in beef and milk from the soil data are provided in Appendix D-1.4.3.

Beef cattle might also be raised by grazing them over a larger area than a remediated waste site. This is analogous to the assessment of exposure via game meat in the Avid Hunter and Nonresident Tribal scenarios, where broad-area soil concentrations are used to model tissue concentrations in game animals. Because exposure via grazing animals is addressed in these scenarios, and because the residential scenarios are defined based on local exposure at a residence, only penned cattle and milk cows are assumed for calculating beef and milk EPCs.

Exposure point concentrations for beef and milk are provided in electronic format through a web-based interface at http://rcbra.gisdt.org and are also tabulated as an electronic attachment to this report.

3.4.3.7 Wild Plant Exposure Point Concentrations. Exposure to COPCs in wild (native) food plants is evaluated for the Nonresident Tribal scenario. Exposure point concentrations for contaminants in the tissues of edible native plants were modeled from representative concentrations measured in upland and riparian surface soil. Although RCBRA investigation upland and riparian plant tissue data were collected for the ecological risk assessment, they were not used to represent COPC concentrations in wild plants. The upland plant data for organic chemical COPCs (which are all pesticides) have elevated detection limits resulting from sample dilutions performed in the analytical laboratory. For example, the dieldrin plant data are censored (i.e., reported as “less than” values) at an average concentration of about 0.04 mg/kg, corresponding to a potential Nonresident Tribal cancer risk of approximately 5×10^{-3}. When EPCs are based on statistical estimates that include these censored results a positive bias results. When there are few detects and plant EPCs are based only on detected concentrations (about 50% of the time), the high detection limits also bias the sensitivity analysis using supplemental EPCs that include nondetect results at the reporting limit (see Section 4.0).

As will be described in Section 4.2, the number of upland surface soil sampling locations does not allow for the calculation of broad-area soil EPCs for each individual ROD decision area. In practice, the harvesting of wild plants over time is likely to integrate plant tissue concentrations over large areas (Ridolfi 2007), and the use of soil data solely from the area around remediated waste sites to estimate upland plant COPC concentrations may overestimate Hanford Site-related contaminant exposure from plants in the upland environment. Riparian plant tissue EPCs are calculated for each individual ROD decision area because there are adequate riparian soil data in each area and this provides risk managers with information to distinguish potential differences in risks among the six ROD areas. Because chronic exposure involves collecting plants over a larger area and many years, wild plant EPCs calculated for a single ROD decision area may produce a bias in the risk results for any individual ROD decision area. If significant risks are calculated for a ROD decision area due to riparian exposures, this will be discussed in the uncertainty analysis for the broad-area risk assessment in Section 4.0. A comparison of measured and modeled concentrations in upland plants is also provided in the uncertainty analysis in Section 4.0.

Exposure point concentrations for wild plants are provided in electronic format through a web-based interface at http://rcbra.gisdt.org and are also tabulated as an electronic attachment to this report.

3.4.3.8 Wild Game Exposure Point Concentrations. For the Avid Hunter and Nonresidential Tribal scenarios, broad-area upland soil concentrations are used to model tissue concentrations in game animals. This modeling, described in Appendix D-1, provides estimates of tissue concentrations based on animal ingestion of soil and grazed (wild) plants. As discussed above, COPC concentrations in wild plants are modeled from surface soil data. The available game animal tissue data for broad-area soil COPCs are limited to deer and rabbit liver data for several metals collected under the SESP. These results are used for a semiquantitative comparison to the modeled concentrations in order to support uncertainty analysis for these calculations. Upland soil MIS data are used rather than riparian soil data because the size of the upland area is considerably larger and because upland samples collected in the vicinity of remediated waste sites are protectively biased.

Existing potential sources of information relating to contaminant concentrations in wild game have been summarized in Appendix C of the *Risk Assessment Work Plan for the 100 Area and 300 Area Component of the River Corridor Baseline Risk Assessment* (DOE/RL-2004-37). Animals for which tissue data were collected under the SESP include elk, deer (mule deer and black-tailed deer), and cottontail rabbit. Surface Environmental Surveillance Program data are also available for game birds including Canada geese, pheasant, and quail. However, the data for soil COPCs is very limited, and these data may not be specifically associated with known areas of residual contamination and, in some cases, may reflect levels of contamination that represent preremediation rather than current conditions.

Exposure point concentrations for game tissue are provided in electronic format through a web-based interface at http://rcbra.gisdt.org and are also tabulated as an electronic attachment to this report.

3.4.4 Methods for Calculation of Intake

3.4.4.1 Overview. As discussed in *Risk Assessment Guidance for Superfund, Human Health Evaluation Manual, Part A* (EPA/540/1-89/002), the RME estimate is generally the principal basis for evaluating potential risks at a Superfund site. Later EPA guidances (OSWER 9285.7-17, "Role of the Ecological Risk Assessment in the Baseline Risk Assessment") recommended including CTE estimates in addition to RME estimates of risk. In general, an RME estimate of risk is at the high end of a risk distribution (90^{th} to 99.9^{th} percentiles), whereas the CTE estimate is associated with the mean or 50^{th} percentile of a risk distribution (EPA 540/R-02/002, *Risk Assessment Guidance for Superfund: Volume 3 - Part A, Process for Conducting Probabilistic Risk Assessment*). An RME scenario assesses risk to individuals whose behavioral characteristics may result in much higher potential exposure than seen in the average individual. A CTE scenario assesses potential risk to an average member of the population. The inclusion of both RME and CTE calculations provides a semiquantitative measure of the range of expected risks that may occur under a particular exposure scenario. Only upper percentile exposure parameter values are available for the Native American scenarios (Harris and Harper 2004, Ridolfi 2007). Central tendency exposure calculations are not conducted for the CTUIR Resident, Yakama Resident, and Nonresidential Tribal scenarios.

The basic structure of the exposure equations used in this assessment were obtained from EPA/540/1-89/002. The general intake equation for chemicals that serves as the basis for the pathway-specific equations is as follows:

$$\text{Chemical Intake} = \frac{C_i \times CR \times EF \times ED}{BW \times AT} \qquad \text{Equation 3-3}$$

where:

Intake = rate of chemical available for uptake at an exchange boundary (mg/kg body weight/day)

C_i = concentration of chemical *i* at exposure point (e.g., mg/kg soil, mg/L water)

CR = contact rate with the environmental medium (e.g., mg soil ingestion per day; L water ingestion per day)

EF = exposure frequency (days/yr)

ED = exposure duration (yr)

BW = body weight (kg)

AT = averaging time for toxicological effects (days).

Intake is specified as the rate at which a chemical becomes available for uptake at an exchange boundary, such as the walls of the gastrointestinal tract or the skin. Hence, it is not equivalent to an absorbed dose, which is the amount of chemical actually entering the bloodstream across an exchange boundary.

Separate intake calculations are performed for adults and children when evaluating noncarcinogenic effects because the averaging time over which effects are assessed is equal to the exposure duration (EPA/540/1-89/002). However, because cancer risk is expressed as a probability averaged over a lifetime, exposure as a child and adult is integrated in intake calculations for carcinogenic effects.

Intake for radiation risk and dose is calculated in a somewhat different manner than either chemical cancer risk or hazard. As described in Chapter 10 of EPA/540/1-89/002, the general intake equation for radiation dose is analogous to that for chemical exposures, except that averaging time and body weight are omitted. These terms are effectively incorporated within the radionuclide cancer slope factors and dose conversion factors used to evaluate radiation health effects (see Sections 3.5.4 and 3.5.5).

Pathway-specific intake equations and associated exposure parameter values are described for each exposure pathway in the following subsections. Instead of chemical mass, radionuclide activity (e.g., pCi) is used to quantify the amount of a radionuclide in an environmental medium. The general intake equation for radiation dose or cancer risk that serves as the basis for the pathway-specific equations is as follows:

$$\text{Radionuclide Intake} = C_i \times CR \times EF \times ED \qquad \text{Equation 3-4}$$

where:

Intake = rate of radionuclide available for uptake at an exchange boundary (pCi)

C_i = concentration of radionuclide *i* at exposure point (e.g., pCi/g soil, pCi/L water)

CR = contact rate with the environmental medium (e.g., mg soil ingestion per day; L water ingestion per day)

EF = exposure frequency (days/yr)

ED = exposure duration (yr).

In practice, radionuclide dose is generally compared with dose thresholds that are specified on an annual basis. Therefore, the exposure duration term is generally omitted from the radionuclide equation when it is applied to radiation dose, resulting in intake expressed in units of pCi/year.

For the exposure scenarios that include both child and adult receptors, the average radionuclide intake across both children and adults is used in the radiation dose calculations. The average radionuclide intake and average external irradiation for these scenarios is used in conjunction with dose conversion factors that integrate dose over a potential 30-year period. The use of these dose conversion factors is discussed in more detail in Section 3.5.5 and Appendix D-3 of this report.

3.4.4.2 Soil or Sediment Ingestion. Chemical intake via soil ingestion is calculated using the following equation. This equation must be modified when calculating intake for carcinogenic chemicals or radionuclide cancer risk by summing child and adult body-weight averaged intakes, expressed as $[(IR_s \times EF \times ED)/BW]$. For radionuclide intake, the equation would be further modified to exclude body weight and averaging time, as indicated in the overview of Section 3.4.4.1.

$$\text{Intake (mg/kg}\cdot\text{d)} = \frac{C_{s,i} \times B_s \times IR_{s,d} \times EF \times ED \times 10^{-6}\ kg/mg}{BW \times AT} \qquad \text{Equation 3-5}$$

where:

- $C_{s,i}$ = concentration of contaminant *i* in exposure area soil (mg/kg soil)
- B_s = relative bioavailability from soil in the gastrointestinal tract (unitless)
- $IR_{s,d}$ = daily soil ingestion rate (mg of soil/day)
- EF = exposure frequency (day/yr)
- ED = exposure duration (year)
- BW = body weight (kg)
- AT = averaging time (day).

The daily soil ingestion rate (IR_d), generally published as a daily rate, may be modified on a scenario-specific basis to account for the fraction of waking hours assumed to be spent within the exposure area (T_{site}). In other words, IR_d is assumed to apply evenly to all waking hours, so the amount of soil ingestion that occurs at a site is proportional to the fraction of waking hours spent at the site. For all exposure scenarios except for those related to Native American receptors, a site-specific soil ingestion rate (IR_s) is calculated as follows:

$$IR_s = IR_{s,d} \times [T_{site}/(24\ hr - T_{sleep})] \qquad \text{Equation 3-6}$$

where:

- IR_s = site-specific soil ingestion rate (mg of soil/day)
- T_{site} = daily time spent in exposure area (hr)
- T_{sleep} = daily time spent sleeping (hr)
- $[T_{site}/(24\ hr - T_{sleep})]$ = constrained to a value of 1.0 or less.

For the Resident National Monument/Refuge scenario, which integrates exposure at a residence with occupational exposure across multiple ROD decision areas, intake is apportioned on a time-dependent basis depending on the fraction of time spent in the two exposure areas. Practically, the adjustment factor for soil ingestion impacts daily soil ingestion rates only for Recreational and Industrial/Commercial exposure scenarios that assume partial daily site use.

Reasonable maximum exposure values for each exposure parameter for each combination of land-use scenario and receptor are provided in Tables 3-18 and 3-19. Values for the CTE calculations are provided in Table 3-20. The basis of the parameter values related to soil and sediment ingestion and the associated references are provided in Appendix D-2.1.1.

The relative bioavailability of a chemical (B_s) in Equation 3-5 represents the fraction of ingested chemical on soil that is absorbed in the gastrointestinal tract relative to the fraction that is absorbed from food or water. This term is introduced to the intake calculation for soil and sediment ingestion because bioavailability of many chemicals on soil tends to be less than that of the chemical when it is administered in food or water, which is generally how a chemical is administered in toxicological studies. In the absence of site-specific data related to bioavailability, the bioavailability of all chemicals in soil has been protectively assumed in the risk assessment to be equivalent to that in the administered dose (B_s equals 1.0).

3.4.4.3 Inhalation. The time-averaged contaminant concentration in air, rather than contaminant intake, is used as the basis for estimating chemical inhalation risks based on guidance described in *Part F, Supplemental Guidance for Inhalation Risk Assessment* (EPA-540-R-070-002). Equation 3-7 is the basic equation, which must be modified when calculating intake for carcinogenic chemicals by summing child and adult exposures, expressed as (ET × EF × ED).

$$\text{Time-averaged exposure concentration } (\text{mg/m}^3) = \frac{C_{a,i} \times ET \times EF \times ED}{AT} \qquad \text{Equation 3-7}$$

where:

$C_{a,i}$ = concentration of contaminant *i* in air (mg/m^3)
ET = exposure time onsite (hr/day)
EF = exposure frequency (day/yr)
ED = exposure duration (yr)
AT = averaging time (hr).

For radionuclides, intake by inhalation is evaluated as a function of a receptor-specific inhalation rate, as follows:

$$\text{Intake (pCi)} = C_{a,i} \times \text{InhR} \times \text{ET} \times \text{EF} \times \text{ED} \qquad \text{Equation 3-8}$$

where:

$C_{a,i}$ = concentration of contaminant *i* in air (pCi/m^3)
InhR= inhalation rate (m^3/hr)
ET = exposure time onsite (hr/day)
EF = exposure frequency (day/yr)
ED = exposure duration (yr).

For inhalation exposure to dust in ambient air, the term $C_{a,i}$ is based on broad-area soil EPCs for the Recreational and Nonresidential Tribal scenarios and the occupational portion of the Resident National Monument/Refuge scenario, and on the sum of EPCs for broad-area and local-area soils for the residential scenarios, the Industrial/Commercial scenario, and the residential component of the Resident National Monument/Refuge Exposure scenario. This is described in more detail in Section 3.4.3.

Reasonable maximum exposure values for each exposure parameter and for each combination of land-use scenario and receptor are provided in Tables 3-18 and 3-19. Values for the CTE calculations are provided in Table 3-20. The basis of the parameter values related to inhalation exposure and the associated references are provided in Appendix D-2.1.2.

3.4.4.4 Dermal Absorption from Soil or Sediment. Contaminant intake from soil via dermal contact is calculated using the following equation. This equation must be modified when calculating intake for carcinogenic chemicals by summing child and adult body-weight averaged intakes, expressed as [(DA_{event} × SA × EF × ED)/BW].

$$DAD = \frac{DA_{event,i} \times SA_s \times EF \times ED}{BW \times AT} \qquad \text{Equation 3-9}$$

where:

DAD = dermally absorbed dose (mg/kg-day)
$DA_{event,i}$ = absorbed dose per event, contaminant *i* (mg/cm^2-event)
SA_s = skin surface area exposed to soil (cm^2)
EF = exposure frequency for soil contact (event/yr)
ED = exposure duration (yr)
BW = body weight (kg)
AT = averaging time (day)

and

$$DA_{event,i} = C_{s,i} \times ABS_{d,i} \times AF \times CF \qquad \text{Equation 3-10}$$

where:

$DA_{event,i}$ = absorbed dose per event, contaminant *i* (mg/cm^2-event)
$C_{s,i}$ = concentration of chemical *i* in soil (mg/kg)
$ABS_{d,i}$ = dermal absorption fraction, contaminant *i* (unitless)
AF = soil adherence factor (mg/cm^2-event)
CF = conversion factor (10^{-6} kg/mg).

Reasonable maximum exposure values for each exposure parameter and for each combination of land use scenario and receptor are provided in Tables 3-18 and 3-19. Values for the CTE calculations are provided in Table 3-20. The basis of the parameter values related to dermal exposure from soil or sediment and the associated references are provided in Appendix D-2.1.3.

3.4.4.5 Dermal Absorption from Water. The methodology for assessing dermal uptake of metals and organic chemicals from water follows the approach described in EPA's *Risk Assessment Guidance for Superfund, Volume I: Human Health Evaluation Manual (Part E, Supplemental Guidance for Dermal Risk Assessment), Final* (EPA/540/R-99/005). The absorbed dose per dermal exposure event (DA_{event}) for metals in water is a function of the water concentration, exposure time, and chemical-specific dermal permeability coefficient.

Calculation of DA_{event} for organic chemicals in water is more complex because absorption is also a function of the absorption and desquamation kinetics in the skin. One of two equations is used for this calculation, depending on the length of time required to achieve a steady-state skin concentration relative to the length of an exposure event. If the exposure event time (T_{event}) is shorter than the time taken to reach steady state (i.e., the length of time required for a chemical to be absorbed into the viable epidermis; t*), a nonsteady-state model is used (EPA/540/R-99/005). If T_{event} is longer than t*, a pseudo-steady-state model is used (EPA/540/R-99/005).

Contaminant uptake via dermal absorption from water is calculated using the following equation. This equation is modified when calculating dermal uptake for carcinogenic chemicals by summing child and adult body-weight averaged exposures, expressed as [$(SA_w \times EV_f \times EF \times ED)/BW$]. Consistent with EPA guidance (EPA/540/1-89/002, Section 10.5.5), radiation dose and risk via dermal absorption will not be quantified, as it is likely to be negligible compared with other exposure pathways of radiation exposure. An exception exists in the case of tritium, for which the inhalation toxicity criteria include an adjustment factor that accounts for dermal absorption of tritium as tritiated water.

$$DAD = \frac{DA_{event,i} \times SA_w \times EV_{f,bath} \times EF \times ED}{BW \times AT} \qquad \text{Equation 3-11}$$

where:

DAD	=	dermally absorbed dose (mg/kg-day)
$DA_{event,i}$	=	absorbed dose per event, contaminant *i* (mg/cm^2-event)
SA_w	=	skin surface area exposed to water (cm^2)
$EV_{f,bath}$	=	event frequency for bathing (events/day)
EF	=	exposure frequency (day/yr)
ED	=	exposure duration (yr)
BW	=	body weight (kg)
AT	=	averaging time (day).

The steady-state, nonsteady-state, and pseudo-steady-state models used to calculate DA_{event} for metals and radionuclides are presented and discussed in Appendix D-2.1.4.

Reasonable maximum exposure values for each exposure parameter and for each combination of land-use scenario and receptor are provided in Tables 3-18 and 3-19. Values for the CTE calculations are provided in Table 3-20. The basis of the parameter values related to dermal exposure from water and the associated references are provided in Appendix D-2.1.4.

3.4.4.6 External Irradiation from Soil or Sediment. The general radionuclide intake equation described in the overview of Section 3.4.4 may also be used to characterize external radiation dose and risk from soil and sediment exposure. However, contact is simply a function of exposure time. Equation 3-12 shows the form of the equation for external irradiation that applies to radionuclide cancer risk. For radionuclide annual dose, which is defined as dose within a 1-year period, the exposure duration term is omitted from Equation 3-12.

$$\text{External Radiation Exposure} = C_{s,i}\ ((ET_{in} \times GSF) + ET_{out})\ EF \times ED \qquad \text{Equation 3-12}$$

where:

Exposure	=	rate of external exposure (pCi-hr/g soil)
$C_{s,i}$	=	concentration of radionuclide *i* in exposure area soil (pCi/g)
ET_{in}	=	indoor exposure time onsite (hr/day)
ET_{out}	=	outdoor exposure time onsite (hr/day)
GSF	=	gamma shielding factor for indoor exposure
EF	=	exposure frequency (day/yr)
ED	=	exposure duration (yr).

Reasonable maximum exposure values for each exposure parameter and for each combination of land-use scenario and receptor are provided in Tables 3-18 and 3-19. Values for the CTE calculations are provided in Table 3-20. The basis of the parameter values related to external irradiation from soil or sediment and the associated references are provided in Appendix D-2.1.5.

3.4.4.7 Ingestion of Water. Chemical intake via water ingestion is calculated using the following equation. This equation must be modified when calculating intake for carcinogenic chemicals by summing child and adult body-weight averaged intakes, expressed as [($IR_w \times EF \times ED$)/BW]. For radionuclide intake, the equation would be modified as indicated in Equation 3-4 and supporting text in the overview of Section 3.4.4.

$$\text{Intake (mg/kg}\cdot\text{d)} = \frac{C_{w,i} \times IR_{gw} \times EF \times ED}{BW \times AT} \qquad \text{Equation 3-13}$$

where:

$C_{w,i}$	=	concentration of contaminant *i* in exposure area water (mg/L)
IR_{gw}	=	groundwater ingestion rate (L/day)
EF	=	exposure frequency (day/yr)
ED	=	exposure duration (yr)
BW	=	body weight (kg)
AT	=	averaging time (day).

Reasonable maximum exposure values for each exposure parameter and for each combination of land-use scenario and receptor are provided in Tables 3-18 and 3-19. Values for the CTE calculations are provided in Table 3-20. The basis of the parameter values related to water ingestion and the associated references are provided in Appendix D-2.1.6.

3.4.4.8 Foodstuffs Ingestion. Contaminant intake from food ingestion is calculated using Equation 3-14. For the Subsistence Farmer, Avid Angler, and Avid Hunter scenarios, this equation applies to both chemical carcinogens (adult and child exposure summed), radionuclides (adult and child exposure averaged), and noncarcinogens (adult and child exposure evaluated separately) because the ingestion rate values for home-raised foodstuffs and hunted game and fish published by EPA (EPA/600/P-95/002Fb, *Exposure Factors Handbook, Volume II – Food Ingestion Factors*) are based on survey data across a general population that includes adults and children. To correct for body weight in the radionuclide calculation for these scenarios, a value of 60 kg is recommended by EPA/600/P-95/002Fb and is used in the HHRA. For the Native American receptors, separate food ingestion rates are defined for child and adult receptors. Body weight values specified in Harris and Harper (2004) and Ridolfi (2007) are used for Native American adults and children.

For radionuclide intake, the equation would be modified as indicated in Equation 3-4 and supporting text in the overview of this section.

$$\text{Intake (mg/kg}\cdot\text{d)} = \frac{C_{food,i} \times IR_{food} \times F_{site} \times EF_{food} \times ED \times 10^{-3}\, kg/g}{AT} \qquad \text{Equation 3-14}$$

where:

$C_{food,i}$ = concentration of contaminant *i* in foodstuff (mg/kg)
IR_{food} = foodstuff ingestion rate (g/kg-d)
F_{site} = fraction of foodstuff grown or gathered onsite
EF_{food} = exposure frequency for ingesting food products (day/yr)
ED = exposure duration (yr)
AT = averaging time (day).

Except for the Native American scenarios, EPA's *Exposure Factors Handbook* (EPA/600/P-95/002Fb) is the reference employed for all food ingestion rate parameter values, including fruits, vegetables, meats, fish, eggs, and milk. The ingestion rates obtained from EPA/600/P-95/002Fb are specific to a subpopulation of individuals who eat home-raised foods and hunted/fished meats, rather than to the general U.S. population. The value of the parameter F_{site} is 1.0 for all foods and scenarios except wild plants in the Nonresident Tribal scenario.

Reasonable maximum exposure values for each exposure parameter for each combination of land-use scenario and receptor are provided in Tables 3-18 and 3-19. Values for the CTE calculations are provided in Table 3-20. The basis of the parameter values related to ingestion of the various foods evaluated in this assessment, and the associated references, are provided in Appendix D-2.1.7.

3.5 METHODOLOGY FOR TOXICITY ASSESSMENT

3.5.1 Overview

Potential health effects related to intake of chemical COPCs are assessed using dose-response information described by cancer slope factors (CSFs) for carcinogenic effects of chemicals and reference doses (RfDs) for systemic (noncarcinogenic) effects of chemicals. These values describe a relationship between the intensity of exposure (i.e., dose) and the likelihood or severity of associated health effects. The EPA has evaluated available dose-response information for many chemicals and has published this information in the form of toxicity values and accompanying information.

The hierarchy of references for chemical toxicity criteria is described in a 2003 memorandum from EPA's Office of Solid Waste and Emergency Response (OSWER 9285.7-53, "Human Health Toxicity Values in Superfund Risk Assessments"). In accordance with this memorandum, the primary source of toxicity values used in the RCBRA is EPA's Integrated Risk Information System database (IRIS) (IRIS 2009). Only toxicity criteria published in IRIS have gone through peer-review and EPA-consensus-review processes. The second tier of toxicity criteria are the provisional peer-reviewed toxicity values (PPRTV) published by the National Center for Environmental Assessment (NCEA) in EPA's Office of Research and Development. These values are developed on a chemical-specific basis when requested by EPA's Superfund program, but the documentation for them is generally not citable. The third tier of references include values published in EPA's *Health Effects Assessment Summary Tables* (HEAST) (EPA/540/R-97/036) and other sources such as California EPA and the Agency for Toxic Substances and Disease Registry. Application of this hierarchy for identifying route-specific toxicity criteria is described in Section 3.4.7.

The potential health effects related to radionuclide exposure are also assessed in the HHRA. Radiation dose is evaluated as the effective dose using radiation dose conversion factors (DCFs) published in *External Exposure to Radionuclides in Air, Water, and Soil* (EPA/402/R-93/081) and *Age-Dependent Doses to Members of the Public from Intake of Radionuclides: Part 5, Compilation of Ingestion and Inhalation Dose Coefficients* (ICRP 1996). Radiation cancer risk is also assessed using radionuclide CSFs published in *Cancer Risk Coefficients for Environmental Exposure to Radionuclides* (EPA 402/R-99/001) and available online from EPA's Office of Radiation and Indoor Air in the HEAST tables (EPA 2006a).

3.5.2 Chemical Hazard

The toxicity values used to evaluate systemic health effects related to long-term chemical exposures are the chronic RfD and reference concentration (RfC). The chronic RfD is an estimate of daily exposure likely to be without appreciable risk of adverse effects for exposure of several years or longer (EPA/540/1-89/002). The chronic RfC is similar but specific to the inhalation exposure route.

The general model of toxicity for noncarcinogenic effects is that there exists a range of exposure from zero to some threshold in which exposure can be tolerated without a significant probability of an adverse effect. An RfD represents an estimate of this threshold and is expressed as a body-weight-normalized rate of exposure with the same units as intake (e.g., mg/kg-day). This model of toxicity is reflected in the averaging time for noncarcinogenic effects, which is equivalent to the exposure duration. The effect is generally assumed to manifest only when exposure exceeds a threshold and not when exposure is less than the threshold or at some time following the exposure.

Systemic inhalation toxicity values are provided by EPA as inhalation reference concentrations (RfCs) with units of mg/m^3. Reference concentrations are used in the risk assessment calculations consistent with the methods described in *Part F, Supplemental Guidance for Inhalation Risk Assessment* (EPA/540/R-070/002). Both RfDs and RfCs share many of the same characteristics. Prior to the recent EPA guidance, RfCs were converted to RfDs for use in a risk assessment by factoring in assumed values of human body weight and daily inhalation rates.

An RfD or an RfC is derived by EPA using human dose-response data from adequate studies if available. If human data are unavailable, dose-response information from animal studies may be employed. The EPA will preferentially base an RfD or an RfC on the highest dose level not associated with adverse effects (no observable adverse effect level [NOAEL]). If such a value was not identified in the literature, the lowest observable adverse effect level (LOAEL) is generally used as the basis of the RfD or RfC. In practice, EPA will generally first identify the critical study and adverse effect for a chemical from a review of the available toxicological data. Once these are specified, the NOAEL or LOAEL is identified. The RfD or RfC is then calculated from the NOAEL or LOAEL using uncertainty factors (UFs) to account for uncertainty in extrapolating from the NOAEL or LOAEL to a chronic RfD. Uncertainty factors may relate to potential variability in sensitivity in the human population, to interspecies variability between humans and test animals, to inadequate dosing periods in a critical study, or to use of a LOAEL instead of a NOAEL. A modifying factor is sometimes also employed to account for additional uncertainties in the derivation of a chronic RfD or an RfC.

Chronic oral RfD and RfC values are presented in Tables 3-21 through 3-24. These tables include accompanying information such as the test species and critical effect, the magnitude of uncertainty and modifying factors, and the references for the values.

3.5.3 Chemical Cancer Risk

The toxicity values used to evaluate chemical carcinogenic health effects are the CSF and the inhalation unit risk (UR). A CSF is a quantitative relationship between dose and carcinogenic response and is usually representative of a plausible upper-bound estimate of the lifetime probability of developing cancer associated with exposure to a specific quantity of a potential carcinogen (EPA/540/1-89/002). The UR is similar but specific to the inhalation exposure route. The units of a chemical CSF are expressed as cancer risk per intake with units of $(mg/kg\text{-}day)^{-1}$. The UR has units of $(\mu g/m^3)^{-1}$.

The EPA's CSFs published in IRIS are associated with a weight-of-evidence classification that indicates the strength of the evidence by which the chemical is suspected to be a human carcinogen. These classifications include the following:

A Human carcinogen
B1 Probable human carcinogen (limited human data available indicating carcinogenicity)
B2 Probable human carcinogen (inadequate or no human data available)
C Possible human carcinogen
D Not classifiable as to human carcinogenicity
E Evidence for noncarcinogenicity in humans.

Because class D and E carcinogens are not classified as human carcinogens, CSFs have not been developed for them; therefore, they are not included in a quantitative analysis of potential carcinogenicity. More recently, EPA has recommended replacing these classifications with a weight-of-evidence narrative analysis (EPA/630/P-03/001F, *Guidelines for Carcinogen Risk Assessment*). For this reason, classifications are not tabulated for individual carcinogenic COPCs in the HHRA.

The great majority of CSFs are based on carcinogenic effects observed at relatively high dose rates that have been extrapolated to lower doses. There are multiple mathematical models used for this extrapolation that relate both to the goodness-of-fit with the dose-response data, as well as theoretical models of carcinogenesis. The CSF is commonly calculated as the 95% UCL on the slope of the dose-response curve, although in some cases where the data are more robust, a best estimate is used instead. Arsenic is a relevant example of a chemical for which the CSF is based on a best estimate from human dose-response data.

Carcinogenic inhalation toxicity values, URs, are the cancer risk associated with lifetime exposure to a unit concentration of a carcinogenic chemical in air and are used in the risk assessment calculations consistent with the methods described in EPA/540/R-070/002. Similar to the RfCs discussed in Section 3.5.2, prior to the recent EPA guidance, URs were converted to CSFs for use in a risk assessment by factoring in assumed values of human body weight and daily inhalation rates.

One of the principal differences in assumptions regarding carcinogenic and noncarcinogenic effects pertains to the presumption of a threshold of exposure for noncarcinogenic effects. As described in Section 3.5.2, it is assumed for noncarcinogenic effects that there exists a range of exposure in which exposure can be tolerated without a significant probability of an adverse effect. By contrast, EPA believes that the underlying mechanisms of carcinogenesis imply that there is no threshold of exposure (EPA/540/1-89/002). That is, any exposure, no matter how small, provides some finite possibility of resulting in a carcinogenic effect. A CSF represents the incremental risk of cancer incidence associated with some finite exposure and is expressed as cancer risk per unit intake [risk/(mg/ kg-day)], or $(mg/kg\text{-}day)^{-1}$. Because there may be a decades-long latency period between exposure and effect (EPA/630/P-03/001F), effects are averaged over an entire lifetime.

In EPA/630/P-03/001F, EPA's Risk Assessment Forum stressed the use of mode-of-action information for evaluating chemical carcinogenicity, where the term "mode-of-action" refers to a sequence of key events and processes, starting with interaction of an agent with a cell, proceeding through operational and anatomical changes, and resulting in cancer formation (EPA/630/P-03/001F). For the great majority of potentially carcinogenic chemicals, dose-response data for carcinogenic response must be extrapolated to the much lower dose rates commonly related to environmental exposures. When a chemical acts to cause cancer by causing mutations in genetic material (i.e., a mutagenic mode of action), EPA assumes there is a linear dose-response function for carcinogenicity at low doses and that the dose-response curve passes through zero. When a chemical does not directly act to cause gene mutations, EPA does not necessarily assume that the slope of the dose-response curve is linear at low doses (i.e., the slope is zero at zero dose, so the dose-response curve does not pass through zero).

Recognizing that exposure to mutagenic chemicals may pose particularly high cancer risk to infants and young children, EPA released a companion document (EPA/630/R-03/003F, *Supplemental Guidance for Assessing Susceptibility from Early-Life Exposure to Carcinogens*) to EPA/630/P-03/001F to provide guidance for adjusting cancer potency estimates for childhood exposures for carcinogens that have a mutagenic mode of action. Table 1b of EPA/630/R-03/003F lists 12 chemicals that have been identified as mutagens based on animal experiments where both early-life and adult exposures were conducted with separate groups of animals. Of particular interest, relative to the COPCs identified in the RCBRA, are the four polycyclic aromatic hydrocarbons (PAH) listed in Table 1b (benzo(a)pyrene, dibenzanthracene, dimethylbenz(a)anthracene, and 3-methylcholanthrene).

The mutagenicity of these and certain other PAH is related to the formation of electrophilic metabolites, leading to the creation of DNA adducts (Klaassen 2001, *Casarett & Doull's Toxicology: The Basic Science of Poisons*). However, the carcinogenicity of these compounds may also be related to other modes of action, including immune system suppression and the tumor-promoting effect of increased mitogenicity (Klaassen 2001). The relative importance of these modes of action with chronic exposure is uncertain. Nevertheless, EPA guidance (EPA 2006b, Memorandum to the Science Policy Council) indicates that, pending reanalysis of the dose-response data for these chemicals, age-dependent adjustment of the CSFs be performed. Among the 12 mutagenic chemicals listed in EPA/630/R-03/003F, only the PAH benzo(a)pyrene and dibenz(a,h)anthracene are identified as COPCs in the HHRA. Consistent with EPA guidance (2006b), the age-dependent adjustment factors are applied to the CSFs for these PAH, to which the cancer potency of other potentially carcinogenic PAH is referenced by a toxicity equivalency factor. The adjustment factors (EPA 2006b) are 10 (ages birth to <2 years) and 3 (ages 2 to <16 years). The possibility and magnitude of risk overestimation related to childhood exposures to mutagenic carcinogens is discussed relative to other sources of uncertainty and bias in the uncertainty analysis of the risk assessments in Sections 4.0, 5.0, and 6.0.

Cancer slope factors for oral exposures and URs for inhalation exposures are presented in Tables 3-25 through 3-28. These tables include accompanying information such as the test species and type of tumor observed, the dose-response model employed, and the references for the CSF and UR values.

3.5.4 Radionuclide Cancer Risk

The toxicity value used to evaluate radionuclide carcinogenic health effects is also referred to as a CSF. The radionuclide CSF is a quantitative relationship between radiation dose and carcinogenic response; however, unlike the chemical CSF, it reflects an average estimate of the lifetime risk of cancer associated with exposure to a specific concentration of a carcinogen in an environmental medium (EPA/402/R-99/001). The units of a radionuclide CSF are expressed as cancer risk per annual intake of radionuclide activity, with units of risk per activity $(pCi)^{-1}$. For external irradiation, radionuclide CSFs define the relationship between annual cancer risk and the radionuclide activity in the source medium (risk/yr per pCi/g).

Radionuclide slope factors published by EPA are preferred to the use of risk factors applied as multipliers to calculated radiation effective dose. Although such dose equivalents are applicable for comparison to dose-based radiation protection standards, they were derived for application to adults in a workplace setting. By contrast, more recent radionuclide slope factors from *Cancer Risk Coefficients for Environmental Exposure to Radionuclides, Federal Guidance Report No. 13* (EPA/402/R-99/001) were derived to pertain to the general U.S. population and are, therefore, applicable for use in estimating cancer risks for a general population composed of adults and children. EPA/402/R-99/002 slope factors are derived using age- and gender-specific values for intake and radionuclide dosimetry. Radionuclide CSFs are presented in Table 3-29.

3.5.5 Radionuclide Dose

Radiation dose is not an effect per se, but rather a measure of the radiation dose absorbed by a tissue. The DCFs used in the dose assessment account for the biological effectiveness of the radiation (e.g., alpha particles, photons) in causing cellular damage in different tissues. For external dose, this "effective dose" is calculated. For internal dose, the committed effective dose is calculated, which accounts for continued dose over time from radionuclides retained in the body.

The effective dose is the weighted sum of the equivalent doses to different organs and tissues, where the weighting factors express an individual tissue dose as an equivalent dose to the whole body, with respect to the probability of developing a fatal cancer. The committed effective dose is a variation on the effective dose and is defined as the total effective dose deposited in the body in a 50-year period (for an adult) or a 70-year period (for a child), following the intake of a radionuclide (ICRP 1996).

Radiation dose calculated in the HHRA is compared to a threshold dose limit of 15 mrem/yr (see Section 3.6.4). The basis of the 15 mrem/yr dose limit is a risk assessment calculation using a risk factor (risk per dose) and an assumed 30-year exposure duration (ISCORS 2002, *A Method for Estimating Radiation Risk from Total Effective Dose Equivalent (TEDE), Final Report*).

For the residential and recreational exposure scenarios that include both child and adult receptors, age-dependent dose coefficients that likewise apply to approximately a 30-year exposure period are used in the risk assessment. In EPA/402/R-99/001, EPA states that linear interpolation between the published age-dependent dose coefficients in ICRP (1996) may be used to calculate a dose coefficient for any particular age class. The 30-year interpolated DCFs used for the residential and recreational exposure scenarios include age-dependent DCFs for the following ages: 1 year, 5 years, 10 years, 15 years, and adult. Adult DCFs published in ICRP (1996) were used for the Industrial/Commercial and Resident National Monument/Refuge exposure scenarios. Details on the calculation and use of internal DCFs in the risk assessment are provided in Appendix D-3. Radionuclide DCFs are presented in Table 3-30.

3.5.6 Development of Dermal Toxicity Values

As discussed in *Dermal Risk Assessment Guidance for Superfund* (EPA/540/R-99/005), dermal toxicity values would ideally incorporate assessment of direct toxicity in the skin and be based on dose-response data for systemic effects via percutaneous absorption. In the absence of such information, EPA/540/R-99/005 recommends the use of dose-response relationships obtained from oral administration studies with adjustment for gastrointestinal absorption efficiency so that the dermal toxicity values reflect absorbed rather than administered dose. Values for gastrointestinal absorption (ABS_{gi}) were obtained from Exhibit 4-1 of EPA/540/R-99/005. ABS_{oral} values are provided in Table 3-31.

For noncarcinogenic effects, the oral RfD is adjusted according to the following:

$$RfD_{dermal} = RfD_{oral} \times ABS_{gi} \qquad \text{Equation 3-15}$$

For carcinogenic effects, the oral CSF is adjusted according to the following:

$$CSF_{dermal} = {CSF_{oral}}/{ABS_{gi}} \qquad \text{Equation 3-16}$$

A potential problem in the use of oral toxicity values for dermal absorption involves chemicals, such as arsenic and carcinogenic PAH, where toxic effects may manifest in the skin at the site of absorption. These issues are addressed on a chemical-specific basis in the risk assessment.

3.5.7 Protocol for Identification of Route-Specific Toxicity Criteria

The toxicity criteria that EPA recommends for use in the HHRA correspond to one of three data quality categories. The categories reflect the confidence in the toxicity studies used to develop the toxicity criteria, as well as their preference for use in the HHRA. The categories, listed from most to least preferred with parenthetical reference to the associated agency or authority, are as follows: Tier 1 (IRIS), Tier 2 (NCEA PPRTV), and Tier 3 (NCEA/HEAST; Agency for Toxic Substances and Disease Registry [ATSDR]). Tier 1 toxicity data are associated with the highest degree of confidence because of rigorous peer review. In general, if a Tier 1 criterion for a particular chemical is available for either the cancer or noncancer effects endpoint, a Tier 2 or Tier 3 criterion will not be used for the other endpoint, even if one exists. Exceptions are noted in Tables 3-21 through 3-28.

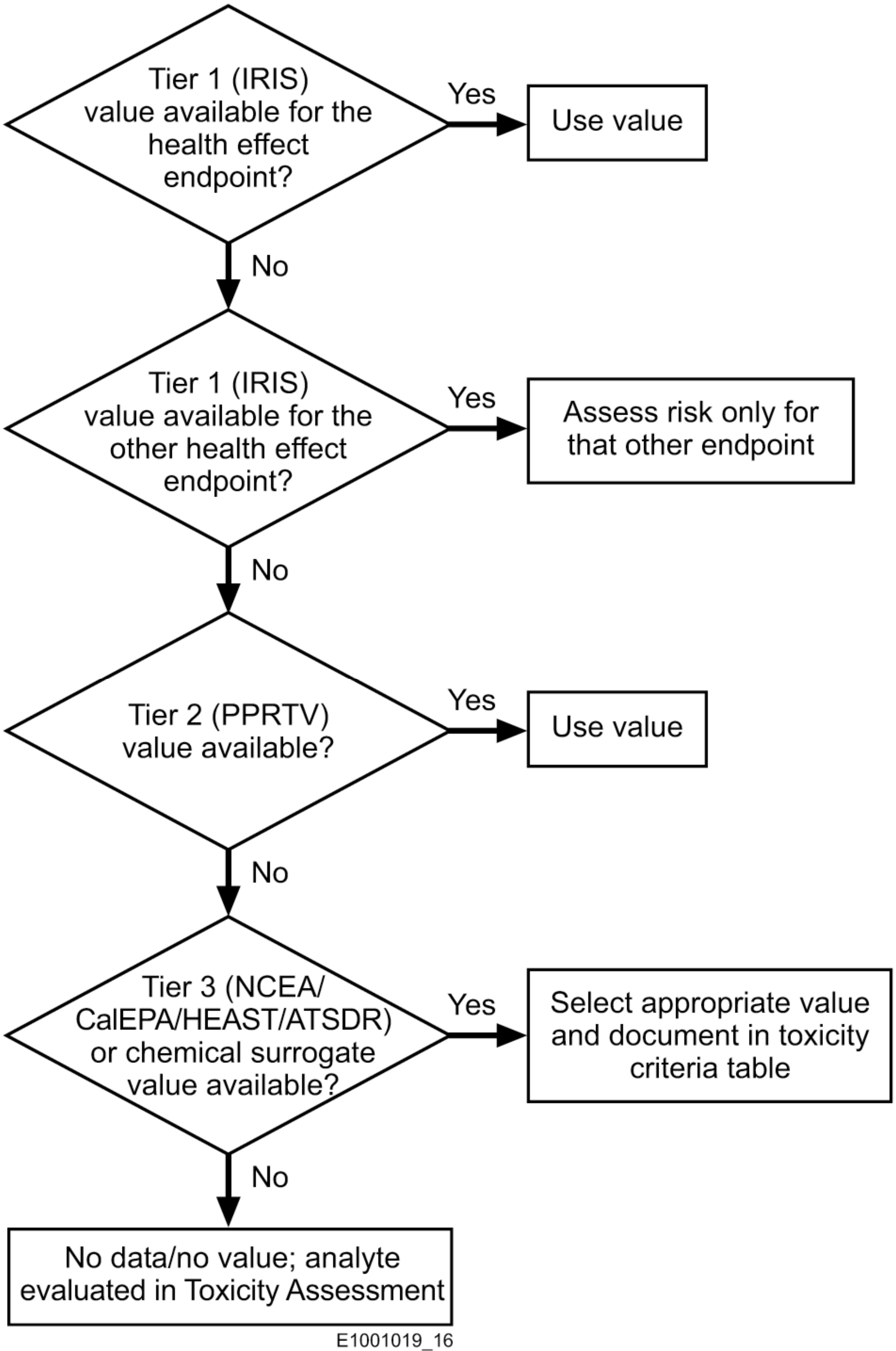
The protocol used to select toxicity criteria is described here and presented in Figure 3-18. If available, a Tier 1 (IRIS) toxicity criterion value is preferentially used for each combination of chemical, health effect endpoint (cancer or noncancer), and exposure route (oral or inhalation). If a Tier 1 toxicity criterion for a health effect endpoint is not available, but there is a Tier 1 value for the other health effect endpoint, then a Tier 2 or Tier 3 value for the first endpoint may not be used even if one is available. In particular, Tier 3 toxicity data may not be used if a Tier 1 value is available for the other health effect endpoint. This minimizes the number of Tier 3 values used in the assessment and still ensures that all chemicals are evaluated for potential health effects that have been adequately documented. If a Tier 1 value is unavailable for any health effect endpoint, then a Tier 2 or Tier 3 value is always used if one is available, in that order of preference. If no toxicity criteria are published for a chemical, the value(s) for a surrogate chemical may be used and will be documented in the tabulation of toxicity criteria. Selection of a surrogate chemical is based on criteria including commonality in mechanism of action or molecular structure. Preferentially, the surrogate value selected is a Tier 1 value.

Route-to-route extrapolation of toxicity criteria is not used in the HHRA. Route-to-route extrapolation is the process of employing a toxicity value based on either oral or inhalation administration to characterize the potential toxicity for exposure via the other exposure route when toxicity data for that route are unavailable. Typically, this involves using toxicity values for oral administration to represent toxicity via inhalation. A route-to-route extrapolation presumes that the critical toxicity of a chemical is systemic. In other words, that the toxic effect is remote from the contact sites of the gastrointestinal or inhalation systems. If this condition is violated, there is a potential for significant under- or over-representation of an agent's potential toxicity when extrapolating between exposure routes. Although this practice was once common for organic chemicals, it is less common now with the growing number of Tier 2 and Tier 3 values available to supplement values published in IRIS. For example, the regional screening values supported by EPA no longer use route-to-route extrapolation as a default method for organic chemicals (http://epa-prgs.ornl.gov/chemicals/faq.shtml).

3.5.8 Toxicity Assessment for Lead

Lead exposure can result in significant health effects, particularly among children, whose physiology and behavior are generally believed to cause them to be more susceptible to the effects of lead in environmental media such as soil and dust. Health effects associated with exposure to inorganic forms of lead include neurotoxicity, developmental delays, hypertension, impaired hearing acuity, impaired hemoglobin synthesis, and male reproductive impairment (IRIS 2009).

Figure 3-18. Protocol for Selecting Route-Specific Toxicity Criteria.



Studies indicate that high-level exposure to lead produces encephalopathy in children, with signs of encephalopathy associated with blood lead concentrations of approximately 90 to 800 μg/dL (mean, 330 μg/dL) (ATSDR 2005). The distribution of blood lead concentrations associated with death was approximately the same as that related to encephalopathy. Effects at low levels of lead exposure in children, (e.g., impaired neurobehavioral development and decreased intelligence) have been observed (ATSDR 2005).

Children are potentially more susceptible to the health effects of lead in soil than adults for several reasons (ATSDR 2005). First, children have more frequent hand-to-mouth behavior than adults, and young children ingest more soil and dust than adults. Second, because the gastrointestinal efficiency of lead absorption is greater in children, a bigger proportion of the amount of lead swallowed will enter the blood in children. Finally, children are physiologically more sensitive to the neurological effects of lead because of their developing nervous systems.

The EPA has not established any toxicity criteria for lead (IRIS 2009). Instead, potential health risks related to lead exposure are evaluated by modeling blood lead concentrations and comparing these concentrations to published blood lead concentration criteria. A 5% probability of a child having a blood lead level exceeding 10 μg/dL is generally used as a criterion to determine whether potential blood lead levels are of concern (EPA/540/F-98-030, "Clarification to the 1994 Revised Interim Soil Lead Guidance for CERCLA Sites and RCRA Corrective Action Facilities"). The blood level of 10 μg/dL derives from a 1991 recommendation by the Centers for Disease Control (CDC) (IRIS 2009). However, because EPA recognizes that health effects are still observable below this level, the CDC convened an advisory committee on childhood lead poisoning prevention to consider whether the level of concern should be changed (IRIS 2009). To date, the CDC has not changed the blood lead level of concern.

The EPA has assigned lead a weight-of-evidence classification for human carcinogenicity of B2, a probable human carcinogen. This designation is based on rodent bioassays that have shown statistically significant increases in renal tumors with dietary and subcutaneous exposure to several soluble lead salts (IRIS 2009). The available human evidence is considered by EPA to be inadequate to refute or demonstrate any potential carcinogenicity for humans from lead exposure.

The EPA has recommended a residential screening level for lead in soil of 400 mg/kg, derived using the integrated exposure uptake biokinetic model (OSWER 9355.4-12, "Revised Interim Soil Lead Guidance for CERCLA Sites and RCRA Corrective Action Facilities"). More recently, this screening level has been associated with bare soil in a play area, and an additional screening level of 1,200 mg/kg defined for other bare-earth portions of a residential yard (EPA 2001c). The 400 mg/kg screening level was developed such that a typical child would have no more than a 5% chance of having a blood lead level exceeding 10 μg/dL, a level associated with health effects in children (OSWER 9355.4-12). Site-related residential exposures contributing to the 400 mg/kg screening level include soil ingestion from the yard and indoor ingestion of house dust contaminated with soil. In addition to these site-related exposures, the 400 mg/kg screening level incorporates background levels of lead exposure from nonsite-related sources (e.g., ambient air, drinking water, and diet). These background exposures were defined using national averages or typical values where suitable (OSWER 9355.4-12). The 400 mg/kg residential screening level, rather than the 1,200 mg/kg value, will be employed as a protective screening value for use in this HHRA. Comparison of representative concentrations of lead in soil to 400 mg/kg screening value is documented in the discussion of results for the Subsistence Farmer exposure scenario in Section 5.0 of this report.

3.5.9 Carcinogenicity Assessment for PCBs

The majority of the PCB analytical data available for use in the RCBRA represent PCB concentrations in the form of PCB Aroclors. The EPA publishes oral cancer slope factors for PCBs that differentiate between three categories of potential risk and environmental persistence (IRIS 2009). For each of the three categories, EPA publishes both an upper-bound and a central-estimate slope factor, which differ by 50% to 100% of each other. The criteria for use of the slope factors for high risk and persistence include several attributes that pertain to the HHRA. These include exposure via foodstuffs, soil ingestion, dust inhalation, dermal absorption, and potential early-life exposure. The upper-bound slope factor for high risk and persistence (2.0 per mg/kg-day) is used for soil and food-mediated exposure pathways in the HHRA (IRIS 2009). For water-mediated exposure pathways, a slope factor of 0.4 per mg/kg-day (IRIS 2009) is used. The slope factors are applied to the sum of all PCB Aroclors in a sample.

PCB data for individual PCB congeners is also available for some fish tissue samples. When both congener and Aroclor data are available, cancer risk calculated for total PCB Aroclors using the slope factor approach is supplemented by analysis of dioxin toxicity equivalency factors (TEFs) to evaluate carcinogenicity via dioxin-like activity. Cancer risks from dioxin-like PCB congeners (evaluated using dioxin TEFs) are summed with cancer risks calculated for each detected PCB Aroclor using the Aroclor slope factors described above. However, there are no individual fish samples for which both Aroclor and congener data are available.

The dioxin TEFs for PCB congeners used in the HHRA are published by the World Health Organization and have been cited by EPA in their December 2003 review draft of a Health Assessment for 2,3,7,8-tetrachlorodibenzo-p-dioxin (2,3,7,8-TCDD) and by Ecology in a 2008 memorandum titled *Evaluating the Toxicity and Assessing the Carcinogenic Risk of Environmental Mixtures Using Toxicity Equivalency Factors*. These TEFs have been reproduced in Table 3-32.

The TEFs for individual PCB congeners are applied by multiplication with a slope factor for 2,3,7,8-TCDD. EPA's December 2003 review draft of a Health Assessment for 2,3,7,8-TCDD provides a cancer slope factor of 10^{+6} per mg/kg-day, but this assessment is still under technical review, and the slope factor cannot presently be used in CERCLA risk assessment. Therefore, the 2,3,7,8-TCDD slope factor of 13,000 per mg/kg-day published by the California EPA (California EPA 2005, *Air Toxics Hot Spot Program Risk Assessment Guidelines, Part II: Technical Support Document for Describing Available Cancer Potency Factors*) is used in this assessment. The difference between the California EPA slope factor and the value in the draft EPA risk assessment for 2,3,7,8-TCDD reflects that high degree of uncertainty in these values.

3.5.10 Toxicity Assessment for Detected Analytes Not Included in the Quantitative Risk Assessment

Among the detected analytes in soil and groundwater, several constituents were excluded from the HHRA because they have negligible toxicological relevance. These included radionuclides with a half-life less than 3 years and essential nutrients present at relatively low concentrations. The list of the chemical analytes includes the following:

- Calcium
- Magnesium
- Sodium
- Potassium.

A number of other detected chemicals do not have toxicity criteria published by the EPA. In some cases, these analytes are structurally and/or toxicologically similar to chemicals having published criteria. These chemicals are then used as toxicological surrogates; these instances are described in Tables 3-21 through 3-28. The remaining chemicals, having no toxicity criteria nor toxicological surrogates, are unlikely to present significant human health risks and, therefore, are not included in the quantitative HHRA. The basis for concluding that these chemicals present negligible potential human health risk is presented in the following paragraphs.

Bismuth. Bismuth has a long history of use as a pharmaceutical in Europe and North America. A well known, clinically used form of bismuth is bismuth subsalicylate, or PeptoBismol™. Most bismuth compounds are insoluble and poorly absorbed from the gastrointestinal tract with less than 1% of an oral dose being absorbed. Bismuth compounds are also poorly absorbed when applied to the skin even when the skin is abraded or burned. Acute renal toxicity is possible with oral administration of bismuth, particularly in children. Chronic toxicity with a broad spectrum of symptoms and manifestations is also possible at clinical doses (Klaassen 2001). Even though bismuth may be toxic at doses related to clinical treatment, effects from exposure to environmental concentrations are unlikely to be seen due to the very low absorption potential of bismuth. The amount of bismuth subsalicylate in a single dose of PeptoBismol is 524 mg, or 7.8 mg/kg bw/d for a 70-kg (154-lb.) adult.

Bromide. Bromide has been used in over-the-counter and prescription formulations as a sedative-hypnotic drug. Currently, the bromide salt is only available in prescription drugs and as part of the antihistamine molecule brompheniramine. Oral bioavailability has been observed to be relatively high, but acute poisoning is rare. In adults, the therapeutic dose is 3,000 to 5,000 mg, while a fatal dose is estimated at 10,000 to 20,000 mg (Schonwald 2001, *Medical Toxicology: A Synopsis and Study Guide*). The therapeutic and fatal doses to a 70-kg (154-lb.) adult correspond to 43 to 71 mg/kg and 143 to 286 mg/kg, respectively. Although bromide, like bismuth, may be toxic in clinical treatment, effects from exposure at environmentally relevant concentrations are unlikely to be seen due to the much lower dosages in such a context.

Chloride. Chloride is an essential mineral for humans. Excessive intake of chloride salts is associated with fluid retention and high blood pressure, but excess chloride is normally excreted in the urine, sweat, and bowels, so overt toxicity is generally not observed. Healthy individuals can tolerate high levels of chloride with adequate intake of fresh water. The average intake of salt from a regular salt-free diet is approximately 100 mg/day, or 1.4 mg/kg for a 70-kg (154-lb.) adult. Adverse effects from exposure to environmental concentrations of chloride are unlikely to be seen due to the relatively low concentrations encountered in the environment and the ability of the body to process excess intake of chloride.

Heptacosane and octacosane. Heptacosane is a long-chain alkane (27 carbons), which is an acyclic chemical compound consisting of carbon and hydrogen linked together by single bonds. Octacosane is a 28 carbon alkane. Alkanes longer than approximately 20 carbons are also known as paraffin, and the solid forms of paraffin are called paraffin wax. Alkanes are not very reactive and have very little biological activity. Food-grade paraffin wax has many uses, and it is edible but nondigestable. It passes through the body without being broken down. Because of the biological inactivity of these compounds, adverse effects from low concentrations in the environment are unlikely.

Hexadecanoic acid. Hexadecanoic acid, or palmitic acid, is one of the most common saturated fatty acids found in animal and plant products. It is a major component of the oil from palm trees. Butter, cheese, milk, and meat also contain this fatty acid. Being an acid, it may cause irritation to the eye and skin, and, upon inhalation or ingestion, it may cause irritation of the mucus membranes. However, these effects are related to exposure to the pure substance and are not relevant to the low environmental concentrations.

Lithium. Lithium has a variety of therapeutic uses, including the treatment of manic-depressive affective disorders, as well as industrial uses in nuclear reactor coolant. Occupational toxicity is rare; however, it does have a narrow toxic therapeutic index with chronic use and has a bioavailability over 95% via ingestion (Schonwald 2001). Detected concentrations in cleanup verification shallow zone soil data in the 100-D/100-H and 100-B/C Areas are within the range observed in reference area and area background data sets.

Octadecanoic acid. Octadecanoic acid, also known as stearic acid, is a saturated fatty acid found in many animal and vegetable fats and oils. Stearic acid is an ingredient in candles, soaps, plastics, oil pastels, and cosmetics and is used as a rubber softener. Like hexadecanoic acid, it may, in a pure form, cause irritation to the eye and skin and, upon inhalation or ingestion, may cause irritation of the mucus membranes. It is unlikely to result in any adverse effects at environmentally relevant concentrations.

Orthophosphate. Orthophosphate, also known as phosphoric acid, is a corrosion inhibitor added to finish drinking water and is added to foods and beverages to control pH. Phosphoric acid is a generally recognized as safe (GRAS) substance by the Food and Drug Administration.

Phosphorus. Phosphorus is an essential nutrient required for the development of bones and teeth, for metabolism of fats and carbohydrates, and for protein synthesis. The recommended dietary intake is 100 mg/day for infants less than 1 year old to 1,250 mg/day for children and pregnant/lactating women younger than 18. It is unlikely to result in any adverse effects at environmentally relevant concentrations.

Silica. Exposure to high concentrations of respirable-size particles of crystalline silica is of concern in occupational settings related to abrasive grinding and cement manufacturing, as well as in mining and quarrying. In the environment, silica (also known as, silicon dioxide) is one of the principal components of many types of sand. Silicate minerals (silica combined with various metals including aluminum, calcium, sodium, and magnesium) are also used as anticaking agents in powdered foods. These silicates are defined as GRAS substances by the Food and Drug Administration and have no known adverse effects at environmentally relevant intake rates.

Sulfate. Sulfate, a commonly detected analyte in groundwater, is generally soluble in water, with the exception of lead, barium, and strontium sulfate compounds. The major health effect associated with sulfate is a laxative action. The acceptably safe levels from various health organizations for infants for the ingestion of water range from 250 to 500 mg/L. However, the aesthetic quality of water is considered poor at a level of 400 mg/L (http://rais.ornl.gov/tox/profiles/sulfate_c_V1.shtml/). The poor taste and color at higher concentrations most likely serves as a deterrent to consumption. However, concentrations in excess of 1,000 mg/L have been measured in groundwater samples used in this assessment. Therefore, although no toxicity criteria are available for quantification of potential health risks, the presence of higher concentrations of sulfate in groundwater is a potentially relevant metric of groundwater quality for both health and aesthetic reasons.

Titanium. Titanium, in either metallic form or as a salt, is relatively nontoxic (Klaassen 2001). As a metal, it is used in surgical implants and prostheses, and, in the form of titanium dioxide, it is used as a whitening agent in foods such as wheat flour and dairy products and in cosmetics such as toothpaste (Klaassen 2001). As a component of particulates, it is categorized by the Occupational Safety and Health Administration as a contributor to nuisance dust with respect to permissible exposure limits, rather than a toxicologically active chemical (http://www.osha.gov/pls/oshaweb/owadisp.show_document?p_table=STANDARDS&p_id=9992/).

3.6 METHODOLOGY FOR RISK CHARACTERIZATION

The risk characterization integrates information from the exposure assessment (Section 3.4) and toxicity assessment (Section 3.5) to characterize risk for each set of exposure scenarios and receptors and each of the health effect endpoints. A discussion of the general protocol for characterizing risk for each of the health effects endpoints addressed in the RCBRA is provided in Sections 3.6.1 through 3.6.4. The risk assessment results for the broad-area calculations, local-area calculations, and groundwater calculations will be presented in Sections 4.0, 5.0, and 6.0, respectively. An assessment of potentially additive risk for exposure scenarios that integrate exposure to soil-related media, groundwater, and fish is provided in Section 7.0.

The risk assessment results for all health effects endpoints and exposure scenarios are calculated using EPCs that represent COPC concentrations in the sample media from potentially affected areas of the River Corridor. A separate set of risk assessment calculations has been conducted using reference area EPCs for individual COPCs, and for EPCs that represent the 50th and 90th percentiles of Washington State and Hanford Area background. The reference area and background risk results are used in Sections 4.0 and 5.0 of the HHRA to discuss the increment of risk from site COPC concentrations to levels of risk related to reference area and background COPC concentrations for the RCBRA exposure scenarios. EPA guidance related to human health risk assessment discusses collection of samples to represent background concentrations (EPA/540/R-01/003). Appendix B to EPA/540/R-01/003 discusses the application of these concentrations in a baseline risk assessment. In accordance with this guidance, the contribution of background-related concentrations to site-related concentrations is distinguished with respect to the risk results in the HHRA.

3.6.1 Carcinogenic Effects of Chemicals

Cancer risk is evaluated as the incremental probability that an individual will develop cancer during their lifetime. This cancer risk is the product of the average daily dose (i.e., chemical intake) and a cancer slope factor. Chemical intake and chemical CSFs are described in Sections 3.4.4 and 3.5.3, respectively. Chemical cancer risk is calculated as follows:

$$\text{Cancer Risk} = \text{Intake}\left(\frac{\text{mg}}{\text{kg - d}}\right) \times \text{CSF}\left(\frac{\text{mg}}{\text{kg - d}}\right)^{-1} \qquad \text{Equation 3-17}$$

where:

Cancer Risk = incremental lifetime cancer risk
Intake = daily intake across all exposure pathways
CSF = cancer slope factor.

As described in Section 3.4.4, current EPA methodology for inhalation risk assessment is based on time-averaged exposure concentrations in air rather than chemical intake. For inhalation, chemical cancer risk is calculated as follows:

$$\text{Cancer Risk} = \text{EC}\left(\frac{\text{mg}}{\text{m}^3}\right) \times \text{UR}\left(\frac{\text{mg}}{\text{m}^3}\right)^{-1} \qquad \text{Equation 3-18}$$

where:

Cancer Risk	= incremental lifetime cancer risk
EC	= time-averaged exposure concentration in air
UR	= unit risk factor.

Cancer risks for each exposure route and COPC (radionuclides and chemicals) are summed to calculate cancer risk to an individual. The acceptability of any calculated incremental cancer risk is generally evaluated relative to the point of departure of 1×10^{-6} and the target risk range of 10^{-6} to 10^{-4} described in the "National Oil and Hazardous Substances Pollution Contingency Plan" (NCP) (40 CFR 300.430). Under the "Model Toxics Control Act" (MTCA) (WAC 173-340), Washington State evaluates cancer risks due to exposure to multiple chemicals and/or multiple exposure pathways using an incremental cancer risk threshold of 1×10^{-5}. As a point of comparison, national cancer statistics have been developed that provide the background incidence of cancer for the U.S. population. For example, national cancer statistics indicate that each male has approximately a 1 in 2 chance of developing cancer during his lifetime and that each female has approximately a 1 in 3 chance of developing cancer in her lifetime (American Cancer Society 2005, Surveillance Research: Probability of Developing Invasive Cancers Over Selected Age Intervals, by Sex). An individual with a theoretical incremental lifetime cancer risk of 1-in-100,000 due to site-related exposure has an approximate total cancer risk of 50,001-in-100,000 (male) or 33,334-in-100,000 (female), where the background levels are 50,000-in-100,000 (male) and 33,333-in-100,000 (female).

In most risk assessments, the carcinogenic risks from all carcinogenic chemicals are treated as additive and summed to produce an overall estimate of carcinogenic risk from the site (EPA/540/1-89/002). Interactions that alter the toxicity may also occur among chemicals in a mixture. That is, the potential exists for synergistic effects or antagonistic effects. Synergistic effects occur when the combined effects are greater than the toxicity of each component of a mixture individually, while antagonistic effects occur when the combined effects are less than the toxicity of each component of a mixture individually. Failure to consider potential synergistic or antagonistic effects on toxicity may result in either an underestimation or an overestimation (similar to the assumption of additivity) of the risk, respectively.

In practice, synergistic or antagonistic effects between the carcinogens are difficult to quantify due to the lack of information on the toxicity of specific chemical mixtures. Because slope factors are upper 95th percentile estimates and are not strictly additive, the total cancer risk estimate becomes increasingly biased in a conservative manner as the number of summed carcinogens increases. Summing the risks from all carcinogens equally also gives as much weight to Class B or C carcinogens as to Class A carcinogens, and gives CSFs derived from animal data the same weight as CSFs derived from human data. Uncertainties and protective biases introduced in the risk characterization for carcinogenic effects are addressed in the Uncertainty Analysis of the risk assessments in Sections 4.0, 5.0, and 6.0.

3.6.2 Noncarcinogenic Effects of Chemicals

Noncarcinogenic effects for individual chemicals are expressed as hazard quotients (HQs). An HQ is the ratio of the average daily dose (i.e., chemical intake) of a chemical to the corresponding RfD for that chemical. Chemical intake and RfDs were discussed in Sections 3.4.4 and 3.5.2 of this methodology, respectively. The HQ is calculated as follows:

$$HQ = \frac{Intake\left(\frac{mg}{kg\text{ - }d}\right)}{RfD\left(\frac{mg}{kg\text{ - }d}\right)} \qquad \text{Equation 3-19}$$

where:

Intake = chronic daily intake
HQ = hazard quotient
RfD = reference dose.

As described in Section 3.4.4, current EPA methodology for inhalation risk assessment is based on time-averaged exposure concentrations in air rather than chemical intake. For inhalation, HQ is calculated as follows:

$$\mathrm{HQ} = \frac{\mathrm{EC}\left(\frac{\mathrm{mg}}{\mathrm{m}^3}\right)}{\mathrm{RfC}\left(\frac{\mathrm{mg}}{\mathrm{m}^3}\right)} \qquad \text{Equation 3-20}$$

where:

EC = time-averaged exposure concentration in air
HQ = hazard quotient
RfC = reference concentration.

Hazard quotients for each chemical may be summed to calculate a hazard index (HI) across chemicals for each exposure pathway if target organs and mechanisms of toxicity are similar. In some cases, summing HQs may be protectively assumed even in situations where target organs and mechanisms of toxicity are dissimilar. Hazard indices across exposure pathways may also be summed to calculate an overall HI. It should be noted that there is also a possibility of synergistic effects among chemicals, where simple additivity may in fact underestimate potential toxicity. An HQ or HI value of greater than 1.0 is indicative of the potential for adverse effects, with higher values being related to greater concern for adverse effects. Unlike cancer risk, the magnitude of an HQ is not a measure of the probability of an effect occurring.

The level of concern does not increase linearly as an HI of unity is approached or exceeded because the individual HQ values do not have equal accuracy or precision and are not based on the same severity of effect. RfDs are also associated with varying levels of confidence due to differences in the uncertainty and modifying factors across chemicals. Uncertainties and protective biases introduced in the risk characterization for noncarcinogenic effects will be addressed in the uncertainty analysis.

Because the uncertainties related to exposure to chemical mixtures affect whether the risk is over- or underestimated, it is important to determine the conditions under which additivity versus synergism may occur. For example, *Supplementary Guidance for Conducting Health Risk Assessment of Chemical Mixtures* (EPA 630-R-00-002) suggests that additivity be assumed as a default approach when mixture components are at low doses and when toxicity occurs via the same mechanism. In this assessment, values of HI are initially calculated across all chemicals and exposure routes. If an HI is potentially significant, the issue of similarity of the toxicological mechanisms of action across the major contributors to the HI is evaluated as part of the discussion of risk assessment results in Sections 4.0, 5.0, and 6.0 of the risk assessment.

3.6.3 Radiation Cancer Risk

Radiation cancer risk, like chemical cancer risk, is evaluated as the incremental probability that an individual will develop cancer during their lifetime. Radiation cancer risk is the product of the average daily radionuclide intake or external dose and a cancer slope factor. Intake and radionuclide CSFs are described in Sections 3.4.4 and 3.5.4, respectively. Radiation cancer risk is calculated as follows:

$$\text{Cancer Risk} = \text{Intake}\left(\text{pCi}\right) \times \text{CSF}\left(\text{pCi}\right)^{-1} \qquad \text{Equation 3-21}$$

where:

Cancer Risk = incremental lifetime cancer risk
Intake = daily intake across all exposure pathways
CSF = cancer slope factor.

The units in the equation for external irradiation differ as described in Sections 3.4.4 and 3.5.4:

$$\text{Cancer Risk} = \text{Intake}\left(\text{pCi - hr/g soil}\right) \times \text{CSF}\left(\frac{\text{risk/yr}}{\text{pCi/g}}\right)^{-1} \times 1.14 \times 10^{-4} \text{ yr/hr} \qquad \text{Equation 3-22}$$

Cancer risks for each exposure route and COPC (radionuclides and chemicals) are summed to calculate cancer risk to an individual. The acceptability of any calculated incremental cancer risk is evaluated relative to the target risk range of 10^{-6} to 10^{-4} described in the NCP (40 CFR 300.430). A context for this level of cancer risk, with respect to national background rates of cancer, is provided in Section 3.6.1.

3.6.4 Radiation Dose

Radiation dose, which is a measure of the amount of energy deposited in body tissues, is calculated as the product of the intake or exposure rate for a single radionuclide and the DCF for that radionuclide and exposure route. The specific radiation dose associated with dose conversion factors from ICRP (1996) and EPA/402/R-93/081 is the annual committed effective dose (internal) or annual effective dose (external). Intake and radionuclide DCFs are described in Sections 3.4.4 and 3.5.5, respectively.

Internal radiation dose is calculated as follows:

$$\text{Dose} = \text{Intake} \times \text{DCF} \qquad \text{Equation 3-23}$$

where:

Dose = annual radiation dose (mrem/yr)
Intake = annual intake across all exposure pathways (pCi/yr)
DCF = dose conversion factor (mrem/pCi).

As described in Section 3.4.4, the units of external irradiation differ from those used for internal dose. External radiation dose is calculated as follows:

$$\text{Dose} = \text{Exposure} \times \text{DCF} \times 1.14\times10^{-4} \text{ yr/hr} \qquad \text{Equation 3-24}$$

where:

Dose = annual radiation dose (mrem/yr)
Exposure = annual exposure to external radiation (pCi-hr per g-yr)
DCF = dose conversion factor (mrem/yr per pCi/g).

Radiation doses for each exposure route and radionuclide are summed to calculate the total annual dose to an individual. The acceptability of a calculated annual dose for exposures related to soil and foodstuffs is evaluated for human receptors in the RCBRA relative to a threshold dose limit of 15 mrem/yr. The origin of this 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, "Establishment of Cleanup Levels for CERCLA Sites with Radioactive Contamination"). Current EPA policy is to employ cancer risk as a basis for CERCLA cleanup levels rather than radiation dose. However, past assessments at the Hanford Site (including cleanups at 100 and 300 Area waste sites under the IARODs) have employed radiation dose as a basis for action. For comparison purposes, radiation dose is also evaluated in this risk assessment. 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 (ALARA) below a primary dose limit of 100 mrem/yr above background.

For groundwater exposures, a threshold of 4 mrem/yr is used to facilitate comparison of results in this risk assessment to historical dose assessments at the Hanford Site. This threshold is consistent with EPA guidance in the "National Primary Drinking Water Regulations" (40 CFR 141.66, "Maximum Contaminant Levels for Radionuclides"). The EPA threshold of 4 mrem/yr is applicable specifically to beta particles and gamma radiation from manmade radionuclides, with different drinking water thresholds for alpha-particle radioactivity and uranium. However, in DOE Order 5400.5, DOE applies the 4 mrem/yr threshold to radiation dose across all radionuclides. The HHRA also uses the 4 mrem/yr threshold as a reference for comparison of radiation dose from exposures to all radionuclides in water.

As a point of comparison to the thresholds of 15 and 4 mrem/yr, the average annual radiation dose in the United States from natural and manmade sources of radiation is about 360 mrem/yr (NRC 2004, *Fact Sheet: Biological Effects of Radiation*). Natural sources of radiation, predominantly radon gas, make up about 80% of this total. The remainder is related to such sources as medical x-rays, nuclear medicine, and consumer products. In some areas of the country, however, background radiation dose may be considerably higher. For example, the average annual dose in Denver, Colorado, is over 1,000 mrem/yr (NRC 2004) due to local differences in altitude and lithology.

3.7 METHODOLOGY FOR CALCULATING PRELIMINARY REMEDIATION GOALS

Preliminary remediation goals (PRGs) for soil are calculated in the HHRA for several of the exposure scenarios described in Section 3.3.1. With the exception of the Resident National Monument/Refuge scenario, discussed below, soil PRGs are calculated using the same methods and input parameter values described in Sections 3.4, 3.5, and 3.6. The intake equations shown in Section 3.3.4 pertain to the "forward" calculation of risk, where EPCs in one or more exposure media related to soil are used as inputs to the intake calculations. As described in Section 3.6, COPC intake is combined with a route-specific toxicity criterion to calculate risk for three assessment endpoints: chemical hazard, cancer risk, and radiation dose. These calculations are performed for each exposure pathway, and the results for all pathways are summed to calculate risk for a particular COPC, exposure scenario, and assessment endpoint.

Equations for the calculation of soil PRGs are developed from those described for the forward calculations for each of the exposure scenarios described in Section 3.3.1by manipulating the equations to solve for the COPC concentration in soil. For example, for a carcinogenic chemical, the generic intake equation shown in Equation 3-3 would be combined with Equation 3-17 for cancer risk. The full equation for cancer risk would then be expressed as follows:

$$risk = \frac{C_i \times CR \times EF \times ED \times CSF}{BW \times AT} \qquad \text{Equation 3-25}$$

where:

C_i = concentration of chemical *i* at exposure point (mg/kg soil)
CR = contact rate with soil (mg soil ingestion / day)
EF = exposure frequency (days/year)
ED = exposure duration (year)
BW = body weight (kg)
AT = averaging time for toxicological effects (days)
CSF = cancer slope factor $(mg/kg\text{-}day)^{-1}$.

Equation 3-25 may then be rearranged as follows to calculate a PRG for each pathway:

$$PRG_{pathway} = \frac{TR \times BW \times AT}{CR \times EF \times ED \times CSF} \qquad \text{Equation 3-26}$$

where:

TR = target risk for carcinogenesis (1×10^{-6}).

To calculate PRGs in an exposure medium across multiple exposure pathways, the inverse of the sum of the reciprocals is used. For example, for a scenario with three exposure pathways (shown as PRG 1 through 3), the following equation would apply:

$$PRG_{scenario} = \frac{1}{1/PRG_1 + 1/PRG_2 + 1/PRG_3} \qquad \text{Equation 3-27}$$

The target hazard quotient and target chemical cancer risk used in the calculation of the PRGs are 1.0 and 1×10^{-6}, respectively. These thresholds are discussed in Section 3.5. For radionuclide cancer risk, a threshold of 1×10^{-4} is used, which equates to a radiation dose of approximately a 5 mrem/yr (OSWER 9200.4-18).

The Resident National Monument/Refuge exposure scenario was developed as a composite scenario where a fraction of an individual's time is spent at a residence located on a remediated waste site and a fraction is spent working within a much larger area equivalent to the size of one or more ROD Areas. For the PRG calculations, which must pertain to a single soil exposure source term, both the residential and occupational components of exposure time were applied to an individual remediated waste site with an assumed area of 1 acre. This assumption affects the calculation of exposure by dust inhalation, because for the PRG calculation dust is assumed to originate with a contaminated source area of 1 acre. In the risk assessment, a Resident National Monument/Refuge receptor was also assumed to be exposed to contaminated dust arising from the large surrounding area over which occupational activities occur.

Another RCBRA exposure scenario that was modified slightly for application to calculation of soil PRGs is the Casual User recreational scenario. This scenario is likely to be used in the RI/FS investigation as a threshold scenario for screening low-intensity land uses at individual remediated waste sites. In order to align more closely with EPA soil screening guidance (EPA/540/R-95/128), the calculation of site-specific soil ingestion rates based on the fraction of waking time spent in the impacted area (see Equation 3-6) has not been used in this PRG calculation. Instead, 100% of daily soil ingestion, as defined in Table 3-18, is assumed to occur within the impacted area.

As discussed in Section 3.4.3, the HHRA includes calculations for the soil source term to evaluate potential exposures at different times in the future. These risk assessment calculations are conducted using approximate present-day radionuclide activities that have been decayed to the years 2075 and 2150 to supplement the assessment of risks using present-day COPC concentrations. Radionuclide soil activities were not decayed over the duration of an exposure scenario, resulting in an overestimation of present-day cancer risk for short-lived radionuclides.

An EPA methodology to account for radiological decay during the period of the exposure duration was used to calculate the RCBRA soil PRGs (http://epa-prgs.ornl.gov/radionuclides/). The method is based on the radiological half-life of each radionuclide COPC, which is shown in Tables 3-29 and 3-30. For each radionuclide, a decay coefficient (λ) is computed as:

$$\lambda = \frac{LN(2)}{t_{1/2}} \qquad \text{Equation 3-28}$$

where $t_{1/2}$ is radiological half-life (yr).

A decay correction factor (CF) for each soil PRG is then calculated according to:

$$CF = \frac{ED \times \lambda}{1 - e^{-\lambda t}} \qquad \text{Equation 3-29}$$

where:

ED = exposure duration (yr)
t = radiological decay period (equivalent to ED).

In addition to soil PRGs for the exposure scenarios described in Section 3.3.1, soil PRGs are also derived for exposure scenarios that are consistent with the basis of the remedial action goals (RAGs) used in the IAROD process. RAGs for radionuclides were calculated using the RESidual RADioactivity (RESRAD) computer code based on exposure pathways including soil ingestion, dust inhalation, external radiation, produce ingestion, beef ingestion, and milk ingestion and leaching to groundwater followed by drinking water ingestion. RAGs for chemicals were based on exposure by soil ingestion, consistent with the State of Washington's "Model Toxics Control Act" (MTCA) published in the 1996 *Washington Administrative Code*

(WAC) Part 173-340. A discussion of these scenarios, and their implementation using a cancer risk threshold for radionuclides and present-day toxicity criteria for chemicals, is presented in Section 2.8. The IAROD-related RESRAD calculation for radionuclides that is the basis of the soil PRGs shown in Section 7.4 of this report is that used for the 100 and 300 Area RI/FS investigation and is fully described in (ECF-Hanford-10-0429, In Preparation). Some differences in exposure assumptions between this scenario and that originally used in the IAROD process (and adopted in Section 2.8) relate to the proportion of time spent in indoor and outdoor environments, and the radiation shielding factor assigned to a building.

3.8 REFERENCES

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4.0 BROAD-AREA RISK ASSESSMENT RESULTS

4.1 INTRODUCTION

The human health risk assessment (HHRA) component of the RCBRA has been conducted in three parts: broad-area risk, local-area risk, and groundwater risk. This section presents the results of the HHRA calculations using “broad-area” environmental data that characterize contaminant of potential concern (COPC) concentrations in upland and riparian surface soils, river water and sediment, and fish. These data sets are listed under the spatial-scale category of “broad area” in Table 3-9 and comprise all the near-shore environment data sets listed in Table 3-10. Results for the risk assessment scenarios evaluated in the RCBRA that use broad-area data are provided in this section of the report. Local-area risk assessment results related to individual remediated waste sites and groundwater operable units (OUs) are presented in Sections 5.0 and 6.0, respectively. The scope of the HHRA, as it relates to exposure scenarios, pathways, and media, is described in Section 3.3 and outlined in Figures 3-15 through 3-17. Potentially complete exposure pathways for scenarios evaluated using the broad-area data are outlined in detail in Table 3-11.

As discussed in Section 1.0 of this report, there are four steps in the baseline HHRA process. These include Data Collection and Evaluation, Exposure Assessment, Toxicity Assessment, and Risk Characterization. Inputs and methodology related to Data Collection and Evaluation, Exposure Assessment, and Toxicity Assessment are the subject of Section 3.0 and supporting appendices. The results sections of the HHRA (Sections 4.0, 5.0, and 6.0) correspond to the Risk Characterization. The results of the implementation of the methods for identification of COPCs, which is an aspect of Data Collection and Evaluation, are also presented in Sections 4.0, 5.0, and 6.0 of the HHRA.

Section 1.0 also lists five questions that the RCBRA is designed to address to provide information needed by risk managers to support final CERCLA decisions in the River Corridor that ensure protection of human health and the environment. Two of those questions are addressed on a broad-area scale in Section 4.0 including the following:

- Are residual conditions for cleanup actions under the interim action records of decision (IARODs) protective of human health and the environment?

- What are the uncertainties associated with the risk results and conclusions?

Risk assessment calculations for the Recreational scenarios (Avid Angler, Avid Hunter, Casual User) and Nonresident Tribal scenario occur over broad spatial scales where exposure concentrations are integrated across one or more record of decision (ROD) decision areas. In the special case of the Resident National Monument/Refuge scenario, the residential portion of the scenario employs the local-area data for an individual waste site, whereas the occupational portion of the scenario employs the broad-area upland surface soil data. The results for this scenario are presented in Section 5.0. In addition to the risk assessment results for the

Recreational and Nonresident Tribal scenarios, this section of the HHRA includes an evaluation of risks related to groundwater contaminants at shoreline springs along the Columbia River, and a screening-level evaluation of risks from exposure to near-shore COPCs in sculpin, clams, crayfish, and juvenile sucker collected along the river banks.

Table 3-11 shows the exposure routes, exposure media, and exposure areas applicable to the Avid Angler, Avid Hunter, Casual User, and Nonresident Tribal scenarios evaluated in the RCBRA. Each recreational scenario is associated with one of the three River Corridor environments (upland, riparian, and near-shore). The Casual User scenario addresses occasional recreational use and is focused on activities such as walking and picnicking in riparian areas near the river. The Avid Hunter is focused on individuals who are recreational hunters, as opposed to those engaged in subsistence hunting of game species such as deer and elk. This application has been associated with upland regions of the River Corridor. The Avid Angler, like the Avid Hunter, is focused on individuals who are not engaged in a subsistence lifestyle. The Avid Angler application is associated with exposure in the near-shore region of the River Corridor. Risks associated with the sediment exposure component of the Avid Angler scenario are presented in Section 4.2, while fish ingestion risks based on COPC concentrations in sculpin and other aquatic species with limited home range are discussed in Section 4.6.

The Nonresident Tribal scenario is focused on individuals engaged in a subsistence lifestyle who reside offsite but use the River Corridor for various activities such as hunting, gathering plants, and fishing. Hunting activities are assumed to occur within upland regions of the River Corridor, while fishing exposures relate to riparian and near-shore regions. It is assumed that individuals reside in temporary dwellings along the river for many consecutive days while catching and drying fish, and that they may participate in sweat lodges during these periods. Plants are assumed to be gathered in both riparian and upland areas. A single individual is assumed to engage in all these activities, so onsite exposures in the upland, riparian, and near-shore environments are additive for the Nonresident Tribal scenario.

4.1.1 Summary of Broad-Area Risk Assessment Results

The results of the risk assessment for the Casual User, Avid Hunter, and Avid Angler scenarios indicate that residual COPC concentrations in the riparian soil, near-shore sediment, and surface soil subsequent to waste site remediation are generally protective of human health under these land uses. Cancer risk and chemical HI results for the screening-level fish ingestion risk assessment were often above risk thresholds, but these results are applicable only for identifying risk-relevant COPCs and for evaluating relative risk among the six ROD decision areas. The risk assessment for fish ingestion being prepared for the RCBRA Columbia River Component (CRC) is intended to address realistic potential human health risks from ingestion of food fish in the Hanford Reach of the Columbia River. Nonresident Tribal risk assessment results above threshold values are driven by exposure to arsenic concentrations that have been modeled in plant tissues based on uptake from soil. The key uncertainties in the Nonresident Tribal results relate to actual plant arsenic concentrations in native plants that could serve as a food source, and whether the cancer and noncancer toxicity criteria for arsenic accurately reflect potential health risks from arsenic ingestion in foods at such concentrations.

The results of the broad-area risk assessment for the Casual User, Avid Hunter, and Avid Angler scenarios indicate minimal potential risks for these types of activities. Cancer risks are generally near or below the 1×10^{-6} de minimis threshold for direct exposures to soil, sediment, and surface water. Radiation dose and chemical hazard are below thresholds for these exposures. Exposure to game meat in the Avid Hunter scenario is associated with modeled reasonable maximum exposure (RME) cancer risks within the 1×10^{-6} to 1×10^{-4} risk management range due to exposures to arsenic, benzo(a)pyrene, and a polychlorinated biphenyl (PCB) Aroclor. The Avid Hunter RME hazard index is approximately equal to the threshold level of 1.0. Central tendency exposure (CTE) Avid Hunter CTE cancer risks are between 1×10^{-6} and 1×10^{-5}. Screening-level fish ingestion cancer risks were only calculated for the 100-K Area. This is the only ROD decision area where carbon-14 (the only carcinogenic COPC in near-shore biota) was detected. The RME screening-level Avid Angler cancer risk in the 100-K Area is 4×10^{-5}. Carbon-14 was detected in one of two sculpin samples from the 100-K Area, so this risk estimate is highly uncertain due to the limited amount of data. The RME screening-level HI values for Avid Angler sculpin ingestion ranged between 1.7 and 4.3.

The results of the broad-area risk assessment for the Nonresident Tribal scenario indicate potential cancer risk greater than 1×10^{-4} and chemical hazard greater than the threshold of 1.0, related primarily to the plant ingestion exposure pathway. Arsenic exposure is the main driver of both cancer risk and hazard due to plant ingestion. It contributes 90% or more to total cancer risk. A total cancer risk of 1×10^{-2} was calculated for most individual ROD areas. The Nonresident Tribal arsenic cancer risk calculated using arsenic concentrations in the upland and riparian reference areas was also 1×10^{-2}, indicating that the relatively small difference of about 50% between site and reference area arsenic concentrations does not impact estimated cancer risk reported using one significant figure. The range of upland and riparian soil CTE and RME arsenic concentrations was approximately 4 to 10 mg/kg, which are up to about twice the 90th percentile background levels for the Hanford Site (6.47 mg/kg) (DOE/RL-92-24, *Hanford Site Background: Part 1, Soil Background for Nonradioactive Analytes*) and Washington State Yakima Basin (5.13 mg/kg) (Ecology 1994, *Natural Background Soil Metals Concentrations in Washington State*). Site broad-area arsenic concentrations are well below the IAROD cleanup value of 20 mg/kg for arsenic, which was established by the Washington State Department of Ecology as an unrestricted land use cleanup value, with adjustment related to background levels of arsenic (http://apps.leg.wa.gov/WAC/default.aspx?cite=173-340-900). The IAROD level does not address exposures related to uptake of arsenic into foodstuffs. Arsenic concentrations in areas where orchards existed prior to Hanford Site operations may be considerably higher than the levels described in this report.

Nonresident Tribal cancer risks for exposure pathways other than ingestion of plants and game were generally within or slightly above the 1×10^{-6} to 1×10^{-4} risk management range. In the 100-B/C, 100-K, 100-D/100-H, 100-F/100-IU-2/100-IU-6, and 300 Areas, external irradiation related to fission products such as cesium-137 and europium-152 was most important. In the 100-N Area, ingestion of strontium-90 in the sweat lodge contributed 85% of the total cancer risk of 2×10^{-4}. Chemical hazard results for exposure to soil, sediment, and surface water were below the threshold of 1.0. Screening-level sculpin ingestion cancer risk and HI values were approximately six times higher than those described for the Avid Angler, reflecting the higher fish ingestion rates used in the Nonresident Tribal scenario.

The results of the risk assessment for groundwater contaminants at shoreline springs along the Columbia River indicate that for the majority of the shoreline springs there is negligible risk related to exposure to key groundwater contaminants being released to the Columbia River at these locations. For a subset of springs, concentrations of radiological contaminants may exceed long-term drinking water criteria by factors of between 2 and 10, but this would present negligible exposure for occasional users of a spring. At only one spring (100-F Spring) are concentrations of a contaminant (nitrate) elevated relative to short-term standards that apply to 1-day or 10-day drinking water exposure. There are no short-term health advisories for uranium; however, at spring 42-2 in the 300 Area, concentrations of uranium are routinely above drinking water standards.

A detailed screening-level assessment of the fish ingestion pathway is presented in Section 4.6. This assessment presents a summary of information on contaminants in fish and related health risk from EPA studies conducted in the Columbia River, and also provides scenario-specific risk results for fish ingestion based on tissue data for near-shore COPCs in sculpin, clams, crayfish, and juvenile sucker collected along the river banks. Two relevant studies have been released by EPA Region 10 during the past 10 years describing contaminant levels in fish tissues in the Columbia River Basin (EPA 910-R-02-006, *Columbia River Basin Fish Contaminant Survey 1996-1998*) and the Upper Columbia River (EPA 2007, *Phase I Fish Tissue Sampling Data Evaluation Upper Columbia River Site CERCLA RI/FS, Final*). Fish ingestion risks were estimated for Native American tribes and the general public in the Columbia River Basin study. The most important contributors to health risks were PCBs, arsenic, and mercury (EPA 910-R-02-006). Tissue concentrations of these contaminants in sculpin captured in the near-shore environment are either comparable to or below concentrations in various game fish reported in EPA 910-R-02-006 and EPA (2007). PCBs, arsenic, and mercury were not identified as near-shore COPCs in biota; therefore, they are not included in the scenario-specific screening-level risk calculations performed using near-shore COPC concentrations in sculpin, clams, crayfish, and juvenile sucker. The fish ingestion risk results for the Subsistence Farmer and Avid Angler scenarios were similar. Fish ingestion risk results for the CTUIR Resident and Yakama Resident scenarios were also roughly equivalent to those described for the Nonresident Tribal scenario.

4.1.1.1 Key Uncertainties. The key uncertainties in the Nonresident Tribal results relate to concentrations of metals and radionuclides in native plants, particularly arsenic. These plants could serve as a food source in this scenario, and risks have been calculated based on modeling of plant concentrations from soil data. Such models have a number of important uncertainties including differences in uptake among various plant species, and differences in concentrations among the parts of plants that may be consumed. There are also uncertainties associated with chemical form and soil properties that affect the uptake and retention of metals and radionuclides in plant tissues. Cancer risk and HI above threshold values were also calculated for COPCs related to sculpin ingestion in the Nonresident Tribal scenario. These are screening-level fish ingestion results, with the critical uncertainty being the applicability of sculpin as a food fish for chronic human consumption.

The remainder of Section 4.0 is organized as follows: The data sets used in the different components of the broad-area risk assessment, results of the COPC selection process, and calculation of representative concentrations are described in Section 4.2. Note that the assumptions and methods employed for defining COPCs are described in Section 3.2.2. Risk assessment results for the three Recreational scenarios and the Nonresident Tribal scenario are presented in Sections 4.3 and 4.4, respectively. An evaluation of potential risks related to exposure to water emanating from shoreline springs is presented in Section 4.5. A screening-level assessment of risks related to ingestion of fish from the Columbia River is presented in Section 4.6. The screening-level fish ingestion assessment evaluates risks related to COPCs in near-shore biota and focuses on localized species with very limited home range along the shoreline including sculpin, clams, and crayfish. This screening assessment will be supplemented in the CRC of the RCBRA using data from the sampling of game fish within the Hanford Reach of the Columbia River that are more representative of food sources for chronic exposure. The uncertainty analysis associated with the broad-area risk results is presented in Section 4.7.

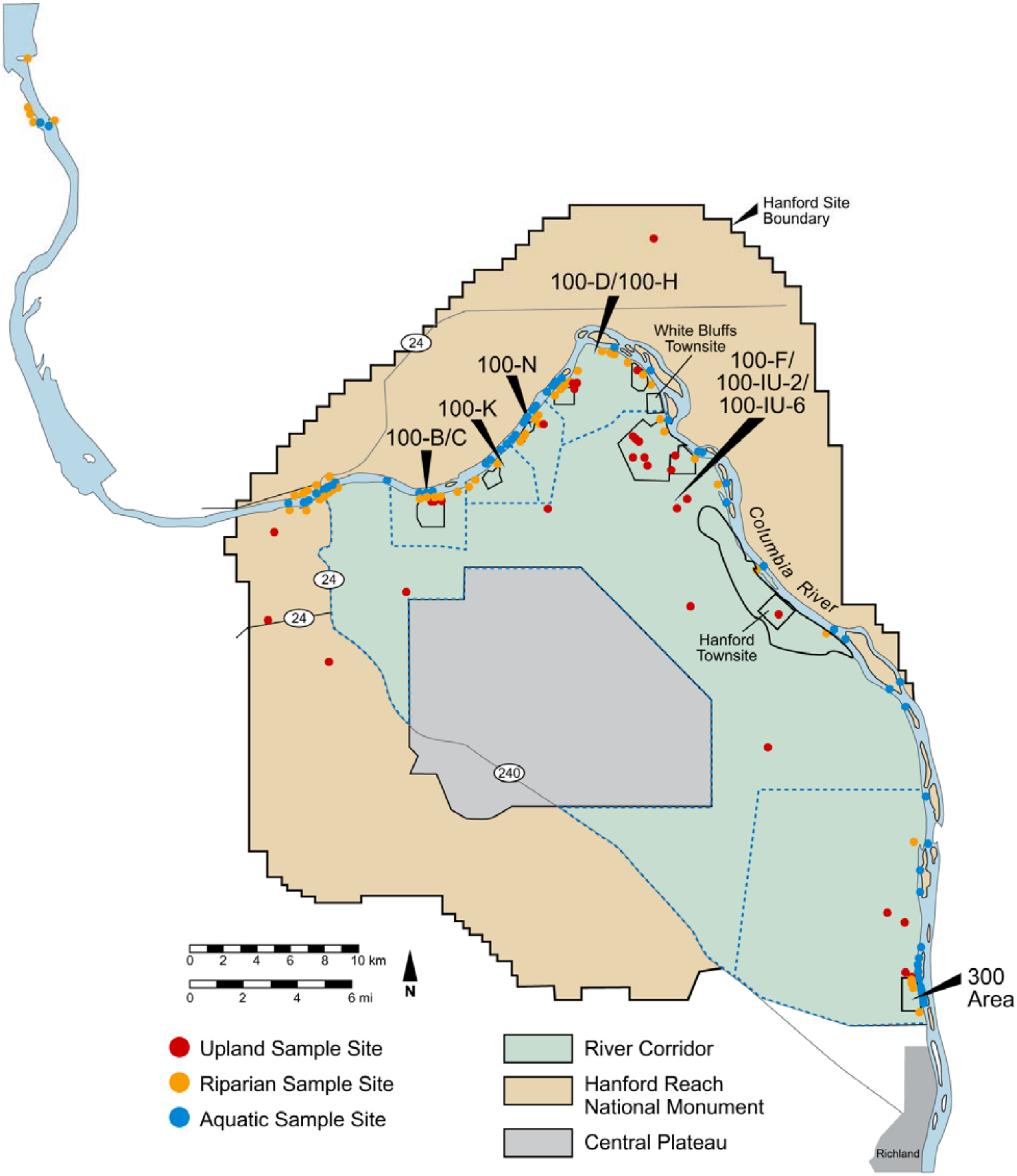
4.2 SITE INVESTIGATION AND DATA ANALYSIS

4.2.1 Broad-Area Data Evaluation

The data used in the HHRA to characterize broad-area exposures were primarily collected under the *100 Area and 300 Area Component of RCBRA Sampling and Analysis Plan* (DOE/RL-2005-42) (RCBRA SAP). The RCBRA SAP originally identified sampling locations within the 100 Area and 300 Area. The SAP was amended during the first year of field sampling to include sampling of the shoreline regions between the operating areas, referred to as the “Inter-Areas.” This assessment uses data from both the operating and the “Inter-Areas” to characterize contaminant concentrations in the River Corridor and does not differentiate between the original sampling distinctions. In addition to the data collected under the RCBRA SAP (DOE/RL-2005-42), environmental data obtained for the 100-B/C Pilot and 100-NR-2 Projects are used in the HHRA. Together, these data allow evaluation of exposures within the upland, riparian, and near-shore environments of the River Corridor.

Particular media and locations were selected for sampling under the RCBRA SAP (DOE/RL-2005-42) as a result of a data quality objectives (DQO) process conducted specifically for the RCBRA (BHI-01757, *DQO Summary Report for the 100 Area and 300 Area Component of the River Corridor Baseline Risk Assessment*). Through this process, gaps in existing data were identified and conceptual models were developed that considered the relationship between contaminant sources and potentially exposed receptors. Figure 4-1 shows the general locations where RCBRA samples were collected. The locations of samples collected for the 100-B/C Pilot and 100-NR-2 Projects are shown in Figure 4-1.

Figure 4-1. RCBRA Sampling Locations for the River Corridor.



Biotic and abiotic samples were collected under the RCBRA SAP (DOE/RL-2005-42) from onsite investigation areas and reference sites between October 2005 and October 2007. Site descriptive information and species/habitat metrics were also recorded for use in Volume I. The following paragraphs describe the types and quantities of biotic and abiotic data collected under the guidance of the SAP that are used in the HHRA. A description of relevant data collected for the 100-B/C Pilot and 100-NR-2 Projects is also provided. Appendix C-1 provides a summary of the data sources used in this report and additional details on specific analyses performed for the RCBRA sample media.

4.2.1.1 Data Collected Under the RCBRA SAP (DOE/RL-2005-42).

Abiotic Data. Data for abiotic sample media collected under the RCBRA SAP (DOE/RL-2005-42) that are used in the HHRA include sediment, upland soils, riparian soils, and surface water from the Columbia River. Most soil samples were collected as *MULTI INCREMENT*® samples (MIS), which are a form of composite samples across a 1-ha (2.47-ac) area, but a subset of soil samples were focused samples that were paired with a corresponding phytotoxicity bioassay. Only the MIS soil samples have been used to calculate representative concentrations in the HHRA.

The MIS methodology uses approximately 50 increments of soil, one from each section of the grid established over the study site. Each soil increment was collected from 0 to 15 cm (0 to 6 in.) depth. Analyses and field measurements conducted for all RCBRA abiotic media are shown below.

Media Sampled	Analyses Performed	Field Measurements
• Sediment • Upland and riparian soils • Pore water • Surface water	• Metals (including mercury) • Radionuclides (including gamma energy analysis, isotopic uranium, plutonium, thorium, tritium, and total beta radiostrontium) • Semivolatile organic analytes • Pesticides/polychlorinated biphenyls	• Soil type • Thermoluminescent dosimetry • Conductivity • pH • Temperature • Turbidity

Some of the abiotic data planned for this project were not obtained. Hexavalent chromium results for 35 river sediment samples collected in 2006 were determined to be unusable based on a quality assurance review of the data. Instrument calibration records were considered inadequate, and holding times were exceeded for some samples. River sediment from 17 locations was resampled in 2007, and the analyses were successfully completed.

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Biotic Data. Biotic samples were collected from upland, riparian, and aquatic investigation areas and reference sites. Analytical results used in the HHRA include tissue concentration data for vegetation, fish, aquatic macroinvertebrates, and clams. Analyses performed for RCBRA biotic media are listed below.

Media Sampled	Analyses Performed
• Upland and riparian vegetation (leaves) • Terrestrial invertebrates • Mice (whole organism, liver, kidney) • Kingbirds (crop contents, carcass) • Aquatic invertebrates • Clams (soft tissue, shells) • Sculpin (whole organism, liver, and kidney)	• Metals (including mercury) • Radionuclides (gamma energy analysis, isotopic uranium and thorium, and total beta radiostrontium) • Semivolatile organic analytes • Pesticides/polychlorinated biphenyl compounds

Data Quality Considerations. To ensure that the laboratory analytical data collected during the RCBRA investigation are adequate to support risk assessment and environmental decision making, EPA guidance pertaining to the appropriate type, quantity, and quality of environmental data was followed. The DQO process has been used throughout the development of the *Risk Assessment Work Plan for the 100 Area and 300 Area Component of the RCBRA* (DOE/RL-2004-37) and RCBRA SAP (DOE/RL-2005-42) to accomplish this objective.

Quality Assurance Requirements. Appendix C-1 describes these quality assurance requirements and provides a detailed discussion of the data reviews conducted for data obtained under RCBRA SAP (DOE/RL-2005-42). Additional information on data quality of data obtained under the RCBRA SAP (DOE/RL-2005-42) and data from other sources, in terms of acceptability and usability, is provided in Appendix C-2. General data quality considerations for the data obtained under RCBRA SAP (DOE/RL-2005-42) include the quality assurance requirements for analytical samples, completeness of analytical sample results, and sample representativeness. Six criteria are used to evaluate the data supporting the HHRA:

Criterion 1: Data Sources – Overall quality and level of detail in report
Criterion 2: Documentation – Formal documentation of procedures
Criterion 3: Analytical Methods – Analytical methods used and detection levels achieved
Criterion 4: Data Quality Indicators – Assessment of data quality indicators
Criterion 5: Data Review – Data review, validation, and quality assurance
Criterion 6: Data Reporting – Data history and overall apparent data quality.

Each data set used in the RCBRA was evaluated for these six criteria described. Each of the data sets was assigned to one of four levels of usability, based on the evaluation:

Level A: Acceptable, unrestricted use
Level B: Acceptable, some use restrictions may apply
Level C: Conditionally acceptable, limited uses
Level D: Conditionally acceptable, use with caution.

Appendix C-2 lists the results of this procedure for each of the data sets used in the RCBRA. The assignment of a level does not preclude or endorse the use of a data set for a particular purpose, but rather is intended to alert users to potential limitations in the data. All of the data collected under RCBRA SAP (DOE/RL-2005-42)2 were assigned Level A, as were the majority of the other data sources; hence, uncertainty related to these quality assurance measures is minimal.

Quality assurance requirements are also provided in the RCBRA SAP (DOE/RL-2005-42) (Section 2.0 of the SAP is the Quality Assurance Project Plan [QAPP]). The QAPP section of the SAP specifies the target quantitation limits (practical quantitation limits [PQLs]) and duplicate requirements for analytical samples. The PQLs are evaluated for nondetected analytes as part of the COPC refinement process. Nondetected analytes that do not meet PQLs are discussed in the uncertainty analysis of the risk assessment. Laboratory duplicate samples were used to validate the data but were not used to quantify exposure concentrations to avoid over-representing certain sample locations in the exposure assessment.

Completeness of Data Collection and Analysis. The primary data obtained under RCBRA SAP (DOE/RL-2005-42) used in the HHRA are the abiotic sample results. In addition, biotic media sample results are used in some of the calculations. Collection of environmental samples presents a number of inherent challenges and, generally, obtaining a complete set of abiotic media samples (e.g., soil, water) is less difficult than sampling biotic media (e.g., fish tissue, plant tissue). In some cases, targeted species will be uncommon or difficult to sample in the amounts needed for all analytical suites. Weather conditions, worker safety, seasonal or annual variations in abundance, and other environmental factors can have unexpected impacts on sample media accessibility. In other cases, the characteristics of the biotic tissue itself may interfere with sample extraction or measurement and make it difficult to obtain planned quantitation limits. Where acquisition of the samples or data was not possible as originally planned, the RCBRA project implemented contingencies and supplemental sampling to meet the requirements for abiotic and biotic data. The contingencies and supplemental sampling that pertain to the HHRA include the following:

- Sampling of aquatic invertebrates in the Inter-Areas shoreline assessment supplemented data for the 100 Area and 300 Area

- Supplemental fish sampling was conducted in 2008 to improve the quantitation limits achieved during PCB analysis

- Supplemental sediment sampling was conducted in 2007 to replace rejected hexavalent chromium data.

At the conclusion of each sampling effort, field activities and details pertaining to sample analysis were documented in a summary report. These reports provide detailed descriptions of conditions encountered during sample collection for each RCBRA field campaign and include the following:

- *100 Area and 300 Area Component of the RCBRA Fall 2005 Data Compilation* (WCH-85)
- *100 Area and 300 Area Component of the River Corridor Baseline Risk Assessment Spring 2006 Data Compilation* (WCH-139)
- *Inter-Areas Component of the River Corridor Baseline Risk Assessment Sampling Summary* (WCH-274).

These reports and other supporting information and documents can be found on the Internet at http://www.washingtonclosure.com/Projects/EndState/risk_library.html.

Sampling Representativeness. Selection of sampling locations and field execution of the sample plan was designed to conservatively characterize exposure of targeted receptors to potential residual Hanford Site-related contamination. As described in Section 3.0 of the RCBRA SAP (DOE/RL-2005-42), upland, riparian, and near-shore aquatic sampling sites were targeted to include areas expected to contain the highest levels of residual contamination. For example, upland sampling plots were configured to overlap excavation side-slope and layback areas in remediated waste sites. Riparian and near-shore aquatic sites were selected based on information from radiological surveys and groundwater plume maps, with the intent of including sites with the most apparent potential for elevated contaminant levels. In addition to choosing locations with the goal of including "worst-case" sites, the sample collection schedule considered seasonal and other environmental factors that can impact optimal exposure characterization.

4.2.1.2 100-B/C Pilot and 100-NR-2 Soil, Sediment, and Biota Data. Upland surface soil data obtained in the 100-B/C Pilot Project at five remediated waste sites are used in the HHRA. These sites include 116-B-11, 116-B-11 N, 116-C-1 N, 116-C-1 S, and 116-C-5 S. Unlike the analyses performed under the RCBRA SAP (DOE/RL-2005-42), the analytical soil data obtained for the 100-B/C Pilot investigation is limited to metals and certain radionuclides. At 116-C-1 N and 116-C-1 S there are both radionuclide and metals data. At 116-B-11 N and 116-C-5 S there are only radionuclide data, and at 116-B-11 there are only metals data.

Riparian surface soil data obtained in the 100-B/C Pilot and 100-NR-2 Projects are also used in the HHRA. Riparian sampling locations in the 100-B/C Pilot Project include 100 B/C Area DR, 100 B/C Intermediate, 100 B/C Outfall 1, 100 B/C Outfall 2, 100 B/C Outfall 3, 100-B Spring 38-3, and 100-B Spring 39-2. Because one MIS sample location was also included under the RCBRA SAP (DOE/RL-2005-42) (2a Riparian), there are a total of eight riparian surface soil locations in the 100-B/C ROD Area. There are a number of riparian soil sampling locations described in the 100-NR-2 report. However, some samples are co-located and soil samples were binned into three location-groups in the 100-NR-2 report. Because one MIS sample location was also included under the RCBRA SAP (DOE/RL-2005-42), a total of four riparian surface soil locations in the 100-NR-2 ROD Area are evaluated in the HHRA. Like the 100-B/C Pilot investigation, analytical soil data at 100-NR-2 is limited to metals and certain radionuclides.

For the 100-B/C Pilot Project, two sediment samples were collected and analyzed for metals and radionuclides. In addition, 17 surface water samples were collected and analyzed for metals, VOCs, tritium, and technetium-99. A subset of eight of these water samples were also analyzed for beta- and gamma-emitting radionuclides. Aquatic biota samples were also obtained for the 100-B/C Pilot Project. Between five and eight samples of clam soft tissue were analyzed for metals, PCB Aroclors, strontium-90, and technetium-99. One of these clam samples was also analyzed for beta- and gamma-emitting radionuclides. Five to eight sculpin samples were also analyzed for metals and PCB Aroclors, with a single sample analyzed for beta- and gamma-emitting radionuclides.

Three sediment samples from the 100-NR-2 Project were analyzed for metals, PCB congeners, strontium-90, and technetium-99. Three water samples were also obtained and analyzed for metals, strontium-90, and technetium-99. Aquatic biota samples were also obtained for the 100-NR-2 Project. Twenty-seven samples of clam soft tissue were collected and analyzed for selected metals, with other metals analyzed in a subset of 7 of 12 of the samples. Five sculpin samples were collected and analyzed for PCB congeners.

4.2.1.3 Shoreline Spring (Seep) Water Data. Water data from seeps along the Columbia River have been obtained as part of the 100-B/C Pilot and 100-NR-2 Projects, as well as under the Surface Environmental Surveillance Program (SESP). These seeps represent surface discharge of the unconfined aquifer that underlies the Hanford Site. The presence and flow rate of water at the seeps is governed by fluctuations in the level of river water. River water moves into the aquifer at high water levels and returns as spring water when the river level decreases. After extended periods, low water level seeps may not be identifiable. These processes also affect concentrations of groundwater-related contaminants in the seep water.

Major groundwater plume contaminants in the seven groundwater OUs underlying the River Corridor are described in *Summary of Hanford Site Groundwater Monitoring for Fiscal Year 2006* (PNNL-16346). A subset of these contaminants including chromium, nitrate, tritium, strontium-90, technetium-99, and uranium are discussed in the *Hanford Site Environmental Report for Calendar Year 2008* (PNNL-18427) as key groundwater contaminants observed in seep water. Tables 4-1 through 4-6 show the number of sample results, detection information, range of detected values, and range of nondetect values for the key contaminants in seep water within each of the six ROD decision areas.

4.2.2 Results of COPC Identification for Broad-Area Data in the Upland, Riparian, and Near-Shore Environments

The human health exposure scenarios address exposures on different spatial scales depending on the receptor, activity, and exposure medium (see Section 3.3). For exposures related to soil, sediment, or biota that are likely to occur over relatively broad areas, analytes are identified as COPCs in the upland, riparian, and near-shore environments. Contaminant of potential concern refinement at this scale is performed with data collected under the RCBRA SAP (DOE/RL-2005-42), supplemented with data collected for the 100-B/C Pilot and 100-NR-2 investigations.

Contaminant of potential concern refinement for the upland, riparian, and near-shore environments has been documented in Volume I, the Ecological Risk Assessment. This documentation includes tables presenting the count of samples, maximum concentration, and the significance (p-value) for the statistical tests of comparison to background and/or reference site concentrations for the various sampled media. Tables documenting the analyte-specific evaluation in cases where the statistical tests were inconclusive are also presented for each of the three environments. To avoid repetition of this information, cross-references to specific tables in Volume I are provided below and tables with the final lists of COPCs in each environment are included in this volume.

The results of statistical tests using data from all River Corridor sampling locations and media were supplemented by review of results for individual sampling locations and decision areas in order to identify COPCs. This additional review focused on identifying anomalous results at individual locations when sample size or detection frequency was such that statistical results were inconclusive. Both MIS and discrete soil sample results were used for COPC identification. The upland and riparian soil results in most ROD decisions areas are primarily from MIS samples, although a subset of the data used for COPC identification are from discrete samples collected at the remediated waste sites. In the 100-B/C and 100-N ROD decisions areas, the majority of the soil data are from discrete samples. As discussed in Section 4.1.3, MIS and discrete samples were not combined for the purpose of calculating representative concentrations.

In some specific cases, the basis of COPC refinement by which an analyte was identified as a COPC for the ecological risk assessment is not applicable to the HHRA. This may occur because the HHRA addresses only a subset of the sampled media considered in the ecological risk assessment. Contaminants of potential concern in the upland, riparian, and near-shore environments for the HHRA are discussed in the following section.

4.2.2.1 Considerations for Defining the Spatial Scale of COPC Refinement. For most of the COPCs in the upland, riparian, and near-shore environments, the Gehan test indicates that there has not been an overall shift in the bulk of the site data in comparison to the reference area data. Instead, a relatively few concentrations close to or above the range of reference area results are what distinguishes a COPC. When statistical tests are inconclusive, a common basis for identifying a COPC in the analyte-specific evaluations is that the higher concentrations are localized at only one or two sampling locations.

In the upland environment, sampling locations are primarily distinguished by native soil (unexcavated) versus backfilled (excavated) remediated waste sites. Although relevant for ecological characteristics of the remediated waste sites, such as species type and diversity, these categories are not significant for the HHRA because COPC concentrations are the only inputs to the assessment. Representative concentrations for upland COPCs are calculated on the same scale (the entire upland environment including native soil and backfilled sites) for which COPC refinement occurs. Therefore, there is accord between the spatial scale of COPC identification and the scale at which representative concentrations are computed for risk results in the upland environment.

In the riparian and near-shore environments, human health risks are evaluated separately for each of the six individual ROD decision areas. However, because site data in individual ROD decision areas are inadequate to conduct statistical tests, COPC identification is performed in each environment using the data from across all six ROD decision areas. The consequence of identifying COPCs for the entire riparian or near-shore environment on the broad-area risk assessment results for an individual ROD decision area may be a masking of the influence of site-related contamination in an individual decision area. For example, arsenic was identified as a COPC in riparian soil based on soil concentrations at riparian site 1 in the 100-D/100-H Area and on soil concentrations in the 100-B/C Pilot data. Arsenic can contribute to significant calculated cancer risk and chemical hazard even when it is present at background concentrations. In other ROD decision areas, risks related to COPCs that are elevated in riparian soils may be difficult to distinguish because they are masked by the contribution of arsenic at concentrations that approximate background. For this reason, the HHRA includes calculations of cancer risk and radiation dose above reference area levels of the COPCs included in a calculation. For chemical hazard, emphasis is placed on the relative hazard in a ROD decision area compared with chemical hazard for COPC concentrations in reference areas. The cancer risk results for all individual chemicals are provided as an attachment to Appendix D-5 of this report.

4.2.2.2 Use of Data for Multiple Environmental Media in COPC Refinement. Frequently, an analyte that is elevated in one sample medium (for example, soil or some biotic tissue) is not elevated in other sample media in that environment. As described in RCBRA Volume I, COPC refinement made use of environmental data from multiple abiotic and biotic media. However, not all of these media were used in the HHRA. If COPCs were identified in RCBRA Volume I solely on the basis of concentrations in media that are not employed in the HHRA, these analytes are not defined as COPCs in RCBRA Volume II. Details are described with respect to each of the three River Corridor environments in the following paragraphs.

Upland Environment. Soil, plant tissue, mammal, and terrestrial invertebrate tissue data collected under the RCBRA SAP (DOE/RL-2005-42) and the 100-B/C Pilot Project were reviewed in RCBRA Volume I to identify COPCs in the upland environment.

Of these media, only soil and plant tissue are identified in the CSM (Section 3.3) as potential human exposure media. However, as discussed in Section 3.4.3, because of detection limit problems for a number of analytes, exposure point concentrations in plants were modeled from the soil data rather than computed directly from the plant tissue data. RCBRA Volume I indicates that the basis of COPC identification for all upland COPCs relates primarily to concentrations in soil.

Volume I upland COPCs that were identified based solely on media other than soil, and are therefore ecological COPCs but not human health COPCs, include the following:

- Aluminum (invertebrate data from 100-B/C Pilot Project)
- Beryllium (plant data from 100-B/C Pilot Project)
- Manganese (invertebrate data from 100-B/C Pilot Project)
- Selenium (mammal organ data from 100-B/C Pilot Project)
- Zinc (invertebrate data from 100-B/C Pilot Project).

No COPCs were identified for the HHRA based solely on analyte concentrations not used to calculate representative concentrations in the HHRA. Doing so could potentially result in masking the influence of media-specific COPCs on the risk results, since soil concentrations of these five metals are not different between operational and reference areas. Therefore, the five metals cited in these bullets are *not* identified as upland COPCs for the HHRA. Box plots of upland soil and plant tissue concentrations are provided in Appendices C-5.0 and C-6.0, respectively.

Documentation of COPC refinement in the upland environment is provided in RCBRA Volume I tables summarizing analytical results and statistical comparisons to background/reference. The COPCs in the upland environment for the HHRA are shown in Table 4-7.

Riparian Environment. Soil, plant tissue, invertebrates, birds, and small mammal data collected under the RCBRA SAP (DOE/RL-2005-42), the 100-B/C Pilot Project, and the 100-NR-2 investigation were reviewed in RCBRA Volume I to identify COPCs in the riparian environment. Of these media, only soil and plant tissue are potential human exposure media. However, as discussed in Section 3.4.3, because of detection limit problems for a number of analytes, exposure point concentrations in plants were modeled from the soil data rather than computed directly from the plant tissue data.

RCBRA Volume I indicates that the basis of COPC identification for most riparian analytes relates primarily to concentrations in soil.

Volume I riparian COPCs that were identified based solely on media other than soil, and are therefore ecological COPCs but not human health COPCs, include the following:

- Barium (plant and invertebrate data)
- Copper (plant and invertebrate data)
- Zinc (plant data).

No COPCs were identified for the HHRA based solely on analyte concentrations not used to calculate representative concentrations in the HHRA. Therefore, the three metals cited in these bullets are *not* identified as riparian COPCs for the HHRA. Box plots of riparian soil and plant tissue concentrations are provided in Appendices C-7.0 and C-8.0, respectively.

The RCBRA sampling of the riparian and near-shore environments was focused foremost on those regions where groundwater plumes of three different contaminants impact the river. These include (1) hexavalent chromium at the 100-K and 100-D/100-H Areas, (2) strontium-90 at the 100-N Area, and (3) uranium at the 300 Area (DOE/RL-2005-42). Only a single location was sampled under the RCBRA SAP (DOE/RL-2005-42) in the 100-N Area, because previous data existed from the 100-NR-2 investigation. A review of the riparian soil box plots in Appendix C-7.0 shows that hexavalent chromium and uranium do appear elevated at 100-D/100-H and 300 Areas, respectively. Strontium-90, however, was not detected in the five riparian soil samples at the 100-N Area.

Documentation of COPC refinement in the riparian environment is provided in RCBRA Volume I tables summarizing analytical results and the statistical comparisons to background/reference. The COPCs in the riparian environment for the HHRA are shown in Table 4-8.

Near-Shore Environment. Sediment, pore water, river water, fish (sculpin and sucker), bivalves (clams and mussels), amphibians, and benthic macroinvertebrates were sampled in the near-shore environment investigation under the RCBRA SAP (DOE/RL-2005-42). Sediment, surface water, sculpin, sucker, benthic macroinvertebrates, and clam tissue were sampled for the 100-B/C Pilot Project. Sediment, surface water, pore water, sculpin, and clam tissue were sampled for the 100-NR-2 Project. Of these media, sediment, surface water, sculpin, clams, and benthic macroinvertebrates (primarily crayfish, by mass) are evaluated in the HHRA as potential human exposure media. Sucker and mussels were not identified as potential exposure media because sampling was limited to only a few locations and these species provide analogous information to the sculpin and clams, respectively. Sculpin and sucker data were obtained for individual organs as well as for the whole organism. Clam data were obtained for the soft tissue and for the shell. In the HHRA, only whole-organism fish data and clam soft tissue data are employed because these are most relevant for human exposures.

The results of the statistical tests and analyte-specific evaluation conducted to identify COPCs in the near-shore media are presented in RCBRA Volume I. The analyte-specific evaluations were reviewed to divide COPCs into those associated with abiotic media, biotic media, or both. Both calculated total uranium and uranium (inorganic) were identified as near-shore COPCs based on sediment concentrations in Volume I. To avoid double-counting, only one of these data sets can be used to quantify sediment-related human health risk for uranium metal. The calculated total uranium data are used because they show significantly greater detection frequencies than the inorganic uranium data.

The analyte-specific evaluation shows that several analytes were identified as COPCs based on review of data that are not employed in the HHRA. Volume I near-shore COPCs that were identified based solely on media other than sediment, surface water, sculpin, clams, and benthic macroinvertebrates include the following:

- Antimony (sculpin liver data)
- Tin (sculpin kidney data)
- Plutonium-238 (pore-water data)
- Tritium (pore-water data).

No COPCs were identified for the HHRA based solely on analyte concentrations not used to calculate representative concentrations in the HHRA, because doing so could potentially result in masking the influence of media-specific COPCs on the risk results. Therefore, the four analytes cited in these bullets are *not* identified as near-shore COPCs for the HHRA. Box plots of near-shore analyte concentrations in sediment and surface water are provided in Appendix C-9.0. Box plots of near-shore analyte concentrations in clams and sculpin are provided in Appendix C-10.0.

As described with respect to riparian environment COPCs, the RCBRA sampling of both the riparian and near-shore environments was focused foremost on those regions where groundwater contaminant plumes impact the river. A review of the sediment box plots in Appendix C-9.0 shows that the soluble contaminant hexavalent chromium does not appear elevated in 100-D/100-H Area sediments. Inorganic and isotopic uranium concentrations in the 300 Area, and strontium-90 sediment concentrations at the 100-N Area, do appear elevated relative to concentrations in other ROD areas.

Documentation of COPC refinement in the near-shore environment is provided in RCBRA Volume I tables summarizing analytical results and the statistical comparisons to background/reference. The COPCs in biotic and abiotic media in the near-shore environment for the HHRA are shown in Table 4-9.

4.2.3 Exposure Point Concentrations for Broad-Area Data in the Upland, Riparian, and Near-Shore Environments

As described in Section 3.4, an exposure point concentration is an estimate of the COPC concentration in a given medium likely to be contacted by a receptor over time within the exposure area. Exposure point concentrations are calculated based on representative concentrations. The distinction between exposure point concentrations and representative concentrations for the HHRA is that representative concentrations pertain to sampled media, such as soil and sediment, whereas exposure point concentrations may also include modeled concentrations in other exposure media such as dust or plants as well as concentrations at future times. Methods for computing representative concentrations for soil were described in Section 3.4.1.

Broad-area exposure point concentrations in the upland environment are associated with the Avid Hunter recreational scenario and with certain activities in the Nonresident Tribal scenario. Additionally, the occupational portion of the Resident National Monument/Refuge scenario uses broad-area exposure point concentrations for the upland environment. Potentially complete exposure pathways for the recreational and Nonresident Tribal scenarios are outlined in detail in Table 3-11. The Resident National Monument/Refuge scenario is discussed in Section 5.0.

The MIS upland surface soil data are used to calculate broad-area representative concentrations, which describe contaminant concentrations that are applicable over the scale of an entire ROD decision area or larger. These data are considered to be broadly representative of present-day surface soil contamination in upland areas around remediated waste sites. As described in Section 4.2.2, identification of COPCs in upland surface soil was performed by aggregating the surface soil data available across all six ROD decision areas of the River Corridor, including the 100-B/C Area where data from the 100-B/C Pilot Project were used. Upland environment MIS sampling locations are primarily distinguished by native soil (unexcavated) versus backfilled (excavated) remediated waste sites. Contaminant of potential concern concentrations for these two types of remediated waste sites are very similar. There are no River Corridor upland surface soil data in the 100-K Decision Area, and only one remediated waste site was sampled at the 100-N Decision Area. For these reasons, calculation of upland environment representative concentrations for each individual ROD decision area is impractical. Only a single set of representative concentrations is calculated for upland surface soil using the MIS data from 20 upland sampling sites. These exposure point concentrations are used to represent upland soil COPC concentrations in all ROD decision areas.

Contaminant of potential concern concentrations between the 100-B/C Pilot soil data and the MIS data show notable differences, particularly in the residual concentrations of certain fission products. For example, isotopic europium, cesium-137, and strontium-90 are higher at the 100-B/C remediated waste sites. These differences may in part reflect the fact that each MIS sampling location is composed of five 1-ha (2.47-ac) composite MIS samples, whereas the 100-B/C locations are characterized by a single discrete soil sample. The MIS sampling method is designed to provide a representative average surface soil concentration at a remediated waste site rather than a concentration representative of a single discrete sample location. Because of these differences, upland surface soil representative concentrations for the upland environment are calculated using only the MIS data.

Broad-area soil representative concentrations in the riparian environment are associated with the Casual User and Nonresident Tribal scenarios, as well as the inhalation exposure route for the Avid Angler exposure scenario (see Table 3-11). Fishing activities within the Nonresident Tribal scenario are assumed to occur predominantly in the riparian and near-shore environments. MIS soil sampling data from 16 riparian locations in the 100-K, 100-D/100-H, 100-F/100-IU-2/100-IU-6, and 300 Areas are available for calculating representative concentrations in riparian soil. Additionally, there is one MIS location plus seven discrete locations sampled for the 100-B/C Pilot Project in the 100-B/C Area. In the 100-N Area, there is one MIS location plus additional discrete soil samples from three locations collected as part of the 100-NR-2 investigation. The MIS soil data from the riparian reference area sites will be used to calculate representative concentrations associated with background in riparian areas.

Riparian soils have different concentrations of certain analytes than the upland soils, as can be observed in the box plots in Appendices C-5.0 and C-7.0. For this reason, and because riparian and upland soils represent different environments that may be used for different kinds of activities, they are kept as discrete potential exposure media for the HHRA. Among the six ROD decision areas, riparian soil concentrations exhibit differences for various metals, radionuclides, and organic chemicals. Unlike the upland environment, there are generally an adequate number of soil samples in each ROD decision area to support calculations of representative concentrations in the riparian environment. Another distinction between riparian and upland environments is that recurring recreational and Nonresident Tribal exposures in individual ROD areas for swimming or fishing is more likely than for the activities such as hunting and gathering plants in the upland environment. Therefore, riparian soil representative concentrations are calculated for each individual ROD decision area rather than for all ROD areas together.

Sediment representative concentrations are related to the Avid Angler and the fishing component of the Nonresident Tribal scenarios (see Table 3-11). Fishing activities within the Nonresident Tribal scenario are assumed to occur predominantly in the riparian and near-shore environments. Sampling sites in the River Corridor were selected based on the locations of known groundwater plumes, results of a 2005 conductivity survey for identifying areas of groundwater discharge to the river, results of past biota sampling locations, areas of fine-grained sediment deposits, and other available information indicating areas of potential contamination. Therefore, sediment locations were selected to provide a conservative estimate of general sediment concentrations along the Hanford Reach. Sampling sites associated with known groundwater plumes were located within the 300 Area (primarily the uranium plume), 100-D and 100-K Areas (primarily the chromium plumes), and 100-N Area (primarily the strontium plume), between the 100-B/C Area and 100-F Area (nitrate plume) and downstream of the Hanford townsite (nitrate, technetium-99, and tritium plumes).

Similar to riparian soil, sediment representative concentrations were calculated for each individual ROD decision area. Among the six ROD decision areas, sediment concentrations exhibit significant differences for various metals and radionuclides. Concentrations of uranium in 300 Area sediments are particularly elevated relative to other ROD decision area and reference area sediments. Plots of the sediment data for the different ROD decision areas are provided in Appendix C-9.0.

To support the uncertainty analysis in Section 4.7, risk results for the recreational scenarios are calculated using both CTE and RME values for representative concentrations and exposure parameter values. For the Nonresident Tribal scenario, a single set of exposure parameter values were obtained from the scenario reports provided by the tribes (Harris and Harper 2004, Ridolfi 2007). Exposure parameter values for the three recreational scenarios are summarized in Tables 3-18 and 3-19.

As discussed in Section 3.4, the protocol for computing representative concentrations depends on the number of detected values in the data set. When there are four or more detected sample results, the mean and a 95% upper confidence limit on the mean (UCL) are calculated using statistical methods. These values are used for the CTE and RME representative concentrations, respectively. When the detect sample size *n′* is four or less, CTE and RME representative concentrations are calculated as follows:

- *n′* = 0; then no CTE, no RME
- *n′* = 1; then the detected result is used for the CTE and RME value
- *n′* = 2; then maximum detect is used for the CTE and RME value
- *n′* = 3 or 4; then the average is the CTE and the maximum detect is used as RME.

Representative concentrations when *n′* is four or less will be based on only a few data points. For the broad-area risk assessment, this means that only a few sample points are used to characterize a large area. However, as discussed in Section 3.4.1, supplemental representative concentrations are also calculated using the sample-specific reporting limit for chemicals and a method-specific minimum detectable activity for radionuclides. The results of these calculations are used in the Uncertainty Analysis in Section 4.7 to identify the practical outcome of uncertainty in representative concentrations calculated from data sets with few detections.

As described in Section 4.2.1, five replicate MIS samples were collected from each soil sampling location in the upland and riparian environments. In the broad-area risk assessment, each site sampling location represents a virtual point in the larger exposure area. For this reason, multiple samples from individual locations are not treated as independent samples of the broad area, but rather as samples that represent COPC concentrations at that particular location. The average value of a COPC concentration at each sampling location, computed using the same protocol as the broad-area CTE representative concentration described above, was used to represent the site COPC concentration at an individual location with multiple samples. Hence, sample size for the upland and riparian site and reference area representative concentrations is equivalent to the number of sampling locations available for the calculation. The broad-area representative concentrations were then computed across all sampling locations as described above. Because COPC concentrations at each MIS sampling location are based on five MIS per 1 ha (2.47 ac), there is little variance in the mean concentration among all sampling locations and, therefore, relatively small differences between CTE and RME representative concentrations.

There are two COPCs addressed in the risk assessment for which multiple types of data may exist: uranium metal and nitrate/nitrite. In addition to assessment of radiological cancer risk for each isotope, uranium as a metal is evaluated in the risk assessment because of its potential toxicity to the kidney. The broad-area soil data include results of inorganic uranium, and also calculated total uranium results based on radioisotope data. Because the inorganic uranium results have relatively low detection frequencies, the calculated total uranium results for upland and riparian surface soil are used in the risk assessment. The broad-area soil data sets also include results for nitrate, nitrite, and the sum of both nitrate and nitrite. Because toxicity criteria are provided by EPA for nitrite and nitrate individually and results for these analytes exist whenever data for the sum of both nitrate and nitrite exist; the data for nitrate and nitrite individually are used, and the results based on the sum of nitrate and nitrite are not used.

The tabulated broad-area results are the sum of the risk values for the individual COPCs included in the risk calculation. These tabulated results and accompanying text summaries present the risks that have been summed across the individual exposure pathways and COPCs. The principal COPCs and exposure pathways contributing to risks above the threshold criteria for cancer risk, radiation dose, and chemical hazard (see Section 3.6) are discussed in the text of the risk assessment.

Many radionuclide sample results were not censored by the analytical laboratory at the minimum detectable activity. These non-censored values, which may include both positive and negative values, are used in the same manner as a detected result to calculate representative concentrations. In some cases, representative concentrations are calculated for a radionuclide even when all results are below the minimum detectable activity. Rarely, a negative value is computed for a radionuclide representative concentration. When this occurs, a value of zero is substituted in the risk calculations so that the COPC risk result is not negative. The use of noncensored radionuclide results may impart a slight protective bias to the total cancer risk and total radiation dose results in this section of the report when a positive representative concentration was calculated and all measurements were below the minimum detectable activity.

Files of the representative concentrations used to calculate exposure point concentrations in the various exposure media are provided as an electronic attachment to Appendix C-3. The exposure point concentrations and results of the risk assessment calculations for each individual COPC and exposure pathway contributing to the sums shown in the Section 4.0 tables are provided in electronic attachments to Appendix D-5.

4.2.4 Contaminant of Potential Concern Concentrations and Risks Related to Background and Reference Areas

The CTE and RME representative concentrations for the upland soil and riparian soil reference areas are shown in Table 4-10. The COPCs include all analytes identified as soil COPCs in the upland and riparian environments. As summarized above, the values were computed using the same protocol as the CTE and RME site broad-area representative concentrations. The reference area CTE and RME concentrations are primarily intended for use in calculating risks that are comparable to those for the site, but representative of areas that were not impacted by Hanford releases. The 90^{th} percentile values for Hanford Area background and Washington State Yakima Basin background are also shown in Table 4-10. These values are commonly used to represent a reasonable upper bound on naturally occurring background soil concentrations. There is an important distinction between the 90^{th} percentile and the representative concentrations used in the risk assessment. As one pools a larger number of sample results in the representative concentration calculation, then the 95% UCL on the mean approaches the mean concentration. There is no such trend for the 90^{th} percentile. One expects that on average the 90^{th} percentile does not change with the number of sample results available to estimate this statistic. In addition, because these 90^{th} percentile values are based on discrete soil samples collected over a much larger geographic area with different soil types, they tend to reflect a more heterogeneous population of background concentrations. In contrast, the reference area RME values are computed as the 95% UCL on the mean using MIS with relatively low variance within and among reference sites.

Tables 4-11 through 4-13 show COPC-specific reference area cancer risk, radiation dose, and hazard quotient (HQ) for each exposure scenario evaluated in the broad-area risk assessment. The values in these tables are referenced in the risk assessment results in Sections 4.3 and 4.4 to provide a context for site-related risks of key COPCs. Because reference area and site risks are computed using the same types of samples and representative concentration methodology, they are highly comparable.

4.3 RECREATIONAL USE SCENARIO RESULTS

The risk assessment results for the Recreational scenarios reflect exposures that are likely to occur over a larger area than that associated with any particular remediated waste site. The Recreational exposure scenario is made up of three separate scenarios: Casual User, Avid Angler, and Avid Hunter. Each of these scenarios includes both an adult and child receptor. In the Avid Angler and Avid Hunter scenarios, children are represented as 7 to 12 years of age rather than the traditional default assumption of a child being 1 to 6 years old, because young children are not anticipated to regularly participate in these activities. The Recreational scenarios are more fully described in the CSM (Section 3.3). Exposure media, data sets, and potentially complete exposure pathways are summarized in Table 3-11. Exposure for the Avid Hunter and Casual User exposure scenarios is evaluated using surface soil data for upland and riparian environments, respectively. For the Avid Angler scenario, sediment and surface water exposure in the near-shore environment is evaluated with dust inhalation included via potential exposure to airborne dusts originating in the nearby riparian environment.

As described in Section 3.4.4, chemical hazard is calculated separately for adult and child receptors. For the Recreational scenarios, child HIs generally exceed those for adults. An exception is the Avid Hunter scenario, where child and adult values are approximately equal because exposure via the game ingestion pathway is shared between these receptors and is assessed using ingestion-rate data across all ages. Because child HIs are either larger than or equal to the HI values for adults, only child HI values are presented in the following subsections.

4.3.1 Casual User Results

Tables 4-14 through 4-16 contain the RME and CTE risk assessment results for the Casual User exposure scenario in the riparian environment. Risk assessment results for the Casual User exposure scenario are also calculated with radionuclide soil concentrations decayed to the years 2075 and 2150. Soil concentrations for these times were calculated using the RESRAD computer code, as described in Appendix D-1.4.1. These results show the effect of radionuclide decay over time on cancer risk and radiation dose, and also include the loss of these radionuclides from soil due to leaching with rainfall.

4.3.1.1 Cancer Risk. Table 4-14 presents the Casual User total site cancer risks with present-day radionuclide concentrations, and with concentrations decayed to years 2075 and 2150. Present-day RME total site cancer risks in the six ROD decision areas were between 2×10^{-6} and 8×10^{-6}. The highest RME cancer risks were in the 100-F/100-IU-2/100-IU-6 Area. Approximately 90% of the total RME present-day risk in this ROD area is related to the short-lived radionuclides cesium-137 and europium-152. Total site RME cancer risk in the 100-F/100-IU-2/100-IU-6 Area at year 2075 is 2×10^{-6}, reflecting the radiological decay of cesium-137 and europium-152. Total RME cancer risk results are all below or within the risk management range of 1×10^{-6} to 1×10^{-4}. Central tendency exposure total cancer risk results for the Casual User scenario are all below the de minimis threshold of 1×10^{-6}.

4.3.1.2 Radiation Dose. Table 4-15 presents the Casual User total site radiation dose with present-day radionuclide concentrations, and with concentrations decayed to years 2075 and 2150. Present-day RME total site radiation dose in the six ROD decision areas was between 0.036 and 0.33 mrem/yr, with the highest value in the 100-F/100-IU-2/100-IU-6 Area. This range of results is well below the 15 mrem/yr threshold.

4.3.1.3 Hazard Index. Table 4-16 presents the Casual User HI results for each ROD decision area. Child RME HI values in the six ROD decision areas were between 0.017 and 0.024, with the highest value in 100-F/100-IU-2/100-IU-6 and the 300 Area. This range of results is well below the threshold HI of 1.0 that indicates a potential for adverse effects.

4.3.2 Avid Angler Results

Tables 4-17 through 4-19 contain the RME and CTE risk assessment results for the Avid Angler exposure scenario in the near-shore environment. Unlike the calculations for soil described in Section 4.3.1, radionuclide concentrations in near-shore aquatic media (surface water, sediment, and fish) were not decayed to year 2075 and 2150. One reason for evaluating only present-day conditions is uncertainty in the influence that radionuclides in groundwater discharging to the near-shore environment may have on sediment concentrations over time. In addition, sediment is relatively mobile in comparison to upland and riparian soil, so COPC concentrations at future dates are less predictable. Because the Avid Angler scenario involves exposures primarily to sediments, surface water, and fish, calculations for cancer risk and radiation dose at years 2075 and 2150 are not performed for this scenario.

Exposure to COPCs in fish is a key aspect of the Avid Angler exposure scenario. The RCBRA, 100-B/C Pilot Project, and 100-NR-2 investigation biota tissue samples for localized species such as sculpin, clams, and crayfish were largely collected in specific areas where groundwater plumes emerge at the Columbia River. The purpose of this sampling was to determine the extent to which Hanford Site-related contaminants are accumulated by biota at these locations. Although these localized species are not plausible sources for chronic human exposure, sculpin in particular were identified in the RCBRA SAP (DOE/RL-2005-42) as an appropriate species for screening human health risks for Hanford Site-related releases. Sampling of game fish that are more representative of food sources for chronic exposure assessment is being conducted within the Hanford Reach of the Columbia River under the RCBRA CRC. In the Avid Angler scenario, screening-level fish ingestion risks for near-shore COPCs detected in sculpin samples are shown in relation to risks from exposure to other near-shore exposure media to indicate the magnitude of potential fish ingestion risks relative to risks for near-shore COPCs in other media. The screening-level fish ingestion risk assessment in Section 4.6 of this report for the exposure scenarios described in the RCBRA will be supplemented under the RCBRA CRC to include calculations based on contaminant concentrations in food fish.

As described in Sections 4.1 and 4.6, appropriate uses for these screening results include identifying risk-relevant COPCs in near-shore aquatic biota and evaluating relative risk among the six ROD decision areas. Therefore, the screening-level risks related to fish ingestion are not summed with the other exposure pathways for the Avid Angler scenario. Risk assessment results for sculpin, a small fish with a limited home range used to evaluate the potential for COPC uptake into fish at each ROD area, are shown next to the results for exposure to sediment, river water, and dust. Screening-level risk assessment results using COPC concentrations in clams and crayfish, and a discussion of potential fish ingestion risks related to contaminants other than Hanford Site COPCs in salmon and other food fish species in the Columbia River, are presented in Section 4.6.

4.3.2.1 Cancer Risk. Table 4-17 presents the Avid Angler total site cancer risks with present-day radionuclide concentrations. Present-day RME total site cancer risks for exposures to sediment, river water, and dust in the six ROD decision areas were between 6×10^{-7} and 3×10^{-6}. The highest RME total cancer risk for exposures to sediment, river water, and dust was in the 100-N Area. Approximately 90% of this total RME risk in the 100-N Area is related to strontium-90 and cesium-137 in sediment. Although radionuclide concentrations in these near-shore media were not decayed to future times, a decrease in these radionuclide concentrations (half-life of approximately 30 years for each) over time is expected. Total RME cancer risk results for exposures to sediment, river water, and dust were 3×10^{-6} or lower, well within the risk management range of 1×10^{-6} to 1×10^{-4}. Central tendency exposure total site cancer risk results for these media are all below the de minimis threshold of 1×10^{-6}.

Carbon-14 is the only carcinogenic COPC in biota in the near-shore environment. As described in Section 4.2.4, COPC identification is based on comparisons of analyte concentrations between site and reference areas. Other carcinogenic chemicals, including arsenic and various persistent organic pollutants, have also been measured in Columbia River fish (EPA 910-R-02-006). Potential risks related to these analytes is discussed in the screening-level assessment of fish ingestion risks in Section 4.6.

Carbon-14 was only measured in sculpin collected in the 100-K and 100-N Areas and only detected in one of two sculpin samples in the 100-K Area. Therefore, with the exception of the 100-K Area, there are no cancer risk and radiation dose results for the screening-level fish ingestion risk assessment based on sculpin data. The Avid Angler fish ingestion risk result related to carbon-14 in sculpin in the 100-K Area is 4×10^{-5}, which is about tenfold higher than the highest total cancer risk for exposure to sediment, surface water, and dust. As discussed in Section 4.6, the presence of carbon-14 in biota in the 100-K Decision Area is based on a single detection in clam tissue and a single detection in sculpin. Carbon-14 was detected in 100-K Area sediments, which may relate to the detections in sculpin and clams, but it was not detected in the four river water samples collected near the river bank in the 100-K Area where carbon-14 is present in groundwater. The absence of carbon-14 in river water at the location where the carbon-14 groundwater plume exists indicates that carbon-14 related to the 100-K Area plume would not be expected to impact fish species with larger home ranges.

4.3.2.2 Radiation Dose. Table 4-18 presents the Avid Angler total site radiation dose with present-day radionuclide concentrations. Present-day RME total site radiation dose for exposures to sediment, river water, and dust in the six ROD decision areas was between 0.017 mrem/yr and 0.11 mrem/yr, with the highest value in the 100-K Area. This range of results is well below the 15 mrem/yr threshold.

As discussed in relation to cancer risk, carbon-14 is the only radionuclide COPC identified in biota in the near-shore environment and it was only detected in the 100-K Decision Area. The screening-level fish ingestion dose for the Avid Angler related to carbon-14 in sculpin in the 100-K Area is 1.6 mrem/yr, which exceeds the highest total radiation dose for exposure to sediment, surface water, and dust but is still well below the threshold of 15 mrem/yr.

4.3.2.3 Hazard Index. Table 4-19 presents the Avid Angler HI results for each ROD decision area. Child RME HI values for exposures to sediment, river water, and dust in the six ROD decision areas were between 0.0055 and 0.015, with the highest value in the 300 Area. This range of results is well below the threshold HI of 1.0 that indicates a potential for adverse effects. The screening-level RME fish ingestion HIs for sculpin ranged between 1.7 and 4.3, with the highest value in the 100-D/100-H Decision Area. The CTE fish ingestion HIs for sculpin ranged between 0.19 and 0.59. As a point of comparison, the RME and CTE Avid Angler HI values calculated for near-shore COPCs in biota using the reference area data were 1.8 and 0.22, as discussed in Section 4.6. In the 100-D/100-H Area, the main contributors to the screening-level sculpin HI result of 4.3 were nickel (60%) and selenium (15%). The main contributors to the HI result of 2.9 in the 100-F/100-IU-2/100-IU-6 Area were copper (30%) and selenium (20%). None of these three metals were identified as COPCs in sediment or surface water, rendering their association with Hanford Site-related releases uncertain.

4.3.3 Avid Hunter Results

Tables 4-20 through 4-22 contain the RME and CTE risk assessment results for the Avid Hunter exposure scenario in the upland environment. As described in Section 4.2.3, representative concentrations for the upland environment are calculated by merging surface soil results across the different ROD decision areas. The Avid Hunter scenario includes exposure to upland soils incurred while hunting plus exposure to COPCs in the meat of hunted game.

Tissue concentrations in game animals are modeled from the RCBRA upland surface soil data. Because the modeled tissue concentrations are a function of soil concentrations, radionuclide concentrations in both soil and game tissue were calculated at years 2075 and 2150 in order to assess the impact of radionuclide decay on future cancer risk and radiation dose. Soil concentrations for these times were calculated using the RESRAD computer code, as described in Appendix D-1.4.1.

4.3.3.1 Cancer Risk. Table 4-20 presents the Avid Hunter total cancer risks with present-day radionuclide concentrations, and with concentrations decayed to years 2075 and 2150. Present-day RME total site cancer risk for exposures to soil and game meat was 5×10^{-5}. Most of the RME cancer risk is related to ingestion of game meat. Arsenic and benzo(a)pyrene (each about 30%), and Aroclor-1254 (about 20%), were the major contributors to the total RME cancer risk from game ingestion. Because radionuclides contribute a relatively small portion of present-day RME total site risk, cancer risk at years 2075 and 2150 is identical to present-day values. The present-day CTE total site cancer risk is 4×10^{-6} and is also related largely to ingestion of arsenic and benzo(a)pyrene from game meat. Total RME and CTE cancer risk results are all within the risk management range of 1×10^{-6} to 1×10^{-4}.

The arsenic RME upland soil concentration of 4.7 mg/kg is about 50% higher than the RME reference area value of 3.2 mg/kg. However, the arsenic value of 4.7 mg/kg is below both the Hanford Area (6.47 mg/kg) and Yakima Basin (5.13 mg/kg) 90th percentile soil background concentrations shown in Table 4-10. In the absence of arsenic, the Avid Hunter RME cancer risk is 3×10^{-5}, due to benzo(a)pyrene and Aroclor-1254 as described above. The contribution of the PAH benzo(a)pyrene to Avid Hunter RME cancer risk from meat ingestion, which is approximately 1×10^{-5}, is likely to be an unrealistic value. Game tissue concentrations of organic chemicals were modeled based on the chemical-specific tendency to partition into fat tissue. However, benzo(a)pyrene is subject to enzymatic degradation in mammals and, therefore, will not accumulate in the body of game animals to any significant extent.

4.3.3.2 Radiation Dose. Table 4-21 presents the Avid Hunter total radiation dose with present-day radionuclide concentrations, and with concentrations decayed to years 2075 and 2150. Present-day RME and CTE total site radiation dose values were 0.18 mrem/yr and 0.071 mrem/yr, respectively. These values are well below the 15 mrem/yr threshold.

The results of the game ingestion dose calculations for radionuclides are consistent with those calculated in "Radiological Risk from Consuming Fish and Wildlife to Native Americans on the Hanford Site (USA)" (Delistraty et al. 2009). In this paper, total radiological dose from consuming game mammals in a Native American scenario was calculated to be 0.11 mrem/yr, which is roughly the same as the value of 0.08 mrem/yr shown in Table 4-21 for the Avid Hunter.

4.3.3.3 Hazard Index. Table 4-22 presents the Avid Hunter HI results for the River Corridor upland environment. Present-day RME and CTE HIs for exposures to soil and game meat were 1.1 and 0.26, respectively. Effectively, all of the HI results are related to ingestion of game meat. About 50% of the RME HI of 1.1 and 20% of the CTE HI of 0.26 is related to Aroclor-1254, which was detected in soil at six of the 20 upland sampling locations. The RME HI value is near the threshold of 1.0 that indicates a potential for adverse effects.

Ideally, the modeled COPC concentrations in game meat would be compared to analogous muscle tissue (meat) data collected under the SESP. However, the available tissue data for upland COPCs are limited to liver tissue data for certain metals in rabbit and deer. Because some metals are known to accumulate in liver, these data are of limited utility for comparing to modeled results in meat. Although the liver may be eaten by some hunters, the amount of liver eaten over time is likely to be much lower than meat ingestion, and it is a meat ingestion rate that is used to calculate risks for the Avid Hunter.

The available tissue data are compared to modeled exposure point concentrations in game meat in Table 4-23. There are six metals shown in Table 4-23 for which measured concentrations are available in both deer and rabbit liver. A review of these metal concentrations reveal that there are frequently significant differences between modeled concentrations in game meat, measured concentrations in deer liver, and measured concentrations in rabbit liver. With the exception of lead, concentrations of metals in deer and rabbit liver were generally within a factor of 5 of each other. However, chromium was the only metal where model concentrations were similarly close to both measured values. With the exceptions of lead and mercury, modeled concentrations in game meat were systematically lower than measured concentrations in deer and rabbit liver.

4.3.4 Risks Related to Lead in Soil

Lead was identified as a COPC in soil in the upland and riparian environments and in sediment in the near-shore environment. Risks related to lead are screened by comparison of lead concentrations in these media to published soil screening criteria in order to determine the need for additional evaluation. The EPA publishes a recommended residential screening level for lead in soil of 400 mg/kg, derived using the Integrated Exposure Uptake Biokinetic model that has been associated with bare soil in a play area (40 CFR 745). An additional screening level of 1,200 mg/kg has been defined for other bare-earth portions of a residential yard (40 CFR 745). More detailed information on lead toxicity and the basis of EPA's soil screening criteria is provided in Section 3.5.8.

The CTE and RME representative concentrations for lead in upland soil are 27 and 45 mg/kg, respectively. Among the six ROD decision areas where individual riparian representative concentrations are calculated, the highest values are in the 100-F/100-IU-2/100-IU-6 Area. These CTE and RME representative concentrations for lead in riparian soil in this ROD area are 51 and 92 mg/kg, respectively. As shown in Table 3-9, reference area soil lead concentrations are much higher in riparian soil than in upland soil. Hence, the difference between site and reference area soil lead concentrations is much more pronounced in the upland environment. Lead concentrations in sediment among the six ROD decision areas are, like riparian soil, highest in the 100-F/100-IU-2/100-IU-6 Area. The CTE and RME representative concentrations for lead in sediment are 27 and 59 mg/kg, respectively. All of these soil and sediment representative concentrations are well below the residential screening criteria of 400 and 1,200 mg/kg.

Although the CTE and RME representative concentrations for lead in upland soil are below residential screening criteria, the RME soil concentration in 1 of the 20 sampled remediated waste sites did exceed the lower of the criteria. The CTE and RME representative concentrations for lead in upland soil at the 600-131 site were 116 mg/kg and 511 mg/kg, respectively. This RME concentration is well in excess of the next-highest RME lead value of 159 mg/kg at the 600-139 site. In *Arsenic, Lead, and Other Trace Elements in Soils Contaminated with Pesticide Residues at the Hanford Site (USA)*, Yokel and Delistraty (2003) have shown that the average lead concentration in surface soil from 31 samples associated with former orchards at the Hanford Site is approximately 220 mg/kg, with a standard deviation of 460 mg/kg and a maximum value of 1,900 mg/kg. These levels are related to the use of lead arsenate pesticides at these locations (Yokel and Delistraty 2003). It is, therefore, reasonable to assume that there may be a number of discrete locations within the River Corridor where lead concentrations exceed one or both of the residential screening criteria, even if average soil concentrations representative of remediated waste sites are below these levels.

4.4 NONRESIDENT TRIBAL SCENARIO RESULTS

Risk assessment results for the Nonresident Tribal scenario reflect exposures that are anticipated to occur over a larger area than that associated with any particular remediated waste site. The Nonresident Tribal scenario evaluates potential risks related to traditional tribal activities, with a particular focus on fishing, hunting, and gathering plants. For this scenario, it is assumed that 50% of wild plants are gathered in the upland environment and 50% in the riparian environment. Hunting activities are assumed to occur within upland regions of the River Corridor, while fishing-related exposures occur in the riparian and near-shore regions. While engaged in fishing, daily use of a sweat lodge using Columbia River water is assumed. Children (beginning at age 2), youths, and adults may be exposed to COPCs while onsite as well as by ingestion of foods obtained from the site and the Columbia River. This scenario is more fully described in Section 3.3, and a summary of exposure media, data sets, and potentially complete exposure pathways is provided in Table 3-9. COPCs in the upland, riparian, and near-shore environments are listed in Tables 4-7, 4-8, and 4-9, respectively.

As described in Section 3.4.4, chemical hazard is calculated separately for adult and child receptors. For the Nonresident Tribal scenario, child HIs generally exceed those for adults. Because child HIs are either larger than or equal to the HI values for adults, only child HI values are presented in the following subsections.

Tables 4-24 through 4-26 contain the risk assessment results for the Nonresident Tribal exposure scenario. Risk assessment results for the Nonresident Tribal exposure scenario were also calculated with radionuclide upland and riparian soil concentrations decayed to the years 2075 and 2150. Soil concentrations for these times were calculated using the RESRAD computer code, as described in Appendix D-1.4.1. These results show the effect of radionuclide decay over time on cancer risk and radiation dose, and also include the loss of these radionuclides from soil due to leaching with rainfall. As described in Section 4.2.2, radionuclide concentrations in near-shore media, which are also incorporated in the Nonresident Tribal calculations, were not decayed due to uncertainties in the contribution of radionuclides in groundwater over time and the relative mobility of these media in comparison to upland and riparian soil. The present-day radionuclide concentrations in sediment and surface water were used to calculate the cancer risk and dose results across exposures in upland, riparian, and near-shore environments at years 2075 and 2150.

As in the Avid Angler exposure scenario discussed in Section 4.3.2, exposure to contaminants in fish is also a key aspect of the Nonresident Tribal scenario. Screening-level fish ingestion risks for near-shore COPCs in sculpin are shown in relation to risks for exposure to other exposure media to indicate the relative magnitude of potential fish ingestion risks in the Nonresident Tribal scenario. Sculpin were identified in the sampling and analysis plan as an appropriate species for screening human health risks for Hanford Site-related releases (DOE/RL-2005-42). Additional sampling of game fish that are more representative of food sources for chronic exposure assessment is being conducted within the Hanford Reach of the Columbia River under the RCBRA CRC. The screening-level fish ingestion risk assessment in Section 4.6 of this report will be supplemented under the RCBRA CRC based on contaminant concentrations in food fish.

The screening-level risks related to fish ingestion are not summed with the other exposure pathways for the Nonresident Tribal scenario risk results because their use is limited to identifying risk-relevant COPCs and for evaluating relative risk among the six ROD decision areas. Risk assessment results for sculpin, a small fish with a limited home range used to evaluate the potential for COPC uptake into fish at each ROD area, are shown next to the results for the other exposure media. Screening-level risk assessment results using COPC concentrations in clams and crayfish, and a discussion of potential fish ingestion risks related to salmon and other food fish species in the Columbia River, are presented in Section 4.6.

4.4.1.1 Cancer Risk. Table 4-24 presents the Nonresident Tribal total site cancer risks with present-day radionuclide concentrations, and with concentrations decayed to years 2075 and 2150. Present-day and future total site cancer risks for exposures to soil, sediment, river water, plants, and meat in the six ROD decision areas was 1×10^{-2} in all ROD areas except 100-K, where it was 2×10^{-2}.

A summary of the key exposure pathways and COPCs contributing 10% or more to present-day total site cancer risk above 1×10^{-4} for the Nonresident Tribal scenario in each ROD area is provided below.

ROD Area	COPC	Pathway	Fraction of Total Risk	Present-Day Total Risk
100-B/C	Arsenic	nativePlantIngestion	0.93	1×10^{-2}
100-K	Arsenic	nativePlantIngestion	0.72	2×10^{-2}
	Carbon-14	nativePlantIngestion	0.25	
100-N	Arsenic	nativePlantIngestion	0.94	1×10^{-2}
100-D/100-H	Arsenic	nativePlantIngestion	0.97	1×10^{-2}
100-F/100-IU-2/100-IU-6	Arsenic	nativePlantIngestion	0.94	1×10^{-2}
300	Arsenic	nativePlantIngestion	0.97	1×10^{-2}

With the exception of the 100-K Area, arsenic by plant ingestion contributes 90% or more to total cancer risk. In the 100-K Area, where the present-day cancer risk was 2×10^{-2}, arsenic contributed 72% and carbon-14 contributed 25% to total cancer risk, again due to plant ingestion. Total cancer risk in the 100-K Area decreases to 1×10^{-2} by year 2075. This decrease relates to carbon-14 and occurs because the ratios of present-day to future soil concentration were calculated with RESRAD, and the RESRAD carbon-14 pathway model accounts for loss by mineralization to carbon dioxide and gas-phase migration over time.

As shown in Table 4-11, Nonresident Tribal arsenic cancer risk calculated for the reference areas is also 1×10^{-2}. Only for present-day cancer risk in the 100-K Area, where carbon-14 is also a significant contributor to total cancer risk, does cancer risk in any ROD area exceed the arsenic reference areas value. The difference between total present-day cancer risk and reference area arsenic cancer risk is largely related to relatively small differences between arsenic soil concentrations from which plant tissue concentrations are modeled. The 95% UCL representative concentration of arsenic in upland soil in the River Corridor is 4.7 mg/kg and in the upland reference area is 3.2 mg/kg. Riparian soil 95% UCL arsenic concentrations in the six ROD decision areas (ranging from 7.1 mg/kg to 9.6 mg/kg) and the riparian reference area (7.4 mg/kg) show slightly less of a difference. The arsenic upland soil 95% UCL representative concentration of 4.7 mg/kg is below both the Hanford Area (6.47 mg/kg) and Yakima Basin (5.13 mg/kg) 90th percentile soil background concentrations shown in Table 4-10.

Because arsenic concentrations in upland and riparian site soils are not very different than reference area values and commonly applied background levels, risks for COPCs other than arsenic may also be of interest for risk management decisions. A summary of the key exposure pathways and COPCs, other than arsenic, contributing 10% or more to present-day Nonresident Tribal cancer risk in the six ROD areas is provided below.

ROD Area	COPC Other than Arsenic	Pathway	Fraction of Total Risk	Present-Day Total Risk (Without Arsenic)
100-B/C	Technetium-99	nativePlantIngestion	0.60	9×10^{-4}
100-K	Carbon-14	nativePlantIngestion	0.93	4×10^{-3}
100-N	Strontium-90	nativePlantIngestion	0.37	7×10^{-4}
	Strontium-90	surfaceWaterIngestion	0.24	
100-D/100-H	Strontium-90	nativePlantIngestion	0.22	3×10^{-4}
	Benzo[a]pyrene	gameIngestion	0.19	
	Aroclor-1254	gameIngestion	0.12	
100-F/100-IU-2/100-IU-6	Technetium-99	nativePlantIngestion	0.46	7×10^{-4}
300	Benzo[a]pyrene	gameIngestion	0.21	3×10^{-4}
	Aroclor-1254	gameIngestion	0.13	
	Strontium-90	nativePlantIngestion	0.10	

In the absence of arsenic, cancer risks are related to several different COPCs and exposure pathways. In the 100-K Area, as described above, calculated cancer risks over time will decrease as carbon-14 mineralizes and escapes as carbon dioxide. At year 2075, total cancer risk in the 100-K Area without arsenic is 2×10^{-4} and is mostly related to exposure to benzo(a)pyrene and Aroclor-1254 from ingestion of game and plants. Carbon-14 was identified as a COPC in riparian soil, but not upland soil, so the calculated risk of 4×10^{-3} is based on the 50% of the total amount of game and plants that are taken in the riparian environment. In the 100-N Area, decay of strontium-90 (half-life of approximately 29 years) results in a total cancer risk of 5×10^{-4} in 2075. Radioactive decay and, in the case of the soluble radionuclide technetium-99, leaching from soil also contribute to the lower 2075 cancer risks shown in Table 4-24 at the 100-B/C, 100-D/100-H, 100-F/100-IU-2/100-IU-6, and 300 Decision Areas.

Total present-day cancer risk, including arsenic, for exposure pathways involving direct soil exposure and exposures to COPCs in sediment and river water ranges between 6×10^{-5} and 2×10^{-4}. Total present-day cancer risks for game ingestion, which employs the upland surface soil data to model concentrations of COPCs in game meat, was 2×10^{-4}. Because the Nonresident Tribal scenario includes exposures to environmental media in the upland, riparian, and near-shore environments for many different exposures routes, the risk for any analyte/pathway combination is a complex function of the relative COPC concentrations and exposure intensity for each of the environmental media and environments.

Carbon-14 is the only carcinogenic COPC identified in biota in the near-shore environment. Carbon-14 was only measured in sculpin collected in the 100-K and 100-N Areas and was only detected in the 100-K Area. Therefore, with the exception of the 100-K Area, there are no cancer risk and radiation dose results for the screening-level fish ingestion risk assessment based on sculpin data. The Nonresident Tribal fish ingestion risk result related to carbon-14 in sculpin in the 100-K Area is 3×10^{-4}. This value is well below the site and reference area risk estimates related to arsenic due to plant ingestion, but is somewhat higher than the range of results for total cancer risks for exposure to soil, sediment, and surface water in the six ROD areas. As discussed in Section 4.6, the identification of carbon-14 as a COPC in near-shore biota in the 100-K Decision Area, as well as the representative concentrations in these tissues, is based on a single detection in clam tissue and a single detection in sculpin.

Carbon-14 was detected in 100-K Area sediments, which may relate to the detections in sculpin and clams, but it was not detected in the four river water samples collected near the river bank in the 100-K Area where carbon-14 is present in groundwater. The absence of carbon-14 in river water at the location where the carbon-14 groundwater plume exists indicates that carbon-14 related to the 100-K Area plume would not be expected to impact fish species with larger home ranges.

4.4.1.2 Radiation Dose. Table 4-25 presents the Nonresident Tribal total site radiation dose with present-day radionuclide concentrations, and with concentrations decayed to years 2075 and 2150. Present-day total site radiation dose values for exposures to soil, sediment, river water, plants, and meat in five of the six ROD decision areas were between 2.3 mrem/yr and

11 mrem/yr, with a much higher value in the 100-K Area (62 mrem/yr). With the exception of 100-K, this range of results is below the 15 mrem/yr threshold.

A summary of the key exposure pathways and COPCs contributing 10% or more to Nonresident Tribal present-day total radiation dose above 15 mrem/year in the 100-K ROD Area is provided below.

ROD Area	COPC	Pathway	Fraction of Total Dose	Present-Day Total Dose (mrem/yr)
100-K	Carbon-14	nativePlantIngestion	0.96	62

Effectively, 100% of the present-day radiation dose of 62 mrem/yr at the 100-K Area is related to carbon-14 and plant ingestion. This dose is based on a modeled carbon-14 plant concentration of 150 pCi/g in the riparian environment. As discussed in relation to cancer risk, carbon-14 is expected to be lost relatively quickly from soil due to conversion into carbon dioxide. Nonresident Tribal radiation dose in the 100-K Area was modeled using the RESRAD computer code to be below 1 mrem/yr by year 2075 due to conversion to carbon dioxide.

As discussed in relation to cancer risk, carbon-14 is the only radionuclide COPC in biota in the near-shore environment and it was only detected in the 100-K Decision Area. The total fish ingestion dose result for the Nonresident Tribal scenario related to carbon-14 in sculpin in the 100-K Area is 3.9 mrem/yr, which is below the threshold of 15 mrem/yr.

4.4.1.3 Hazard Index. Table 4-26 presents the Nonresident Tribal HI results for each of the six ROD decision areas. Child HIs for exposures to soil, sediment, river water, plants, and meat in the six ROD decision areas were between 73 and 95.

A summary of the key exposure pathways and COPCs contributing 10% or more to an HI value above 1.0 in a ROD area is provided below.

ROD Area	COPC	Pathway	Fraction of Child Hazard	Child Hazard
100-B/C	Arsenic	nativePlantIngestion	0.68	79
	Cadmium	nativePlantIngestion	0.11	
100-K	Arsenic	nativePlantIngestion	0.61	84
	Cadmium	nativePlantIngestion	0.14	
100-N	Arsenic	nativePlantIngestion	0.67	73
100-D/100-H	Arsenic	nativePlantIngestion	0.66	81
	Cadmium	nativePlantIngestion	0.10	
100-F/100-IU-2/100-IU-6	Arsenic	nativePlantIngestion	0.59	95
	Cadmium	nativePlantIngestion	0.16	
300	Arsenic	nativePlantIngestion	0.69	84

Arsenic by plant ingestion was responsible for 60% to 70% of the child HI at all ROD decision areas, with cadmium also contributing 10 to 15% by plant ingestion in some areas. The critical effects related to the arsenic noncancer toxicity value are in the integumentary (skin) and vascular systems, while those for cadmium are in the renal system (IRIS 2009). Because they have critical effects in different organs, the HQs for these two metals are not likely to be additive, and the HI values that combine these HQs may be biased high by approximately 10%.

Table 4-13 shows that Nonresident Tribal child HQ values for 95 UCL reference area concentrations of arsenic and cadmium are approximately 44 and 7.4, respectively. As discussed with respect to Nonresident Tribal cancer risk, the arsenic 95 UCL representative concentration in upland surface soil (4.7 mg/kg) is about 50% higher than it is in the upland reference area (3.2 mg/kg). Riparian soil 95 UCL arsenic concentrations in the six ROD decision areas (ranging from 7.1 mg/kg to 9.6 mg/kg) and the riparian reference area (7.4 mg/kg) are more similar. As shown in Table 4-10, these site values are comparable to both the Hanford Area (6.47 mg/kg) and Yakima Basin (5.13 mg/kg) 90^{th} percentile soil background concentrations.

Because arsenic concentrations in upland and riparian site soils are not very different than reference area values and commonly applied background levels, child hazard for COPCs other than arsenic may also be of interest for risk management decisions. A summary of the key exposure pathways and COPCs, other than arsenic, contributing 10% or more to present-day Nonresident Tribal child HI in the six ROD areas, is provided below.

ROD Area	COPC Other than Arsenic	Pathway	Fraction of Child HI	Present-Day Total Child HI (Without Arsenic)
100-B/C	Cadmium	nativePlantIngestion	0.34	25
	Zinc	nativePlantIngestion	0.20	
100-K	Cadmium	nativePlantIngestion	0.37	32
	Zinc	nativePlantIngestion	0.25	
100-N	Cadmium	nativePlantIngestion	0.28	24
	Zinc	nativePlantIngestion	0.23	
	Copper	nativePlantIngestion	0.10	
100-D/100-H	Cadmium	nativePlantIngestion	0.31	27
	Zinc	nativePlantIngestion	0.25	
100-F/100-IU-2/100-IU-6	Cadmium	nativePlantIngestion	0.39	38
	Zinc	nativePlantIngestion	0.23	
300	Cadmium	nativePlantIngestion	0.29	25
	Zinc	nativePlantIngestion	0.20	
	Mercury	nativePlantIngestion	0.10	

In the absence of arsenic, about 60% of child HI in each of the six ROD areas is related to cadmium and zinc by the plant ingestion exposure pathway. As shown in Table 4-13, Nonresident Tribal child HQ values for 95% UCL reference area concentrations of cadmium and zinc are 7.4 and 6.5, respectively. These reference area nonresident Tribal HQ calculations for arsenic (44), cadmium (7.4), and zinc (6.5) indicate that even soil concentrations in unaffected areas are associated with high HQ values. These high HQ values for naturally occurring levels of metals in soil are a function of the high plant ingestion rates for this scenario, coupled with protective models for estimating the uptake of metals from soil by plants.

The screening-level fish ingestion child HI for sculpin ranged between 10 and 25, with the highest values in the 100-D/100-H ROD decision area (25) and 100-F/100-IU-2/100-IU-6 ROD decision area (17). As a point of comparison, the child HI values for the COPC concentrations in reference area sculpin was 11. In the 100-D/100-H Area, the main contributors to the screening-level sculpin HI result of 4.3 were nickel (60%) and selenium (15%). The main contributors to the HI result of 17 in the 100-F/100-IU-2/100-IU-6 Area were copper (30%) and selenium (20%).

4.5 RISKS RELATED TO INGESTION OF WATER FROM SHORELINE SPRINGS (SEEPS)

As described in Section 4.2.1, water data from seeps along the Columbia River have been obtained as part of the 100-B/C Pilot and 100-NR-2 Projects, as well as under the SESP. These seeps represent surface discharge of the unconfined aquifer that underlies the Hanford Site. As discussed in Section 10.5 of PNNL-18427, the presence and flow rate of water at the seeps, as well as concentrations of groundwater COPCs in the seep water, are governed by fluctuations in the level of river water and, therefore, may vary widely over time. River water moves into the aquifer at high water levels and returns as spring water when the river level decreases. The height of the river is dependent upon upstream flow conditions and operation of upstream dams and in the 300 Area may also be influenced by the operation of the downstream McNary Dam.

The objective of this analysis is to provide a screening-level assessment of potential risks related to the use of water from shoreline springs (seeps). It is infeasible to calculate scenario-specific chronic health risks for the Avid Angler and Nonresident Tribal scenarios using contaminant concentrations for one or more seeps because most seeps are accessible only intermittently during conditions of low river flow (PNL-5289, *Investigation of Ground-Water Seepage from the Hanford Shoreline of the Columbia River*; DOE/RL-2008-46, *Integrated 100 Area Remedial Investigation/Feasibility Study Work Plan*). These scenarios presume exposure will occur for between 30 and 60 days/yr for many years, as long as a full lifetime in the case of the Nonresident Tribal scenario (see Section 3.3). Because the presence of seeps is intermittent, chronic exposure over time is unlikely to be realized.

Risks due to short-term exposures to seep water are evaluated in this screening-level assessment by comparing seep water concentrations to EPA 1-day and 10-day drinking water health advisories (EPA 822-R-09-011, *2009 Edition of the Drinking Water Standards and Health Advisories*). These comparisons are considered to be the most relevant for potential exposures involving Recreational and Nonresident Tribal uses of the River Corridor. Both the 1-day and

10-day health advisories are based solely on noncarcinogenic effects, since chemical exposure for such a limited time at levels that do not result in acute effects will have a negligible contribution to lifetime cancer risk. Short-term health advisories are not published for radionuclides including uranium.

Risks due to potential longer term exposures to seep water are evaluated by comparing seep water concentrations to Washington State and EPA maximum contaminant levels (MCLs) for drinking water. Washington State Department of Health (DOH) drinking water standards are published in *Washington Administrative Code* (WAC) 246-290, "Public Water Supplies." The EPA MCLs are available in EPA 822-R-09-011 and online at http://www.epa.gov/safewater/contaminants/index.html. With the exception of uranium, MCLs are not published for individual radionuclides. However, radionuclide-specific MCLs may be calculated based on the 4 mrem/yr standard for beta particle and gamma radioactivity published by DOH and EPA (EPA 822-R-09-011).

The beta-gamma MCL was calculated using dosimetric methods published in the National Bureau of Standards Handbook *Maximum Permissible Body Burdens and Maximum Permissible Concentrations of Radionuclides in Air and in Water for Occupational Exposure* (NBS 1963). To simplify the evaluation of beta-gamma dose with respect to the MCL, EPA has also published water concentration values equivalent to 4 mrem/yr for each beta-gamma emitter. Activity concentrations of beta- and gamma-emitting radionuclides associated with a dose of 4 mrem/yr (40 µSv/yr) are available in Appendix I of the *Implementation Guidance for Radionuclides* (EPA 816-F-00-002) and are used in this screening assessment. These values have also been applied in the *Hanford Site Environmental Reports* and values for selected radionuclides commensurate with the 4 mrem/yr MCL cited in PNNL-18427, Table D.4.

Key groundwater plume contaminants in the seven groundwater operable units (OUs) underlying the River Corridor are described in PNNL-16346 and are employed in Section 6.0 of the HHRA to support identification of groundwater COPCs. A subset of these contaminants, including chromium, nitrate, tritium, strontium-90, technetium-99, and uranium, are discussed in PNNL-18427 as key contaminants observed in seep water. A crosswalk of these six key seep contaminants with the ROD decision areas where they have been measured at levels above drinking water standards include the following:

- Dissolved chromium in the 100-D/100-H, 100-B/C, 100-K, 100-N, and 100-F/100-IU-2/100-IU-6 Areas
- Nitrate in the 100-D/100-H, 100-B/C, 100-K, 100-F/100-IU-2/100-IU-6, and 300 Areas
- Tritium in the 100-D/100-H, 100-B/C, 100-K, 100-N, 100-F/100-IU-2/100-IU-6, and 300 Areas
- Strontium-90 in the 100-D/100-H, 100-B/C, and 100-K Areas
- Uranium in the 100-D/100-H, 100-F/100-IU-2/100-IU-6, and 300 Areas.

Strontium-90 was historically measured in water flowing from certain springs in the 100-N Area, but flow from these springs has not been observed since 1997 due to lowering of the water table that followed the cessation of water discharges from the 105-N Reactor when the reactor was shut down (PNNL-18427).

4.5.1 Analysis of Seep Data

The seep data for the six key contaminants described in the bullets above were used to generate box plots, which show the range of concentrations for each analyte by seep relative to short-term and/or long-term drinking water criteria. These box plots are shown in Figures 4-2 through 4-7. The box plot for each contaminant consists of side-by-side plots for every seep where the contaminant was sampled. The individual plots display the minimum (25th percentile), median (75th percentile), and maximum concentrations for the seep listed beneath the plot. The ratio of detects (n′) to the number of samples (n) is provided above each seep plot. Also provided above each plot is the range of years for which seep samples were collected. Finally, threshold lines are provided on the plots that correspond to the drinking water criteria.

Note that only the 100 µg/L (0.1 mg/L) long-term criterion line is plotted for dissolved chromium because all values lie below the 1,000 µg/L criteria. A threshold line does not appear on the technetium-99 plot because all values are far below the criterion. The screening criteria are shown in Table 4-27.

The seep water data contain many instances where two or more samples were recorded on the same day for a particular contaminant at a seep. These instances were considered to be duplicate samples, and only the maximum value is plotted. Seep water samples were analyzed for either total chromium or hexavalent chromium. Results for both of these analytes were used to create the dissolved chromium box plot. The seep data for nitrate are based on analyses of nitrate and of nitrogen in nitrate. Data reported as nitrate were converted to an equivalent “nitrogen in nitrate” value by multiplying the result by (14 g N/mol per 62 g NO_3/mol). Finally, uranium in seep water was primarily measured as isotopic uranium (uranium-233/234, uranium-235, and uranium-238) with units of pCi/L. In order to compare these data to uranium drinking water criteria with units of µg/L, the isotopic data were converted to a mass basis and the mass of the three isotopes was summed.

Samples have been collected in the seep over multiple years. The box plots in Figures 4-2 through 4-7 were used to focus the analyte/seep combinations for evaluation of temporal trends in contaminant concentrations. Chromium in the 100-D spring; nitrate in the 100-D, 100-F, and 100-H springs; strontium-90 in the 100-H springs; tritium in Spring 28-2; and uranium in Spring 42-2 all have results above the respective screening levels. Time series plots for these combination of contaminants and locations were generated to review the variation of concentrations over time. If more than one result was reported for a particular day, the maximum concentration was plotted on the time trend figures.

Figure 4-2. Box Plots Displaying Chromium Concentrations in Seep Water Collected Along the Hanford Shoreline.

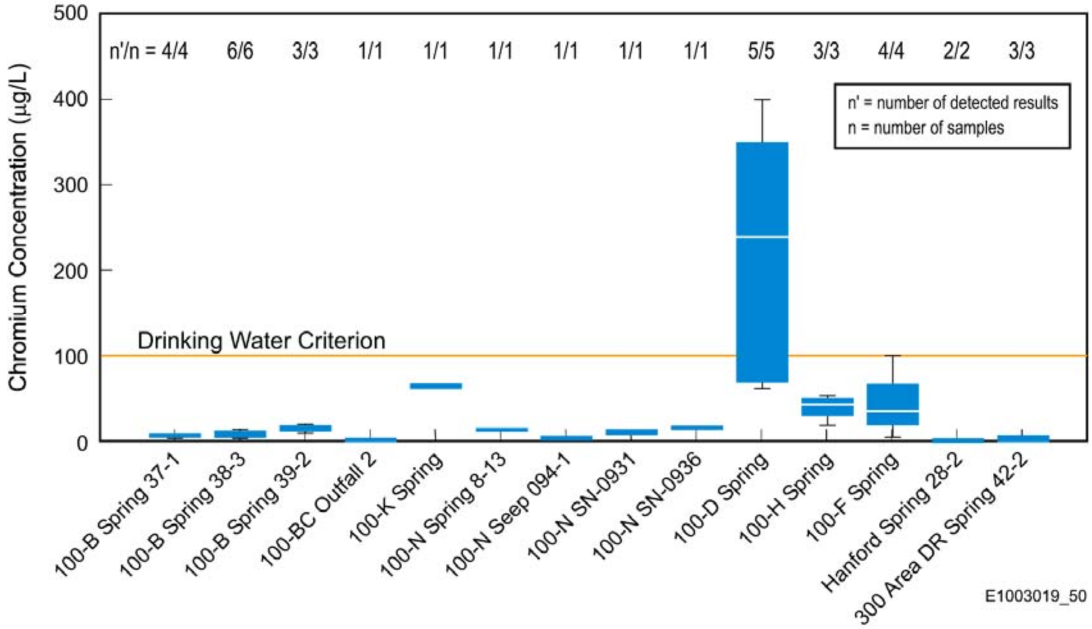


Figure 4-3. Box Plots Displaying Nitrate Concentrations in Seep Water Collected Along the Hanford Shoreline.

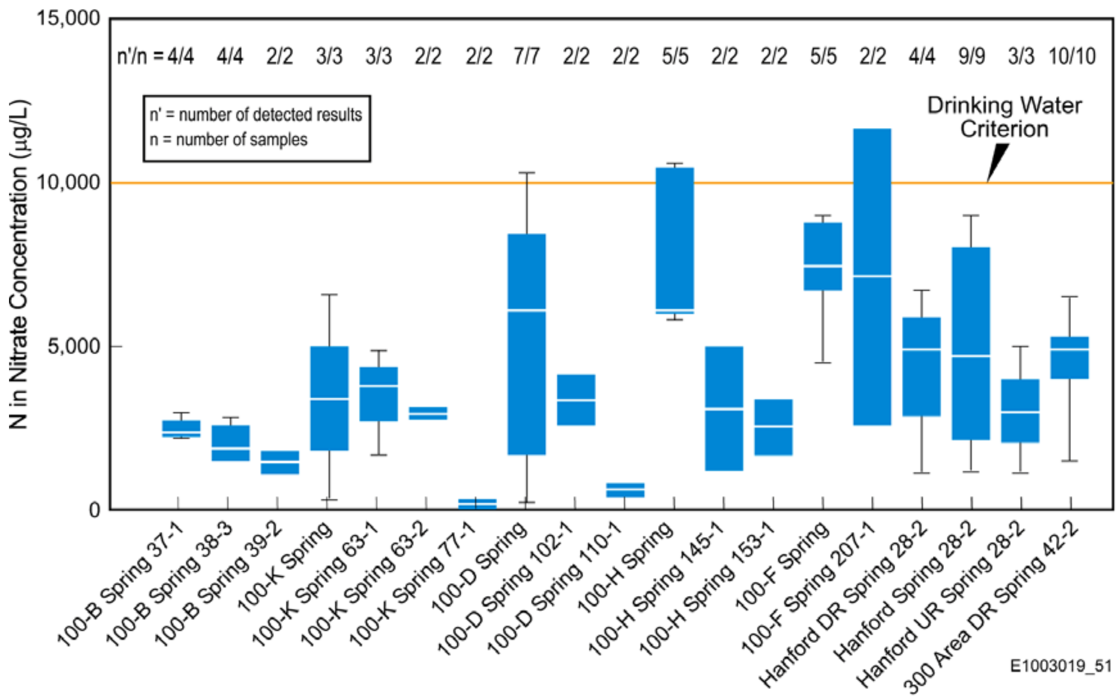


Figure 4-4. Box Plots Displaying Strontium-90 Concentrations in Seep Water Collected Along the Hanford Shoreline.

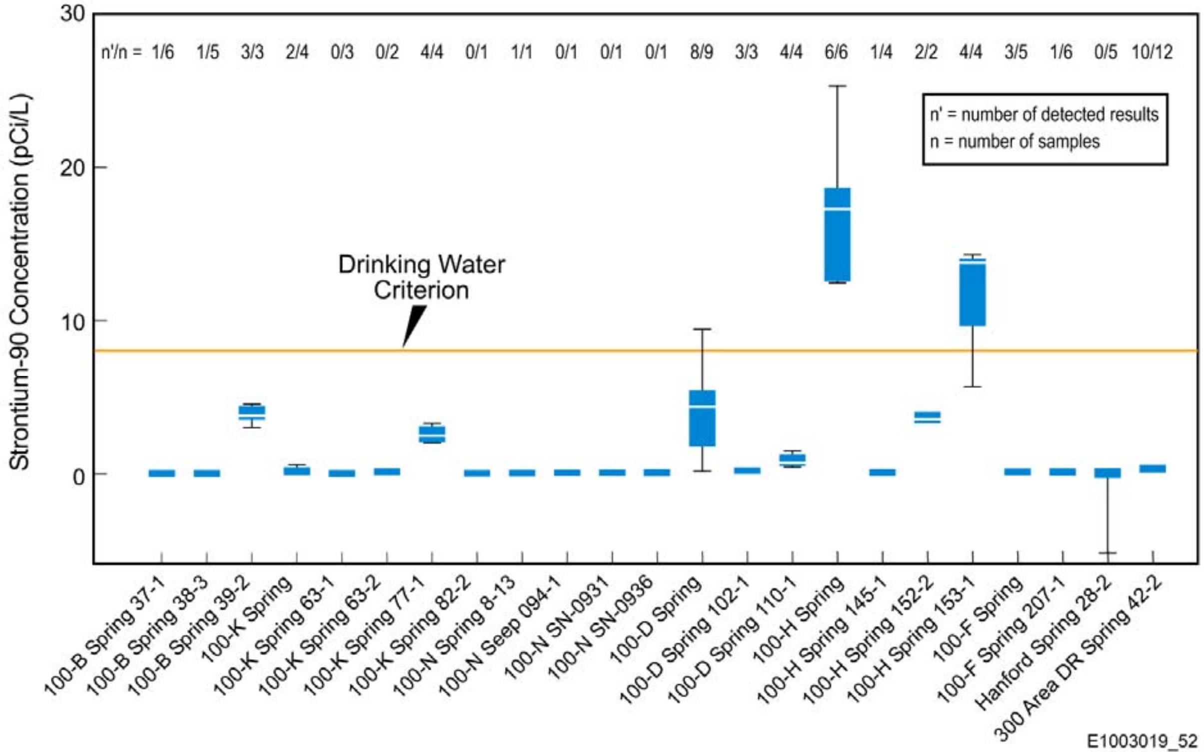


Figure 4-5. Box Plots Displaying Technetium-99 Concentrations in Seep Water Collected Along the Hanford Shoreline.

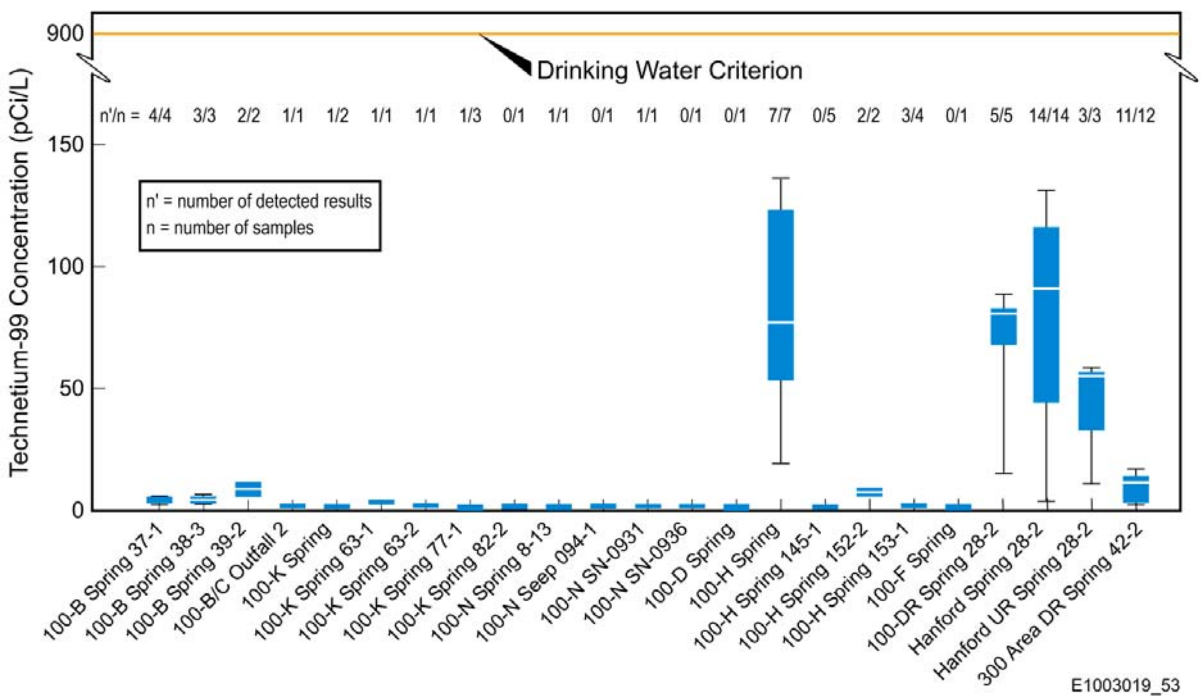


Figure 4-6. Box Plots Displaying Tritium Concentrations in Seep Water Collected Along the Hanford Shoreline.

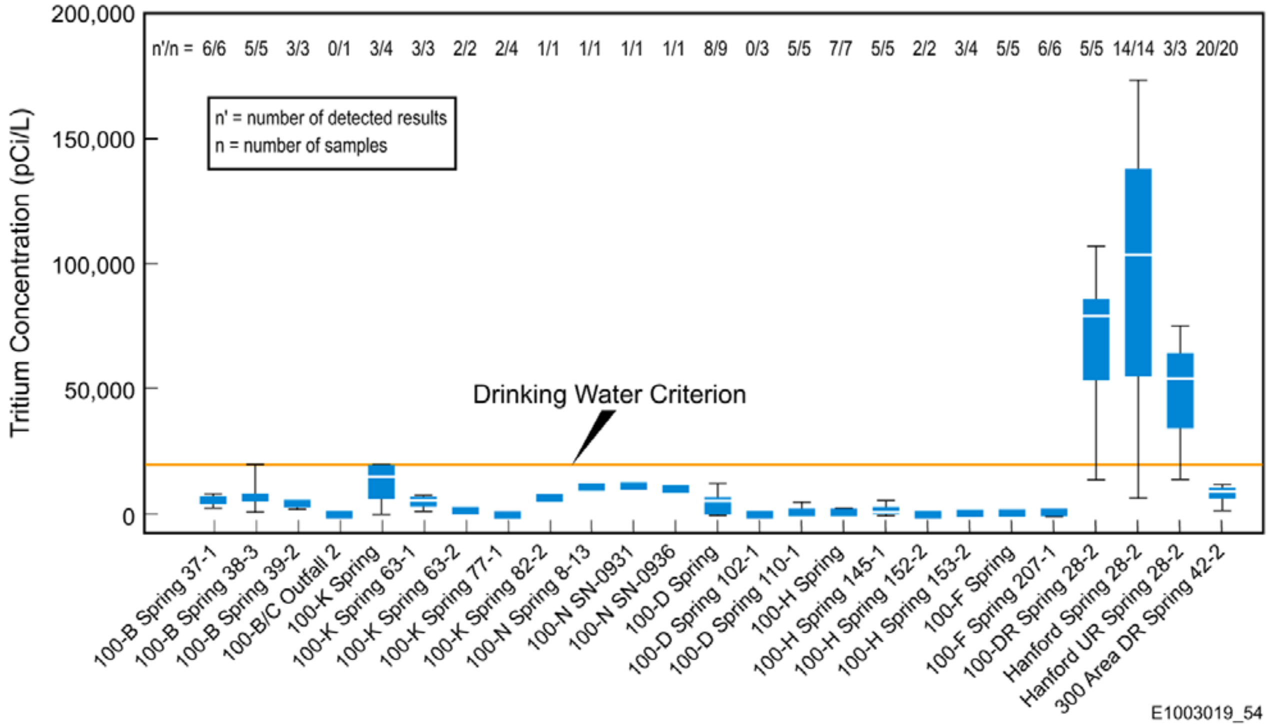


Figure 4-7. Box Plots Displaying Total Uranium Concentrations in Seep Water Collected Along the Hanford Shoreline.

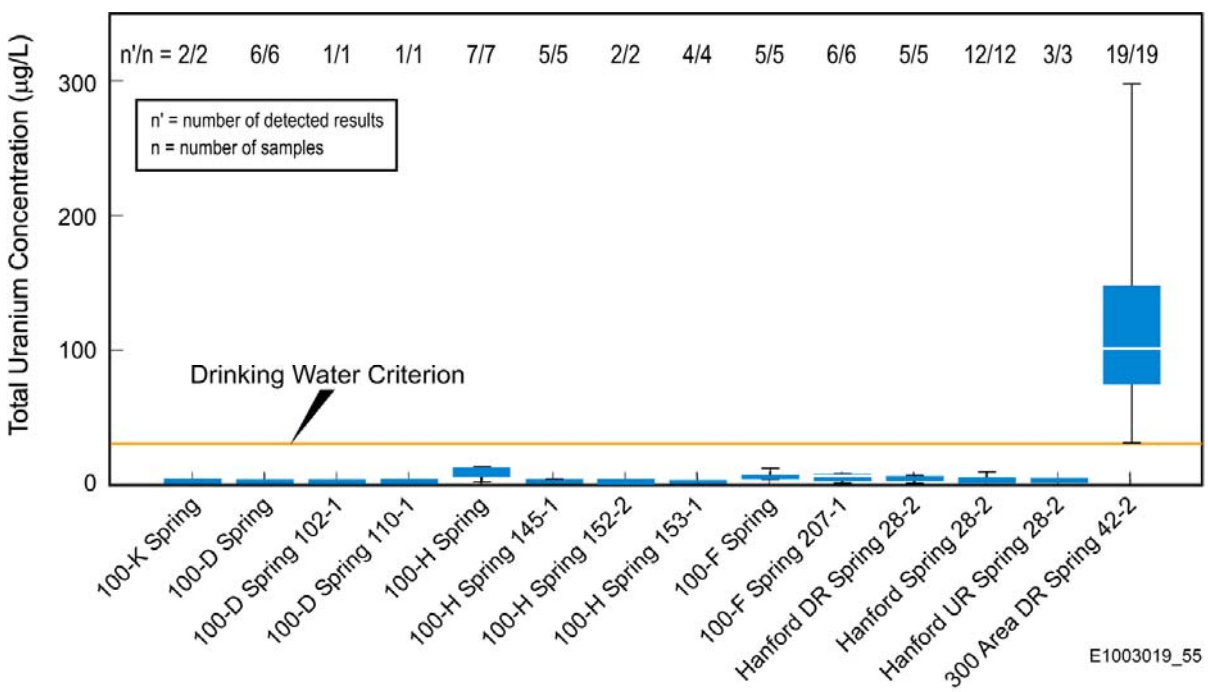


Figure 4-8 shows the concentrations of chromium and hexavalent chromium that were collected between 1994 and 1996 in the 100-D spring. There is a decreasing trend shown in these data. The last values of chromium and hexavalent chromium (approximately 60 µg/L to 70 µg/L) are below the chronic drinking water MCL of 100 µg/L.

Figure 4-9 shows the concentrations of nitrogen in nitrate from the 100-F spring (1994-1996) and 100-F spring 207-1 (1999 and 2003). The concentration of nitrate shows a steadily increasing trend from 1994 to 1999. When the 100-F spring was sampled again in 2004, the concentration of 2,600 µg/L was lower than it had been at any previous sampling date and well below the threshold of 10,000 µg/L. Although not shown in time series plots, there were two measurements of nitrate in 100-H Area springs at approximately 10,500 µg/L, slightly above the threshold of 10,000 µg/L. These measurements were made in 1995 and 1998. Two subsequent samples in 1999 and 2003 were at or below 5,000 µg/L.

Figure 4-10 shows strontium-90 concentrations in the 100-H spring over a period of 11 years. Samples collected between 1993 and 1997 are associated with 100-H spring, whereas samples from between 1999 and 2003 are associated with 100-H Spring 145-1, 152-2, and 153-1. There is a general slight decreasing trend observable in these data. Strontium-90 concentrations range between approximately 10 to 25 pCi/L between 1993 and 1999, and between zero and 15 pCi/L between 2000 and 2004.

Figure 4-8. Concentration of Chromium and Hexavalent Chromium in Seep Water Collected from 100-D Spring.

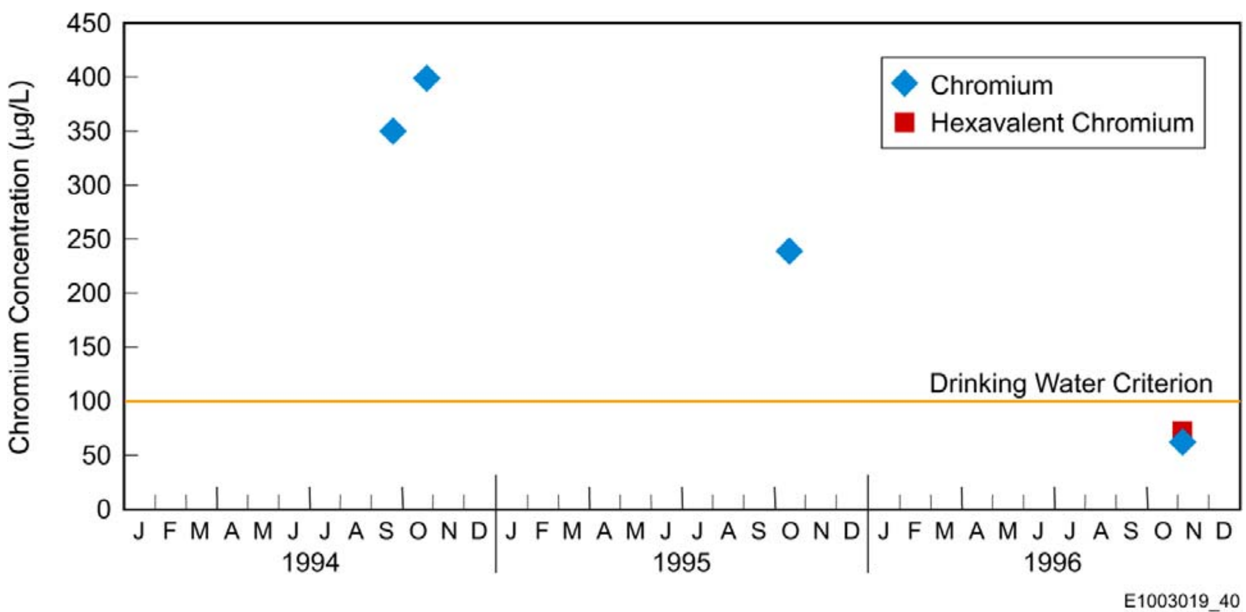


Figure 4-9. Concentration of Nitrate in Seep Water Collected from 100-F Spring 207-1.

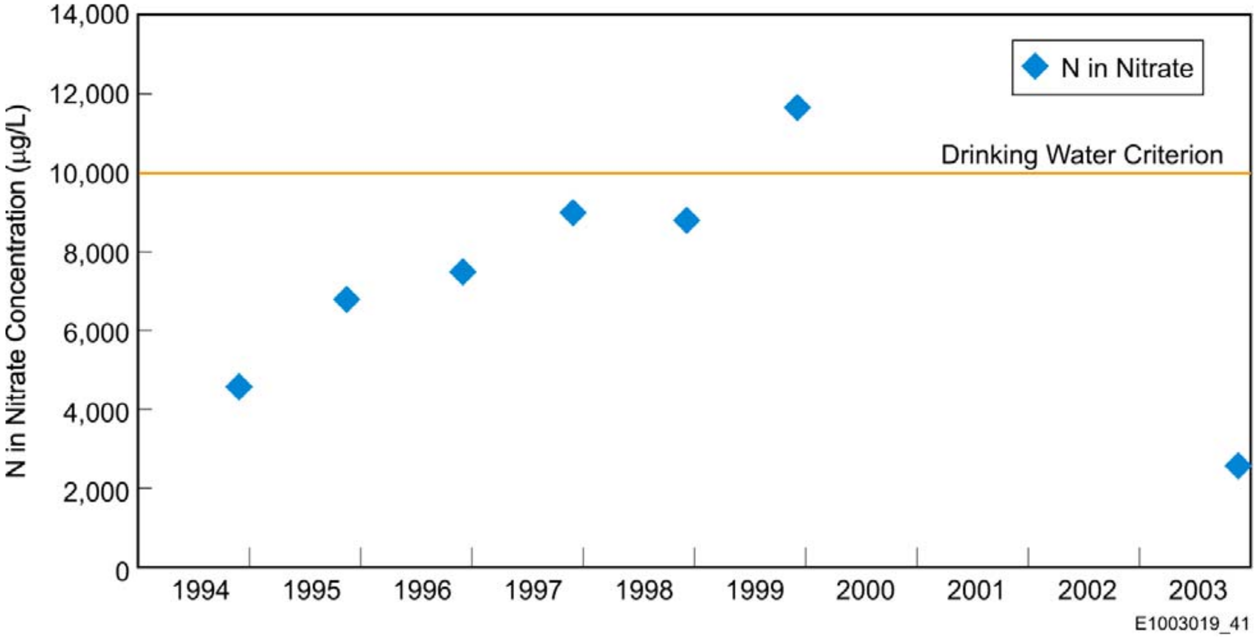


Figure 4-10. Concentration of Strontium-90 in Seep Water Collected from 100-H Spring.

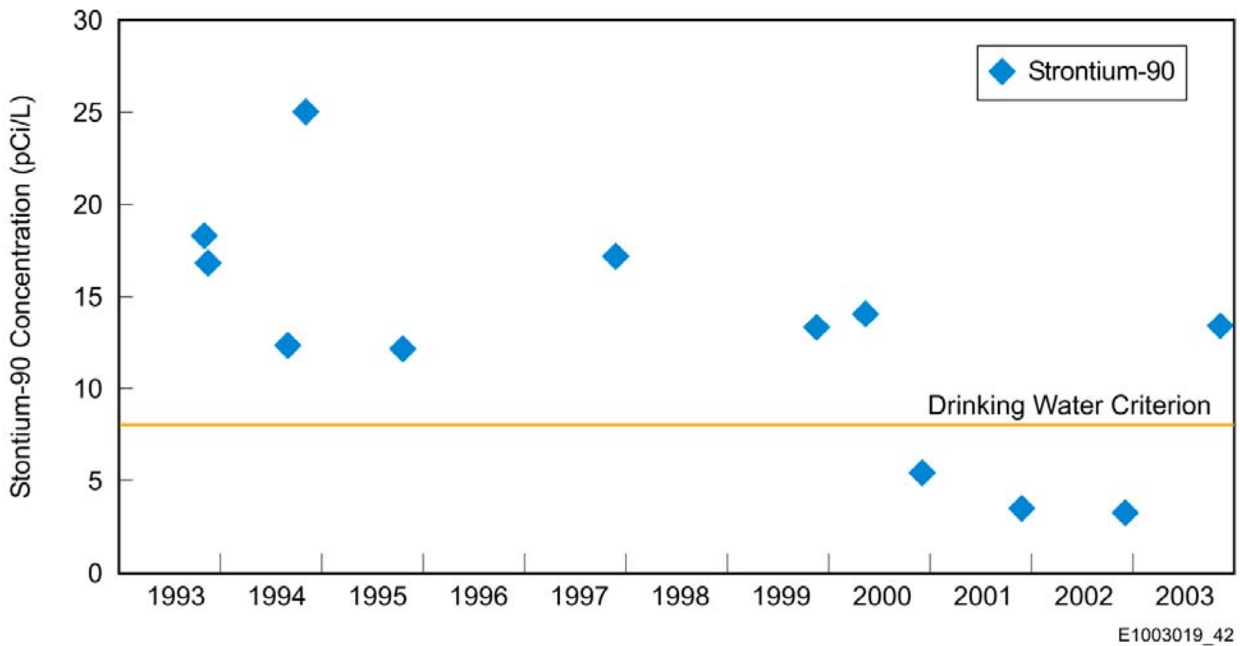


Figure 4-11 shows tritium concentrations collected between 1991 and 2004 at Spring 28-2 in the 100-F/100-IU-2/100-IU-6 ROD decision area. The tritium concentrations were highest in the years 1991 through 1994, reaching as high as 173,000 pCi/L in a sample collected on September 12, 1994. An anomalously low sample of 6,340 pCi/L was measured on September 2, 1994. By September 1995 the tritium concentration had dropped to approximately 20,000 pCi/L and then began slowly increasing again through 1998. The last two samples, collected in October 2002 and November 2003, show lower tritium concentrations with a last value of 14,000 pCi/L.

Uranium was analyzed and reported as isotopic uranium in Spring 42-2 in the 300 Area. Figure 4-12 shows the concentration of total uranium between 1991 and 2003.

4.5.2 Conclusions of the Risk Evaluation for Shoreline Springs

For the majority of the shoreline springs for which data have been made available, there is negligible risk related to exposure to key groundwater contaminants being released to the Columbia River at these locations. For a subset of springs, concentrations of a contaminant may exceed long-term drinking water criteria. At only one spring (100-F spring) is concentrations of a contaminant (nitrate) elevated relative to short-term standards that apply to 1-day or 10-day drinking water exposure. There are no short-term health advisories for uranium.

Figure 4-11. Concentration of Tritium in Seep Water Collected from Spring 28-2.

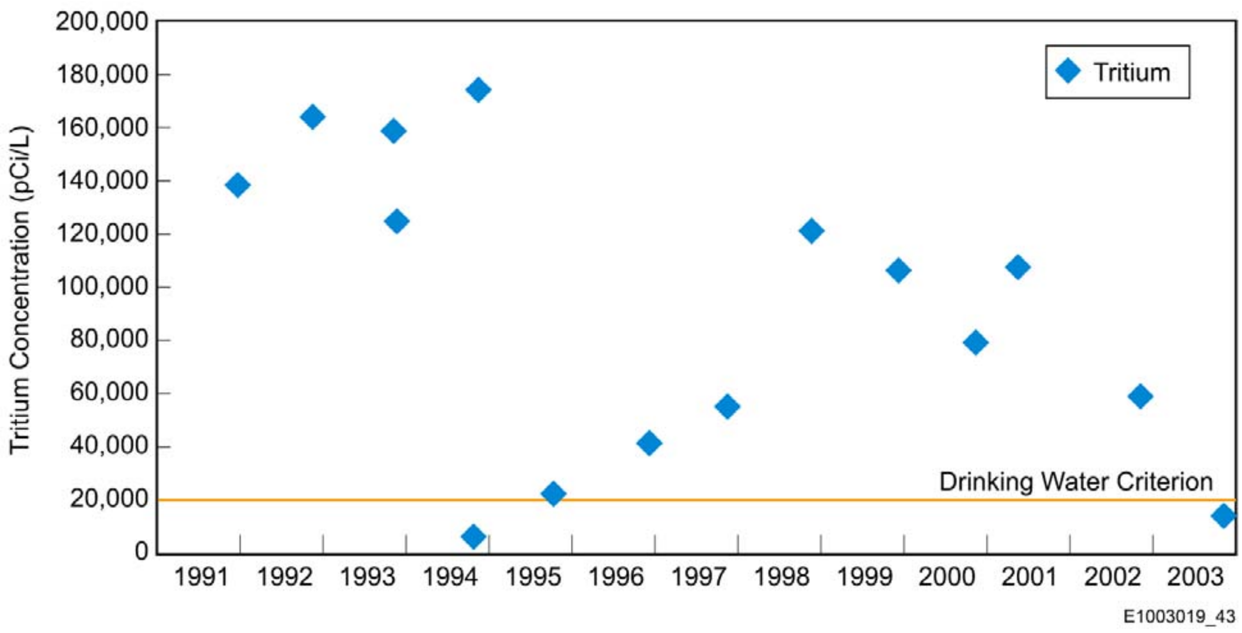
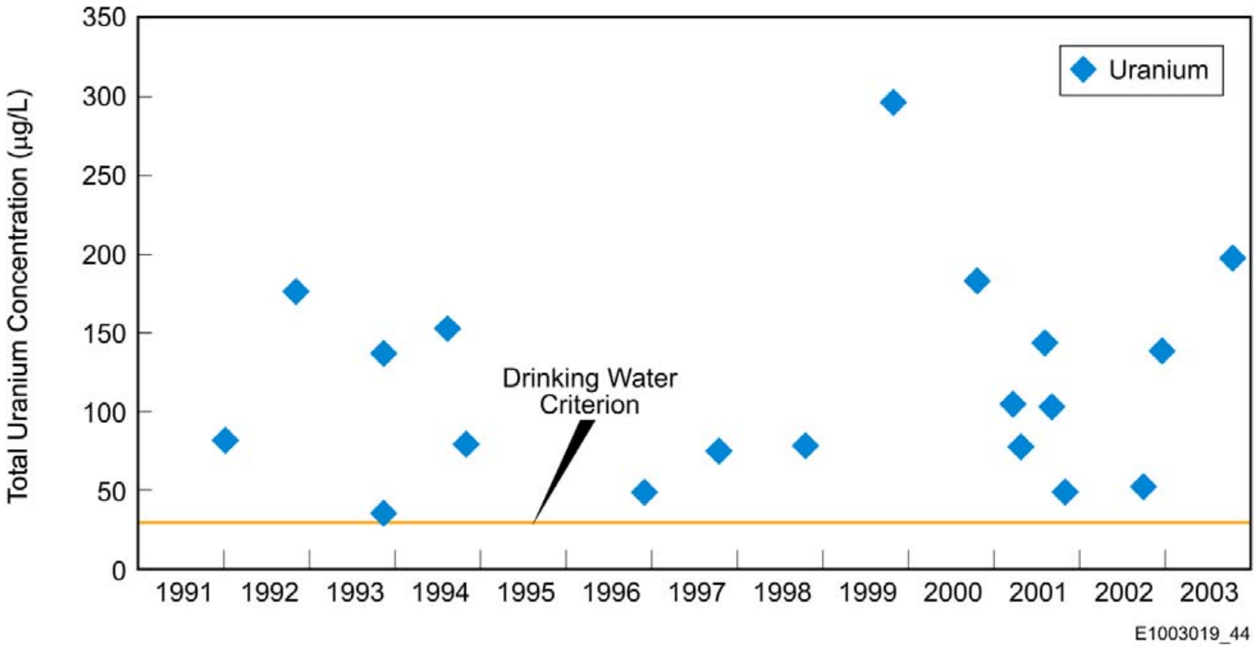


Figure 4-12. Concentration of Total Uranium in Seep Water Collected from Spring 42-2.



A summary of the major findings of this evaluation is as follows.

1. Concentrations of chromium at 100-D spring are up to four times higher than the long-term exposure screening criterion, but below the 1-day and 10-day health advisory levels used to evaluate short-term exposures. This analysis indicates minimal hazard from exposure to chromium at the spring. However, as discussed in Section 4.7.3, the states of California and New Jersey have recently published oral cancer slope factor criteria for hexavalent chromium.

2. Maximum nitrate values at the 100-F spring are at most 10% elevated above the short-term and long-term drinking water threshold and more recent measurements are all below. There is high confidence in the toxicological basis of the nitrate standards, which are related to epidemiological surveys of effects in infants. Therefore one may conclude there is minimal risk from occasional use of the water, particularly for adults. Caution is appropriate if young children might be exposed, such as in the Nonresident Tribal scenario, because they are particularly at risk for methemoglobinemia, the critical effect for nitrate exposure (IRIS 2009).

3. Concentrations of strontium-90 in the 100-H seeps are between two and three times the long-term threshold of 8 pCi/L. Since the threshold is related to long-term exposure at 2 L/day, one may conclude that there is negligible radiation dose for short-term, occasional exposures.

4. Concentrations of tritium in the 28-2 seeps in the 100-F/100-IU-2/100-IU-6 Area have been variable over time and have sometimes been almost 10 times as high as the long-term threshold of 20,000 pCi/L. As for the case of strontium-90 in the 100-H seeps, because the threshold is related to long-term exposure at 2 L/day, one may conclude that there is negligible radiation dose for short-term, occasional exposures.

5. Concentrations of uranium in the Spring 42-2 in the 300 Area have been between two to five times higher than the long-term threshold of 30 µg/L, with a maximum value in 1999 that was approximately tenfold higher. There is no short-term health advisory for uranium. The basis of EPA oral RfD for uranium (IRIS 2009) is moderate kidney toxicity observed in a 30-day rabbit study. Because there are no short-term health advisories, and on the basis of the study length for EPA's oral RfD, it is possible that short-term risks may exist for uranium exposures at the Spring 42-2.

4.6 RISKS RELATED TO THE FISH INGESTION EXPOSURE PATHWAY

4.6.1 Context and Scope

The aquatic biota tissue samples collected under the RCBRA, 100-B/C Pilot, and 100-NR-2 investigations focused on species with very small home ranges (e.g., sculpin), shellfish, and macroinvertebrates (e.g., crayfish). This sampling was largely conducted in specific areas where groundwater plumes emerge at the Columbia River for the purpose of determining the extent to which Hanford Site-related contaminants are accumulated by biota at these locations. Sculpin were identified in the RCBRA SAP (DOE/RL-2005-42) as an appropriate species for screening human health risks for Hanford Site-related releases.

Localized species such as sculpin, shellfish, and crayfish are not plausible food sources for chronic human exposure. Not only are these species not commonly eaten, local populations of these species in the region of a groundwater plume would be unable to support continuous harvesting for food over several decades, as envisioned in a chronic exposure assessment. Consequently, chronic health risks based on such an exposure assumption are unrealistic because individuals gathering these species would have to quickly move on to other locations once local stocks at a sampling location were depleted. Because the screening-level fish ingestion risk results are not based on plausible estimates of chronic exposure, they have not been summed with risks related to other exposure pathways for the exposure scenarios that include fish ingestion.

A survey conducted by the Columbia River Intertribal Fish Commission (EPA 910-R-02-006) identified salmon and trout as the primary food fish in the Columbia River basin. However, tissue concentrations of near-shore COPCs in anadromous fish like salmon and steelhead trout are unlikely to have any association with Hanford releases because these fish are captured as they return to spawn. Other game fish with large home ranges are resident within the Hanford Reach of the Columbia River. These may include rainbow trout, whitefish, sturgeon, and other species. Sampling of game fish that are more credible food sources for chronic exposure assessment is being conducted within the Hanford Reach of the Columbia River under the RCBRA CRC.

The screening-level risk assessment results using sculpin, clam, and crayfish data may be used for identifying risk-relevant COPCs and for evaluating relative risk among the six ROD decision areas. The screening-level results, although they are largely hypothetical, may still be useful in risk communication. The risk assessment for fish ingestion being prepared for the RCBRA CRC is intended to address realistic potential human health risks from ingestion of food fish in the Hanford Reach of the Columbia River, although such risks may not be related to releases from the Hanford Site.

Two relevant studies have been released by EPA Region 10 during the past 10 years describing contaminant levels in fish tissues in the upper Columbia River (EPA 2007) and in the entire Columbia River Basin (EPA 910-R-02-006). The Columbia River Basin study addressed levels of pesticides, metals including arsenic, dioxin-like chemicals such as PCBs, and other organic chemicals in 10 species of anadromous and resident fish from 24 study sites. The upper Columbia River study, which focused on a 241-km (150-mi) stretch between the U.S.-Canada border and the Grand Coulee Dam, addressed levels of the same types of contaminants in five resident fish species from six fish sample collection areas.

Summaries of the sample collection, preparation, and analysis for the EPA and RCBRA studies are provided in Section 4.6.2. The results of these two EPA studies are summarized in Section 4.6.3. Concentrations of key contaminants in the fish sampled by EPA are compared to concentrations measured in sculpin in the six ROD decision areas. Additionally, the health risks related to ingestion of these game fish are compared to those for sculpin in Section 4.6.3.

The results of screening-level fish ingestion risk calculations for the Avid Angler, Nonresident Tribal, Subsistence Farmer, CTUIR Resident, and Yakama Resident scenarios are presented in Section 4.6.4. These scenario-specific calculations are focused on COPCs in near-shore biota and employ data for sculpin, clams, and crayfish collected under the RCBRA, 100-B/C Pilot, and 100-NR-2 investigations.

4.6.2 Sample Collection, Preparation, and Analysis

Tissue contaminant concentration data from the RCBRA 100-B/C Pilot and 100-NR-2 data sets exist for a variety of fish including adult and juvenile sucker, sculpin, clams, and crayfish. For sculpin, fillet or muscle data were not obtained and contaminant concentrations for the whole organism and multiple whole organisms (including the skeleton and viscera) were combined and used instead. Results for clams are based on clam tissue and do not include the shell. The sampling for benthic macroinvertebrates involved placement of rock baskets in shallow water to provide unconsolidated substrate for colonization. These biota have been described as crayfish in the HHRA because crayfish were the dominant species by mass. Clams were deployed at the sampling locations by putting live organisms in plastic mesh sleeving (i.e., clam tubes). Asiatic clams (*Corbicula fluminea*) of relatively uniform size (15 mm [0.6 in.] to 22 mm [0.8 in.])

were collected from within the left-bank fish ladder upstream of the Hanford Site at Priest Rapids Dam. Sculpins (*Cottus bairdi*) were collected by electro-fishing in sampling reaches that overlapped other near-shore aquatic locations. Additional details of the near-shore sampling and analysis are provided in RCBRA Volume I.

Biota tissue samples were analyzed for a broad suite of possible contaminants including metals, radionuclides, semivolatile organic analytes, pesticides, and PCB Aroclors. Subsequent to review of the initial tissue data, an additional 10 sculpin tissue samples from the operational area, and one from a reference area were analyzed for total arsenic and inorganic arsenic and for PCB congeners. The intention of this sampling was to distinguish the relative contribution of inorganic and organic species of arsenic in sculpin tissues. Organic arsenic species such as arseno-sugars, monomethylarsenic acid, and dimethylarsenic acid are generally considered to be less toxic than inorganic arsenic. Detections of PCB Aroclors in some sculpin samples, particularly two high values in sculpin collected as part of the 100-B/C Pilot Project, led to supplemental sampling of PCB congeners in sculpin. The 10 additional sculpin samples supplement 5 sculpin samples from the 100-NR-2 investigation that were analyzed for PCB congeners. The PCB congener analytical method has much lower quantitation levels compared to the Aroclor mixture method and allows for better estimation of potential cancer risks based on the TCDD dioxin-like properties of certain congeners.

Five anadromous species (Pacific lamprey, smelt, Coho salmon, fall and spring Chinook salmon, steelhead) and six resident species (largescale sucker, bridgelip sucker, mountain whitefish, rainbow trout, white sturgeon, walleye) were sampled for the Columbia River Basin study (EPA 910-R-02-006). The types of samples collected included whole-body with scales, fillet with skin and scales, fillet without skin (white sturgeon only), and eggs. Although 24 locations were sampled, bridgelip sucker, Coho salmon, smelt, lamprey, and walleye were only collected at one or two locations. Therefore, evaluation of these results will focus on the remaining six species.

Columbia River Basin fish tissues were collected from 24 study sites throughout the basin between 1996 and 1998. A total of 281 samples of fish and fish eggs were collected for the Columbia River Basin study (EPA 910-R-02-006). Tissues were analyzed for 132 chemicals including 26 pesticides, 18 metals, 7 PCB Aroclors, 13 dioxin-like PCBs, 7 dioxin congeners, 10 furan congeners, and 51 miscellaneous organic chemicals (EPA 910-R-02-006). Of these 132 chemicals, the most frequently detected were 14 metals, DDT and related compounds, chlordane and related compounds, PCBs, and chlorinated dioxin and furans.

Five fish species (walleye, rainbow trout, lake or mountain whitefish, largescale sucker, and burbot) were targeted for collection in the upper Columbia River study (EPA 2007). Whole-body samples were collected for whitefish, sucker, burbot, and at three of the six collection areas for walleye and rainbow trout. At the remaining three collection areas, fillet samples were obtained for walleye and rainbow trout. Five composite samples for each target species and tissue type were planned for collection within each of the six fish sampling areas.

Upper Columbia River fish tissues were collected from six fish sample collection areas between the U.S.-Canada border and the Grand Coulee Dam in September and October 2005. A total of 165 fish tissue composites and 15 QA replicates were prepared and submitted for chemical analysis (EPA 2007). Tissues were analyzed for target analyte list metals, 9 PCB Aroclors, all 209 PCBs congeners, chlorinated dioxins and dibenzofurans, and multiple species of inorganic and organic arsenic (EPA 2007).

4.6.3 Risks Related to Key Contaminants in Columbia River Fish

Human health risks from fish ingestion related to contaminants in fish tissue were evaluated in the Columbia River Basin study (EPA 910-R-02-006). Risks were evaluated for both the general U.S. population and for Tribal populations along the Columbia River. Mercury and PCBs were the most common contributors to HI results for both resident and anadromous fish species. Figure 6-17 of EPA 910-R-02-006 shows adult HI results for both the general public and Tribal member for each fish species. Reasonable maximum exposure adult HI results for high-consuming individuals ranged between approximately 5 and 15 for salmon and trout species, which are favored food fish in the Columbia Basin (EPA 910-R-02-006). The highest adult HI values for the general public and Tribal member were approximately 25 and 65 for mountain whitefish, respectively.

Arsenic was a major contributor to cancer risk for anadromous fish species such as salmon and steelhead trout, but was relatively unimportant for resident species. PCB congeners/Aroclors were important contributors to cancer risk for both resident and anadromous species. Figure 6-34 of EPA 910-R-02-006 shows adult cancer risk results for both the general public and Tribal member for each fish species. Reasonable maximum exposure adult cancer risk results for high-consuming individuals ranged between approximately 5×10^{-4} and 1×10^{-3} for salmon and trout species, with a high Tribal member RME value of approximately 1×10^{-2} for mountain whitefish.

A comparison of tissue concentrations of the key contaminants driving HI and cancer risk (arsenic, mercury, and PCBs) in the Columbia River Basin study (EPA 910-R-02-006) is provided in Table 4-28. Average tissue concentrations in sculpin (whole body) collected at operational areas along the Hanford Reach are shown in relation to average tissue concentrations for different species measured in the Columbia River Basin study (EPA 910-R-02-006) and the upper Columbia River investigation (EPA 2007).

Whole body basin-wide average contaminant concentrations in different fish species are tabulated in Section 2 of the Columbia River Basin study (EPA 910-R-02-006). Equivalent values were obtained for arsenic and mercury from the upper Columbia River investigation (EPA 2007). Congener data were not summarized for the toxicologically relevant dioxin-like congeners in the upper Columbia River investigation.

Review of Table 4-28 indicates that there is no evidence that arsenic or mercury concentrations in sculpin from operational areas along the Hanford Reach are elevated with respect to other resident fish species in the upper Columbia River or throughout the Columbia River Basin.

Neither arsenic nor mercury were identified as a COPC in near-shore biota, indicating that concentrations of these metals in sculpin and other sampled species were not elevated relative to concentrations in the reference area. Concentrations of dioxin-like PCB congeners in sculpin were on the low end of the range of concentrations measured in Columbia River Basin resident fish species. With the exception of Pacific lamprey, the sculpin tissue concentrations of these congeners exceeded those measured in anadromous fish. The relatively low concentrations of PCBs in anadromous fish other then lamprey was observed for other persistent organic chemicals as well (EPA 910-R-02-006). Similarly, the high concentrations of PCBs in white sturgeon and mountain whitefish was also observed for other persistent organic chemicals. This systematic difference between resident and anadromous fish suggest that resident fish are the appropriate ones for comparison of sculpin PCB congener concentrations.

A subset of the sculpin data (11 samples) was analyzed for total inorganic arsenic. These data were acquired to allow comparison with the total arsenic data so that the relative quantities of inorganic and organic species of arsenic could be known. Organic arsenic species are generally considered to be less toxic than inorganic arsenic. However, all of the inorganic arsenic results were U-qualified (not detected), with detection limit values ranging between 0.014 and 0.075 mg/kg (mean of 0.02 mg/kg). These results are similar to those reported in EPA Region 10 (2002), where the majority of the data obtained for arsenic speciation were either nondetect or qualified as estimated.

Arsenic in fish tissue is generally present in the form of various organic species of arsenic, with only a minor contribution from inorganic forms. However, arsenic in the elemental form is the basis for the oral CSF used in the calculation of cancer risk via fish ingestion. Most of the arsenic in fish tissue is present as organic arsenic species such as arseno-sugars, monomethylarsenic acid, and dimethylarsenic acid, which are generally considered to be less toxic than inorganic arsenic. In a survey of fish contamination in the Columbia River Basin, EPA Region 10 reports the average percentage of inorganic arsenic in anadromous fish species, such as salmon and steelhead, at about 1% (EPA 910-R-02-006). For resident fish species, including sucker and sturgeon, the average was about 9% (EPA 910-R-02-006). Hence, the arsenic-related fish ingestion cancer risks summarized above from EPA 910-R-02-006 for ingestion of anadromous fish may be significantly overestimated to the extent that the organic species of arsenic are indeed less toxic than the elemental forms.
Fifteen sculpin tissue samples have analytical results for PCB congeners. These samples include 5 related to the 100-NR-2 investigation, as well as the 11 samples (10 operational and 1 reference area) related to the sampling for arsenic speciation described above. The sum of the concentrations of the detected congeners from operational area sculpin were lower than those in the single reference site sculpin sample.

4.6.4 Results for the Screening-Level Fish Ingestion Risk Assessment

In order to differentiate scenario specific risks related to soil and sediment exposures from screening risks due to fish ingestion, the screening-level risk assessment results for exposures related to fish ingestion are calculated and presented independently. Screening results using

COPC concentrations in tissue samples for sculpin, clams, crayfish, and juvenile sucker are presented in this section for the Avid Angler, Nonresident Tribal, Subsistence Farmer, CTUIR Resident, and Yakama Resident scenarios, which have fish ingestion exposure pathways. The risk assessment calculations presented in Sections 4.3 and 4.4 pertain to exposures based on soil, sediment, and surface water data.

Tables 4-29 through 4-33 show the risk assessment results for near-shore COPCs for the Avid Angler, Nonresident Tribal, and the three Residential scenarios in each ROD decision area using the RCBRA data for sculpin, clams, crayfish, and juvenile sucker. In all cases, risk assessment results for individual species presume that all fish ingestion over the exposure duration period is limited to this species of fish. As described in Section 4.6.1, not only are these species not commonly eaten, but the local populations of these species in the region of a groundwater plume would be unable to support continuous harvesting for food over several decades, as envisioned in a chronic exposure assessment. Therefore, the risk assessment results with the sculpin, clam, crayfish, and juvenile sucker data should only be used for identifying risk-relevant COPC concentrations among the different fish species, for evaluating relative risk among the ROD decision areas, and for risk communication. The RCBRA CRC risk assessment for fish ingestion being prepared with newly acquired data is intended to address realistic potential human health risks from ingestion of food fish in the Hanford Reach of the Columbia River.

4.6.4.1 Cancer Risk. The only carcinogenic COPC identified for any ROD area in near-shore biota is carbon-14. Carbon-14 was identified as a COPC in sculpin and clam in the 100-K Area based on detects in one of two samples and one of four samples, respectively. Carbon-14 was not detected in the single reference-area clam tissue sample collected, and there are no carbon-14 reference-area data for sculpin. Carbon-14 was detected in two of three sediment samples in the 100-K Area but was not detected in the three river water samples collected near the river bank in the 100-K Area where carbon-14 is present in groundwater. The low detection frequency in localized aquatic biota, and the absence of carbon-14 in river water at the location where the carbon-14 groundwater plume exists, indicates that carbon-14 related to the 100-K Area plume would not be expected to impact food fish with large home ranges.

As described in RCBRA Volume I, Section 6.0, carbon-14 was identified as a COPC in near-shore biota because it was either not measured (sculpin) or not detected (in one sample) in reference-area samples of these biota. Carbon-14 data for sculpin and clams are uncensored, so CTE and RME representative concentrations for the 100-K Area were computed using all six (sculpin) or eight (clam) data points.

Cancer risk results for the Avid Angler, Nonresident Tribal, and Residential scenarios using the near-shore environment sculpin and clam data are shown in Tables 4-29 through 4-33. The 100-K ROD decision area CTE cancer risk calculated using the sculpin data is 2×10^{-6} for both the Avid Angler and Subsistence Farmer scenarios. The 100-K Area RME cancer risks calculated using the sculpin data are 4×10^{-5} and 9×10^{-6} for the Avid Angler and Subsistence Farmer scenarios, respectively. Cancer risks for the Nonresident Tribal, CTUIR Resident, and Yakama Resident scenarios using the sculpin data are higher (2×10^{-4} or 3×10^{-4}), reflecting the higher fish ingestion rates for these scenarios and a full 70- to 75-year lifetime (rather than 30-year) exposure duration. The cancer risk estimates using the clam tissue data are approximately 10 times higher than when using the sculpin data. Carbon-14 CTE and RME values for sculpin and clam in the 100-K Area are both 8.1 pCi/g and 93 pCi/g due to the low detection frequencies. Because carbon-14 was also identified as a COPC in sediment from 100-K Area samples, with a representative concentration of 80 pCi/g, the higher concentration in the single clam detect (93 pCi/g) versus the single sculpin detect (8.1 pCi/g) may reflect the greater physical association of clams with sediment.

Representative concentrations may be calculated for a radionuclide COPC even when all results are below the minimum detectable activity if the data are uncensored. This is the case for carbon-14 in sculpin in the 100-N ROD decision area and juvenile sucker in the 100-K Area, where the sample results were positive but below the minimum detectable activity. Although representative concentrations for carbon-14 were calculated for sculpin in the 100-N Area and juvenile sucker in the 100-K Area, cancer risk and dose calculations were not performed with these nondetect values.

4.6.4.2 Radiation Dose. As summarized with respect to cancer risk, carbon-14 was detected in both sculpin and clam tissue in the 100-K decision area. Radiation dose results for the Avid Angler, Nonresident Tribal, and the three Residential scenarios using the near-shore sculpin and clam data are shown in Tables 4-29 through 4-33. The 100-K Area CTE radiation dose calculated using the sculpin data is 0.24 mrem/yr for both the Avid Angler and Subsistence Farmer scenarios. The 100-K Area RME radiation dose values calculated using the sculpin data are 1.6 mrem/yr and 0.43 mrem/yr for the Avid Angler and Subsistence Farmer scenarios, respectively. Radiation dose for the Nonresident Tribal, CTUIR, and Yakama scenarios using the sculpin data are higher (3.7 mrem/yr and 3.9 mrem/yr), reflecting the higher fish ingestion rates for these scenarios. The radiation dose estimates using the clam tissue data are approximately seven to eight times higher than when using the sculpin data. Carbon-14 representative concentrations in sculpin and clam in the 100-K Area are 8.1 pCi/g and 93 pCi/g, respectively. Because carbon-14 was also identified as a COPC in sediment from 100-K decision area samples, the higher concentration in the single clam detect versus the sculpin detect may reflect the greater physical association of clams with sediment.

The results of the CTUIR Resident and Yakama Resident fish ingestion dose calculations for carbon-14 in sculpin in the 100-K Area (3.7 mrem/yr and 3.9 mrem/yr) are higher than those calculated for food fish in Delistraty et al. (2009). In this paper, total radiological dose for 10 radionuclides (not including carbon-14) in a Native American scenario was calculated to be 0.23 mrem/yr. These fish included bass, carp, sucker, and whitefish. The higher results for sculpin may reflect the actual association of a Hanford Site-related radionuclide in these tissue samples from the location of a groundwater contaminant plume compared to nominal background levels in the more widely-ranging species evaluated in Delistraty et al. (2009).

4.6.4.3 Chemical Hazard. Chemical hazard results for the Avid Angler, Nonresident Tribal, and the three Residential scenarios using the near-shore data for sculpin, clam, crayfish and juvenile sucker are shown in Tables 4-29 through 4-33. Near-shore biota COPCs potentially contributing to chemical hazard for fish ingestion include aluminum, barium, copper, manganese, nickel, selenium, strontium, vanadium, and zinc. For any scenario, results vary considerably depending on both the ROD decision area and the individual fish species. In general, the highest calculated HI results are associated with clams and crayfish, with lower results for sculpin and juvenile sucker.

The main contributor to the crayfish HI in each ROD decision area is copper, except in the 100-F/100-IU-2/100-IU-6 Area, where copper and manganese contribute in about equal proportion. That copper is a major contributor to the crayfish HI is likely due to its presence in the molecule hemocyanin, which carries oxygen in the blood of crayfish in the same manner as hemoglobin does in mammals. Reference-area HI values for crayfish are approximately equal to or above HI values for each ROD decision area except 100-F/100-IU-2/100-IU-6. Both copper and manganese are essential micronutrients whose uptake from the gastrointestinal tract is normally under homeostatic control. For this reason, toxicity via uptake of these metals from foods would not be expected.

Hemocyanin is also present in clams, and copper is the primary contributor to the clam HI in all ROD decision areas. Unlike the case with crayfish, clam HI results are generally higher in the ROD decision areas than in the reference area. The reference area copper RME value of 10 mg/kg in clams is below the range of RME clam representative concentrations in the ROD decision areas (8.7 mg/kg in the 100-N Area to 27 mg/kg in the 100-K Area). However, the copper oral RfD is based on gastrointestinal irritation by free copper ions and is related to the drinking water standard for copper (IRIS 2009), local irritation due to the action of these metal ions on the walls of the gastrointestinal tract is an effect that may not be relevant for a food ingestion exposure pathway.

Some of the highest calculated HI results are for sculpin in the 100-D/100-H ROD decision area. Nearly 90% of these HI results are due to nickel. The nickel RME representative concentration in sculpin in the 100-DH Area is 15 mg/kg. The 100-D/100-H Area nickel RME value for sculpin is well above the range of RME values in the other ROD decision areas (0.47 in the 300 Area to 2.2 mg/kg in the 100-K Area). Nickel is not known to be an essential nutrient. As discussed in Section 4.7.3, the oral reference dose for nickel includes an uncertainty factor of 300 due to limitations in the rodent study that is the basis for this toxicity criterion.

4.7 UNCERTAINTY ANALYSIS FOR THE BROAD-AREA RISK ASSESSMENTS

The uncertainty analysis provides quantitative and qualitative information for evaluating the level of confidence, including the potential for over- or underestimating risk, in the broad-area risk assessment results described in Sections 4.3 through 4.6. The main purpose of the uncertainty analysis is to put the numerical results in perspective with regard to the assumptions and uncertainties in the risk assessment process. Another purpose of the uncertainty analysis is to provide a basis for recommendations relating to additional information that could be of value for refining the risk estimates.

The discussion of uncertainties is organized according to the steps comprising a baseline risk assessment that lead to risk characterization. These steps include Data Collection and Evaluation Exposure Assessment, and Toxicity Assessment. The principal tools applied in the HHRA for quantifying uncertainties in the risk estimates are as follows:

- The use of RME and CTE exposure parameter values in the risk calculations
- The use of RME and CTE estimates of exposure concentrations in the risk calculations
- The use of multiple exposure scenarios to address a range of low- and high-intensity land uses.

The range of exposure parameter values related to behavioral and/or physiological characteristics (i.e., ingestion and inhalation rates, exposure frequency) provides a measure of uncertainty related to the attributes of individual receptors within a receptor population. The use of best estimate and upper-bound exposure concentrations in the CTE and RME calculations, respectively, provides a measure of the importance of uncertainty in the COPC concentrations in exposure media to the risk estimates. The range of the RME and CTE results across the various exposure scenarios provides information on the importance of the different exposure pathways and receptor characteristics on the risk estimates.

The quantitative measures of uncertainty evaluated via the CTE and RME calculations, and the use of multiple exposure scenarios as described here, can only address those aspects of uncertainty that relate to the choice of specific input parameter values and exposure pathways. A semiquantitative or qualitative assessment of uncertainty inherent to these types of assessments is provided for other aspects of the risk assessment that affect the final estimates. These include the following:

- Uncertainty in data collection and evaluation including analytical data quality and data representativeness (Section 4.7.1)
- Uncertainty in the exposure assessment, particularly the various transport models used in developing the exposure point concentrations (Section 4.7.2)
- Uncertainty in the toxicity assessment, particularly the dose-response models underlying the chemical and radionuclide toxicity criteria and extrapolation from effects observed at high dose rates to effects at much lower environmental dose rates (Section 4.7.3).

Both the quantitative and qualitative assessments of uncertainty are directed towards identifying key assumptions and parameters that have the most potential to contribute to significant human health exposures and effects.

With the exception of the Avid Angler scenario, where exposures are assumed to occur primarily to sediment and river water in the near-shore environment, soil-related exposure pathways are the basis of most of the risk assessment results presented in Sections 4.3 and 4.4. Like the Avid Angler scenario, the Nonresident Tribal scenario discussed in Section 4.4 also includes exposure to sediments and surface water in the Columbia River, but potential risks from exposures to these media are negligible compared to risks related to soil-based exposure pathways.

4.7.1 Uncertainty in Data Collection and Evaluation

Uncertainty pertaining to data collection and evaluation encompasses sample collection activities, laboratory sample preparation and analysis, and data preparation and analysis. Uncertainty related to laboratory sample preparation and analysis is generally not a significant contributor to overall uncertainty in risk assessment results. A major reason for this is that quality control samples are used to assure that analytical results are within acceptable levels of precision and accuracy. However, key uncertainties are associated with other aspects of sample collection and data analysis. These uncertainties are discussed below.

4.7.1.1 Combining Environmental Data from Multiple Sampling Programs. The use of environmental data from three different sampling programs (RCBRA, 100-B/C Pilot Project, and 100-NR-2) for broad-area risk calculations in the riparian and near-shore environments introduces uncertainty related to the consistency of the analytical results. This is due to differences among the programs in sample acquisition methods, sample preparation techniques, target analyte lists, and analytical methods from one study to another.

Because data were available from various sampling programs that had different objectives as noted above, all analytical data used in the human health and ecological risk assessments were subjected to a process for ascertaining their usability to support such assessments. All data were required to have, at a minimum, the following attributes in order to be considered usable:

- An analyte name or Chemical Abstracts Service identification number
- A numerical result without a rejected (R) qualifier
- Associated units for the results
- A media type
- Definitive locational information.

Even in cases where all five attributes were present, analytical data were at times labeled not usable for 1 or more of 29 reasons where usability codes have been assigned in the database (see Section 4.4 of Appendix C-1). Some of these reasons include inappropriate analytical method, nonstandard units that cannot be converted, physically infeasible results, and mixed media type, such as paint chips or concrete. A complete discussion of data usability is provided in Appendix C-2.

4.7.1.2 Comparison of Detection Limits to Practical Quantitation Limits and Reference Values. The QAPP within the RCBRA SAP (DOE/RL-2005-42) specified the analytical performance requirements for soil, sediment, water, and biota data. Data from other investigations used to quantify health risks, including the 100-B/C Pilot and 100-NR-2 data, have not been individually reevaluated for analytical performance requirements or other specific quality criteria for this risk assessment. However, all data sets used in the RCBRA were evaluated for general usability in the risk assessment according to six criteria, as described in Section 4.2.

A comparison of RCBRA sample-specific detection limits for RCBRA soil and surface water data versus target PQLs and human health reference values from the QAPP is provided in a spreadsheet attachment to Appendix C-3. The results are also summarized in Tables 4-34 and 4-35. There are four PCB Aroclors that had detection limits in upland surface soil exceed the target PQLs and both direct contact and groundwater protection screening levels. Four pesticides also had detection limits in upland surface soil that exceeded the target PQLs, although there were no screening levels in the QAPP for these analytes. The highest upland nondetect PCB value of 1.3 mg/kg is higher than the Aroclor-1254 RME upland soil concentration of 0.59 mg/kg used in the risk assessment. In the absence of arsenic, this RME concentration would still only contribute approximately 5% to 10% of the "above reference" cancer risk for the Nonresident Tribal scenario in the six ROD decision areas. Relative to arsenic, potential contributions to risk from the Aroclors and pesticides with detection limits above PQL values are negligible for upland soil.

Toxaphene, 2,4,6-trichlorophenol, and pentachlorophenol had highest detection limits in riparian surface soil that exceeded the target PQLs by factors of between 2 and 5. The detection limits of the phenolic compounds (0.67 mg/kg and 1.7 mg/kg) exceed only the groundwater protection screening levels and are below the direct contact screening values. Widespread riparian soil contamination with these compounds is inconsistent with the CSM and with no detections above 1 or 2 mg/kg. The possibility of an actual threat to groundwater for these compounds is remote. Similar phenolic compounds were also identified with detection limits above PQLs in surface water, but the highest detection limits were less than 10% higher than PQLs.

4.7.1.3 Spatial Distribution of Contaminants.

Upland Surface Soil MIS Data. These data are employed in the broad-area risk assessment to represent average COPC concentrations in upland surface soil across the entire River Corridor. These data were obtained from 1-ha (2.47-ac) MIS soil samples at the 10 remediated native soil waste sites and 10 backfilled waste sites sampled for the RCBRA investigation. The principal uncertainty related to their use is the relationship of average soil concentrations from these 20 remediated waste sites to average soil concentrations across the entire upland environment of the River Corridor. Because soils at and adjacent to remediated waste sites are expected to have higher concentrations of Hanford Site-related COPCs than soils in other upland areas, COPC-related risks for the Avid Hunter and Nonresident Tribal exposure scenarios are likely to have a protective bias.

Reference Area Surface Soil MIS Data. The reference site data were used in the COPC identification process and for computing COPC-specific cancer risk, radiation dose, and HQ for the reference area soils. The term "reference area" is used in the RCBRA in a manner consistent with its use in *Guidance for Comparing Background and Chemical Concentrations in Soil at CERCLA Sites* (EPA/540/R-01/003), where it is called a "background reference area." The selection of reference areas for the RCBRA was based on a number of factors that included ecological conditions in addition to factors such as soil chemistry and the absence of contamination. Although discrimination based on ecological characteristics is not relevant to the HHRA, the other reference site selection criteria ensure that these sites are applicable as reference areas for both the human health and ecological risk assessments.

Soil data from 10 upland reference sites were used to compute the COPC-specific reference area risk results shown in Tables 4-11, 4-12, and 4-13. Five of the upland reference sites were located within "native" (mostly undisturbed) habitat areas, and the other five upland reference areas were located within uncontaminated borrow pits, which are disturbed areas used to acquire backfill material for remediated waste sites. The reference area samples from uncontaminated borrow bits may have lower concentration radionuclides such as cesium-137, strontium-90, and isotopic plutonium that are associated with radioactive atmospheric fallout from historical above-ground atomic weapons testing. These radionuclides are distributed within surface soils, but have near-zero background levels in deeper soils unaffected by atmospheric deposition. Upland reference area cancer risk and radiation dose results for these radionuclides may be biased high as a result.

Riparian and Near-Shore Data. The riparian and near-shore aquatic sampling sites were targeted to include areas expected to contain the highest levels of residual contamination. These sites were selected based on information from radiological surveys and groundwater plume maps, with the intent of collecting samples from locations with the highest apparent potential for elevated contaminant levels. Therefore, the riparian and near-shore samples are expected to have higher concentrations of Hanford Site-related COPCs than would samples collected from random locations along the river within each ROD area. Contaminant of potential concern-related risks for the Avid Angler, Casual User, and Nonresident Tribal exposure scenarios for the six ROD areas are likely to have a protective bias because 100% of exposure is assumed to occur in the areas represented by the riparian and near-shore sampling.

Identification of COPCs. Section 3.2 of this assessment describes the process that was developed to identify COPCs in different environmental media that are related to Hanford Site operations. The results of this process for the broad-area data are described in Section 4.2.4. Key factors related to COPC identification include the magnitude and significance of the results of statistical comparisons with background data, process knowledge, and comparison of results among different media and for similar analytes. Because the COPC identification process was systematic, it is unlikely that Hanford Site-related analytes that could contribute to significant health risks were eliminated in this process. As discussed below in relation to arsenic; although it is unlikely that Hanford Site-related analytes were excluded as COPCs there may be considerable uncertainty regarding whether any particular COPC that had elevated concentrations compared to the reference area is in fact related to Hanford operations.

A key COPC in the broad-area risk assessment is arsenic, which was responsible for a total cancer risk of approximately 1×10^{-2} in the Nonresident Tribal scenario due to plant ingestion. The 95% UCL concentrations of arsenic in upland soil in the operational areas (4.7 mg/kg) are about 50% higher then the 95% UCL value of 3.2 mg/kg in the upland reference area. Smaller differences in site and reference area arsenic concentrations are observed in riparian soils in the six ROD areas. Across both upland and riparian soil, the range of CTE and RME arsenic concentrations was approximately 4 to 10 mg/kg. The lower end of this range is comparable to the 50th percentile Hanford Site background level of 3.6 mg/kg (DOE/RL-92-24). The upper end of this range is above the 90th percentile Hanford Site background level of 6.5 mg/kg. Although site RME arsenic concentrations exceed reference area and sometimes Hanford Site 90th percentile values, 10 mg/kg is one-half of the IAROD cleanup value of 20 mg/kg for arsenic, which was established by the Washington State Department of Ecology as an unrestricted land use cleanup value, with adjustment related to background levels of arsenic (http://apps.leg.wa.gov/WAC/default.aspx?cite=173-340-900).

Yokel and Delistraty (2003) have shown that the average arsenic concentration in surface soil from 31 samples associated with former orchards at the Hanford Site is approximately 30 mg/kg, with a standard deviation of 61 mg/kg and a maximum value of 270 mg/kg. These levels are related to the use of lead arsenate pesticides at these locations (Yokel and Delistraty 2003). This indicates that there is a substantial source of arsenic contamination in the 100 Area of the River Corridor other than Hanford Site operations.

4.7.1.4 Calculation of Representative Concentrations. As described in Section 4.2.3, representative concentrations for the CTE and RME calculations are based on the mean, and the 95% UCL on the mean when there were five or more detected values available to calculate a statistic. When there are between one and four detected values, then either the average or maximum value is used for the representative concentration. A summary of the number of representative concentrations based on 1, 2, 3, 4, or more than 4 detected values for the different environmental media used in the risk assessment is provided in Tables C3-1 through C3-5 in Appendix C-3. These tables can be used to quickly ascertain the number of samples supporting the representative concentrations used in the risk assessment. In general, there is an inverse relationship between the number of detected samples and the uncertainty associated with a representative concentration.

The methods used to calculate representative concentrations distinguish between cases where the number of detected values is below five, or is equal to five or more. When there are fewer than five detects, the maximum detected concentration is used as the RME representative concentration. Reasonable maximum exposure representative concentrations would be expected to be lower if samples where an analyte was not detected were accounted for in the calculation. To evaluate the degree of protective bias, supplemental risk assessment calculations were performed using the sample-specific reporting limit for chemicals and the method-specific minimum detectable activity for radionuclides for a U-qualified (nondetect) result. The supplemental calculations were performed for the Subsistence Farmer and Industrial/Commercial scenarios for cancer risk and chemical hazard, and are discussed in detail in Section 5.9.

The supplemental cancer risk and HI results were never more than a factor of two lower than the original value, indicating that the degree of protective bias is usually minimal.

When there are five or more detected values, the median of three separate estimates of the mean and UCL, based on different statistical methods, was used as the CTE and RME concentration, respectively. A summary of the number of representative concentrations calculated by the different methods is provided in Tables C3-7 and C3-8 in Appendix C-3. The variability between the results of the UCL methods, and the relationship of the statistical UCL to the maximum detected concentration is provided in Tables C3-9 and C3-10 in Appendix C-3. There were no occurrences where the maximum of the three UCL values was more than five times the median.

The upland and riparian reference area samples are also used to calculate representative concentrations. These values are used in calculating risks that are comparable to those for the broad-area risk assessment, but representative of areas that were not impacted by Hanford releases. With the exception of 100-B/C Pilot and 100-NR-2 riparian soil samples, site and reference area soil samples used to calculate representative concentrations were collected using the same sampling protocol.

4.7.2 Uncertainty in the Exposure Assessment

There are inherent uncertainties in the application of hypothetical exposure scenarios to the as-yet unknown future conditions of the Hanford Site. To an extent, these uncertainties are addressed by the use of multiple exposure scenarios in the broad-area and local-area assessments to provide a range of potential exposure intensities in support of remedial decision making.

4.7.2.1 Values of Behavioral Variables. Within each of the Recreational exposure scenarios, there is uncertainty in the representative concentrations of contaminants in environmental media and in the exposure parameter values that is assessed by the calculation of CTE and RME risks. For any given exposure scenario, the CTE risks represent a hypothetical individual with approximately best-estimate levels of contact with contaminants in the exposure media across the various exposure pathways. The RME risk calculations represent a hypothetical individual with an RME condition of contact with contaminants in the exposure media. Exposure parameter values for the Nonresident Tribal scenario are based on values provided in Harris and Harper (2004) and Ridolfi (2007). Selection of specific values between Harris and Harper (2004) and Ridolfi (2007) is documented in Appendix D-2. These values are used in conjunction with RME representative concentrations to calculate a single set of risk results.

Differences between CTE and RME risks vary as a function of exposure scenario and receptor depending on the magnitude of the differences between CTE and RME exposure parameter values and the type and number of such parameters included in the exposure equation. Differences between CTE and RME calculations are also related to the magnitude of differences between the CTE and RME exposure point concentration. The range between CTE and RME risk results was generally about a factor of 10 or less for calculations relating to the soil source term.

4.7.2.2 Modeled Concentrations of COPCs in Foods. Another aspect of uncertainty related to the exposure assessment involves the models used to estimate exposure point concentrations in various exposure media. In particular, these include models to estimate concentrations in foodstuffs for the Avid Hunter scenario (game meat) and the Nonresident Tribal scenario (plants and game meat).

As discussed in Section 3.4, both the plant-soil concentration ratios ($K_{p\text{-}s}$) and feed-to-tissue transfer factors for meat (B_a) used to model COPC concentrations in foods reflect an assumption that there is a linear and unchanging relationship between soil (or feed) and tissue concentrations. Because this assumption ignores physiological mechanisms controlling the accumulation of toxic substances including metals and radionuclides, tissue concentrations of these COPCs are susceptible to overestimation when soil concentrations are elevated. Because surface soil concentrations of metals and radionuclides in the upland and riparian environments are not greatly elevated above background levels, the assumption of linearity may not be a significant source of uncertainty in the Avid Hunter and Nonresident Tribal results. However, even at soil concentrations that are near naturally occurring levels, variability in plant uptake related to soil conditions, plant species and tissue type, harvest time, and other variables contribute to a high level of intrinsic variability in $K_{p\text{-}s}$ values (Sheppard 2005, "Transfer Parameters – Are On-Site Data Really Better?").

The $K_{p\text{-}s}$ values used in the HHRA are those employed in Version 6.5 the RESRAD computer code (http://web.ead.anl.gov/resrad/home2/). RESRAD has historically been used at the Hanford Site for performing radiation dose assessments, and is the basis of the IAROD cleanup levels for radionuclides used to document interim closure of the remediated waste sites evaluated in this report. Arsenic exposure through plant ingestion is the primary risk driver for the Nonresident Tribal scenario, resulting in calculated risks of approximately 1×10^{-2} in each ROD decision area. The arsenic $K_{p\text{-}s}$ value of 0.53 from the RESRAD code is higher than the values cited in a number of other references commonly used in HHRAs. For example, values from EPA/530-R-05-006, *Human Health Risk Assessment Protocol for Hazardous Waste Combustion Facilities* (0.00633); ORNL-5786, *A Review and Analysis of Parameters for Assessing Transport of Environmentally Released Radionuclides Through Agriculture* (0.006 and 0.04); and BJC/OR-133, *Empirical Models for the Uptake of Inorganic Chemicals by Plants* (median; 0.0375) are all a factor of 10 to 100 below the RESRAD value. Use of these $K_{p\text{-}s}$ values would result in a significant decrease in the Nonresident Tribal cancer risk results.

Because the plant ingestion exposure pathway is the key contributor to potential human health cancer risk and chemical hazard above threshold criteria for the Nonresident Tribal scenario, a sensitivity analysis was conducted focusing on two key variables: $K_{p\text{-}s}$ values (range of values based on published ratios) and the percentage of diet that comes from wild plant foods gathered on the Hanford Site (range varied between 5% and 95%). The range of plant ingestion pathway cancer risk (approximately four orders of magnitude) and chemical hazard (approximately three orders of magnitude) was overwhelmingly related to the range of $K_{p\text{-}s}$ values. Nonresident Tribal plant ingestion results described in Section 4.4 are near the upper end of the range of high-end and low-end cases from the sensitivity analysis. The differences between broad-area Site soil concentrations and reference area soil concentrations for COPCs that drive cancer and/or chemical hazard high-end sensitivity analysis results (arsenic, uranium, and zinc) are about a factor of 1.5. The small difference between Site and reference soil concentrations means that there is little difference in calculated risks whether wild plants are gathered in the River Corridor or in a reference area. By contrast, the range of published K_{p-s} values vary by factors of 190 (arsenic), 2,200 (uranium), and 500 (zinc). This analysis indicates that Nonresident Tribal plant ingestion risks are most sensitive to variability in potential plant concentrations, and that the range of this variability is very great relative to the magnitude of the increment of Hanford Site soil concentrations above reference area levels. This sensitivity analysis is documented in Section 4.1.1 of Appendix D-1.

Measurements of paired soil and plant tissue samples collected for the RCBRA show an arsenic K_{p-s} value close to the RESRAD number of 0.53. The ratio of the average plant concentration to the average soil concentration from RCBRA Volume I is 1.1. The soil and plant sampling for these data pairs occurred over a 1-ha (2.47-ac)-area and therefore provides only a spatially-averaged measure of the relationship between concentrations in these two media. Also, the RCBRA plant tissue data are related to shrubs and grasses that may have little relevance to the edible tissues of garden crops or native plants that might serve as a food source. Detection limits for certain COPCs are elevated in the available plant tissue data, which further limits their utility. This subject is discussed in detail in Section 5.9.2 of this report. For these reasons, literature K_{p-s} values were used rather than measured values in plant tissues.

Exposure point concentrations of volatile and semivolatile COPCs in sweat lodge air were calculated for the Nonresident Tribal scenario from river water COPC concentrations using the methods described in Appendix 4 of Harris and Harper (2004). Section 3.4.3, Equation 14 of Harris and Harper (2004) provides a method for calculating the concentration of nonvolatile COPCs in air as a function of the amount of water needed to create a saturated atmosphere of water vapor in a sweat lodge. Because nonvolatile contaminants have no vapor pressure, this equation is physically implausible and the resulting risk estimates are of limited value. For this reason, only inhalation of volatile and semivolatile COPCs in a sweat lodge was evaluated in the Nonresident Tribal scenario in Section 4.4.

COPCs in river water in the six ROD decision areas include hexavalent chromium, cadmium, and mercury, and a number of radionuclides including uranium isotopes, strontium-90, technetium-99, and cesium-137. The chemical hazard values calculated using Equation 14 of Harris and Harper (2004) for the sweat lodge inhalation exposure pathway were below 0.1, and the radiation dose results were below 1 mrem/yr. Cancer risk results ranged from 3×10^{-6} (100-K and 100-B/C Areas) to approximately 5×10^{-5} in the 100-D/100-H, 100-F/100-IU-2/100-IU-6, and 300 Areas. The cancer risks of approximately 5×10^{-5} were related to inhalation of hexavalent chromium.

A comparison of nonvolatile exposure concentrations in sweat lodge air modeled with Equation 14 of Harris and Harper (2004) with aerosol measurements from boiling and flashing of aqueous solutions is presented in Appendix D-1. A high-end estimate of respirable aerosol formation from boiling water (DOE-HDBK-3010-94, *Airborne Release Fractions/Rates and*

Respirable Fractions for Nonreactor Nuclear Facilities, Volume 1 – Analysis of Experimental Data) is approximately a factor of 100 lower than the air concentrations calculated using Equation 14 of Harris and Harper (2004). For an experiment that measured aerosols formed by the flashing of superheated (pressurized) water, a high-end estimate is lower than the Harris and Harper (2004) model by approximately a factor of 14 (DOE-HDBK-3010-94). However, the application of these experimental values presumes that all the aerosol created from the 4 L (1.05 gal) of water used in a sweat lodge is available for inhalation during the event. In reality, water must condense over time as new steam and aerosol are created in the sweat lodge. These experimental factors are, therefore, likely to overestimate the time-averaged quantity of respirable aerosols available in the sweat lodge. This analysis indicates that the exposure to nonvolatile COPCs based on Equation 14 of Harris and Harper (2004) may be greatly overestimated.

4.7.3 Uncertainty in the Toxicity Assessment

4.7.3.1 Updating of Toxicity Criteria. Toxicity criteria for chemical COPCs are continually revised by EPA and state agencies. Therefore, risk assessment calculations based on these criteria may become outdated over time as our understanding of chemical toxicity improves. The chemical toxicity criteria for this risk assessment were retrieved in Spring and Summer 2009. As this report was finalized (winter 2009/2010), toxicity criteria published by EPA in December 2009 in support of their Regional Screening Levels were reviewed (http://epa-prgs.ornl.gov/radionuclides/) to identify newly-published values. For the broad-area COPCs listed in Tables 4-7, 4-8, and 4-9, revised toxicity criteria were only discovered for hexavalent chromium and cadmium. The cadmium inhalation unit risk PPRTV has changed from 0.0028 per $\mu g/m^3$ to 0.009 per $\mu g/m^3$. Because cadmium risk via inhalation exposure is a negligible contributor to calculated risks in the broad-area risk assessment, this threefold change in the PPRTV will not affect the results of this assessment.

The oral CSF newly published in the December 2009 Regional Screening Levels for hexavalent chromium could potentially have an impact on the results of the broad-area risk assessment. Hexavalent chromium is a COPC in the upland, riparian, and near-shore environments. The largest impact is likely to be in the results of the groundwater risk assessment (Section 6.0) because hexavalent chromium is a major groundwater plume contaminant. However, chromium has also been measured in seeps on the river shoreline as discussed in Section 4.5. An oral CSF for hexavalent chromium has been independently derived by the states of New Jersey and California in 2009 based on the results of a 2-yr rodent study performed by the National Toxicology Program, which is an interagency program within the Public Health Service of the Department of Health and Human Services. Only inhalation exposure was historically considered to be important for hexavalent chromium carcinogenicity, due to reduction of chromium from the hexavalent to trivalent state in the lungs.

4.7.3.2 Cancer Risk and Radiation Dose. General sources of uncertainty pertaining to the assessment of carcinogenicity include (1) high-to-low dose extrapolation, (2) uncertainty in the applicability of the no-threshold model of carcinogenicity, (3) the common use of a UCL (typically 95%) on the slope of the dose-response curve for chemical CSFs, and (4) uncertainty in whether a particular chemical is in fact a human carcinogen. As discussed in Section 3.5.3, most chemical CSFs are based on carcinogenic effects observed at relatively high dose rates in test animals that have been extrapolated to lower dose rates in humans. The human data on which radionuclide CSFs are based are also based on high-to-low-dose extrapolation. The underlying assumption for both chemicals and radionuclides is that even a very low level of exposure carries some risk of carcinogenesis.

Uncertainties in extrapolating carcinogenic response from high-to-low dose rates and in the no-threshold model of carcinogenicity are related. The human body has the capability to repair DNA damage that may lead to cancer, and such repair is constantly under way due to damage that is naturally incurred independent of any exposure to environmental contaminants. There are also a variety of cellular mechanisms for halting the proliferation of cells that, due to DNA damage, have developed characteristics of uncontrolled growth. At the low dose rates common to environmental exposures, it becomes impossible in either human or test animal populations to observe the slight potential increase in cancer incidence that may be related to exposure. As discussed in Section 3.5.3, there are some chemical carcinogens that do not directly act on DNA for which the no-threshold cancer model is generally understood to be inaccurate. At low dose rates, it is unlikely that a carcinogenic response would be invoked by such chemicals. Scientific positions on the validity of calculating cancer risks at low levels of exposure vary.

Cancer risks for each exposure route and COPC (radionuclides and chemicals) are summed to calculate cancer risk to an individual. This methodology is consistent with CERCLA guidance in "Establishment of Cleanup Levels for CERCLA Sites with Radioactive Contamination" (OSWER No. 9200.4-18). The EPA reiterated their recommendation for summing chemical and radionuclide cancer risks in a 1999 memorandum titled "Distribution of OSWER Radiation Risk Assessment Q&A's Final Guidance" (EPA 1999). Separate sums of cancer risk for radionuclides and chemicals are provided in the electronic results files in Appendix D-5, but summary tables in the main report provide cancer risk results for the sum of all COPCs.

An important distinction between chemical and radionuclide CSFs is that chemical CSFs are commonly calculated as the 95% UCL on the slope of the dose-response curve, and radionuclide CSFs reflect an average estimate of the lifetime risk of cancer. Although chemical and radionuclide cancer risk estimates have been summed in this assessment, the intentional bias associated with chemical CSFs does not strictly allow for simple summation of chemical and radionuclide cancer risks. Additionally, many chemical CSFs are based on animal studies and therefore incorporate uncertainties that do not pertain to radionuclide CSFs, which are based on human epidemiological studies. Ultimately, chemical and radiological CSFs both involve a similarly high degree of uncertainty related to extrapolation of effects at high doses to hypothetical effects at low (environmental) dose rates. Also, EPA is moving towards a mode-of-action approach for evaluating chemical carcinogenicity that stresses whether a chemical is carcinogenic by mutagenic or nonmutagenic processes (EPA/630/P-03/001F, *Guidelines for Carcinogen Risk Assessment*). From that perspective, additivity of cancer risk for mutagenic chemicals and radionuclides may be more defensible in principle than additivity of risk among chemicals that operate by mutagenic and nonmutagenic modes of action.

There are numerous instances where chemicals and radionuclides both contribute significantly to a cancer risk result. The most common occurrence is the summing of cancer risks for arsenic and radionuclides. Although the different bases of the chemical and radionuclide slope factors makes the general summation of these cancer risks uncertain, arsenic is (like ionizing radiation) a known human carcinogen and has a slope factor based on human epidemiological data. Therefore, uncertainty introduced by the addition of chemical (arsenic) and radionuclide cancer risks is not as large in this assessment as might more generally be the case for other chemicals.

As discussed in Section 3.5.3, *Supplemental Guidance for Assessing Susceptibility from Early-Life Exposure to Carcinogens* (EPA/630/R-03/003F) contains recently published guidance for adjusting cancer potency estimates for childhood exposures to mutagenic carcinogens. Among the 12 chemicals listed in Table 1b of EPA/630/R-03/003F that EPA calls out as having been identified as mutagens, the four PAHs (benzo(a)pyrene, dibenzanthracene, dimethylbenz(a)anthracene, and 3-methylcholanthrene) are of particular relevance in this assessment because carcinogenic PAHs are among the COPCs addressed in the HHRA. A quantitative adjustment of the CSFs for carcinogenic PAHs was performed for child receptors in this assessment. The adjustment factors for CSFs described in EPA/630/R-03/003F are 10 for ages 0 through 2 years, and 3 for ages 2 through 16 years. Over a 30-year residential RME exposure duration, these changes amount to an increase in estimated lifetime cancer risk of approximately 2.5 times.

EPA guidance related to childhood exposure to mutagens is contained in a companion document to *Guidelines for Carcinogen Risk Assessment* (EPA/630/P-03/001F). In these guidelines, EPA distinguishes between mutagenic and nonmutagenic carcinogens. Although CSFs for both types of carcinogens are currently developed using a no-threshold model of dose-response (see Section 3.5.3), EPA/630/P-03/001F stresses the importance of differentiating these mechanistically-different types of carcinogens because there is likely to be a threshold dose for the latter. Hence, a complete implementation of EPA's 2005 cancer risk assessment guidelines may have the effect of reducing the estimated cancer incidence risk for chemicals with laboratory evidence of carcinogenicity that are not mutagens.

As described in Section 3.5.5, age-dependent internal dose coefficients published in *Age-Dependent Doses to Members of the Public from Intake of Radionuclides: Part 5, Compilation of Ingestion and Inhalation Dose Coefficients* (ICRP 1996) were used for the radiation dose calculations. For the Residential exposure scenarios that include both child and adult receptors, age-dependent dose coefficients that apply to a 30-year exposure period were calculated because the dose threshold (15 mrem/yr) used in the local-area risk assessment also pertains to a 30-year period. However, when calculating radiation dose for a child receptor of ages 1 to 6 years, these DCFs will tend to underestimate the actual dose, because effective dose is usually lower for an adult than a child. A discussion of the calculation of age-dependent DCFs, including a comparison of DCFs for specific age classes to the 30-year exposure DCFs used in this assessment, is provided in Appendix D-3.

4.7.3.3 Systemic Toxicity. General sources of uncertainty pertaining to assessment of systemic toxicity include (1) the application of uncertainty factors on the dose-response data, (2) relying on a single critical effect to measure toxicity, and (3) toxicological interactions among the various COPCs. Uncertainty factors are used to account for several possible sources of uncertainty in developing an RfD, including extrapolating from the NOAEL or LOAEL to a chronic RfD, variability in sensitivity in the human population, interspecies variability between humans and test animals, and inadequate dosing periods in a critical study. These uncertainty factors (and related modifying factors) are designed to introduce a protective bias in the toxicity criteria such that the potential for adverse effects in sensitive human subpopulations will not be underestimated. There is also considerable uncertainty, and usually a protective bias, in the screening-level practice adopted in this assessment of summing HQs across chemicals to estimate an HI.

Some of the metal COPCs for which chemical hazard is quantified in the risk assessment (i.e., copper, total chromium, and zinc) are also essential micronutrients required by the body for normal functioning. The body normally exerts a degree of homeostatic control over the body burdens of these metals following ingestion exposures, meaning that uptake is regulated depending on need. In order for uptake followed by toxicity to occur, the body’s control mechanisms on uptake must be overwhelmed or incapacitated. Thus, chronic toxicity expressed as HQs below or only a few times higher than 1.0 for exposure to these metals is not realistic. Specific examples are discussed in relation to the risk assessment results presented in Sections 4.3 and 4.4.

As described in Section 3.4.4, each chemical RfD value has associated uncertainty and modifying factors, used as protective multipliers on the observed dose-response data to account for various uncertainties when applying these data to the human population. The extent to which uncertainty and modifying factors in the chemical RfD values impact the results of the risk assessment is chemical specific. Significant contributions to a chemical hazard above 1.0 was calculated for arsenic and cadmium in the Nonresident Tribal scenario as well as for copper, nickel, and selenium in sculpin in the Avid Angler scenario. Uncertainty and modifying factors in oral RfDs for these chemicals are as follows:

COPC	Uncertainty Factor	Modifying Factor
Arsenic	3	1
Cadmium	10	1
Copper	1	1
Nickel	300	1
Selenium	3	1
Zinc	3	1

Calculated HQ values for nickel are, therefore, associated with a much higher degree of protective bias than those for the remaining metals. The uncertainty factor for nickel is composed of individual factors of 10 for interspecies extrapolation from rodent data to humans, 10 for protection of sensitive subpopulations, and 3 to account for inadequacies in the characterization of reproductive effects (IRIS 2009).

In this assessment, HQs for individual COPCs were summed to generate an HI for all chemicals (see Section 3.6.2). In the Nonresident Tribal scenario where HI results were greatest, the contribution of arsenic overwhelmed that of cadmium. Therefore, the summation of chemical-specific HQ values for chemicals whose toxic effects are manifested in different organ systems has not contributed to an appreciable protective bias in the risk assessment results.

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5.0 LOCAL-AREA RISK ASSESSMENT RESULTS

5.1 INTRODUCTION

The Human Health Risk Assessment (HHRA) component of the RCBRA has been conducted in three parts: broad-area risk, local-area risk, and groundwater risk. This part presents the results of the HHRA calculations using "local-area" environmental data that characterize contaminant of potential concern (COPC) concentrations in shallow zone soil (0 to 4.6 m [0 to 15 ft] depth) in the individual remediated waste sites. These data sets are listed under the spatial-scale categories of "Local-area" in Table 3-9. Results for risk assessment scenarios that employ the local-area soil data are provided in this section of the report. "Broad-area" risk assessment results related to exposures occurring across one or more record of decision (ROD) decision areas were presented in Section 4.0. Risks related to groundwater exposure are presented in Section 6.0. The scope of the HHRA, as it relates to exposure scenarios, pathways, and media is described in Section 3.3 and outlined in Figures 3-15 through 3-17. Potentially complete exposure pathways for scenarios evaluated using the local-area soil data are outlined in detail in Tables 3-12 and 3-13.

As discussed in Section 1.0 of this report, there are four steps in the baseline HHRA process. These include Data Collection and Evaluation, Exposure Assessment, Toxicity Assessment, and Risk Characterization. Inputs and methodology related to Data Collection and Evaluation, Exposure Assessment, and Toxicity Assessment are the subject of Section 3.0 and supporting appendices. The results sections of the HHRA (Sections 4.0, 5.0, and 6.0) correspond to the Risk Characterization. The results of the implementation of the methods for identification of COPCs, which are an aspect of Data Collection and Evaluation, are also presented in Sections 4.0, 5.0, and 6.0 of the HHRA.

Section 1.0 also lists five questions that the RCBRA is designed to address to provide information needed by risk managers to support final *Comprehensive Environmental Response, Compensation, and Liability Act of 1980* (CERCLA) decisions in the River Corridor that ensure protection of human health and the environment. Two of those questions are addressed directly in the results of the local-area risk assessment in Section 5.0:

- Are residual conditions for cleanup actions under the interim action records of decision (IARODs) protective of human health and the environment?
- What are the uncertainties associated with the risk results and conclusions?

The Industrial/Commercial scenario and the Resident National Monument/Refuge scenario are the two occupational scenarios evaluated in the local-area risk assessment. The potentially exposed population for these scenarios is limited to adult workers. The main distinction between these scenarios is that the Industrial/Commercial receptor is assumed to work at a building located on a remediated waste site, but resides off site. The National Monument workers are assumed to be exposed across the entire upland environment as they lead tours, conduct ecological education, or engage in similar activities. When not working, these receptors are envisioned to live in an onsite residence at the location of a remediated waste site. Potentially complete exposure pathways to contaminants in surface soil for both scenarios include external irradiation, incidental soil ingestion, dermal absorption, and inhalation.

All three of the Residential scenarios describe exposures related to a rural land-use pattern that involves home-produced foods. The Subsistence Farmer scenario envisions a substantial quantity of home-produced foods, but not a diet composed solely of such foods. The Native American Resident scenarios, however, envision a complete subsistence lifestyle where all foods are grown at the home or (in the case of fish) caught in the Columbia River. Exposure parameter values for the Confederated Tribes of the Umatilla Indian Reservation (CTUIR) Resident and Yakama Resident scenarios were provided by the Tribes, as documented in *Exposure Scenario for the CTIUR Traditional Subsistence Lifeways* (Harris and Harper 2004) and *Yakama Nation Exposure Scenario for Hanford Site Risk Assessment* (Ridolfi 2007). Residential receptors are assumed to spend effectively all of their time in the area around a residence located on a remediated waste site in order to protectively assign all soil-related exposures to that site.

Risk assessment calculations for the Industrial/Commercial scenario, the residential portion of the Resident National Monument/Refuge scenario, and the three Residential scenarios (Subsistence Farmer, CTUIR Resident, and Yakama Resident) are performed for each individual remediated waste site using the shallow zone soil cleanup verification data. As discussed in Section 3.3, soil exposures for these scenarios occur primarily in the vicinity of a residence or commercial building and may involve localized exposure to residual contamination related to an individual waste site. In the special case of the Resident National Monument/Refuge scenario, the residential portion of the scenario employs the local-area data for an individual waste site, whereas the occupational portion of the scenario employs broad-area upland surface soil data. The risk assessment calculations pertaining to the individual remediated waste sites are organized by the six ROD decision areas within the River Corridor. The specific waste sites where risks are quantified in the HHRA are those for which remedial activities, if necessary, have been completed and for which soil verification data exist through 2005. These waste sites, organized according to ROD decision area, are listed in Table 5-1.

There are a total of 156 remediated waste sites across all ROD decision areas for which risk calculations are performed in the RCBRA for the various exposure scenarios. Waste site soil data are available for a total of 164 sites, as shown in Table 5-1. However, only deep zone soil data exist for six waste sites (100-B-14:3, 100-C-9:3, 100-F-4, 116-F-7, 116-F-11, and the 300 Ash Pits). These deep zone data will be included in an evaluation of potential groundwater impacts from soil leaching in the remedial investigation (RI) reports. For the 168-K-1 and 1607-H4 waste sites, no COPCs were detected in shallow zone soil. The distribution of individual waste sites within ROD decision areas for which shallow zone soil data are available is shown below.

ROD Decision Area	Number of Waste Sites
100-B/C	45
100-K	13
100-N	2
100-D and 100-H	35
100-F, 100-IU-2, and 100-IU-6	44
300	17

5.1.1 Summary of Local-Area Risk Assessment Results

A summary of the results of the local-area risk assessment for all ROD decision areas and exposure is shown in Tables 5-2 through 5-7. The primary purpose of these summary tables is to allow the reader to quickly see how the remediated waste site risk assessment results for the different exposure scenarios compare to the thresholds used for risk management decisions. These tables also provide a perspective of how the waste site risk assessment results vary among the six ROD decision areas, and how cancer risk and radiation dose will change over time due to radioactive decay.

A summary of the results of the local-area risk assessment for all ROD decision areas for the Occupational scenarios is provided in Tables 5-2, 5-3, and 5-4. Table 5-2 shows results using present-day radionuclide soil concentrations. Tables 5-3 and 5-4 show the results with radionuclide concentrations at years 2075 and 2150. The results of the local-area risk assessment for the Industrial/Commercial scenario indicate that present-day reasonable maximum exposure (RME) cancer risk is very rarely above 1×10^{-4} (4 of 156 sites), and that present-day RME radiation dose is always below 15 mrem/yr. There are no remediated waste sites where Industrial/Commercial chemical hazard index (HI) results exceed the threshold of 1.0. Because the highest Industrial/Commercial cancer risk and dose results are related to external irradiation from short-lived radionuclides, including cesium-137, cobalt-60, and europium-152, these results decrease with time. There are no sites where the Industrial/Commercial cancer risk is above 1×10^{-4} at year 2075. The number of sites where risks are within the 1×10^{-6} to 1×10^{-4} risk management range also decreases with time at most of the ROD decision areas (the 300 Area, where the highest risks are primarily due to external irradiation from long-lived uranium isotopes, is an exception). By the year 2150, Industrial/Commercial RME cancer risks at or above 1×10^{-5} are limited to five remediated waste sites in the 300 Area (316-5, 316-2, 316-1, 300-50, and 618-12).

The results of the local-area risk assessment for the Resident National Monument/Refuge scenario indicate that present-day RME cancer risk and radiation dose are rarely above 1×10^{-4} (11 of 156 sites) and 15 mrem/yr (3 of 156 sites), respectively. There are no remediated waste sites where Resident National Monument/Refuge chemical HI results exceed the threshold of 1.0. Like the Industrial/Commercial scenario, the highest cancer risk and dose results are related to short-lived radionuclides (except in the 300 Area) and therefore decrease with time. By the year 2150, Resident National Monument/Refuge RME cancer risks at or above 1×10^{-5} are limited to the five remediated waste sites described for the Industrial/Commercial scenario (316-5, 316-2, 316-1, 300-50, and 618-12). Risks for the Resident National Monument/Refuge scenario at these five sites are primarily due to external irradiation from long-lived uranium isotopes.

A summary of the results of the local-area risk assessment for all ROD decision areas for the Residential scenarios is provided in Tables 5-5, 5-6, and 5-7. Table 5-5 shows results using present-day radionuclide soil concentrations. Tables 5-6 and 5-7 show the results with radionuclide concentrations at years 2075 and 2150. The results of the local-area risk assessment for the Residential scenarios indicate that present-day RME cancer risk is frequently greater than 1×10^{-4} and that RME chemical HI frequently exceeds the threshold of 1.0. Present-day RME radiation dose results are below the threshold of 15 mrem/yr at most remediated waste sites. For sites where RME radiation dose exceeds 15 mrem/yr, the pattern is similar to that described for the highest cancer risks under the Occupational scenarios. Short-lived radionuclides contribute most to present-day radiation dose above 15 mrem/yr, except in the 300 Area where uranium isotopes are most important.

There are 19 remediated waste sites where present-day RME Subsistence Farmer radiation dose exceeds 15 mrem/yr, but only 2 sites (316-5 and 316-2) where Subsistence Farmer radiation dose exceeds 15 mrem/yr at year 2075. An important difference between the Occupational and Subsistence Farmer radiation dose results above 15 mrem/yr is that food ingestion exposure pathways frequently dominate the dose results for the Subsistence Farmer.

Present-day RME cancer risks greater than 1×10^{-4} for the Subsistence Farmer exposure scenario are almost entirely related to one of three factors:

1. External irradiation from short-lived radionuclides including europium-152, cesium-137, and cobalt-60
2. Exposure to arsenic from ingestion of garden produce
3. Exposure to the short-lived radionuclide strontium-90 from ingestion of produce and livestock products.

By the year 2075, Subsistence Farmer RME cancer risks above 1×10^{-4} are related overwhelmingly to arsenic exposure from produce ingestion. The only exceptions are the 116-B-6A and 116-DR-9 waste sites, where concentrations of cesium-137 and strontium-90 are still high enough at year 2075 to dominate cancer risks. Because the CTUIR Resident and Yakama Resident scenarios use very high (subsistence level) site-raised food ingestion rates, strontium-90 still plays a significant role in food-related exposures at year 2075 for these scenarios. By year 2150, however, CTUIR Resident and Yakama Resident cancer risks above 1×10^{-4} are dominated by arsenic exposure from ingestion of garden produce.

Present-day cancer risk results for the RME Subsistence Farmer (5×10^{-4}), CTUIR Resident (8×10^{-3}), and Yakama Resident (8×10^{-3}) were well above 1×10^{-4} for COPC concentrations in the upland reference area. Exposure to arsenic from produce ingestion was the main contributor to each of these background cancer risk levels. Because arsenic is the primary contributor to reference area cancer risk for the three Residential scenarios, these results do not vary for years 2075 and 2150.

Chemical HI results for children for the RME Subsistence Farmer (4.8), CTUIR Resident (42), and Yakama Resident (27) scenarios were also well above 1.0 for COPC concentrations in the upland reference area. Exposure to arsenic from produce ingestion was the main contributor to these reference HI levels. There were 15 remediated waste sites where Subsistence Farmer RME child HI results were more than twice the reference level. Although arsenic was also the main risk driver at some of these 15 sites (100-F-37, 100-H-21, 300-10, 618-12, and 300-50), the results at sites where Subsistence Farmer RME child HI was highest were related to food ingestion exposure pathways for other metals including mercury (100-K-30, 1607-H2, 100-K-31, 100-K-33, 100-K-32, 100-B-14:6, 116-B-10), uranium (316-5 and 316-2), and copper (316-1).

5.1.1.1 Relationship of RCBRA Results to IAROD Cleanup Actions. Waste sites in the River Corridor were interim closed using remedial action goals (RAGs) related to direct contact of soil and protection of groundwater from contaminants leaching from soil. Remedial action goals for direct contact in the 100 Area of the River Corridor were based on a Residential exposure scenario. Interim closure of sites in the 300 Area was based on either a Residential or Industrial land use scenario.

As summarized above, the RCBRA Industrial/Commercial scenario results show that radiation dose and chemical hazard were below threshold criteria at all remediated waste sites. However, present-day RME Industrial/Commercial cancer risk results were sometimes within (or in a few cases, above) the upper portion of the 1×10^{-6} to 1×10^{-4} risk management range. The RCBRA Subsistence Farmer cancer risk and chemical HI results were frequently above threshold criteria. There are two major differences between the risk assessment methods used in the RCBRA and the basis of the IAROD residential cleanup levels. These differences largely explain why some waste sites remediated to meet the IAROD residential cleanup levels still appear to present high levels of residual risk under the Subsistence Farmer scenario:

1. Residential IAROD cleanup levels for chemicals are the "Model Toxics Control Act – Cleanup" (MTCA) Method B Soil Cleanup Levels for Unrestricted Land Use (WAC 173-340-740), which are based on incidental soil ingestion and do not address the food exposure pathways evaluated for radionuclides.

2. Residential IAROD cleanup levels for radionuclides are based on a 15 mrem/yr dose threshold, not on a cancer risk threshold. Although it varies by radionuclide and pathway, a 15 mrem/yr radiation dose will generally correspond to a cancer risk of approximately 3×10^{-4} (as discussed in EPA 1997, *Establishment of Cleanup Levels for CERCLA Sites with Radioactive Contamination*).

3. The IAROD cleanup level for arsenic is 20 mg/kg, which is an “adjusted” value established by the State of Washington to address a range of natural background levels (http://apps.leg.wa.gov/WAC/default.aspx?cite=173-340-900).

At any particular waste site, other methodological differences may also contribute to a discrepancy between IAROD and RCBRA results. For example, representative concentrations in soil were calculated differently in the RCBRA than they were during the cleanup verification process.

5.1.1.2 Key Uncertainties. There are two key uncertainties related to the site-specific results of the local-area risk assessment. The first of these is how applicable the cleanup verification soil data at excavated and backfilled sites are for future human exposure. Residual contamination at backfilled sites is characterized by verification soil samples collected on the sidewalls and (if the excavation depth is <4.6 m) bottom of an excavation before the site is backfilled. There may be a strong conservative bias when using cleanup verification data from excavated sites to estimate exposure concentrations in surface soil after the sites have been backfilled. This is of particular significance because the great majority of remediated waste sites with the highest levels of residual risk are sites that have been excavated and backfilled.

The second key uncertainty related to the site-specific results relates to modeled exposure concentrations in foods, particularly garden produce. In the case of the produce ingestion chemical HI results for mercury, uranium, and copper, a large conservative bias is anticipated because a linear plant uptake model was applied to soil concentrations that are far above naturally occurring levels. For arsenic, where the range of site soil concentrations is relatively small, uncertainty in produce concentrations is attributable to intrinsic variability related to soil conditions, plant species and tissue type, harvest time, and other variables. A review of recommended plant-soil ratios from a number of sources, described in Section 5.9, shows that the range of ratios for arsenic (from 0.006 to 1.125) is approximately a factor of 200. The value of 0.53 used in the HHRA, from the RESidual RADioactivity (RESRAD) computer code that has been used to perform dose assessment at the Hanford Site and other DOE facilities, is near the upper end of this range.

The remainder of Section 5.0 is organized as follows: The data sets used in the different components of the local-area risk assessment, results of the COPC selection process, and calculation of representative concentrations are described in Section 5.2. Note that the assumptions and methods employed for defining COPCs are described in Section 3.2.2. Risk assessment results for the two Occupational and three Residential scenarios are presented for each of the six ROD decision areas in Sections 5.3 through 5.8. The uncertainty analysis associated with these results is presented in Section 5.9.

5.2 SITE INVESTIGATION AND DATA ANALYSIS

5.2.1 Local-Area Data Evaluation

As described in Section 3.2, cleanup verification soil samples were collected at individual CERCLA waste sites within the River Corridor to document completion of remedial actions according to the IARODs. Most of the shallow zone verification samples were collected by combining numerous individual sub-samples into a single composite sample that was submitted for laboratory analysis. These are referred to as "statistical" samples because they were intended to be used to produce statistical estimates of average soil concentrations at the waste site. The other type of verification sample is referred to as a "focused" sample. The focused samples are individual samples collected from a single location. The locations of focused samples were usually selected based on where contamination was either evident or expected, or to document the effectiveness of a targeted soil removal. Analyses to examine the influence of variability between statistical and focused sampling results are described in Section 5.2.3.

The cleanup verification soil data are distinguished by whether they were collected in shallow zone soil (0 to 4.6 m [0 to 15 ft] below ground surface [bgs]) or deep zone soil (greater than 4.6 m [15 ft] bgs). Potential human exposure to contaminants in deep zone soil is limited to possible diffusion of volatile organic compounds (VOCs) to the ground surface and leaching of soluble contaminants to groundwater (see Section 3.3). The locations of the remediated waste sites evaluated in the RCBRA in each of the six ROD decision areas are shown in Figures 5-1 through 5-8.

In addition to cleanup verification soil data, other data were collected during waste site remediation for the purpose of waste characterization. To ensure that all data represent residual soil concentrations, the cleanup verification data used in the RCBRA have been extensively reviewed so that only data pertaining to residual, post-remediation soil contamination (rather than materials that were removed) are used in the RCBRA. The assignment of shallow zone and deep zone designations was also confirmed by verifying the RCBRA database against sampling records documented in the individual cleanup verification package (CVP) and remaining sites verification package (RSVP) reports.

Occasionally, duplicative reporting of analytical results occurred between the cleanup verification sampling results and results from other data sources used in the RCBRA. In these instances, the cleanup verification data source was identified as the preferred source and duplicate results were appropriately identified in the Guided Interactive Statistical Decision Tools (GiSdT) database to ensure they were not used twice. Issues related to data duplication are discussed in Section 3.2. Additional information on the criteria used to make these determinations and other details on the data importing process is provided in Appendix C-1.

Figure 5-1. Remediated Waste Sites in the 100-B/C Area Included in the RCBRA Human Health Risk Assessment.

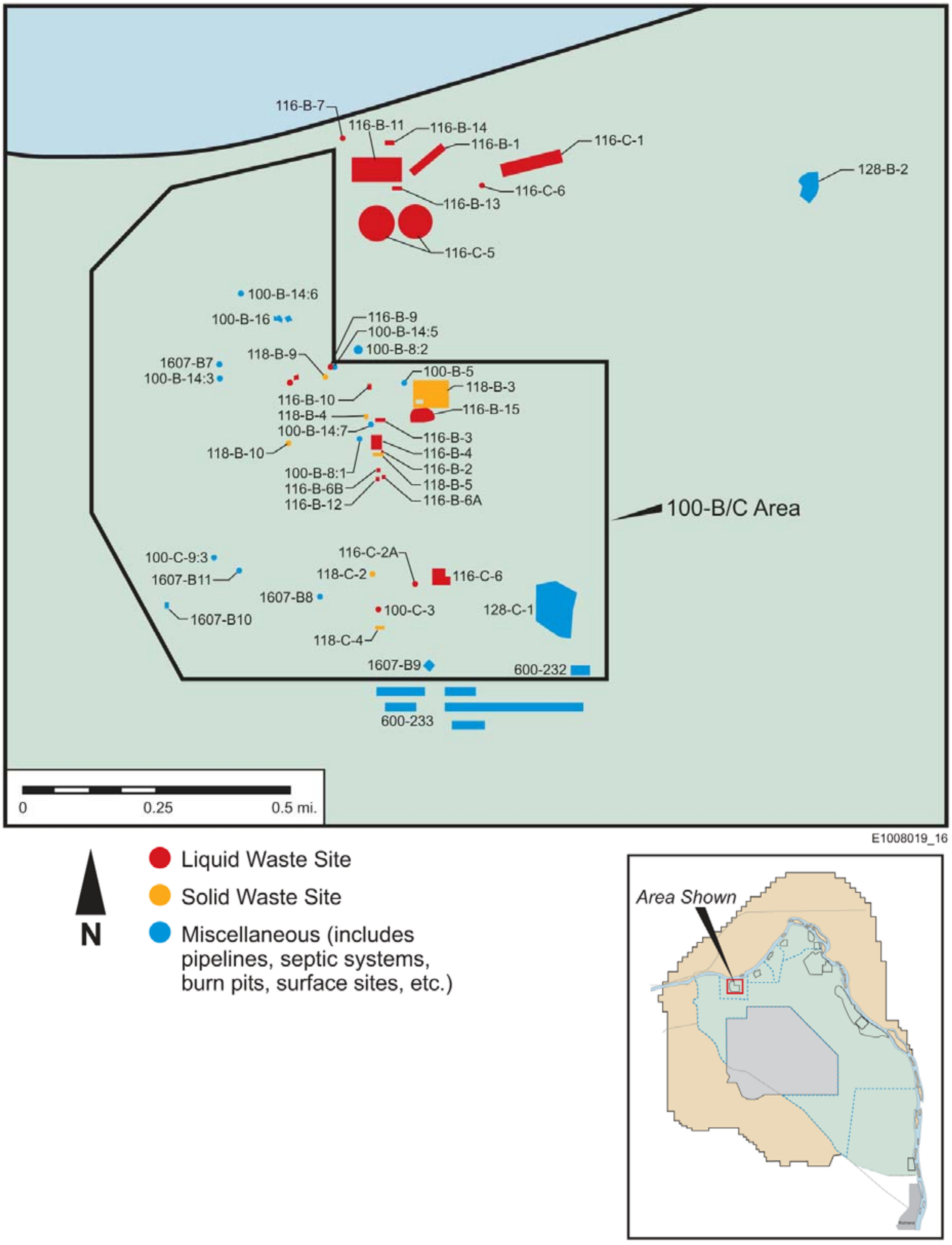

Figure 5-2. Remediated Waste Sites in the 100-K Area Included in the RCBRA Human Health Risk Assessment.

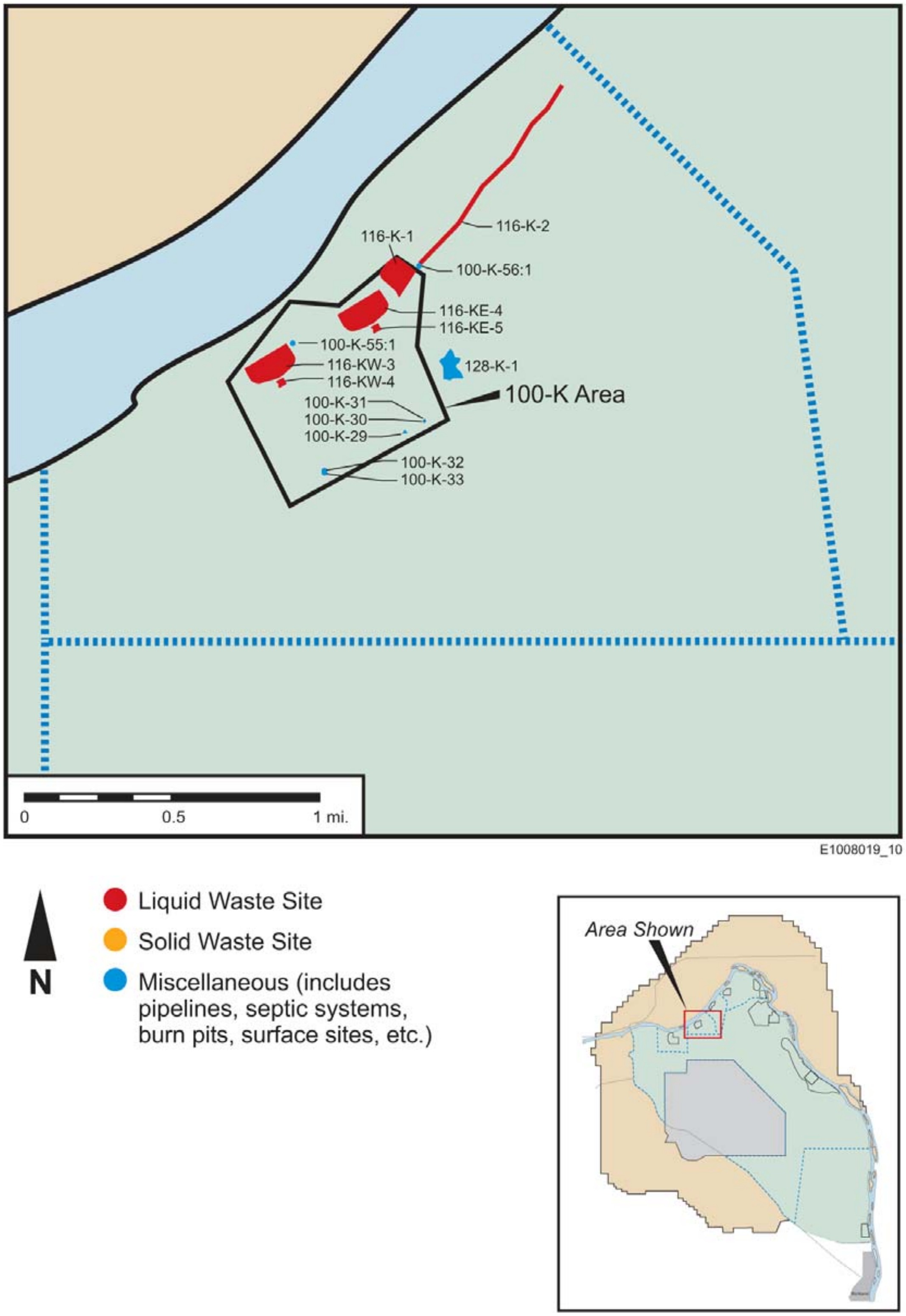

Figure 5-3. Remediated Waste Sites in the 100-N Area Included in the RCBRA Human Health Risk Assessment.

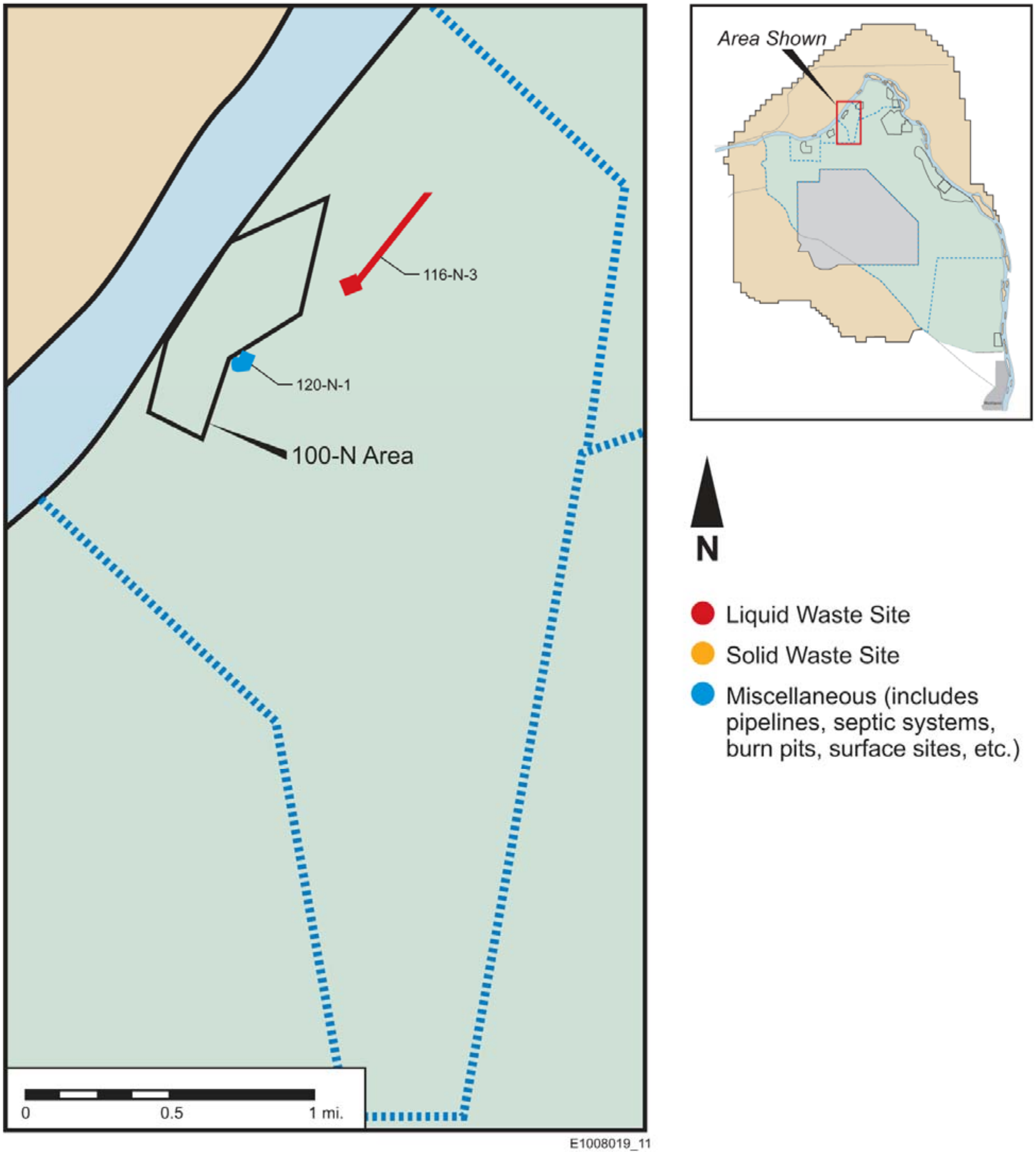

Figure 5-4. Remediated Waste Sites in the 100-D Area Included in the RCBRA Human Health Risk Assessment.

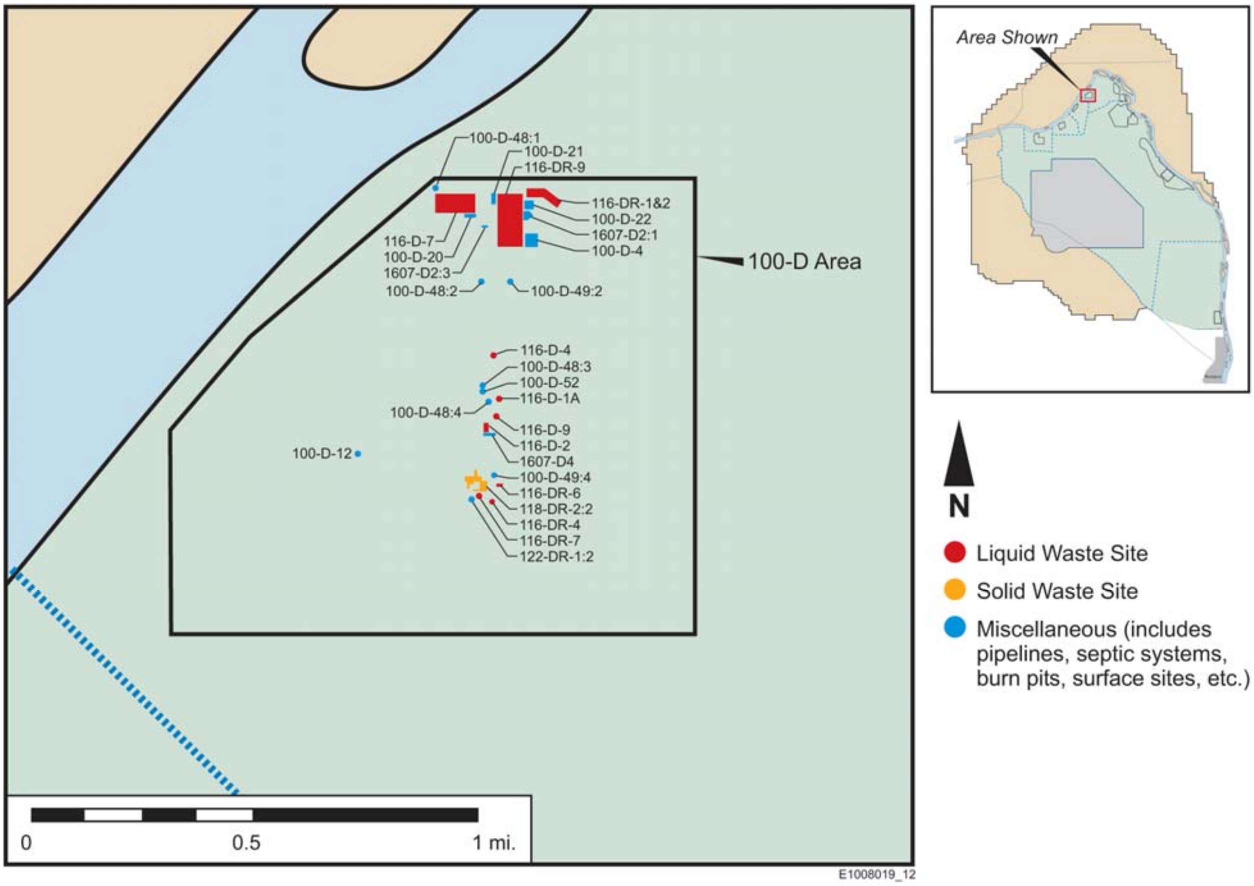

Figure 5-5. Remediated Waste Sites in the 100-H Area Included in the RCBRA Human Health Risk Assessment.

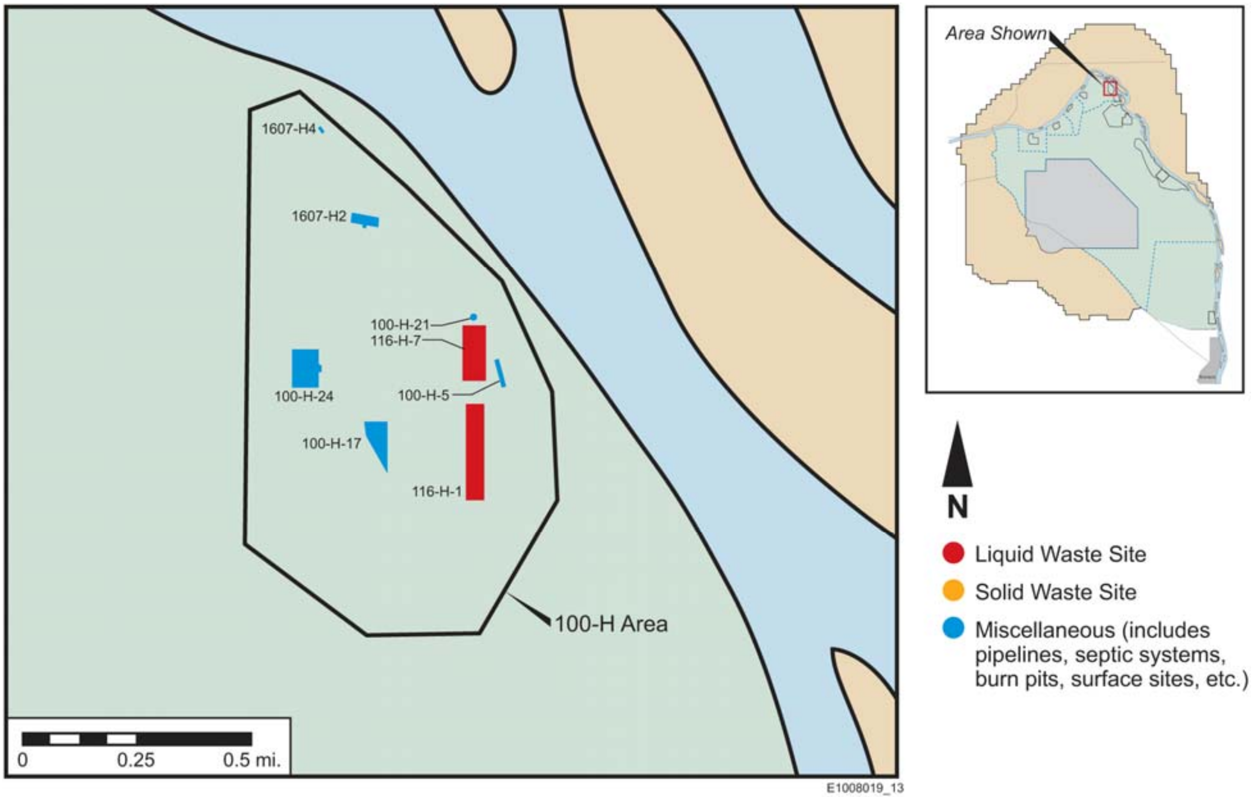

Figure 5-6. Remediated Waste Sites in the 100-F Area Included in the RCBRA Human Health Risk Assessment.

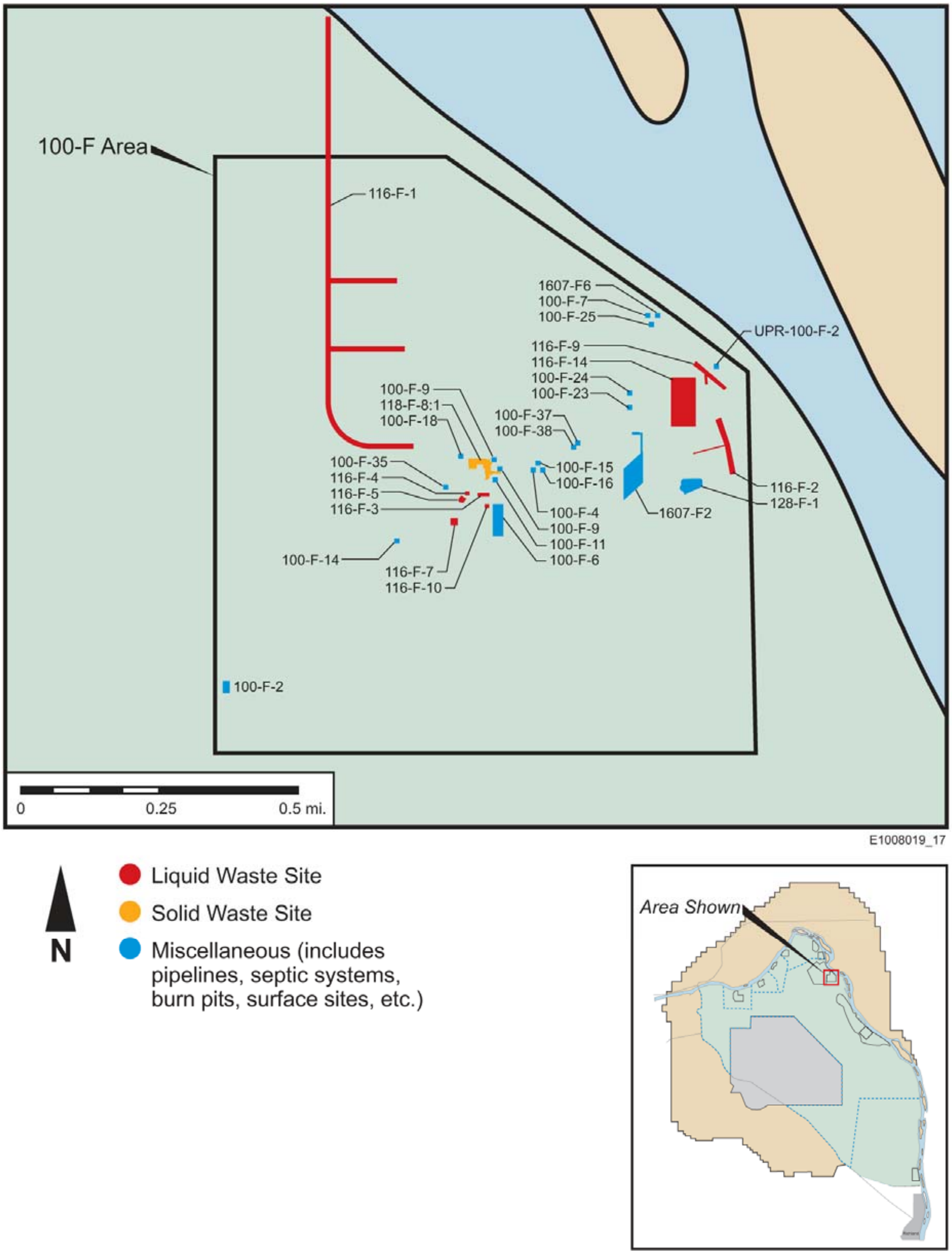

Figure 5-7. Remediated Waste Sites in the 100-F/100-IU-2/100-IU-6 Operable Units Included in the RCBRA Human Health Risk Assessment.

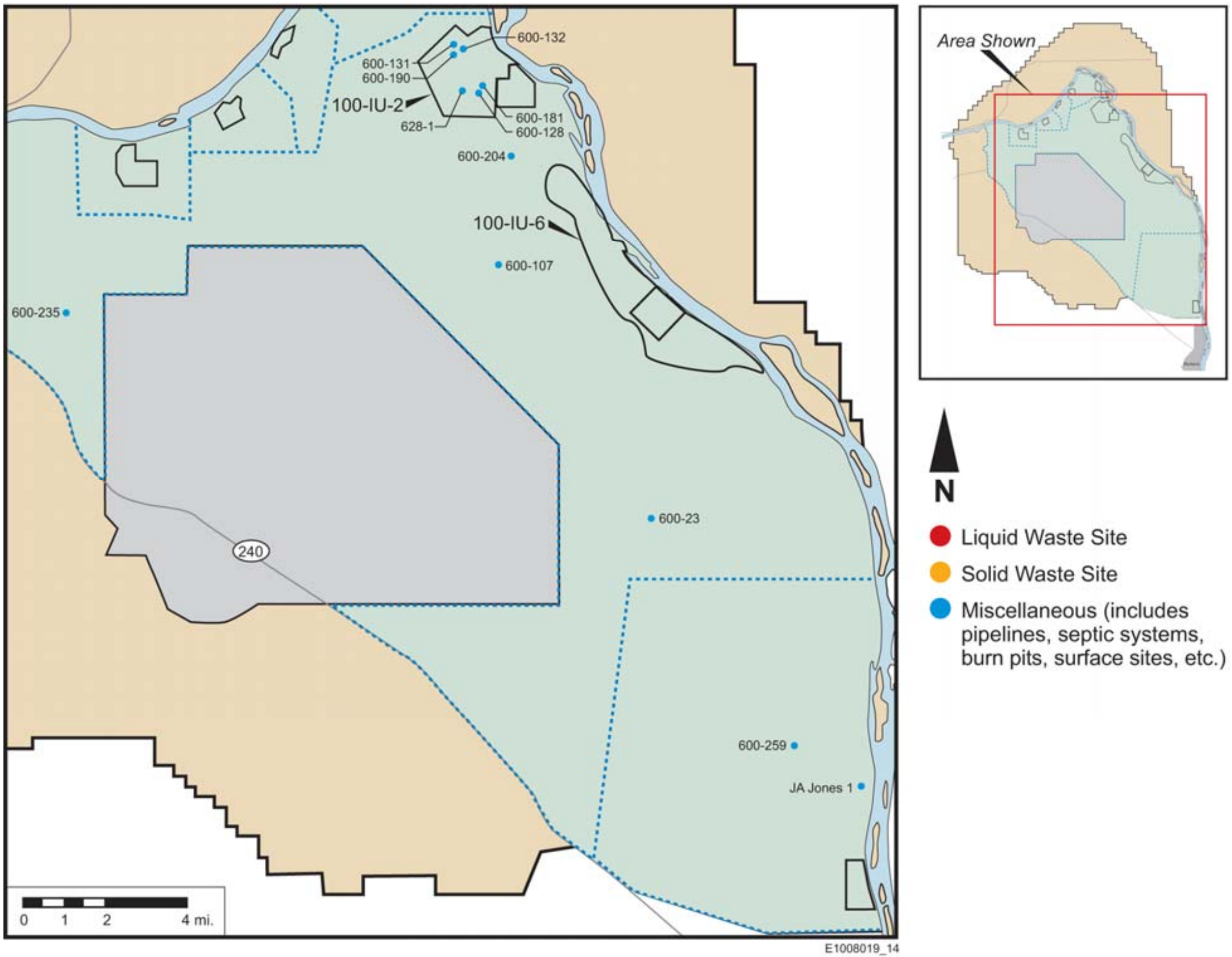

Figure 5-8. Remediated Waste Sites in the 300 Area Included in the RCBRA Human Health Risk Assessment.

The quality assurance (QA) and quality control measures performed in sample collection and laboratory analysis are described in the remedial design reports/remedial action work plans for the 100 and 300 Areas, as well as sampling and analysis plans (SAPs) for particular cleanup verification actions. The cleanup verification data were collected under a QA project plan and were subjected to review of applicable acceptability criteria for completeness, comparability, representativeness, precision, and accuracy. Additional information on QA and data usability is provided in Appendix C-2.

5.2.2 Results of COPC Identification for Local-Area Soil Data in the Six ROD Decision Areas

5.2.2.1 Shallow Zone Soil Data. For soil-mediated exposures that may occur in the area immediately surrounding a home or workplace, the cleanup verification shallow zone soil data are used to identify COPCs. The COPCs are identified independently for each of the six ROD decision areas using the shallow zone cleanup verification data. Data for the individual waste sites are grouped by ROD decision areas for COPC identification primarily because a single list of COPCs is desirable in a ROD decision area and because many of the waste sites in an individual ROD area are related to the same historical operations. For many remediated waste sites, there are also too few verification soil samples to perform statistical comparisons with background and reference area data. The use of statistical and graphical methods to identify outlying values ensures that analytes that are elevated at only one or two waste sites are identified as COPCs for the ROD decision area. However, a consequence of identifying COPCs on the scale of a ROD decision area when risks are calculated for each individual remediated waste site is that the influence of site-related contamination in an individual site may be masked by other COPCs not elevated above background levels at that location. For example, arsenic was identified as a COPC in the 100-D/100-H Area, but was not present at elevated levels in every individual remediated waste site. Because arsenic can contribute to significant calculated cancer risk and chemical hazard, even when it is present at background concentrations, risks related to COPCs that are elevated in soils at an individual site may be masked by the contribution of arsenic at concentrations that approximate background.

As discussed in Section 3.2, a set of four statistical tests were employed to identify analytes with soil concentrations that were clearly consistent with reference area and (when available) Hanford Area background data. However, in situations where only a few samples were obtained, it is possible that significant p-values (less than 0.05) may not be calculated with the slippage test when a site sample result exceeds the maximum of the reference area or background data. This may occur when the number of reference area or background samples is also relatively small. Also, data review may indicate that concentrations at a specific waste site are elevated even when the concentrations from all waste sites in an ROD decision area are not different than reference or background. For this reason, all results where statistical tests indicated an analyte was not a COPC were reviewed. Analytes that statistical tests indicated were not COPCs that were reclassified as COPCs based on data review include strontium-90 (100-B/C, 100-D/100-H, and 100-F/100-IU-2/100-IU-6), mercury (100-D/100-H and 100-F/100-IU-2/100-IU-6), and cesium-137 (100-F/100-IU-2/100-IU-6).

The number of soil samples collected at the individual remediated waste sites differs, as does the number of remediated waste sites within each of the six ROD decision areas. For this reason the number of sample results used to conduct the COPC refinement varies considerably. The number of remediated waste sites having shallow zone soil data within each ROD decision area is described in Section 5.1.

Tables 5-8 through 5-13 show the number of sample results, detection information, range of detected values, nondetection information, and range of nondetect values for all analytes measured in shallow zone soil samples in each of the six ROD decision areas. The range of shallow zone soil concentrations for detected analytes in each of the six ROD decision areas is also shown in relation to RCBRA upland surface soil, reference area, and background soil data sets in box plots provided in Appendix C-5.0.

Tables 5-14 through 5-19 present the results of statistical comparisons of shallow zone soil data to background and reference site concentrations in each of the six ROD decision areas. These tables present the count of samples, maximum concentration, and the significance (p-value) for up to four statistical tests. Tests with p-values less than 0.05 indicate a statistical difference. If detection frequencies are adequate to permit multiple tests to be run and the results of the p-values are concordant (all less than 0.05 or all greater than 0.05), then the analyte is grouped as a COPC or not a COPC based on these tests. In cases where no tests or only one test could be run, or the p-values are discordant, then the analyte is identified for further evaluation. These analyte-specific evaluations to support the assignment of COPC status in shallow zone soil are provided for each of the six ROD decision areas in Tables 5-20 through 5-25. The COPCs in shallow zone soil for each ROD decision area are summarized in Table 5-26.

5.2.2.2 Deep Zone Soil Data. As described in Section 3.3, potential human exposure to contaminants in deep zone soil (>4.6 m [>15 ft] depth) is limited to two exposure pathways: diffusion of VOCs to the ground surface and leaching of soluble contaminants to groundwater. However, the results of statistical comparisons of deep zone soil concentrations to background and reference area data may be of interest for establishing future data needs. Additional site characterization to describe the vertical concentration profile of COPCs will be proposed in the RI work plans. These results will be used in an assessment of potential impacts due to leaching from soil to groundwater that will be presented in the RI reports.

Tables 5-27 through 5-32 show the number of sample results, detection information, range of detected values, nondetection information, and range of nondetect values for all analytes measured in deep zone soil samples in each of the six ROD decision areas. The range of deep zone soil concentrations for detected analytes in each of the six ROD decision areas is also shown in relation to other soil data sets in Appendix C-5.0.

Tables 5-33 through 5-38 present the results of statistical comparisons of deep zone soil data to background and reference site concentrations in the six ROD decision areas. These tables present the count of samples, minimum and maximum concentration, and the significance (p-value) for up to four statistical tests. Tests with p-values less than 0.05 indicate a statistical difference. If detection frequencies are adequate to permit multiple tests to be run and the results of the p-values are concordant (all <0.05 or all >0.05), then the analyte is grouped as a COPC or not a COPC based on these tests.

The limitation of human exposure to deep zone soil analytes that may potentially migrate by gas-phase diffusion or leaching to groundwater greatly restricts the number of possible deep zone COPCs. Leaching analysis conducted using the RESRAD computer code, and documented in Appendix D-1, indicates that even under conditions of future irrigation an analyte must have a soil-water partition coefficient (K_d) below 10 mL/g to reach groundwater within 1,000 years. Positive COPC status based on statistical tests (Tables 5-33 through 5-38) is then further evaluated based on migration potential. These analyte-specific evaluations to support the assignment of COPC status in deep zone soil are provided for each of the six ROD decision areas in Tables 5-39 through 5-44. The COPCs in deep zone soil for the five ROD decision areas with deep zone data are summarized in Table 5-45.

No volatile analytes were identified as COPCs in deep zone soil. Therefore, the COPCs shown in Table 5-45 were identified based on potential leachability, which is a function of the chemical-specific K_d value. As described above, only COPCs with K_d values equal to or below 10 mL/g reached groundwater within a 1,000-year modeling period from either shallow or deep zone soil even with active irrigation. The subset of COPCs in shallow and/or deep zone soil with K_d values equal to or below 10 mL/g is as follows:

- Antimony (K_d = 1.4 mL/g)
- Arsenic (K_d = 3 mL/g)
- Boron (K_d = 3 mL/g)
- Hexavalent chromium (K_d = 0 mL/g)
- Nitrate (K_d = 0 mL/g)
- Technetium-99 (K_d = 0 mL/g)
- Tritium (K_d = 0 mL/g)
- Uranium metal (K_d = 2 mL/g)
- Uranium-233/234 (K_d = 2 mL/g)
- Uranium-235 (K_d = 2 mL/g)
- Uranium-238 (K_d = 2 mL/g).

Table 5-46 shows the breakdown of shallow zone and deep zone COPCs as a subset of the analytes that may potentially leach to groundwater. The potential for groundwater impacts due to residual concentrations of these COPCs in soil will be further evaluated in the RI reports.

5.2.3 Representative Concentrations for Local-Area Soil Data in the Six ROD Decision Areas

Methods for computing representative concentrations for soil were described in Section 3.4.1. As described in Section 3.4, an exposure point concentration (EPC) is an estimation of the COPC concentration in a given medium likely to be contacted by a receptor over time within the exposure area. Exposure point concentrations are calculated based on representative concentrations. The distinction between EPCs and representative concentrations for the HHRA is that representative concentrations pertain to sampled media, such as soil, whereas EPCs may also include modeled concentrations in other exposure media, such as dust or plants, as well as concentrations at future times.

To support the Uncertainty Analysis in Section 5.9, risk results for the Occupational and Residential scenarios are calculated using both the central tendency exposure (CTE) and RME values for EPCs and exposure parameter values. For the CTUIR Resident and Yakama Resident scenarios, a single set of exposure parameter values were obtained from the scenario reports provided by the tribes (Harris and Harper 2004, Ridolfi 2007). Exposure parameter values for the Occupational and Residential scenarios are summarized in Tables 3-18 and 3-19.

As discussed in Section 3.4, when there are four or more detected sample results, the mean and a 95% upper confidence limit (UCL) on the mean are calculated using statistical methods. These values are used for the CTE and RME representative concentrations, respectively. When the detect sample size n' is four or less, representative concentrations in the HHRA are defined as follows:

- If $n' = 0$; then there is no CTE or RME value
- If $n' = 1$; then the detected result is used for the CTE and RME value
- If $n' = 2$; then the maximum detect is used for the CTE and RME value
- If $n' = 3$ or 4; then the average is used as the CTE and the maximum detect is used as the RME.

As described in Section 3.4.1, regression-on-order statistical methods were used to estimate the mean and standard deviation of data that included nondetect or "censored" values. Representative concentrations calculated by these methods have been supplemented with calculations for censored COPCs based on all sample results. Nondetects were used "as is" and were not replaced for these supplemental calculations, implying the use of a sample-specific reporting limit for chemicals and a method-specific minimum detectable activity for radionuclides.

When there are five or more detected results, RME representative concentrations are based on the middle value of three computed UCLs. Comparisons of the three UCL values computed for the estimation of RME representative concentrations were performed to evaluate the variability between the calculated UCLs. These comparisons determined whether use of the highest, instead of the middle, UCL for the RME representative concentration would have often resulted in greatly different RME representative concentration values. These comparisons were performed for both the broad-area and local-area data sets and are discussed in Section 3.4.1. The results of this supplemental analysis are discussed in the context of the risk assessment results in the Uncertainty Analysis in Section 5.9.

An additional supplemental analysis was conducted specifically for the local-area cleanup verification soil samples. As described above, cleanup verification soil samples were collected as either statistical or focused samples. At some waste sites there are analytical data from both kinds of samples. When both focused and statistical samples exist for an analyte at a waste site, only the statistical samples were used to calculate the representative concentrations. This approach emphasizes the use of data that were collected for the purpose of calculating a statistical estimate of the average concentration at a remediated waste site. To evaluate the uncertainty that selection of focused and/or statistical samples has on the risk assessment results, representative concentrations for these waste sites are also calculated using the combined focused and statistical samples. A comparison of statistical representative concentrations with representative concentrations calculated by combining focused and statistical samples is shown in Table C3-11 in Appendix C, Section C-3, "Representative Concentrations." The results of this supplemental analysis are discussed in the context of the risk assessment results in the Uncertainty Analysis in Section 5.9.

Two COPCs addressed in the risk assessment for which multiple types of data exist are uranium metal and nitrate/nitrite. In addition to assessment of radiological cancer risk for each isotope, uranium as a metal is evaluated in the risk assessment because of its potential toxicity to the kidney. The cleanup verification soil data include few results for inorganic uranium. Results for calculated total uranium, derived from the isotopic uranium results (as activity per mass of soil), are used in the risk calculations in lieu of data for the direct measurement of uranium (as mass of uranium per mass of soil). The single exception is at the 1607-D4 waste site in the 100-D/100-H Decision Area, where inorganic uranium data were obtained, but no isotopic uranium results are available. The cleanup verification data sets include results for nitrate, nitrite, and the sum of both nitrate and nitrite. Because toxicity criteria are provided by the U.S. Environmental Protection Agency (EPA) for nitrite and nitrate individually, these data are used preferentially in the risk calculations. Only at the 116-N-3 waste site are data available for only the sum of nitrate and nitrite; this situation is discussed in Section 5.6.

The tabulated results shown in Sections 5.4 through 5.9 are the sum of the risk values for the individual COPCs from the local-area shallow zone soil data used in the risk calculation. These tabulated results and accompanying text summaries present the risks that have been summed across the individual exposure pathways and COPCs. The principal COPCs and exposure pathways contributing to risks above the threshold criteria for cancer risk, radiation dose, and chemical hazard (see Section 3.6) are discussed in the text of the risk assessment.

Files of the representative concentrations used to calculate EPCs in the various exposure media are provided as an electronic attachment to Appendix C-3. The EPCs and the results of the risk assessment calculations for each individual COPC and exposure pathway contributing to the sums shown in the Section 5.0 tables are provided in electronic format as an attachment to Appendix D-5 of this report.

5.2.4 Contaminant of Potential Concern Concentrations and Risks Related to Background and Reference Areas

The basic protocol for calculating the representative concentrations of COPCs is summarized in Section 5.2.3. This protocol was modified for calculating representative concentrations in soil for upland and riparian reference areas. These soil samples were collected as *MULTI INCREMENT*® sampling (MIS) samples, which are a form of composite sample across approximately a 1-ha area. Five replicate MIS samples were collected from each 1-ha location. In order not to inflate the reference areas sample sizes, the reference area representative concentrations were calculated using the average (CTE) value for each sampling location. Hence, sample size for the upland and riparian reference area calculations is equivalent to the number of sampling locations. The location-averaged CTE and RME representative concentrations for the upland and riparian reference areas were calculated using the identical protocol summarized in Section 5.2.3. Representative concentrations for the reference areas are provided as an electronic attachment to Appendix C-3.

Both CTE and RME representative concentrations for the upland reference areas are shown in Table 5-47. The COPCs include all analytes identified as COPCs in each of the six ROD decision areas. As summarized above, the values were computed using the same protocol as the CTE and RME representative concentrations for shallow zone soil at the remediated waste sites. The majority of the shallow zone soil samples used for the remediated waste site risk calculations are also composite samples, although usually with fewer subsamples. The upland reference area CTE and RME concentrations are primarily intended for use in calculating risks that are comparable to those for the individual remediated waste sites, but representative of areas that were not impacted by Hanford releases. The 90th percentile values for Hanford Area background and Washington State Yakima Basin background are also shown in Table 5-47. These values are commonly used to represent a reasonable upper bound on naturally occurring background soil concentrations. Because these 90th percentile values are based on discrete soil samples collected over a much larger geographic area with different soil types, they tend to be higher than the reference area RME values, which are computed as the 95% UCL on the mean using composite samples with relatively low variance.

Tables 5-48 through 5-50 show COPC-specific reference area cancer risk, radiation dose, and hazard quotient (HQ) for each exposure scenario evaluated in the local-area risk assessment. The values in these tables are referenced in the risk assessment results in Sections 5.3 through 5.8 to provide a context for site-related risks of key COPCs. Because these reference area risks are computed using representative concentrations comparable to those calculated for the remediated waste sites, there is relatively little bias between the site and reference area risk results.

5.3 100-B/C DECISION AREA RESULTS

Section 5.3 is the first of six sections of the local-area risk assessment (5.3 through 5.8) that present the results of the soil risk assessment for remediated waste sites in an individual ROD decision area. A summary of the results of the waste site soils risk assessment for all ROD decision areas is shown in Tables 5-2 through 5-7. Tables 5-2, 5-3, and 5-4 show the number of remediated waste sites with risk assessment results above or below decision thresholds for the Occupational scenarios. Table 5-2 shows results using present-day radionuclide soil concentrations, and the latter two tables show the results with radionuclide concentrations at years 2075 and 2150. Tables 5-5, 5-6, and 5-7 show these same summaries for the Residential exposure scenarios. The primary purpose of these summary tables is to allow the reader to quickly see how the waste sites risk assessment results for the different exposure scenarios compare to the thresholds used for risk management decisions. These tables also provide a perspective of how the waste site risk assessment results vary among the six ROD decision areas, and how cancer risk and radiation dose will change over time due to radioactive decay.

As described in Section 3.3, broad-area EPCs in the upland environment are associated with the occupational portion of the Resident National Monument/Refuge scenario and are also used to supplement the local-area cleanup verification data to calculate EPCs for dust inhalation for all scenarios evaluated in Sections 5.3 through 5.8. The COPCs for the broad-area components of the waste site risk calculations are those related to upland surface soil, while the COPCs for the local-area components of the waste site risk calculations are those related to the cleanup verification data for a particular ROD decision area. Risk assessment results for the Resident National Monument/Refuge scenario are provided for the local-area and broad-area components separately. Broad-area representative concentrations for upland surface soil are calculated using the MIS soil data from all 20 upland sampling locations, using the same protocol described for MIS reference area data in Section 5.2.4.

5.3.1 Industrial/Commercial Scenario Results for the 100-B/C Area

The Industrial/Commercial exposure scenario encompasses potential occupational exposures related to shallow zone soils around remediated waste sites. Like the Resident National Monument/Refuge scenario, it focuses on direct contact exposure pathways and does not encompass the raising of produce or animals, nor the ingestion of Columbia River fish. It is similar to the Industrial/Commercial scenario used by EPA to calculate soil screening values using direct contact exposure pathways such as inadvertent soil ingestion, dermal contact with soil, and inhalation. Of the scenarios evaluated on the scale of an individual remediated waste site, the Industrial/Commercial scenario has the lowest exposure intensity. The scenario is more fully described in the conceptual site model (CSM) in Section 3.3 and is summarized in Table 3-12 and Figure 3-16. The COPCs evaluated in shallow zone soil in the 100-B/C Decision Area are shown in Table 5-26.

5.3.1.1 Cancer Risk. Table 5-51 shows the RME and CTE total cancer risk results for the Industrial/Commercial exposure scenario for all remediated waste sites in the 100-B/C Decision Area. The tabulated results are the sum of the risk values for the individual COPCs included in the risk calculation. Total risk results use EPCs based on measured soil concentrations in the individual waste sites. Risk assessment results are shown for present-day radionuclide concentrations, and with radionuclide concentrations decayed to the years 2075 and 2150.

Soil concentrations for these times were calculated using the RESRAD computer code, as described in Appendix D-1.4.1.

The range of present-day RME and CTE total cancer risk results for the Industrial/Commercial scenario (radionuclides + chemicals) is from 5×10^{-9} to 2×10^{-4} and from 1×10^{-9} to 2×10^{-5}, respectively. No carcinogenic COPCs were detected at the 1607-B-11 waste site. Per EPA guidance (EPA/540/1-89/002, *Risk Assessment Guidance for Superfund Volume I Human Health Evaluation Manual*), cancer risk results are displayed with only one significant figure. As shown in Table 5-2, 2 of the 45 waste sites evaluated in the 100-B/C Decision Area had calculated total RME cancer risks above 1×10^{-4}, but no waste site had a CTE total cancer risk above 1×10^{-4}.

A summary of the key exposure pathways and COPCs contributing 10% or more to present-day Industrial/Commercial RME cancer risk at 100-B/C Decision Area sites with total cancer risk above 1×10^{-4} is provided below.

Site	COPC	Pathway	Fraction of Total Risk	Present-Day Total Risk
116-B-11	Europium-152	External Irradiation	0.62	2×10^{-4}
	Cobalt-60	External Irradiation	0.18	
	Europium-154	External Irradiation	0.10	
116-B-14	Europium-152	External Irradiation	0.84	2×10^{-4}

About 85% or more of the cancer risk of 2×10^{-4} at the 116-B-14 and 116-B-11 waste sites is due to external irradiation from short-lived radionuclides such as europium-152, europium-154, and cobalt-60 at 116-B-11. Because the half-life of these COPCs is 13.5 years or less, no waste sites exceed 1×10^{-4} risk for the Industrial/Commercial scenario at year 2075. The highest RME total cancer risk values at years 2075 and 2150 are 2×10^{-5} and 4×10^{-6}, respectively, due primarily to cesium-137 at the 116-B-6A waste site.

The current conditions of the 116-B-14 and 116-B-11 remediated waste sites are summarized in Table 2-8. The 116-B-11 site was excavated to a depth of 5 m (16.4 ft), and the 116-B-14 waste site was excavated to a depth of 6 m (19.7 ft). Both sites were backfilled to grade following remediation. The shallow zone soil verification data used to calculate local-area risks at these sites are from samples collected along the sidewalls of the excavations at depths between 0 and 4.6 m (15 ft). The use of these data to represent surface soil for current and future exposures may contribute to a significant protective bias in the Industrial/Commercial risk results for these sites. Uncertainties in the use of shallow zone soil verification data from excavated waste sites are discussed in more detail in Section 5.9 and Appendix D-1.

5.3.1.2 Radiation Dose. Table 5-52 shows the RME and CTE total radiation dose results for the Industrial/Commercial exposure scenario for all remediated waste sites in the 100-B/C Decision Area. The tabulated results are the sum of the dose values for the individual COPCs included in the calculation. Risk assessment results are shown for present-day radionuclide concentrations and radionuclide concentrations decayed to the years 2075 and 2150.

Soil concentrations for these times were calculated using the RESRAD computer code, as described in Appendix D-1.4.1.

The range of present-day RME and CTE total radiation dose results for the Industrial/Commercial scenario is from 0.0018 to 9.2 mrem/yr and from 0.0016 to 4.0 mrem/yr, respectively. No radionuclide COPCs were detected at 12 waste sites.

There are no waste sites in the 100-B/C Area where the present-day radiation dose for the Industrial/Commercial scenario exceeds the threshold of 15 mrem/yr.

5.3.1.3 Chemical Hazard. Table 5-53 shows the RME and CTE HI results for the Industrial/Commercial exposure scenario for all remediated waste sites in the 100-B/C Decision Area. The tabulated results are the sum of the risk values for the individual COPCs included in the risk calculation.

The range of RME and CTE HI results for the Industrial/Commercial scenario is from approximately 0.0001 to 0.020 and from 0.0001 to 0.0029, respectively. No COPCs contributing to chemical hazard were detected at two waste sites (118-B-4 and 118-C-2).

There are no waste sites in the 100-B/C Area, where the HI for the Industrial/Commercial scenario approached the threshold of 1.0.

5.3.2 Resident National Monument/Refuge Scenario Results for the 100-B/C Area

The Resident National Monument/Refuge exposure scenario encompasses potential residential exposures related to shallow zone soils around remediated waste sites, as well as potential occupational exposures to surface soil across the upland environment of the River Corridor. It focuses on direct contact exposure pathways and does not encompass the raising of produce or animals, nor the ingestion of Columbia River fish. This scenario also includes exposures related to groundwater via a domestic water well. Risks related to groundwater use are addressed in Section 6.0. With respect to soil exposure pathways, this scenario is similar to the traditional urban/suburban residential scenario used by EPA to calculate soil screening values using direct contact exposure pathways (e.g., inadvertent soil ingestion, dermal contact with soil, and inhalation). It may be viewed as akin to the Subsistence Farmer scenario for an adult receptor, minus the ingestion of foodstuffs and with an exposure model that averages exposure to both individual waste site soils (cleanup verification soil data) and surface soils over large areas (upland surface soil data). The scenario is more fully described in the CSM in Section 3.3 and is summarized in Table 3-12 and Figure 3-16. The COPCs evaluated in shallow zone soil in the 100-B/C Decision Area are shown in Table 5-26.

As described in the introductory text for Section 5.3, COPCs for the broad-area components of the waste site risk calculations are those related to upland surface soil, while the COPCs for the local-area components of the waste site risk calculations are those related to the cleanup verification data for a particular ROD decision area. Results for the local-area portion of the

Resident National Monument/Refuge exposure scenario are calculated for each individual remediated waste site where a residence might be situated. Risks related to the broad-area occupational component of the Resident National Monument/Refuge exposure scenario, which are additive to each of the local-area results, are shown in the first row of the summary tables for cancer risk, radiation dose, and chemical hazard. As discussed in Section 4.1.3, broad-area representative concentrations are calculated using upland surface soil data collected under the *100 Area and 300 Area Component of the RCBRA Sampling and Analysis Plan* (DOE/RL-2005-42).

5.3.2.1 Cancer Risk. Table 5-54 shows the RME and CTE total cancer risk results for the Resident National Monument/Refuge exposure scenario for all remediated waste sites in the 100-B/C Decision Area. The tabulated results are the sum of the risk values for the individual COPCs included in the risk calculation. Total risk results use EPCs based on measured soil concentrations in the individual waste sites (residential exposure) and River Corridor upland surface soil (occupational exposure). Risk assessment results are shown for present-day radionuclide concentrations, and with radionuclide concentrations decayed to the years 2075 and 2150. Soil concentrations for these times were calculated using the RESRAD computer code, as described in Appendix D-1.4.1.

The range of present-day RME and CTE total cancer risk results for the local-area component of the Resident National Monument/Refuge scenario (radionuclides + chemicals) is from 7×10^{-9} to 3×10^{-4} and from 1×10^{-9} to 2×10^{-5}, respectively. No carcinogenic COPCs were detected at the 1607-B-11 waste site. The broad-area present-day RME and CTE cancer risks are 2×10^{-5} and 4×10^{-6}, respectively. About 80% to 90% of the CTE and RME broad-area cancer risk values are related to external exposure to europium-152 . Per EPA guidance (EPA/540/1-89/002), cancer risk results are displayed with only one significant figure. As shown in Table 5-2, 5 of the 45 remediated waste sites evaluated in the 100-B/C Decision Area had a combined local and broad total RME cancer risks above 1×10^{-4}, but no waste site had a combined local and broad CTE total cancer risk above 1×10^{-4}.

Because the broad-area component of cancer risk is the same across all calculations, the summary of the key exposure pathways and COPCs contributing 10% or more to present-day RME cancer risk at 100-B/C Decision Area sites focuses on the local-area component of the Resident National Monument/Refuge scenario. The local-area calculations with total cancer risk above 1×10^{-4} are summarized below.

Site	COPC	Pathway	Fraction of Total Risk	Present-Day Total Risk
116-B-11	Europium-152	External Irradiation	0.62	3×10^{-4}
	Cobalt-60	External Irradiation	0.18	
116-B-14	Europium-152	External Irradiation	0.84	3×10^{-4}
116-B-6A	Cesium-137	External Irradiation	0.98	2×10^{-4}

Site	COPC	Pathway	Fraction of Total Risk	Present-Day Total Risk
132-B-6	Europium-152	External Irradiation	0.57	2×10^{-4}
	Cesium-137	External Irradiation	0.29	
116-C-5	Europium-152	External Irradiation	0.52	2×10^{-4}
	Cesium-137	External Irradiation	0.20	
	Cobalt-60	External Irradiation	0.14	
	Europium-154	External Irradiation	0.12	

For the waste sites with present-day RME cancer risk above 1×10^{-4}, most of the local-area cancer risk is related to just a few radionuclides, including cobalt-60, cesium-137, and isotopic europium. The half-lives of all of these radionuclides range from approximately 5 years (cobalt-60) to approximately 30 years (cesium-137). Therefore, much of the local-area cancer risk will decrease naturally over time as a function of the decay of these radionuclides. There are no waste sites in the 100-B/C Area where combined local and broad-area RME cancer risk is above 1×10^{-5} at year 2075 for the Resident National Monument/Refuge exposure scenario.

As described in relation to Industrial/Commercial cancer risk results, the use of shallow zone soil verification data used to calculate local-area risks at the 116-B-14 and 116-B-11 waste sites may contribute to a significant protective bias in the risk results for these sites. Both of these sites were excavated to a depth of greater than 4.6 m (15 ft), the limit of shallow zone data used for these calculations. The 116-B-6A and 116-C-5 waste sites (4.6 m [15 ft] excavation depths), and the 132-B-6 waste site (8.3 m [27.2 ft] excavation depth) were similarly excavated to great depths. The use of these data to represent surface soil for current and future exposures may contribute to a significant protective bias in the Resident National Monument/Refuge risk results for all these sites. Uncertainties in the use of shallow zone soil verification data from excavated waste sites are discussed in more detail in Section 5.9 and Appendix D-1.

5.3.2.2 Radiation Dose. Table 5-55 shows the RME and CTE total radiation dose results for the Resident National Monument/Refuge exposure scenario for all remediated waste sites in the 100-B/C Decision Area. The tabulated results are the sum of the dose values for the individual COPCs included in the calculation. Risk assessment results are shown for present-day radionuclide concentrations and with radionuclide concentrations decayed to the years 2075 and 2150. Soil concentrations for these times were calculated using the RESRAD computer code, as described in Appendix D-1.4.1.

The range of present-day RME and CTE total radiation dose results for the local-area component of the Resident National Monument/Refuge scenario is from 0.049 mrem/yr to 16 mrem/yr and from 0.036 mrem/yr to 6.6 mrem/yr, respectively. Dose results at the 100-B-11 waste site were well below this range, indicating little anthropogenic radionuclide contamination. No radionuclide COPCs were detected at 12 waste sites. The broad-area present-day RME and CTE radiation doses are 0.80 mrem/yr and 0.78 mrem/yr, respectively. As shown in Table 5-2, only 2 of the 45 waste sites evaluated in the 100-B/C Decision Area had a combined local and broad total RME radiation dose value above 15 mrem/yr. No waste sites had a CTE total radiation dose above 15 mrem/yr.

Because the broad-area component of cancer risk is the same across all calculations, the summary of the key exposure pathways and COPCs contributing 10% or more to present-day RME radiation dose at 100-B/C Decision Area sites focuses on the local-area component of the Resident National Monument/Refuge. The local-area calculations with total dose above 15 mrem/yr are summarized below.

Site	COPC	Pathway	Fraction Of Total Dose	Present-Day Total Dose (mrem/yr)
116-B-11	Europium-152	External Irradiation	0.62	16
	Cobalt-60	External Irradiation	0.18	

The local-area radiation dose of 16 mrem/yr at the 116-B-11 waste site was mostly due to external irradiation from europium-152. Because this is a very short-lived radionuclide (half-life of 13.5 years), no waste sites exceeded the 15 mrem/yr threshold for the Resident National Monument/Refuge scenario at year 2075. The present-day RME local-area dose at the 116-B-14 waste site was 15 mrem/yr. When the broad-area component of exposure (0.80 mrem/yr) is added to the local-area dose, the resulting dose of 16 mrem/yr is also slightly above the threshold criterion. The short-lived radionuclide europium-152 is also the major contributor to present-day dose at 116-B-14. The highest local-area RME radiation dose at year 2075 was 2.0 mrem/yr at the 116-B-6A waste site, due primarily to cesium-137.

5.3.2.3 Chemical Hazard. Table 5-56 shows the RME and CTE HI results for the Resident National Monument/Refuge exposure scenario for all remediated waste sites in the 100-B/C Decision Area. The tabulated results are the sum of the risk values for the individual COPCs included in the risk calculation.

The range of RME and CTE HI results for the local-area component of the Resident National Monument/Refuge scenario is from 0.011 to 0.046 and 0.008 to 0.014, respectively. No COPCs contributing to chemical hazard were detected at two waste sites (118-B-4 and 118-C-2). The broad-area RME and CTE HI results are 0.025 and 0.0079, respectively.

There are no waste sites in the 100-B/C Area where the combined local-area and broad-area exposures result in an HI for the Resident National Monument/Refuge scenario that approaches the threshold of 1.0.

5.3.3 Subsistence Farmer Scenario Results for the 100-B/C Area

The Subsistence Farmer exposure scenario encompasses potential exposures related to shallow zone soils around remediated waste sites, including both direct contact exposure pathways and exposures related to raising produce and animals in the potentially affected area of the waste site. This scenario also includes exposures related to ingestion of Columbia River fish and exposure to groundwater via a domestic water well. Risks related to fish ingestion and groundwater use are addressed in Sections 4.4 and 6.0, respectively. The scenario is more fully described in the CSM in Section 3.3 and is summarized in Table 3-13 and Figure 3-17. The COPCs evaluated in shallow zone soil in the 100-B/C Decision Area are shown in Table 5-26.

5.3.3.1 Cancer Risk. Table 5-57 shows the RME and CTE total cancer risk results for the Subsistence Farmer exposure scenario for all remediated waste sites in the 100-B/C Decision Area. The tabulated results are the sum of the risk values for the individual COPCs included in the risk calculation. Total risk results use EPCs based on measured soil concentrations in the individual waste sites. Risk assessment results are shown for present-day radionuclide concentrations, and with radionuclide concentrations decayed to the years 2075 and 2150. Soil concentrations for these times were calculated using the RESRAD computer code (see Appendix D-1.4.1).

The range of present-day RME and CTE total cancer risk results for the Subsistence Farmer scenario (radionuclides + chemicals) is from 2×10^{-8} to 1×10^{-3} and from 6×10^{-9} to 8×10^{-5}, respectively. No carcinogenic COPCs were detected at the 1607-B11 waste site. Per EPA guidance (EPA/540/1-89/002), cancer risk results are displayed with only one significant figure. As shown in Table 5-5, 10 of the 45 waste sites evaluated in the 100-B/C Decision Area had present-day total RME cancer risks above 1×10^{-4}, but no waste site had a CTE total cancer risk above 1×10^{-4}.

A summary of the key exposure pathways and COPCs contributing 10% or more to Subsistence Farmer present-day RME cancer risk at 100-B/C Decision Area sites with total cancer risk above 1×10^{-4} is provided below.

Site	COPC	Pathway	Fraction of Total Risk	Present-Day Total Risk
116-B-6A	Cesium-137	External Irradiation	0.30	1×10^{-3}
	Strontium-90	Milk Ingestion	0.26	
	Strontium-90	Produce Ingestion	0.21	
	Strontium-90	Beef Ingestion	0.12	
116-B-14	Europium-152	External Irradiation	0.58	8×10^{-4}
	Strontium-90	Milk Ingestion	0.12	
116-B-11	Europium-152	External Irradiation	0.55	7×10^{-4}
	Cobalt-60	External Irradiation	0.16	
116-C-5	Europium-152	External Irradiation	0.41	4×10^{-4}
	Cesium-137	External Irradiation	0.16	
	Cobalt-60	External Irradiation	0.11	
132-B-6	Europium-152	External Irradiation	0.49	4×10^{-4}
	Cesium-137	External Irradiation	0.25	

Site	COPC	Pathway	Fraction of Total Risk	Present-Day Total Risk
116-C-2A	Strontium-90	Milk Ingestion	0.26	4×10^{-4}
	Strontium-90	Produce Ingestion	0.22	
	Cobalt-60	External Irradiation	0.14	
	Europium-152	External Irradiation	0.13	
	Strontium-90	Beef Ingestion	0.12	
116-C-1	Europium-152	External Irradiation	0.41	4×10^{-4}
	Cesium-137	External Irradiation	0.18	
116-C-6	Cesium-137	External Irradiation	0.42	2×10^{-4}
	Strontium-90	Milk Ingestion	0.12	
	Europium-152	External Irradiation	0.11	
	Strontium-90	Produce Ingestion	0.10	
100-B-8:2	Europium-154	External Irradiation	0.22	2×10^{-4}
	Cobalt-60	External Irradiation	0.21	
	Europium-152	External Irradiation	0.18	
116-B-1	Europium-152	External Irradiation	0.50	2×10^{-4}
	Cobalt-60	External Irradiation	0.17	
	Europium-154	External Irradiation	0.13	

For a large number of the waste sites with present-day RME cancer risk above 1×10^{-4}, most of the cancer risk is related to just a few radionuclides, including cobalt-60, cesium-137, isotopic europium, and strontium-90. The half-lives of these radionuclides range from approximately 5 years (cobalt-60) to approximately 30 years (strontium-90 and cesium-137). Therefore, much of the site-related cancer risk will decrease naturally over time as a function of the decay of these radionuclides. Only the 116-B-6A waste site (2×10^{-4}) has a total RME cancer risk above 1×10^{-4} in year 2075, due primarily to strontium-90 and cesium-137. By year 2150, the highest total RME cancer risk value is 5×10^{-5} at the 116-B-6A waste site, and the next highest value is 2×10^{-5} at the 116-B-14 waste site.

As described in relation to Industrial/Commercial and Resident National Monument/Refuge cancer risk results, the use of shallow zone soil verification data to calculate local-area risks at the 116-B-14, 116-B-11, 116-B-6A, 116-C-5, and 132-B-6 waste sites may contribute to a significant protective bias in the risk results for these sites. All of these sites were excavated to a depth of greater than 4.6 m (15 ft), the limit of shallow zone data used for these calculations. Similarly, the 116-C-2A, 116-C-1, 100-B-8:2, and 116-B-1 waste sites were excavated to depths at or below 4.6 m (15 ft). No excavation depth was recorded for the 116-C-6 waste site in Table 2-8. The use of these data to represent surface soil for current and future exposures may contribute to a significant protective bias in the Subsistence Farmer risk results for all these sites. Uncertainties in the use of shallow zone soil verification data from excavated waste sites are discussed in more detail in Section 5.9 and Appendix D-1.

5.3.3.2 Radiation Dose. Table 5-58 shows the RME and CTE total radiation dose results for the Subsistence Farmer exposure scenario for all remediated waste sites in the 100-B/C Decision Area. The tabulated results are the sum of the dose values for the individual COPCs included in the calculation. Total dose results use EPCs based on measured soil concentrations in the individual waste sites. Risk assessment results are shown for present-day radionuclide concentrations and with radionuclide concentrations decayed to the years 2075 and 2150. Soil concentrations for these times were calculated using the RESRAD computer code, as described in Appendix D-1.4.1.

The range of present-day RME and CTE total radiation dose results for the Subsistence Farmer scenario is from 0.13 to 76 mrem/yr and from 0.11 to 14 mrem/yr, respectively. No radionuclide COPCs were detected at 12 waste sites. As shown in Table 5-5, 7 of the 45 waste sites evaluated in the 100-B/C Decision Area had calculated total RME radiation dose values above 15 mrem/yr, but no waste sites had a CTE total radiation dose above 15 mrem/yr.

A summary of the key exposure pathways and COPCs contributing 10% or more to Subsistence Farmer present-day RME radiation dose at 100-B/C Decision Area sites with total dose above 15 mrem/yr is provided below.

Site	COPC	Pathway	Fraction Of Total Dose	Present-Day Total Dose (mrem/yr)
116-B-6A	Strontium-90	Milk Ingestion	0.31	76
	Strontium-90	Produce Ingestion	0.26	
	Cesium-137	External Irradiation	0.22	
	Strontium-90	Beef Ingestion	0.14	
116-B-14	Europium-152	External Irradiation	0.49	42
	Strontium-90	Milk Ingestion	0.16	
	Strontium-90	Produce Ingestion	0.14	
116-B-11	Europium-152	External Irradiation	0.53	31
	Cobalt-60	External Irradiation	0.15	
116-C-2A	Strontium-90	Milk Ingestion	0.31	24
	Strontium-90	Produce Ingestion	0.26	
	Strontium-90	Beef Ingestion	0.14	
	Cobalt-60	External Irradiation	0.10	
116-C-5	Europium-152	External Irradiation	0.38	20
	Cesium-137	External Irradiation	0.15	
	Cobalt-60	External Irradiation	0.10	
116-C-1	Europium-152	External Irradiation	0.35	20
	Cesium-137	External Irradiation	0.16	
	Strontium-90	Milk Ingestion	0.14	
	Strontium-90	Produce Ingestion	0.12	

Site	COPC	Pathway	Fraction Of Total Dose	Present-Day Total Dose (mrem/yr)
132-B-6	Europium-152	External Irradiation	0.48	18
	Cesium-137	External Irradiation	0.25	

Similar to the findings for cancer risk, the present-day radiation dose at sites with calculated dose above the threshold is driven by relatively short-lived radionuclides. As shown in Table 5-6, no 100-B/C waste sites exceed 15 mrem/yr at year 2075. At year 2075, the highest total RME radiation dose value is 15 mrem/yr at the 116-B-6A waste site, due primarily to strontium-90 and cesium-137. By year 2150, the highest total RME radiation dose value is 2.9 mrem/yr at the 116-B-6A waste site. As described in relation to cancer risk, all of the remediated waste sites where radiation dose exceeded 15 mrem/yr were excavated to a depth of greater than 4.6 m (15 ft). The use of shallow zone soil verification data to calculate local-area risks at these sites may contribute to a significant protective bias in the radiation dose results for these sites.

5.3.3.3 Chemical Hazard. Table 5-59 shows the RME and CTE HI results for the Subsistence Farmer exposure scenario for all remediated waste sites in the 100-B/C Decision Area. The tabulated results are the sum of the risk values for the individual COPCs included in the risk calculation. As described in Section 3.4.4, chemical hazard is calculated separately for adult and child receptors. For the Subsistence Farmer scenario, RME and CTE child HIs are usually equal to or above those for adults. For this reason, only child HI values are presented here.

The range of RME and CTE child HI results for the Subsistence Farmer scenario is from 0.14 to 19 and from 0.064 to 9.4 , respectively. HI results at remediated the 118-B-5 waste site were well below this reported range, indicating essentially no residual site-related contamination contributing to radiation dose. No COPCs contributing to chemical hazard were detected at two waste sites (118-B-4 and 118-C-2). As shown in Table 5-5, 2 of the 45 waste sites in the 100-B/C Decision Area had calculated RME HI values above 10 and another 14 sites had HI values between 1.0 and 10. Eight remediated waste sites had CTE HI values above 1.0.

A summary of the key exposure pathways and COPCs contributing 10% or more to Subsistence Farmer present-day RME HI in the 100-B/C Decision Area is provided below. For brevity, only sites with an HI at or above 2.0 are shown here.

Site	COPC	Pathway	Fraction Of Child Hazard	Present-Day Child Hazard [a]
100-B-14:6	Mercury	Beef Ingestion	0.65	19
	Mercury	Produce Ingestion	0.24	
116-B-10	Mercury	Beef Ingestion	0.69	17
	Mercury	Produce Ingestion	0.26	

Site	COPC	Pathway	Fraction Of Child Hazard	Present-Day Child Hazard [a]
118-B-9	Mercury	Beef Ingestion	0.29	6.9
	Zinc	Beef Ingestion	0.16	
	Zinc	Milk Ingestion	0.14	
	Zinc	Produce Ingestion	0.13	
	Mercury	Produce Ingestion	0.11	
1607-B10	Mercury	Beef Ingestion	0.66	5.0
	Mercury	Produce Ingestion	0.25	
116-B-9	Mercury	Beef Ingestion	0.68	5.0
	Mercury	Produce Ingestion	0.26	
128-C-1	Cadmium	Produce Ingestion	0.17	5.0
	Mercury	Beef Ingestion	0.12	
132-B-6	Mercury	Beef Ingestion	0.65	3.2
	Mercury	Produce Ingestion	0.24	
100-B-14:7	Mercury	Beef Ingestion	0.23	3.1
	Cadmium	Produce Ingestion	0.13	
	Zinc	Beef Ingestion	0.11	
	Zinc	Milk Ingestion	0.10	
118-B-10	Mercury	Beef Ingestion	0.69	2.9
	Mercury	Produce Ingestion	0.26	
1607-B8	Mercury	Beef Ingestion	0.44	2.1
	Mercury	Produce Ingestion	0.16	
	Aroclor-1254	Soil Ingestion	0.11	
1607-B9	Mercury	Beef Ingestion	0.52	2.0
	Mercury	Produce Ingestion	0.20	
	Cadmium	Produce Ingestion	0.19	

[a] Values for mercury hazard at 100-B-14:6, 116-B-10, and 118-B-9, zinc hazard at 118-B-9, and cadmium hazard at 128-C-1 are likely overestimated; see discussion in the text.

Exposures to mercury via produce and/or beef ingestion are the most important exposure pathways contributing to HI values above 2.0 at many of these waste sites. Zinc is also a significant contributor to child HI values at the 118-B-9 and 100-B-14:7 waste sites. Cadmium is an important contributor to child HI values at the 128-C-1 and 100-B-14:7 waste sites. The HI values are calculated as the sum of chemical-specific HQs, as described in Section 3.4.4.

Critical effects underlying the oral reference dose (RfD) value for inorganic forms of mercury include effects on the immune system, but the toxic effects of mercury also include kidney toxicity (Integrated Risk Information System [IRIS], Office of Research and Development and National Center for Environmental Assessment [IRIS 2009]). Cadmium toxicity following ingestion is primarily expressed in the kidneys (IRIS 2009). Therefore, adding of HQs for cadmium and mercury is reasonable. The toxic effects of chronic exposure to relatively low doses of zinc, as associated with environmental exposures, are primarily related to antagonism of copper absorption and effects on the hematological system (Klaassen 2001, *Casarett & Doull's Toxicology: The Basic Science of Poisons*). Additivity of zinc HQs with those of cadmium or mercury is, therefore, likely to overestimate chemical hazard.

Child hazards related to the key exposure pathways and COPCs may be compared to child HQs calculated using COPC concentrations in the upland reference area, shown in Table 5-50. The RME and CTE child HQs calculated using the upland reference area data for mercury are 0.28 and 0.14. The RME soil mercury concentration at the 100-B-14:6, 116-B-10, 116-B-9, 1607-B10, 132-B-6, 118-B-10, and 118-B-9 waste sites are between approximately 10 and 60 times higher than the upland reference area RME mercury soil concentration of 0.022 mg/kg. The RME soil concentration of cadmium at the 128-C-1 waste site (1.1 mg/kg) is also six times larger than the reference area RME value of 0.18 mg/kg. And at the 118-B-9 remediated waste site, the RME zinc value in soil of 250 mg/kg is approximately a factor of 5 greater than the reference area RME value of 50 mg/kg.

As discussed in the Uncertainty Analysis (Section 5.9), the linear models used in this risk assessment to estimate plant tissue concentrations from soil data are prone to overestimation of plant concentrations at higher soil concentrations. At the 100-B-14:6 waste site, for example, the mercury plant-soil regression model, *Empirical Models for the Uptake of Inorganic Chemicals by Plants,* published by Bechtel (BJC/OR-133) predicts a mercury plant concentration of 0.44 mg/kg based on the site shallow zone soil concentration of 1.4 mg/kg. By contrast, the dry-weight produce and fodder concentrations using the linear plant-soil ratios project plant concentrations of 3.5 and 1.4 mg/kg, respectively. Assuming the nonlinear model is more accurate at the higher soil mercury concentration of 1.4 mg/kg, this indicates mercury exposure at the 100-B-14:6 waste site may be overestimated by a factor of about 8 and 3 for produce ingestion and beef ingestion, respectively. In that case, the mercury HQ at 100-B-14:6 for produce ingestion would both be about 0.59, and for beef ingestion about 4.1. This results in an HI value of about 4.7 rather than 19. An additional aspect of the uncertainty in the HI for 100-B-14:6 is that the site soil concentration for mercury is based on only two soil samples and one positive detection.

As described in relation to cancer risk and radiation dose, the remediated waste sites where cancer risk exceeded 1×10^{-4} and radiation dose exceeded 15 mrem/yr were excavated to a depth of 4.6 m (15 ft) or more. These include the 116-B-14, 116-B-11, 116-B-6A, 116-C-5, 132-B-6, 116-C-2A, 116-C-1, 100-B-8:2, and 116-B-1 waste sites. However, the sites where child HI was highest are not the same as the sites where cancer risk and radiation dose were highest. The 100-B-14:6 waste site was not excavated and the 116-B-10 waste site was excavated to a depth of 2.44 m (8 ft). The 118-B-9 waste site was not excavated, but the 1607-B10, 116-B-9, and 128-C-1 waste sites were excavated to depths of between approximately 2 and 4 m (6.6 and 13.1 ft). The use of shallow zone soil verification data to calculate local-area child HI at these excavated sites may contribute to a protective bias in these results, though potentially to a lesser degree than that related to cancer risk and radiation dose. Uncertainties in the use of shallow zone soil verification data from excavated waste sites are discussed in more detail in Section 5.9 and Appendix D-1.

5.3.4 CTUIR Resident Scenario Results for the 100-B/C Area

The CTUIR Resident exposure scenario encompasses potential exposures related to shallow zone soils around remediated waste sites, including both direct contact exposure pathways and exposures related to raising produce and animals in the potentially affected area of the waste site. This scenario also includes exposures related to ingestion of Columbia River fish (see Section 4.4) and exposure to groundwater via a domestic water well (see Section 6.0). It is similar to the Subsistence Farmer scenario, although exposure intensity is higher for many exposure pathways. The scenario is more fully described in the CSM in Section 3.3 and is summarized in Table 3-13 and Figure 3-17. The COPCs evaluated in shallow zone soil in the 100-B/C Decision Area are shown in Table 5-26.

5.3.4.1 Cancer Risk. Table 5-60 shows total cancer risk results for the CTUIR Resident exposure scenario for all remediated waste sites in the 100-B/C Decision Area. The tabulated results are the sum of the risk values for the individual COPCs included in the risk calculation. Total risk results use EPCs based on measured soil concentrations in the individual waste sites. Risk assessment results are shown for present-day radionuclide concentrations, and with radionuclide concentrations decayed to the years 2075 and 2150. Soil concentrations for these times were calculated using the RESRAD computer code (see Appendix D-1.4.1).

The range of present-day total cancer risk results for the CTUIR Resident scenario (radionuclides + chemicals) is from 6×10^{-8} to 7×10^{-3}. No carcinogenic COPCs were detected at the 1607-B11 waste site. Per EPA guidance (EPA/540/1-89/002), cancer risk results are displayed with only one significant figure. As shown in Table 5-5, 20 of the 45 waste sites evaluated in the 100-B/C Decision Area had present-day total cancer risks above 1×10^{-4}.

A summary of the key exposure pathways and COPCs contributing 10% or more to present-day CTUIR Resident cancer risk at 100-B/C Decision Area sites with total cancer risk above 1×10^{-4} is provided below.

Site	COPC	Pathway	Fraction of Total Risk	Present-Day Total Risk
116-B-6A	Strontium-90	Produce Ingestion	0.74	7×10^{-3}
	Cesium-137	External Irradiation	0.14	
116-B-14	Strontium-90	Produce Ingestion	0.47	3×10^{-3}
	Europium-152	External Irradiation	0.39	
116-C-2A	Strontium-90	Produce Ingestion	0.75	2×10^{-3}
116-B-11	Europium-152	External Irradiation	0.49	2×10^{-3}
	Cobalt-60	External Irradiation	0.15	
	Strontium-90	Produce Ingestion	0.12	

Site	COPC	Pathway	Fraction of Total Risk	Present-Day Total Risk
116-C-1	Strontium-90	Produce Ingestion	0.42	2×10^{-3}
	Europium-152	External Irradiation	0.29	
	Cesium-137	External Irradiation	0.12	
116-C-5	Europium-152	External Irradiation	0.33	1×10^{-3}
	Strontium-90	Produce Ingestion	0.27	
	Cesium-137	External Irradiation	0.13	
132-B-6	Europium-152	External Irradiation	0.45	1×10^{-3}
	Cesium-137	External Irradiation	0.23	
116-C-6	Strontium-90	Produce Ingestion	0.47	1×10^{-3}
	Cesium-137	External Irradiation	0.27	
	Cesium-137	Produce Ingestion	0.10	
100-B-8:2	Strontium-90	Produce Ingestion	0.41	8×10^{-4}
	Europium-154	External Irradiation	0.15	
	Cobalt-60	External Irradiation	0.15	
	Europium-152	External Irradiation	0.12	
116-B-2	Strontium-90	Produce Ingestion	0.53	6×10^{-4}
	Cesium-137	External Irradiation	0.28	
	Cesium-137	Produce Ingestion	0.11	
116-B-13	Strontium-90	Produce Ingestion	0.69	6×10^{-4}
	Europium-152	External Irradiation	0.13	
116-B-1	Europium-152	External Irradiation	0.45	5×10^{-4}
	Cobalt-60	External Irradiation	0.15	
	Strontium-90	Produce Ingestion	0.12	
	Europium-154	External Irradiation	0.12	
118-B-10	Cobalt-60	External Irradiation	0.69	5×10^{-4}
	Nickel-63	Produce Ingestion	0.20	
100-B-8:1	Europium-152	External Irradiation	0.59	4×10^{-4}
	Strontium-90	Produce Ingestion	0.25	
132-C-2	Cobalt-60	External Irradiation	0.50	3×10^{-4}
	Strontium-90	Produce Ingestion	0.30	
118-B-3	Strontium-90	Produce Ingestion	0.30	3×10^{-4}
	Europium-152	External Irradiation	0.19	
	Cobalt-60	External Irradiation	0.19	
	Europium-154	External Irradiation	0.19	

Site	COPC	Pathway	Fraction of Total Risk	Present-Day Total Risk
116-B-7	Strontium-90	Produce Ingestion	0.63	3×10^{-4}
	Cesium-137	External Irradiation	0.15	
100-B-11	Strontium-90	Produce Ingestion	0.94	3×10^{-4}
118-B-4	Europium-152	External Irradiation	0.61	2×10^{-4}
	Cesium-137	External Irradiation	0.27	
	Cesium-137	Produce Ingestion	0.10	
118-C-2	Nickel-63	Produce Ingestion	0.81	2×10^{-4}
	Cesium-137	External Irradiation	0.12	

For a large number of the waste sites with present-day cancer risk above 1×10^{-4}, most of the cancer risk is related to just a few radionuclides, including cobalt-60, cesium-137, isotopic europium, and strontium-90. The half-lives of these radionuclides range from approximately 5 years (cobalt-60) to approximately 30 years (strontium-90 and cesium-137). Therefore, much of the site-related cancer risk will decrease naturally over time as a function of the decay of these radionuclides. There are seven waste sites with a total cancer risk above 1×10^{-4} in year 2075, with risks mostly related to strontium-90 and cesium-137. By year 2150, the highest total cancer risk value is 2×10^{-4} at the 116-B-6A waste site and the next highest value is 1×10^{-4} at the 116-B-14 waste site.

The waste sites where cancer risk was highest for the CTUIR Resident scenario are the same as those described for the Subsistence Farmer exposure scenario. These remediated waste sites were excavated to a depth of 4.6 m (15 ft) or more. The use of shallow zone soil verification data to calculate local-area risks at these sites may contribute to a significant protective bias in the cancer risk results for these sites.

5.3.4.2 Radiation Dose. Table 5-61 shows the total radiation dose results for the CTUIR Resident exposure scenario for all remediated waste sites in the 100-B/C Decision Area. The tabulated results are the sum of the dose values for the individual COPCs included in the calculation. Total dose results use EPCs based on measured soil concentrations in the individual waste sites. Risk assessment results are shown for present-day radionuclide concentrations, with radionuclide concentrations decayed to the years 2075 and 2150. Soil concentrations for these times were calculated using the RESRAD computer code, as described in Appendix D-1.4.1.

The range of present-day total radiation dose results for the CTUIR scenario is from 0.15 to 156 mrem/yr. No radionuclide COPCs were detected at 12 waste sites. As shown in Table 5-5, 9 of the 45 waste sites evaluated in the 100-B/C Decision Area had calculated total radiation dose values above 15 mrem/yr.

A summary of the key exposure pathways and COPCs contributing 10% or more to CTUIR Resident present-day radiation dose at 100-B/C Decision Area sites with total dose above 15 mrem/yr is provided below.

Site	COPC	Pathway	Fraction of Total Dose	Present-Day Total Dose (mrem/yr)
116-B-6A	Strontium-90	Produce Ingestion	0.79	156
	Cesium-137	External Irradiation	0.12	
116-B-14	Strontium-90	Produce Ingestion	0.53	68
	Europium-152	External Irradiation	0.34	
116-C-2A	Strontium-90	Produce Ingestion	0.79	50
116-B-11	Europium-152	External Irradiation	0.48	39
	Strontium-90	Produce Ingestion	0.15	
	Cobalt-60	External Irradiation	0.14	
116-C-1	Strontium-90	Produce Ingestion	0.48	31
	Europium-152	External Irradiation	0.26	
	Cesium-137	External Irradiation	0.11	
116-C-5	Strontium-90	Produce Ingestion	0.32	28
	Europium-152	External Irradiation	0.31	
	Cesium-137	External Irradiation	0.12	
132-B-6	Europium-152	External Irradiation	0.45	22
	Cesium-137	External Irradiation	0.23	
116-C-6	Strontium-90	Produce Ingestion	0.54	20
	Cesium-137	External Irradiation	0.24	
100-B-8:2	Strontium-90	Produce Ingestion	0.47	16
	Europium-154	External Irradiation	0.13	
	Cobalt-60	External Irradiation	0.13	
	Europium-152	External Irradiation	0.11	

Similar to the findings for cancer risk, the present-day radiation dose at sites with calculated dose above the threshold is driven by relatively short-lived radionuclides. The only 100-B/C Decision Area remediated waste site to exceed 15 mrem/yr at year 2075 for the CTUIR Resident scenario is 116-B-6A (30 mrem/yr), due primarily to strontium-90 and cesium-137. Total radiation dose results are below 15 mrem/yr at all remediated waste sites by year 2150.

The waste sites where radiation dose was highest for the CTUIR Resident scenario are the same as those described for the Subsistence Farmer exposure scenario. These remediated waste sites were excavated to a depth of 4.6 m (15 ft) or more. The use of shallow zone soil verification data to calculate risks at these sites may contribute to a significant protective bias in the radiation dose results for these sites.

5.3.4.3 Chemical Hazard. Table 5-62 shows HI results for the CTUIR Resident exposure scenario for all remediated waste sites in the 100-B/C Decision Area. The tabulated results are the sum of the risk values for the individual COPCs included in the risk calculation.

As described in Section 3.4.4, chemical hazard is calculated separately for adult and child receptors. For the CTUIR Resident scenario, child HIs are usually equal to or above those for adults. For this reason, only child HI values are presented here.

The range of child HI results for the CTUIR Resident scenario is from 0.28 to 95. The HI results at the 118-B-5 remediated waste site were well below this reported range, indicating essentially no residual site-related contamination contributing to radiation dose. No COPCs contributing to chemical hazard were detected at two waste sites (118-B-4 and 118-C-2). As shown in Table 5-5, 10 of the 45 waste sites in the 100-B/C Decision Area had calculated HI values above 10 and another 24 sites had HI values between 1.0 and 10.

A summary of the key exposure pathways and COPCs contributing 10% or more to CTUIR Resident child HI in the 100-B/C Decision Area is provided below. For brevity, only sites with a total HI at or above 3.0 are shown here.

Site	COPC	Pathway	Fraction of Child Hazard	Present-Day Child Hazard
100-B-14:6	Mercury	produce Ingestion	0.68	95
	Mercury	Beef Ingestion	0.22	
116-B-10	Mercury	Produce Ingestion	0.75	80
	Mercury	Beef Ingestion	0.24	
118-B-9	Zinc	Produce Ingestion	0.34	36
	Mercury	Produce Ingestion	0.29	
	Cadmium	Produce Ingestion	0.13	
128-C-1	Cadmium	Produce Ingestion	0.41	29
	Copper	Produce Ingestion	0.20	
	Zinc	Produce Ingestion	0.12	
	Mercury	Produce Ingestion	0.11	
1607-B10	Mercury	Produce Ingestion	0.69	25
	Mercury	Beef Ingestion	0.23	
116-B-9	Mercury	Produce Ingestion	0.75	24
	Mercury	Beef Ingestion	0.24	
100-B-14:7	Cadmium	Produce Ingestion	0.31	18
	Zinc	Produce Ingestion	0.22	
	Mercury	Produce Ingestion	0.21	
	Copper	Produce Ingestion	0.14	
132-B-6	Mercury	Produce Ingestion	0.73	15
	Mercury	Beef Ingestion	0.24	

Site	COPC	Pathway	Fraction of Child Hazard	Present-Day Child Hazard
118-B-10	Mercury	Produce Ingestion	0.73	15
	Mercury	Beef Ingestion	0.24	
1607-B9	Mercury	Produce Ingestion	0.43	12
	Cadmium	Produce Ingestion	0.42	
	Mercury	Beef Ingestion	0.14	
128-B-2	Zinc	Produce Ingestion	0.24	9.2
	Cadmium	Produce Ingestion	0.23	
	Copper	Produce Ingestion	0.18	
	Mercury	Produce Ingestion	0.15	
100-B-11	Zinc	Produce Ingestion	0.35	9.1
	Cadmium	Produce Ingestion	0.27	
	Copper	Produce Ingestion	0.27	
1607-B8	Mercury	Produce Ingestion	0.54	8.9
	Mercury	Beef Ingestion	0.17	
	Cadmium	Produce Ingestion	0.13	
100-B-14:5	Zinc	Produce Ingestion	0.43	8.7
	Copper	Produce Ingestion	0.24	
	Cadmium	Produce Ingestion	0.21	
116-C-2A	Mercury	Produce Ingestion	0.73	7.6
	Mercury	Beef Ingestion	0.24	
116-B-11	Mercury	Produce Ingestion	0.67	7.6
	Mercury	Beef Ingestion	0.22	
600-233	Cadmium	Produce Ingestion	0.47	6.6
	Zinc	Produce Ingestion	0.25	
	Copper	Produce Ingestion	0.21	
100-B-16	Cadmium	Produce Ingestion	0.68	5.8
	Mercury	Produce Ingestion	0.24	
118-B-3	Cadmium	Produce Ingestion	0.99	4.8
100-C-3	Cadmium	Produce Ingestion	0.88	3.5
100-B-8:1	Mercury	Produce Ingestion	0.68	3.2
	Mercury	Beef Ingestion	0.22	
116-C-5	Mercury	Produce Ingestion	0.61	3.0
	Mercury	Beef Ingestion	0.20	
	Uranium	Produce Ingestion	0.14	

Exposure to mercury, cadmium, and zinc via produce and/or beef ingestion are the most important analyte and exposure pathway combinations contributing to HI values above 3.0 at many of these waste sites. The HI values are calculated as the sum of chemical-specific HQs, as described in Section 3.4.4. As described in relation to Subsistence Farmer HI, critical effects underlying the oral RfD value for inorganic forms of mercury include effects on the immune system, but the toxic effects of mercury also include kidney toxicity. Cadmium toxicity following ingestion is primarily expressed in the kidneys. Therefore, adding of HQs for cadmium and mercury is reasonable. The toxic effects of chronic exposure to relatively low doses of zinc, as associated with environmental exposures, are primarily related to antagonism of copper absorption and effects on the hematological system. Additivity of zinc HQs with those of cadmium or mercury is, therefore, likely to overestimate chemical hazard.

Child hazards related to the key exposure pathways and COPCs may be compared to child HQs calculated using COPC concentrations in the upland reference area, shown in Table 5-50. The CTUIR child HQs calculated using the upland reference area data for mercury, cadmium, and zinc are 1.9, 1.4, and 2.8, respectively. CTUIR child HIs are highest at 100-B-14:6 and 116-B-10. As discussed in relation to the Subsistence Farmer child HI, the soil mercury concentrations at these sites are much higher than in the upland reference area. The high values of child HIs at these sites are due to the linear models used in this risk assessment to estimate plant tissue concentrations from soil data, which are likely to overestimate plant concentrations at higher soil concentrations. This is discussed in the Uncertainty Analysis (Section 5.9).

The waste sites where child HI was highest for the CTUIR Resident scenario are the same as those described for the Subsistence Farmer exposure scenario. Some of these sites were not excavated, and others were excavated to depths of between approximately 2 and 4 m (6.6 and 13.1 ft) (see Table 2-8). The use of shallow zone soil verification data to calculate local-area child HI at these excavated sites may contribute to a protective bias in these results.

5.3.5 Yakama Resident Scenario Results for the 100-B/C Area

The Yakama Resident exposure scenario encompasses potential exposures related to shallow zone soils around remediated waste sites, including both direct contact exposure pathways and exposures related to raising produce and animals in the potentially affected area of the waste site. This scenario also includes exposures related to ingestion of Columbia River fish (see Section 4.4) and exposure to groundwater via a domestic water well (see Section 6.0). It is similar to the Subsistence Farmer scenario, although exposure intensity is higher for many exposure pathways. The scenario is more fully described in the CSM in Section 3.3 and is summarized in Table 3-13 and Figure 3-17. The COPCs evaluated in shallow zone soil in the 100-B/C Decision Area are shown in Table 5-26.

5.3.5.1 Cancer Risk. Table 5-63 shows total cancer risk results for the Yakama Resident exposure scenario for all remediated waste sites in the 100-B/C Decision Area. The tabulated results are the sum of the risk values for the individual COPCs included in the risk calculation. Total risk results use EPCs based on measured soil concentrations in the individual waste sites. Risk assessment results are shown for present-day radionuclide concentrations, and with radionuclide concentrations decayed to the years 2075 and 2150. Soil concentrations for these times were calculated using the RESRAD computer code (see Appendix D-1.4.1).

The range of present-day total cancer risk results for the Yakama Resident scenario (radionuclides + chemicals) is from 6×10^{-8} to 9×10^{-3}. No carcinogenic COPCs were detected at the 1607-B-11 waste site. Per EPA guidance (EPA/540/1-89/002), cancer risk results are displayed with only one significant figure. As shown in Table 5-5, 22 of the 45 waste sites evaluated in the 100-B/C Decision Area had present-day total cancer risks above 1×10^{-4}.

A summary of the key exposure pathways and COPCs contributing 10% or more to present-day Yakama Resident cancer risk at 100-B/C Decision Area sites with total cancer risk above 1×10^{-4} is provided below.

Site	COPC	Pathway	Fraction of Total Risk	Present-Day Total Risk
116-B-6A	Strontium-90	Produce Ingestion	0.55	9×10^{-3}
	Strontium-90	Beef Ingestion	0.18	
	Cesium-137	External Irradiation	0.10	
116-B-14	Strontium-90	Produce Ingestion	0.39	4×10^{-3}
	Europium-152	External Irradiation	0.32	
	Strontium-90	Beef Ingestion	0.13	
116-C-2A	Strontium-90	Produce Ingestion	0.57	3×10^{-3}
	Strontium-90	Beef Ingestion	0.18	
116-B-11	Europium-152	External Irradiation	0.45	2×10^{-3}
	Cobalt-60	External Irradiation	0.13	
	Strontium-90	Produce Ingestion	0.11	
116-C-1	Strontium-90	Produce Ingestion	0.34	2×10^{-3}
	Europium-152	External Irradiation	0.23	
	Strontium-90	Beef Ingestion	0.11	
116-C-5	Europium-152	External Irradiation	0.28	2×10^{-3}
	Strontium-90	Produce Ingestion	0.23	
	Cesium-137	External Irradiation	0.11	
132-B-6	Europium-152	External Irradiation	0.39	1×10^{-3}
	Cesium-137	External Irradiation	0.20	

Site	COPC	Pathway	Fraction of Total Risk	Present-Day Total Risk
116-C-6	Strontium-90	Produce Ingestion	0.36	1×10^{-3}
	Cesium-137	External Irradiation	0.20	
	Strontium-90	Beef Ingestion	0.12	
100-B-8:2	Strontium-90	Produce Ingestion	0.34	9×10^{-4}
	Europium-154	External Irradiation	0.12	
	Cobalt-60	External Irradiation	0.12	
	Strontium-90	Beef Ingestion	0.11	
	Europium-152	External Irradiation	0.10	
116-B-2	Strontium-90	Produce Ingestion	0.40	8×10^{-4}
	Cesium-137	External Irradiation	0.21	
	Strontium-90	Beef Ingestion	0.13	
116-B-13	Strontium-90	Produce Ingestion	0.53	7×10^{-4}
	Strontium-90	Beef Ingestion	0.17	
116-B-1	Europium-152	External Irradiation	0.41	5×10^{-4}
	Cobalt-60	External Irradiation	0.14	
	Strontium-90	Produce Ingestion	0.12	
	Europium-154	External Irradiation	0.11	
118-B-10	Cobalt-60	External Irradiation	0.60	5×10^{-4}
	Nickel-63	Produce Ingestion	0.18	
100-B-8:1	Europium-152	External Irradiation	0.51	5×10^{-4}
	Strontium-90	Produce Ingestion	0.22	
116-B-7	Strontium-90	Produce Ingestion	0.47	4×10^{-4}
	Strontium-90	Beef Ingestion	0.15	
	Cesium-137	External Irradiation	0.11	
132-C-2	Cobalt-60	External Irradiation	0.42	4×10^{-4}
	Strontium-90	Produce Ingestion	0.26	
100-B-11	Strontium-90	Produce Ingestion	0.68	3×10^{-4}
	Strontium-90	Beef Ingestion	0.22	
118-B-3	Strontium-90	Produce Ingestion	0.26	3×10^{-4}
	Europium-152	External Irradiation	0.16	
	Cobalt-60	External Irradiation	0.16	
	Europium-154	External Irradiation	0.16	
118-C-2	Nickel-63	Produce Ingestion	0.55	2×10^{-4}
	Nickel-63	Milk Ingestion	0.26	

Site	COPC	Pathway	Fraction of Total Risk	Present-Day Total Risk
116-B-9	Strontium-90	Produce Ingestion	0.61	2×10^{-4}
	Strontium-90	Beef Ingestion	0.20	
116-B-3	Strontium-90	Produce Ingestion	0.56	2×10^{-4}
	Strontium-90	Beef Ingestion	0.18	
118-B-4	Europium-152	External Irradiation	0.55	2×10^{-4}
	Cesium-137	External Irradiation	0.24	

For a large number of the waste sites with present-day cancer risk above 1×10^{-4}, most of the cancer risk is related to just a few radionuclides, including cobalt-60, cesium-137, isotopic europium, and strontium-90. The half-lives of these radionuclides range from approximately 5 years (cobalt-60) to approximately 30 years (strontium-90 and cesium-137). Therefore, much of the site-related cancer risk will decrease naturally over time as a function of the decay of these radionuclides. There are 10 waste sites with a total cancer risk above 1×10^{-4} in year 2075, with risks mostly related to strontium-90 and cesium-137. By year 2150, the highest total cancer risk value is 3×10^{-4} at the 116-B-6A waste site and the next highest value is 1×10^{-4} at the 116-B-14 waste site.

The waste sites where cancer risk was highest for the Yakama Resident scenario are the same as those described for the Subsistence Farmer exposure scenario. These remediated waste sites were excavated to a depth of 4.6 m (15 ft) or more. The use of shallow zone soil verification data to calculate local-area risks at these sites may contribute to a significant protective bias in the cancer risk results for these sites.

5.3.5.2 Radiation Dose. Table 5-64 shows the total radiation dose results for the Yakama Resident exposure scenario for all remediated waste sites in the 100-B/C Decision Area. The tabulated results are the sum of the dose values for the individual COPCs included in the calculation. Total dose results use EPCs based on measured soil concentrations in the individual waste sites. Risk assessment results are shown for present-day radionuclide concentrations, and with radionuclide concentrations decayed to the years 2075 and 2150. Soil concentrations for these times were calculated using the RESRAD computer code, as described in Appendix D-1.4.1.

The range of present-day total radiation dose results for the Yakama Resident scenario is from 0.18 to 193 mrem/yr. No radionuclide COPCs were detected at 12 waste sites. As shown in Table 5-5, 10 of the 45 waste sites evaluated in the 100-B/C Decision Area had calculated total radiation dose values above 15 mrem/yr.

A summary of the key exposure pathways and COPCs contributing 10% or more to Yakama Resident present-day radiation dose at 100-B/C Decision Area sites with total dose above 15 mrem/yr is provided below.

Site	COPC	Pathway	Fraction of Total Dose	Present-Day Total Dose (mrem/yr)
116-B-6A	Strontium-90	Produce Ingestion	0.56	193
	Strontium-90	Beef Ingestion	0.19	
116-B-14	Strontium-90	Produce Ingestion	0.41	78
	Europium-152	External Irradiation	0.30	
	Strontium-90	Beef Ingestion	0.14	
116-C-2A	Strontium-90	Produce Ingestion	0.57	60
	Strontium-90	Beef Ingestion	0.19	
116-B-11	Europium-152	External Irradiation	0.45	41
	Cobalt-60	External Irradiation	0.13	
	Strontium-90	Produce Ingestion	0.12	
116-C-1	Strontium-90	Produce Ingestion	0.37	35
	Europium-152	External Irradiation	0.22	
	Strontium-90	Beef Ingestion	0.12	
116-C-5	Europium-152	External Irradiation	0.27	32
	Strontium-90	Produce Ingestion	0.26	
	Cesium-137	External Irradiation	0.11	
116-C-6	Strontium-90	Produce Ingestion	0.40	24
	Cesium-137	External Irradiation	0.20	
	Strontium-90	Beef Ingestion	0.13	
132-B-6	Europium-152	External Irradiation	0.42	24
	Cesium-137	External Irradiation	0.21	
100-B-8:2	Strontium-90	Produce Ingestion	0.36	19
	Strontium-90	Beef Ingestion	0.12	
	Europium-154	External Irradiation	0.12	
	Cobalt-60	External Irradiation	0.11	
116-B-13	Strontium-90	Produce Ingestion	0.54	16
	Strontium-90	Beef Ingestion	0.18	

Similar to the findings for cancer risk, the present-day radiation dose at sites with calculated dose above the threshold is driven by relatively short-lived radionuclides. The only 100-B/C remediated waste site to exceed 15 mrem/yr at year 2075 for the Yakama Resident scenario is 116-B-6A (37 mrem/yr), due primarily to strontium-90 and cesium-137. Total radiation dose results are below 15 mrem/yr at all remediated waste sites by year 2150.

The waste sites where radiation dose was highest for the Yakama Resident scenario are the same as those described for the Subsistence Farmer exposure scenario. These remediated waste sites were excavated to a depth of 4.6 m (15 ft) or more. The use of shallow zone soil verification data to calculate risks at these sites may contribute to a significant protective bias in the radiation dose results for these sites.

5.3.5.3 Chemical Hazard. Table 5-65 shows HI results for the Yakama Resident exposure scenario for all remediated waste sites in the 100-B/C Decision Area. The tabulated results are the sum of the risk values for the individual COPCs included in the risk calculation. As described in Section 3.4.4, chemical hazard is calculated separately for adult and child receptors. For the Yakama Resident scenario, child HIs are usually equal to or above those for adults. For this reason, only child HI values are presented here.

The range of child HI results for the Yakama Resident scenario is from 0.25 to 118. The HI results at the 118-B-5 remediated waste site were well below this reported range, indicating essentially no residual site-related contamination contributing to chemical hazard. No COPCs contributing to chemical hazard were detected at two waste sites (118-B-4 and 118-C-2). As shown in Table 5-5, 11 of the 45 waste sites in the 100-B/C Decision Area had calculated HI values above 10 and another 23 sites had HI values between 1.0 and 10.

A summary of the key exposure pathways and COPCs contributing 10% or more to Yakama Resident present-day HIs in the 100-B/C Decision Area is provided below. For brevity, only sites with a total HI at or above 3.0 are shown here.

Site	COPC	Pathway	Fraction of Child Hazard	Present-Day Child Hazard [a]
100-B-14:6	Mercury	Beef Ingestion	0.63	118
	Mercury	Produce Ingestion	0.30	
116-B-10	Mercury	Beef Ingestion	0.67	103
	Mercury	Produce Ingestion	0.32	
118-B-9	Mercury	Beef Ingestion	0.31	38
	Zinc	Beef Ingestion	0.17	
	Zinc	Produce Ingestion	0.17	
	Mercury	Produce Ingestion	0.15	
1607-B10	Mercury	Beef Ingestion	0.65	31
	Mercury	Produce Ingestion	0.30	
116-B-9	Mercury	Beef Ingestion	0.67	31
	Mercury	Produce Ingestion	0.31	
128-C-1	Cadmium	Produce Ingestion	0.26	25
	Mercury	Beef Ingestion	0.15	
	Copper	Produce Ingestion	0.12	
132-B-6	Mercury	Beef Ingestion	0.66	19
	Mercury	Produce Ingestion	0.31	

Site	COPC	Pathway	Fraction of Child Hazard	Present-Day Child Hazard [a]
118-B-10	Mercury	Beef Ingestion	0.66	18
	Mercury	Produce Ingestion	0.31	
100-B-14:7	Mercury	Beef Ingestion	0.25	17
	Cadmium	Produce Ingestion	0.18	
	Zinc	Beef Ingestion	0.13	
	Zinc	Produce Ingestion	0.12	
	Mercury	Produce Ingestion	0.12	
1607-B9	Mercury	Beef Ingestion	0.51	12
	Mercury	Produce Ingestion	0.24	
	Cadmium	Produce Ingestion	0.23	
1607-B8	Mercury	Beef Ingestion	0.51	11
	Mercury	Produce Ingestion	0.24	
116-C-2A	Mercury	Beef Ingestion	0.66	10
	Mercury	Produce Ingestion	0.31	
116-B-11	Mercury	Beef Ingestion	0.62	10
	Mercury	Produce Ingestion	0.29	
128-B-2	Mercury	Beef Ingestion	0.18	8.7
	Zinc	Beef Ingestion	0.14	
	Zinc	Produce Ingestion	0.14	
	Cadmium	Produce Ingestion	0.13	
	Copper	Produce Ingestion	0.10	
100-B-11	Zinc	Beef Ingestion	0.22	7.8
	Zinc	Produce Ingestion	0.22	
	Cadmium	Produce Ingestion	0.17	
	Copper	Produce Ingestion	0.17	
100-B-14:5	Zinc	Beef Ingestion	0.26	7.8
	Zinc	Produce Ingestion	0.26	
	Copper	Produce Ingestion	0.15	
	Cadmium	Produce Ingestion	0.13	
600-233	Cadmium	Produce Ingestion	0.32	5.2
	Zinc	Beef Ingestion	0.17	
	Zinc	Produce Ingestion	0.17	
	Copper	Produce Ingestion	0.14	

Site	COPC	Pathway	Fraction of Child Hazard	Present-Day Child Hazard [a]
100-B-16	Cadmium	Produce Ingestion	0.46	4.7
	Mercury	Beef Ingestion	0.34	
	Mercury	Produce Ingestion	0.16	
100-B-8:1	Mercury	Beef Ingestion	0.62	4.0
	Mercury	Produce Ingestion	0.29	
116-C-5	Mercury	Beef Ingestion	0.58	3.7
	Mercury	Produce Ingestion	0.27	
116-B-6B	Mercury	Beef Ingestion	0.61	3.5
	Mercury	Produce Ingestion	0.29	
116-B-14	Mercury	Beef Ingestion	0.51	3.1
	Mercury	Produce Ingestion	0.24	
	Uranium	Produce Ingestion	0.11	

[a] Values for mercury hazard at 100-B-14:6, 116-B-10, and other sites are likely overestimated; see discussion in the text.

Exposure to mercury, cadmium, and zinc via beef ingestion and produce ingestion are the most important analyte and exposure pathway combinations contributing to HI values above 3.0 at many of these waste sites. The HI values are calculated as the sum of chemical-specific HQs, as described in Section 3.4.4. As described in relation to Subsistence Farmer HI, critical effects underlying the oral RfD value for inorganic forms of mercury include effects on the immune system, but the toxic effects of mercury also include kidney toxicity. Cadmium toxicity following ingestion is primarily expressed in the kidneys. Therefore, adding of HQs for cadmium and mercury is reasonable. The toxic effects of chronic exposure to relatively low doses of zinc, as associated with environmental exposures, are primarily related to antagonism of copper absorption and effects on the hematological system. Additivity of zinc HQs with those of cadmium or mercury is, therefore, likely to overestimate chemical hazard.

Child hazards related to the key exposure pathways and COPCs may be compared to child HQs calculated using COPC concentrations in the upland reference area, shown in Table 5-50. The Yakama child HQs calculated using the upland reference area data for mercury, cadmium, and zinc are 1.1, 1.7, and 3.0, respectively. Yakama child HIs are highest at 100-B-14:6 and 116-B-10. As discussed in relation to the Subsistence Farmer HI, the soil mercury concentrations at these sites are much higher than in the upland reference area. The high values of child HIs at these sites is due to the linear models used in this risk assessment to estimate plant tissue concentrations from soil data, which are likely to overestimate plant concentrations at higher soil concentrations. This is discussed in the Uncertainty Analysis (Section 5.9).

The waste sites where child HI was highest for the Yakama Resident scenario are the same as those described for the Subsistence Farmer exposure scenario. Some of these sites were not excavated, and others were excavated to depths of between approximately 2 and 4 m

(6.6 and 13.1 ft) (see Table 2-8). The use of shallow zone soil verification data to calculate local-area child HI at these excavated sites may contribute to a protective bias in these results.

5.3.6 Risks Related to Lead in Soil for the 100-B/C Area

The highest calculated RME representative concentration for lead in shallow zone soil at any remediated waste site is approximately 210 mg/kg. This value applies to the 100-F-37 remediated waste site in the 100-F/100-IU-2/100-IU-6 Area. The highest RME representative concentration for lead in shallow zone soil in the 100-B/C Area is 206 mg/kg at the 116-B-4 waste site. No representative concentrations for lead at any waste site in the 100-B/C Area exceed EPA's recommended residential screening level for lead in soil of 400 mg/kg, derived using the Integrated Exposure Uptake Biokinetic (IEUBK) model that has been associated with bare soil in a play area ("Lead; Identification of Dangerous Levels of Lead; Final Rule," *Code of Federal Regulations* [40 CFR 745]). Because soil concentrations for lead are well below the most restrictive of EPA's soil screening criteria, no additional evaluation of lead is included in the HHRA. More detailed information on lead toxicity and the basis of EPA's soil screening criteria is provided in Section 3.5.8.

5.4 100-K DECISION AREA RESULTS

Section 5.4 is the second of six sections of the local-area risk assessment (5.3 through 5.8) that present the results of the soil source term risk assessment for remediated waste sites in an individual ROD decision area. A summary of the results of the waste site soil risk assessment for all ROD decision areas is shown in Tables 5-2 through 5-7. Tables 5-2, 5-3, and 5-4 show the number of remediated waste sites with risk assessment results above or below decision thresholds for the Occupational scenarios. Table 5-2 shows results using present-day radionuclide soil concentrations, and the latter two tables show the results with radionuclide concentrations at years 2075 and 2150. Tables 5-5, 5-6, and 5-7 show these same summaries for the Residential exposure scenarios. The primary purpose of these summary tables is to allow the reader to quickly see how the waste site risk assessment results for the different exposure scenarios compare to the thresholds used for risk management decisions. These tables also provide a perspective of how the waste site risk assessment results vary among the six ROD decision areas, and how cancer risk and radiation dose will change over time due to radioactive decay.

As described in Section 3.3, broad-area EPCs in the upland environment are associated with the occupational portion of the Resident National Monument/Refuge scenario and are also used to supplement the local-area cleanup verification data to calculate EPCs for dust inhalation for all scenarios evaluated in Sections 5.3 through 5.8. The COPCs for the broad-area components of the waste site risk calculations are those related to upland surface soil, while the COPCs for the local-area components of the waste site risk calculations are those related to the cleanup verification data for a particular ROD decision area. Risk assessment results for the Resident National Monument/Refuge scenario are provided for the local-area and broad-area components separately. Broad-area representative concentrations for upland surface soil are calculated using the MIS soil data from all 20 upland sampling locations, using the same protocol described for MIS reference area data in Section 5.2.4.

5.4.1 Industrial/Commercial Scenario Results for the 100-K Area

The Industrial/Commercial exposure scenario encompasses potential occupational exposures related to shallow zone soils around remediated waste sites. Like the Resident National Monument/Refuge scenario, it focuses on direct contact exposure pathways and does not encompass the raising of produce or animals, nor the ingestion of Columbia River fish. It is similar to the Industrial/Commercial scenario used by EPA to calculate soil screening values using direct contact exposure pathways such as inadvertent soil ingestion, dermal contact with soil, and inhalation. Of the scenarios evaluated on the scale of an individual remediated waste site, the Industrial/Commercial scenario has the lowest exposure intensity. The scenario is more fully described in the CSM in Section 3.3 and is summarized in Table 3-12 and Figure 3-16. The COPCs evaluated in shallow zone soil in the 100-K Decision Area are shown in Table 5-26.

5.4.1.1 Cancer Risk. Table 5-66 shows the RME and CTE total cancer risk results for the Industrial/Commercial exposure scenario for all remediated waste sites in the 100-K Decision Area. The tabulated results are the sum of the risk values for the individual COPCs included in the risk calculation. Total risk results use EPCs based on measured soil concentrations in the individual waste sites. Industrial/Commercial risk assessment results are shown for present-day radionuclide concentrations, and with radionuclide concentrations decayed to the years 2075 and 2150. Soil concentrations for these times were calculated using the RESRAD computer code, as described in Appendix D-1.4.1.

The range of present-day RME and CTE total cancer risk results for the Industrial/Commercial scenario (radionuclides + chemicals) is from 7×10^{-7} to 1×10^{-4} and from 7×10^{-8} to 1×10^{-5}, respectively. No carcinogenic COPCs were detected at the 100-K-29 waste site. Per EPA guidance (EPA/540/1-89/002), cancer risk results are displayed with only one significant figure.

As shown in Table 5-2, none of the 13 waste sites evaluated in the 100-K Decision Area had calculated total RME or CTE cancer risks for the Industrial/Commercial scenario above 1×10^{-4}. The highest RME total cancer risk in the 100-K Area is 1×10^{-4} at the 100-K-56:1 waste site, mostly due to external irradiation from europium-152. The highest total RME cancer risk in the 100-K Area for the Industrial/Commercial scenario at year 2075 is 6×10^{-6} at the 116-K-2 waste site, related primarily to cesium-137.

5.4.1.2 Radiation Dose. Table 5-67 shows the RME and CTE total radiation dose results for the Industrial/Commercial exposure scenario for all remediated waste sites in the 100-K Decision Area. The tabulated results are the sum of the dose values for the individual COPCs included in the risk calculation. Risk assessment results are shown for present-day radionuclide concentrations, and with radionuclide concentrations decayed to the years 2075 and 2150. Soil concentrations for these times were calculated using the RESRAD computer code, as described in Appendix D-1.4.1.

The range of present-day RME and CTE total radiation dose results for the Industrial/Commercial scenario is from 1.7 to 6.1 mrem/yr and from 0.59 to 2.9 mrem/yr, respectively. No radionuclide COPCs were detected at seven waste sites in the 100-K Area.

There are no waste sites in the 100-K Area where the present-day radiation dose for the Industrial/Commercial scenario exceeded the threshold of 15 mrem/yr. At year 2075, the highest RME total radiation dose for the Industrial/Commercial exposure scenario is 0.30 mrem/yr at the 116-K-2 remediated waste site.

5.4.1.3 Chemical Hazard. Table 5-68 shows the RME and CTE HI results for the Industrial/Commercial exposure scenario for all remediated waste sites in the 100-K Decision Area. The tabulated results are the sum of the risk values for the individual COPCs included in the risk calculation.

The range of RME and CTE HI results for the Industrial/Commercial scenario is from approximately 0.00012 to 0.034 and from 0.00010 to 0.012, respectively. There are no waste sites in the 100-K Area where the HI for the Industrial/Commercial scenario approaches the threshold of 1.0.

5.4.2 Resident National Monument/Refuge Scenario Results for the 100-K Area

The Resident National Monument/Refuge exposure scenario encompasses potential residential exposures related to shallow zone soils around remediated waste sites, as well as potential occupational exposures to surface soil across the upland environment of the River Corridor. It focuses on direct contact exposure pathways and does not encompass the raising of produce or animals nor the ingestion of Columbia River fish. This scenario also includes exposures related to groundwater via a domestic water well. Risks related to groundwater use are addressed in Section 6.0. With respect to soil exposure pathways, this scenario is similar to the traditional urban/suburban residential scenario used by EPA to calculate soil screening values using direct contact exposure pathways such as inadvertent soil ingestion, dermal contact with soil, and inhalation. It may be viewed as akin to the Subsistence Farmer scenario for an adult receptor, minus the ingestion of foodstuffs and with an exposure model that averages exposure to both individual waste site soils, cleanup verification soil data, and surface soils over large areas, upland surface soil data. The scenario is more fully described in the CSM in Section 3.3 and is summarized in Table 3-12 and Figure 3-16. The COPCs evaluated in shallow zone soil in the 100-K Decision Area are shown in Table 5-26.

As described in the introductory text for Section 5.4, COPCs for the broad-area components of the waste site risk calculations are those related to upland surface soil, while the COPCs for the local-area components of the waste site risk calculations are those related to the cleanup verification data for a particular ROD decision area. Results for the local-area portion of the Resident National Monument/Refuge exposure scenario are calculated for each individual remediated waste site where a residence might be situated. Risks related to the broad-area occupational component of the Resident National Monument/Refuge exposure scenario, which are additive to each of the local-area results, are shown in the first row of the summary tables for cancer risk, radiation dose, and chemical hazard. As discussed in Section 4.1.3, broad-area representative concentrations are calculated using upland surface soil data collected under DOE/RL-2005-42.

5.4.2.1 Cancer Risk. Table 5-69 shows the RME and CTE total cancer risk results for the Resident National Monument/Refuge exposure scenario for all remediated waste sites in the 100-K Decision Area. The tabulated results are the sum of the risk values for the individual COPCs included in the risk calculation. Total risk results use EPCs based on measured soil concentrations in the individual remediated waste sites (residential exposure) and River Corridor upland surface soil (occupational exposure). Risk assessment results are shown for present-day radionuclide concentrations and with radionuclide concentrations decayed to the years 2075 and 2150. Soil concentrations for these times were calculated using the RESRAD computer code, as described in Appendix D-1.4.1.

The range of present-day RME and CTE total cancer risk results for the local-area component of the Resident National Monument/Refuge scenario (radionuclides + chemicals) is from 7×10^{-9} to 2×10^{-4} and 1×10^{-9} to 2×10^{-5}, respectively. Per EPA guidance (EPA/540/1-89/002), cancer risk results are displayed with only one significant figure. As shown in Table 5-2, 1 of the 13 waste sites evaluated in the 100-K Decision Area had a combined local and broad total RME cancer risk above 1×10^{-4}, but no waste site had a combined local and broad CTE total cancer risk above 1×10^{-4}.

Because the broad-area component of cancer risk is the same across all calculations, the summary of the key exposure pathways and COPCs contributing 10% or more to present-day RME Resident National Monument/Refuge cancer risk at 100-K Decision Area sites focuses on the local-area component of the Resident National Monument/Refuge scenario. The local-area calculations with total cancer risk above 1×10^{-4} are summarized below.

Site	COPC	Pathway	Fraction of Total Risk	Present-Day Total Risk
100-K-56:1	Europium-152	External Irradiation	0.66	2×10^{-4}
	Cobalt-60	External Irradiation	0.20	
	Cesium-137	External Irradiation	0.13	

The highest present-day RME total cancer risk in the 100-K Area is 2×10^{-4} at the 100-K-56:1 waste site, due to external irradiation from short-lived radionuclides including europium-152, cobalt-60, and cesium-137. Total present-day RME cancer risks were 1×10^{-4} at 100-K-55:1, 100-K-2, and 116-KW-3, also due to short-lived radionuclides. There are no waste sites in the 100-K Area where combined local and broad-area RME cancer risk is above 1×10^{-5} at year 2075 for the Resident National Monument/Refuge exposure scenario.

The current condition of the 100-K-56:1 remediated waste site is summarized in Table 2-8. The 100-K-56:1 waste site was grouped for cleanup with the 100-K-55:1 waste site, which was excavated to a depth of 8.8 m (28.9 ft) and then backfilled to grade following remediation. The shallow zone soil verification data used to calculate local-area risk are from samples collected along the sidewalls of the excavation at depths between 0 and 4.6 m (15 ft). The use of these data to represent surface soil for current and future exposures may contribute to a significant protective bias in the Resident National Monument/Refuge risk results for this site. Uncertainties in the use of shallow zone soil verification data from excavated waste sites are discussed in more detail in Section 5.9 and Appendix D-1.

5.4.2.2 Radiation Dose. Table 5-70 shows the RME and CTE total radiation dose results for the Resident National Monument/Refuge exposure scenario for all remediated waste sites in the 100-K Decision Area. The tabulated results are the sum of the risk values for the individual COPCs included in the risk calculation. Risk assessment results are shown for present-day radionuclide concentrations, and with radionuclide concentrations decayed to the years 2075 and 2150. Soil concentrations for these times were calculated using the RESRAD computer code, as described in Appendix D-1.4.1.

The range of present-day RME and CTE total radiation dose results for the local-area component of the Resident National Monument/Refuge scenario is from 2.9 to 11 mrem/yr and from 0.98 to 5.6 mrem/yr, respectively. No radionuclide COPCs were detected at seven waste sites. The broad-area present-day RME and CTE radiation doses are 0.80 and 0.78 mrem/yr, respectively.

No remediated waste site evaluated in the 100-K Decision Area had a combined local and broad-area RME or CTE radiation dose value in excess of 15 mrem/yr. The local-area RME radiation dose of 11 mrem/yr at waste site 100-K-56:1 was mostly due to external irradiation from europium-152. The highest local-area RME radiation dose for the Resident National Monument/Refuge scenario at year 2075 was 0.57 mrem/yr at the 116-K-2 remediated waste site.

5.4.2.3 Chemical Hazard. Table 5-71 shows the RME and CTE HI results for the Resident National Monument/Refuge exposure scenario for all remediated waste sites in the 100-K Decision Area. The tabulated results are the sum of the risk values for the individual COPCs included in the risk calculation.

The range of RME and CTE HI results for the local-area component of the Resident National Monument/Refuge scenario is from 0.0099 to 0.059 and from 0.0093 to 0.027, respectively.

There are no waste sites in the 100-K Area where combined local and broad exposures result in an HI for the Resident National Monument/Refuge scenario that approaches the threshold of 1.0.

5.4.3 Subsistence Farmer Scenario Results for the 100-K Area

The Subsistence Farmer exposure scenario encompasses potential exposures related to shallow zone soils around remediated waste sites, including both direct contact exposure pathways and exposures related to raising produce and animals in the potentially affected area of the waste site. This scenario also includes exposures related to ingestion of Columbia River fish and exposure to groundwater via a domestic water well. Risks related to fish ingestion and groundwater use are addressed in Sections 4.4 and 6.0, respectively. The scenario is more fully described in the CSM in Section 3.3 and is summarized in Table 3-13 and Figure 3-17. The COPCs evaluated in shallow zone soil in the 100-K Decision Area are shown in Table 5-26.

5.4.3.1 Cancer Risk. Table 5-72 shows the RME and CTE total cancer risk results for the Subsistence Farmer exposure scenario for all remediated waste sites in the 100-K Decision Area. The tabulated results are the sum of the risk values for the individual COPCs included in the risk calculation. Total risk results use EPCs based on measured soil concentrations in the individual waste sites. Risk assessment results are shown for present-day radionuclide concentrations, and with radionuclide concentrations decayed to the years 2075 and 2150. Soil concentrations for these times were calculated using the RESRAD computer code, as described in Appendix D-1.4.1.

The range of present-day RME and CTE total cancer risk results for the Subsistence Farmer scenario (radionuclides + chemicals) is from 2×10^{-4} to 6×10^{-4} and 3×10^{-5} to 7×10^{-5}, respectively. No carcinogenic COPCs were detected at the 100-K-29 waste site. As shown in Table 5-5, 12 of the 13 waste sites evaluated in the 100-K Decision Area had calculated total RME cancer risks above 1×10^{-4}, but no waste sites had a CTE total cancer risk above 1×10^{-4}.

A summary of the key exposure pathways and COPCs contributing 10% or more to present-day RME Subsistence Farmer cancer risk at 100-K Decision Area sites with total cancer risk above 1×10^{-4} is provided below.

Site	COPC	Pathway	Fraction of Total Risk	Present-Day Total Risk
116-KE-5	Arsenic	Produce Ingestion	0.85	6×10^{-4}
100-K-56:1	Europium-152	External Irradiation	0.54	5×10^{-4}
	Cobalt-60	External Irradiation	0.16	
	Cesium-137	External Irradiation	0.11	
100-K-33	Arsenic	Produce Ingestion	0.85	5×10^{-4}
100-K-31	Arsenic	Produce Ingestion	0.85	4×10^{-4}
116-K-1	Strontium-90	Milk Ingestion	0.31	4×10^{-4}
	Strontium-90	Produce Ingestion	0.26	
	Cesium-137	External Irradiation	0.15	
	Strontium-90	Beef Ingestion	0.14	
100-K-30	Arsenic	Produce Ingestion	0.85	4×10^{-4}
100-K-32	Arsenic	Produce Ingestion	0.85	4×10^{-4}
116-KW-4	Arsenic	Produce Ingestion	0.85	4×10^{-4}
100-K-55:1	Europium-152	External Irradiation	0.38	3×10^{-4}
	Europium-154	External Irradiation	0.19	

Site	COPC	Pathway	Fraction of Total Risk	Present-Day Total Risk
	Cobalt-60	External Irradiation	0.14	
116-K-2	Cesium-137	External Irradiation	0.31	3×10^{-4}
	Europium-152	External Irradiation	0.24	
	Europium-154	External Irradiation	0.13	
116-KW-3	Europium-152	External Irradiation	0.60	3×10^{-4}
	Europium-154	External Irradiation	0.12	
	Cobalt-60	External Irradiation	0.11	
116-KE-4	Europium-152	External Irradiation	0.38	2×10^{-4}
	Europium-154	External Irradiation	0.16	
	Cobalt-60	External Irradiation	0.11	

With two exceptions (100-K-56:1 and 116-K-1), most of the present-day RME cancer risk at the remediated waste sites with the highest risks (4×10^{-4} and above) is related to arsenic by produce ingestion. As shown in Table 5-48, Subsistence Farmer RME arsenic cancer risk calculated for the upland reference area is 5×10^{-4}. The only remediated waste site where RME arsenic-related risk exceeded the reference area level is 116-KE-5. Arsenic was analyzed in only one soil sample at 116-KE-5, at a concentration of 3.7 mg/kg. This concentration is only slightly higher than the upland reference area RME concentration of 3.2 mg/kg, and is below both the Hanford Area (6.47 mg/kg) and Yakima Basin (5.13 mg/kg) 90th percentile soil background concentrations.

Short-lived radionuclides including cobalt-60, cesium-137, isotopic europium, and strontium-90 are the main contributors to present-day cancer risk exceeding 1×10^{-4} at several 100-K Area remediated waste sites. These waste sites include 100-K-56:1, 116-K-1, 100-K-55:1, 116-K-2, 116-KW-3, and 116-KE-4. The half-lives of these radionuclides range from approximately 5 years (cobalt-60) to 30 years (strontium-90 and cesium-137). Therefore, much of the site-related cancer risk will decrease naturally over time as a function of the decay of these radionuclides.

The only remediated waste sites having RME cancer risks above 1×10^{-4} at year 2075 are those where cancer risk is primarily related to arsenic. Because arsenic concentrations at the remediated waste sites where arsenic is a risk driver are similar to background and reference area levels, risks for COPCs other than arsenic are also of interest for risk management at these sites. As shown in Table 2-8, target analytes at the 116-KE-5, 100-K-30, 100-K-31, 100-K-32, 100-K-33, and 116-KW-4 waste sites were limited to metals and ethylene glycol. Radionuclides and organic chemicals other than ethylene glycol were not sampled at these sites. Carcinogenic COPCs other than arsenic for which data were obtained at these sites are limited to hexavalent chromium and cadmium, which are carcinogenic only by inhalation exposure, and for which RME Subsistence Farmer cancer risks were well below 1×10^{-6}.

As described in relation to Resident National Monument/Refuge cancer risk results, the use of shallow zone soil verification data may contribute to a significant protective bias in the risk results for excavated sites. The 100-K Area remediated waste sites with Subsistence Farmer RME cancer risks above 1×10^{-4} due to the presence of short-lived radionuclides were generally excavated to significant depths. Excavation depths listed in Table 2-8 were near or deeper than 4.6 m (15 ft) at the 100-K-56:1, 116-K-1, 100-K-55:1, 116-K-2, 116-KW-3, and 116-KE-4 waste sites. By contrast, sites where arsenic was the main contributor to Subsistence Farmer RME cancer risks above 1×10^{-4} were either unexcavated or excavated to relatively shallow depths of 1 m (3.3 ft) or less. Uncertainties in the use of shallow zone soil verification data from excavated waste sites are discussed in more detail in Section 5.9 and Appendix D-1.

As shown in Table 2-8, the estimated surface area of the 100-K-30, 100-K-31, 100-K-32, and 100-K-33 waste sites is approximately 40 m^2 (430 ft^2). As described in Section 3.3, the risk assessment calculations do not use any weighting factors to apportion a fraction of exposure to a site. When the actual area of contamination may be relatively small, this may result in grossly overestimated risks. At these particular sites, where risks are related to the produce ingestion exposure pathway, it may not be possible to raise 100% of annual fruits and vegetables in such a small area.

5.4.3.2 Radiation Dose. Table 5-73 shows the RME and CTE total radiation dose results for the Subsistence Farmer exposure scenario for all remediated waste sites in the 100-K Decision Area. The tabulated results are the sum of the dose values for the individual COPCs included in the dose calculation. Total dose results use EPCs based on measured soil concentrations in the individual waste sites. Risk assessment results are shown for present-day radionuclide concentrations, and with radionuclide concentrations decayed to the years 2075 and 2150. Soil concentrations for these times were calculated using the RESRAD computer code, as described in Appendix D-1.4.1.

The range of present-day RME and CTE total radiation dose results for the Subsistence Farmer scenario is from 10 to 29 mrem/yr and 5.0 to 10 mrem/yr, respectively. No radionuclide COPCs were detected at seven waste sites. As shown in Table 5-5, 3 of the 13 waste sites evaluated in the 100-K Decision Area had calculated total RME radiation dose values above 15 mrem/yr, but no waste sites had a CTE total radiation dose above 15 mrem/yr.

Similar to the findings for cancer risk at some sites, the present-day radiation dose results at sites with calculated dose above the threshold is driven by relatively short-lived radionuclides. As shown in Table 5-73, no 100-K Area remediated waste sites exceed 15 mrem/yr at year 2075. At year 2075, the highest total RME radiation dose value is 5.1 mrem/yr at the 116-K-1 waste site.

A summary of the key exposure pathways and COPCs contributing 10% or more to present-day RME Subsistence Farmer radiation dose results at 100-K Decision Area sites with total dose above 15 mrem/yr is provided below.

Site	COPC	Pathway	Fraction of Total Dose	Present-Day Total Dose (mrem/yr)
116-K-1	Strontium-90	Milk Ingestion	0.35	29
	Strontium-90	Produce Ingestion	0.29	
	Strontium-90	Beef Ingestion	0.16	
	Cesium-137	External Irradiation	0.10	
100-K-56:1	Europium-152	External Irradiation	0.49	24
	Cobalt-60	External Irradiation	0.14	
100-K-55:1	Europium-152	External Irradiation	0.34	16
	Europium-154	External Irradiation	0.17	
	Cobalt-60	External Irradiation	0.13	
	Strontium-90	Milk Ingestion	0.10	

As described in relation to cancer risk, remediated waste sites where short-lived radionuclides were important contributors to significant cancer risk (100-K-56:1, 116-K-1, 100-K-55:1) were excavated to a depth of greater than 4.6 m (15 ft). The use of shallow zone soil verification data to calculate local-area risks at these sites may contribute to a significant protective bias in the radiation dose results for these sites.

5.4.3.3 Chemical Hazard. Table 5-74 shows the RME and CTE HI results for the Subsistence Farmer exposure scenario for all remediated waste sites in the 100-K Decision Area. The tabulated results are the sum of the risk values for the individual COPCs included in the risk calculation. As described in Section 3.4.4, chemical hazard is calculated separately for adult and child receptors. For the Subsistence Farmer scenario, RME and CTE child HIs are usually equal to or above those for adults. For this reason, only child HI values are presented here.

The range of RME and CTE child HI results for the Subsistence Farmer scenario is from approximately 0.0018 to 222 and 0.00086 to 77, respectively. As shown in Table 5-5, 4 of the 13 waste sites in the 100-K Area had calculated total RME and CTE HI values above 10, and two sites had RME and CTE HI values between 1.0 and 10.

A summary of the key exposure pathways and COPCs contributing 10% or more to the RME Subsistence Farmer HI at 100-K Decision Area sites with HI values above 1.0 is provided below.

Site	COPC	Pathway	Fraction of Child Hazard	Present-Day Child Hazard [a]
100-K-30	Mercury	Beef Ingestion	0.69	222
	Mercury	Produce Ingestion	0.26	
100-K-31	Mercury	Beef Ingestion	0.68	68
	Mercury	Produce Ingestion	0.25	

Site	COPC	Pathway	Fraction of Child Hazard	Present-Day Child Hazard [a]
100-K-33	Mercury	Beef Ingestion	0.65	35
	Mercury	Produce Ingestion	0.24	
100-K-32	Mercury	Beef Ingestion	0.65	32
	Mercury	Produce Ingestion	0.24	
116-KE-5	Ethylene glycol	Produce Ingestion	0.38	7.5
	Arsenic	Produce Ingestion	0.34	
	Mercury	Beef Ingestion	0.12	
116-KW-4	Arsenic	Produce Ingestion	0.82	2.2

[a] Values for mercury hazard at 100-K-30, 100-K-31, 100-K-32, and 100-K-33 are likely overestimated; see discussion in the text.

Exposure to mercury via beef and produce ingestion is the most important exposure pathway and analyte combination at each of these sites, with the exception of 116-KE-5 and 116-KW-4, where exposure to arsenic (and for 116-KE-5, ethylene glycol) via produce ingestion is the largest contributor to total risk. As described in relation to cancer risk, the relatively small surface area of some sites may result in overestimated risks when exposures are related to food ingestion exposure pathways.

More than 95% of the estimated RME HI values at the 100-K-30, 100-K-31, 100-K-33, and 100-K-32 waste sites are related to mercury exposure via beef and produce ingestion. As discussed in the Uncertainty Analysis (Section 5.9), the linear models used to estimate tissue concentrations from soil data are prone to overestimation of plant and animal tissue concentrations at higher soil concentrations. The RME soil EPC values at these sites range from 2.4 mg/kg (100-K-32) to 17.5 mg/kg (100-K-30), which are much higher than the reference area RME mercury concentration of 0.022 mg/kg. Because mercury soil concentrations at these sites are approximately 100 to 1,000 times higher than background, the confidence in the RME mercury HI values based on modeled tissue concentrations is low.

At a mercury soil concentration of 17.5 mg/kg (100-K-30), the equivalent dry-weight concentration in produce is about 44 mg/kg, calculated using the mercury produce-soil ratio (2.5) described in Appendix D-1.4.1. For animal fodder, the grass-soil ratio is 1.0, which would result in a dry-weight concentration in grass of 17.5 mg/kg. As discussed in Appendix D-1.4.1 and in the Uncertainty Analysis (Section 5.9), nonlinear regression models also exist for predicting plant concentrations of certain metals. The mercury plant-soil regression model published by Bechtel (BJC/OR-133) predicts a mercury plant concentration of 1.75 mg/kg when the soil concentration is 17.5 mg/kg. Assuming the nonlinear model is more accurate at the higher soil mercury concentration of 17.5 mg/kg, this suggests mercury exposure at the 100-K-30 waste site may be overestimated by factors of about 25 and 10 for produce ingestion and beef ingestion, respectively. In that case, the mercury HQs at 100-K-30 for produce and beef ingestion would both be about six, resulting in an HI value of 12 rather than 220.

As discussed in Section 5.4.2, the HI values for the Resident National Monument/Refuge scenario at these four sites were all below 0.1. This monument worker scenario is also based in part on residential land use, but without evaluation of home-grown foods. The enormous difference in the HI values between the Resident National Monument/Refuge scenario and the Subsistence Farmer scenario at these sites is an indication of the overarching importance of the modeled mercury concentrations in foods.

As described in relation to cancer risk, 100-K Area remediated waste sites where short-lived radionuclides were drivers of cancer risk were excavated to greater depths than sites where this was not the case. The 100-K-30, 100-K-31, 100-K-33, and 100-K-32 waste sites were excavated to depths between 0.6 and 1.0 m (2 and 3.3 ft). The 116-KE-5 and 116-KW-4 waste sites were not excavated prior to interim closure. Therefore, the protocol of using shallow zone soil data to represent current site conditions at these remediated waste sites may introduce some protective bias at the 100-K-30, 100-K-31, 100-K-33, and 100-K-32 waste sites, but to a lesser degree than for sites with short-lived radionuclides where excavation depths were much greater.

5.4.4 CTUIR Resident Scenario Results for the 100-K Area

The CTUIR Resident exposure scenario encompasses potential exposures related to shallow zone soils around remediated waste sites, including both direct contact exposure pathways and exposures related to raising produce and animals in the potentially affected area of the waste site. This scenario also includes exposures related to ingestion of Columbia River fish (see Section 4.4) and exposure to groundwater via a domestic water well (see Section 6.0). It is similar to the Subsistence Farmer scenario, although exposure intensity is higher for many exposure pathways. The scenario is more fully described in the CSM in Section 3.3 and is summarized in Table 3-13 and Figure 3-17. The COPCs evaluated in shallow zone soil in the 100-K Decision Area are shown in Table 5-26.

5.4.4.1 Cancer Risk. Table 5-75 shows the total cancer risk results for the CTUIR Resident exposure scenario for all remediated waste sites in the 100-K Decision Area. The tabulated results are the sum of the risk values for the individual COPCs included in the risk calculation. Total risk results use EPCs based on measured soil concentrations in the individual waste sites. Risk assessment results are shown for present-day radionuclide concentrations, and with radionuclide concentrations decayed to the years 2075 and 2150. Soil concentrations for these times were calculated using the RESRAD computer code, as described in Appendix D-1.4.1.

The range of present-day total cancer risk results for the CTUIR Resident scenario (radionuclides + chemicals) is from 8×10^{-4} to 9×10^{-3}. No carcinogenic COPCs were detected at the 100-K-29 waste site. As shown in Table 5-5, 12 of the 13 waste sites evaluated in the 100-K Decision Area had calculated total cancer risks above 1×10^{-4}.

A summary of the key exposure pathways and COPCs contributing 10% or more to present-day CTUIR Resident cancer risk at 100-K Decision Area sites with total cancer risk above 1×10^{-4} is provided below.

Site	COPC	Pathway	Fraction of Total Risk	Present-Day Total Risk
116-KE-5	Arsenic	Produce Ingestion	0.99	9×10^{-3}
100-K-33	Arsenic	Produce Ingestion	0.99	7×10^{-3}
100-K-31	Arsenic	Produce Ingestion	0.99	7×10^{-3}
100-K-30	Arsenic	Produce Ingestion	0.99	7×10^{-3}
100-K-32	Arsenic	Produce Ingestion	0.99	7×10^{-3}
116-KW-4	Arsenic	Produce Ingestion	0.99	7×10^{-3}
116-K-1	Strontium-90	Produce Ingestion	0.81	3×10^{-3}
100-K-56:1	Europium-152	External Irradiation	0.43	2×10^{-3}
	Strontium-90	Produce Ingestion	0.29	
	Cobalt-60	External Irradiation	0.13	
100-K-55:1	Strontium-90	Produce Ingestion	0.32	1×10^{-3}
	Europium-152	External Irradiation	0.29	
	Europium-154	External Irradiation	0.14	
	Cobalt-60	External Irradiation	0.11	
116-K-2	Strontium-90	Produce Ingestion	0.32	1×10^{-3}
	Cesium-137	External Irradiation	0.24	
	Europium-152	External Irradiation	0.18	
	Europium-154	External Irradiation	0.10	
116-KW-3	Europium-152	External Irradiation	0.52	8×10^{-4}
	Strontium-90	Produce Ingestion	0.19	
	Europium-154	External Irradiation	0.10	
116-KE-4	Strontium-90	Produce Ingestion	0.36	8×10^{-4}
	Europium-152	External Irradiation	0.28	
	Europium-154	External Irradiation	0.12	

Almost 100% of the present-day total cancer risk at the remediated waste sites with the highest risks (7×10^{-3} to 9×10^{-3}) related to arsenic by produce ingestion. However, CTUIR Resident arsenic cancer risk calculated for the upland reference area is 8×10^{-3}. Only at the 116-KE-5 waste site is arsenic-related cancer risk higher than arsenic risk levels in the upland reference areas. As described in relation to Subsistence Farmer cancer risk, arsenic was analyzed in only one soil sample at the 116-KE-5 remediated waste site, at a concentration of 3.7 mg/kg. The upland reference areas RME value for arsenic is 3.2 mg/kg. Cancer risks related to COPCs other than arsenic at the six remediated waste sites where arsenic is a risk driver for results greater than 1×10^{-4} are negligible.

The six waste sites with present-day cancer risks between 8×10^{-4} and 3×10^{-3} include 116-K-1, 100-K-56:1, 100-K-55:1, 116-K-2, 116-KW-3, and 116-KE-4. At these sites, short-lived radionuclides including cobalt-60, cesium-137, isotopic europium, and strontium-90 are the main contributors to cancer risk. The half-lives of these radionuclides range from approximately 5 years (cobalt-60) to 30 years (strontium-90 and cesium-137). Therefore, much of the site-related cancer risk will decrease naturally over time as a function of the decay of these radionuclides. Only the waste sites where cancer risk is related to arsenic have total risks above 1×10^{-4} at year 2150.

The 100-K Area waste sites where cancer risk was highest for the CTUIR Resident scenario are the same as those described for the Subsistence Farmer exposure scenario in Section 5.4.3. The remediated waste sites with Subsistence Farmer RME cancer risks above 1×10^{-4} due to the presence of short-lived radionuclides were generally excavated to significant depth, whereas sites where arsenic was the main contributor to Subsistence Farmer RME cancer risks above 1×10^{-4} were either unexcavated or excavated to relatively shallow depths of 1 m (3.3 ft) or less. The use of shallow zone soil verification data to calculate local-area risks at those sites where short-lived radionuclides are the primary risk drivers may contribute to a significant protective bias in the cancer risk results for these sites. As described in relation to Subsistence Farmer risks in Section 5.4.3, the relatively small surface area of some sites may result in overestimated risks when exposures are related to food ingestion exposure pathways.

5.4.4.2 Radiation Dose. Table 5-76 shows the total radiation dose results for the CTUIR Resident exposure scenario for all remediated waste sites in the 100-K Decision Area. The tabulated results are the sum of the dose values for the individual COPCs included in the dose calculation. Total dose results use EPCs based on measured soil concentrations in the individual waste sites. Risk assessment results are shown for present-day radionuclide concentrations, and with radionuclide concentrations decayed to the years 2075 and 2150. Soil concentrations for these times were calculated using the RESRAD computer code, as described in Appendix D-1.4.1.

The range of present-day total radiation dose results for the CTUIR Resident scenario is from 15 to 62 mrem/yr. The range of present-day above-reference radiation dose results is from 12 to 59 mrem/yr. No radionuclide COPCs were detected at seven waste sites. As shown in Table 5-5, 5 of the 13 waste sites evaluated in the 100-K Decision Area had calculated total radiation dose values above 15 mrem/yr.

Similar to the findings for cancer risk at some sites, the present-day radiation dose at sites with calculated dose above the threshold is driven by relatively short-lived radionuclides. As shown in Table 5-6, no 100-K waste sites exceed 15 mrem/yr at year 2075. At year 2075, the highest total radiation dose value is 11 mrem/yr at the 116-K-1 waste site.

A summary of the key exposure pathways and COPCs contributing 10% or more to present-day CTUIR Resident radiation dose results at 100-K Decision Area sites with total dose above 15 mrem/yr is provided below. As described for CTUIR cancer risk, the use of shallow zone soil verification data to calculate local-area risks at those sites where short-lived radionuclides are the primary risk drivers may contribute to a significant protective bias in the radiation dose results for these sites.

Site	COPC	Pathway	Fraction of Total Dose	Present-Day Total Dose (mrem/yr)
116-K-1	Strontium-90	produceIngestion	0.84	62
100-K-56:1	Europium-152	External Irradiation	0.40	34
	Strontium-90	Produce Ingestion	0.34	
	Cobalt-60	External Irradiation	0.12	
100-K-55:1	Strontium-90	Produce Ingestion	0.37	23
	Europium-152	External Irradiation	0.27	
	Europium-154	External Irradiation	0.13	
116-K-2	Strontium-90	Produce Ingestion	0.38	18
	Cesium-137	External Irradiation	0.23	
	Europium-152	External Irradiation	0.17	
116-KW-3	Europium-152	External Irradiation	0.49	16
	Strontium-90	Produce Ingestion	0.23	

5.4.4.3 Chemical Hazard. Table 5-77 shows the HI results for the CTUIR Resident exposure scenario for all remediated waste sites in the 100-K Decision Area. The tabulated results are the sum of the risk values for the individual COPCs included in the risk calculation. As described in Section 3.4.4, chemical hazard is calculated separately for adult and child receptors. For the CTUIR Resident scenario, child HIs are usually equal to or above those for adults. For this reason, only child HI values are presented here.

The range of child HI results for the CTUIR Resident scenario is from approximately 0.0017 to 1,100. As shown in Table 5-5, 6 of the 13 waste sites in the 100-K Area had calculated total HI values above 10, but the remaining sites had HI values below 1.0.

A summary of the key exposure pathways and COPCs contributing 10% or more to the CTUIR Resident child HI at 100-K Decision Area sites with HI values above 1.0 is provided below.

Site	COPC	Pathway	Fraction of Child Hazard	Present-Day Child Hazard [a]
100-K-30	Mercury	Produce Ingestion	0.73	1,100
	Mercury	Beef Ingestion	0.24	
100-K-31	Mercury	Produce Ingestion	0.69	349
	Mercury	Beef Ingestion	0.22	
100-K-33	Mercury	Produce Ingestion	0.64	189
	Mercury	Beef Ingestion	0.21	
	Arsenic	Produce Ingestion	0.15	
100-K-32	Mercury	Produce Ingestion	0.63	176

Site	COPC	Pathway	Fraction of Child Hazard	Present-Day Child Hazard [a]
	Mercury	Beef Ingestion	0.21	
	Arsenic	Produce Ingestion	0.14	
116-KE-5	Ethylene glycol	Produce Ingestion	0.47	85
	Arsenic	Produce Ingestion	0.42	
116-KW-4	Arsenic	Produce Ingestion	0.99	26

[a] Values for mercury hazard at 100-K-30, 100-K-31, 100-K-32, and 100-K-33 are likely overestimated; see discussion in the text.

Exposure to mercury via produce and beef ingestion is the most important exposure pathway and analyte combination at each of these sites, with the exception of 116-KE-5 and 116-KW-4, where exposure to arsenic (and for 116-KE-5, ethylene glycol) via produce ingestion is the largest contributor to total risk. As discussed in the Uncertainty Analysis (Section 5.9), the linear models used to estimate tissue concentrations from soil data are prone to overestimation of plant and animal tissue concentrations at higher soil concentrations. The soil EPC values at these sites range from 2.4 mg/kg (100-K-32) to 17.5 mg/kg (100-K-30), which are much higher than the reference area 95% UCL mercury concentration of 0.022 mg/kg. Because mercury soil concentrations at these sites are approximately 100 to 1,000 times higher than background, the confidence in the 95% UCL mercury HI values based on modeled tissue concentrations is low. Additional discussion on this subject is provided in relation to the Subsistence Farmer HI in Section 5.4.3. As described in relation to Subsistence Farmer risks in Section 5.4.3, the relatively small surface area at the 100-K-30, 100-K-31, 100-K-32, and 100-K-33 waste sites may result in overestimated risks when exposures are related to food ingestion exposure pathways.

5.4.5 Yakama Resident Scenario Results for the 100-K Area

The Yakama Resident exposure scenario encompasses potential exposures related to shallow zone soils around remediated waste sites, including both direct contact exposure pathways and exposures related to raising produce and animals in the potentially affected area of the waste site. This scenario also includes exposures related to ingestion of Columbia River fish (see Section 4.4) and exposure to groundwater via a domestic water well (see Section 6.0). It is similar to the Subsistence Farmer scenario, although exposure intensity is higher for many exposure pathways. The scenario is more fully described in the CSM in Section 3.3 and is summarized in Table 3-13 and Figure 3-17. The COPCs evaluated in shallow zone soil in the 100-K Decision Area are shown in Table 5-26.

5.4.5.1 Cancer Risk. Table 5-78 shows the total cancer risk results for the Yakama Resident exposure scenario for all remediated waste sites in the 100-K Decision Area. The tabulated results are the sum of the risk values for the individual COPCs included in the risk calculation. Total risk results use EPCs based on measured soil concentrations in the individual waste sites. Risk assessment results are shown for present-day radionuclide concentrations, and with radionuclide concentrations decayed to the years 2075 and 2150. Soil concentrations for these times were calculated using the RESRAD computer code, as described in Appendix D-1.4.1. The range of present-day total cancer risk results for the Yakama Resident scenario (radionuclides + chemicals) is from 9×10^{-4} to 9×10^{-3}. No carcinogenic COPCs were detected at the 100-K-29 waste site. As shown in Table 5-5, 12 of the 13 waste sites evaluated in the 100-K Decision Area had calculated total cancer risks above 1×10^{-4}.

A summary of the key exposure pathways and COPCs contributing 10% or more to present-day Yakama Resident cancer risk at 100-K Decision Area sites with total cancer risk above 1×10^{-4} is provided below.

Site	COPC	Pathway	Fraction of Total Risk	Present-Day Total Risk
116-KE-5	Arsenic	Produce Ingestion	0.97	9×10^{-3}
100-K-33	Arsenic	Produce Ingestion	0.97	7×10^{-3}
100-K-31	Arsenic	Produce Ingestion	0.97	7×10^{-3}
100-K-30	Arsenic	Produce Ingestion	0.97	6×10^{-3}
100-K-32	Arsenic	Produce Ingestion	0.97	6×10^{-3}
116-KW-4	Arsenic	Produce Ingestion	0.97	6×10^{-3}
116-K-1	Strontium-90	Produce Ingestion	0.60	4×10^{-3}
	Strontium-90	Beef Ingestion	0.19	
100-K-56:1	Europium-152	External Irradiation	0.36	2×10^{-3}
	Strontium-90	Produce Ingestion	0.25	
	Cobalt-60	External Irradiation	0.11	
100-K-55:1	Strontium-90	Produce Ingestion	0.27	1×10^{-3}
	Europium-152	External Irradiation	0.25	
	Europium-154	External Irradiation	0.12	
116-K-2	Strontium-90	Produce Ingestion	0.26	1×10^{-3}
	Cesium-137	External Irradiation	0.19	
	Europium-152	External Irradiation	0.14	
116-KW-3	Europium-152	External Irradiation	0.47	9×10^{-4}
	Strontium-90	Produce Ingestion	0.18	
116-KE-4	Strontium-90	Produce Ingestion	0.30	9×10^{-4}
	Europium-152	External Irradiation	0.23	

Almost 100% of the present-day total cancer risk at the remediated waste sites with the highest risks (6×10^{-3} to 9×10^{-3}) is related to arsenic by produce ingestion. However, Yakama Resident arsenic cancer risk calculated for the upland reference area is 8×10^{-3}. Only at the 116-KE-5 waste site is arsenic-related cancer risk higher than arsenic risk levels in the upland reference areas. Arsenic was analyzed in only one soil sample at the 116-KE-5 remediated waste site, at a concentration of 3.7 mg/kg. The RME arsenic concentration for the upland reference areas is

3.2 mg/kg. Cancer risks related to COPCs other than arsenic at the six remediated waste sites where arsenic is a risk driver for results greater than 1×10^{-4} are negligible.
The six waste sites with present-day cancer risks between 9×10^{-4} and 4×10^{-3} include 116-K-1, 100-K-56:1, 100-K-55:1, 116-K-2, 116-KW-3, and 116-KE-4. At these sites, short-lived radionuclides including cobalt-60, cesium-137, isotopic europium, and strontium-90 are the main contributors to cancer risk. The half-lives of these radionuclides range from approximately 5 years (cobalt-60) to 30 years (strontium-90 and cesium-137). Therefore, much of the site-related cancer risk will decrease naturally over time as a function of the decay of these radionuclides. Only the waste sites where cancer risk is related to arsenic have risks above 1×10^{-4} at year 2150.

The 100-K Area waste sites where cancer risk was highest for the Yakama Resident scenario are the same as those described for the Subsistence Farmer exposure scenario in Section 5.4.3. The remediated waste sites with Subsistence Farmer RME cancer risks above 1×10^{-4} due to the presence of short-lived radionuclides were generally excavated to significant depth, whereas sites where arsenic was the main contributor to Subsistence Farmer RME cancer risks above 1×10^{-4} were either unexcavated or excavated to relatively shallow depths of 1 m (3.3 ft) or less. The use of shallow zone soil verification data to calculate local-area risks at those sites where short-lived radionuclides are the primary risk-drivers may contribute to a significant protective bias in the cancer risk results for these sites. As described in relation to Subsistence Farmer risks in Section 5.4.3, the relatively small surface area at some 100-K Area waste sites may result in overestimated risks when exposures are related to food ingestion exposure pathways.

5.4.5.2 Radiation Dose. Table 5-79 shows the total radiation dose results for the Yakama Resident exposure scenario for all remediated waste sites in the 100-K Decision Area. The tabulated results are the sum of the dose values for the individual COPCs included in the dose calculation. Total dose results use EPCs based on measured soil concentrations in the individual waste sites. Risk assessment results are shown for present-day radionuclide concentrations, and with radionuclide concentrations decayed to the years 2075 and 2150. Soil concentrations for these times were calculated using the RESRAD computer code, as described in Appendix D-1.4.1.

The range of present-day total radiation dose results for the Yakama Resident scenario is from 17 to 77 mrem/yr. No radionuclide COPCs were detected at seven waste sites.

As shown in Table 5-5, 6 of the 13 waste sites evaluated in the 100-K Decision Area had calculated total radiation dose values above 15 mrem/yr.

Similar to the findings for cancer risk at some sites, the present-day radiation dose results at sites with calculated dose values above the threshold is driven by relatively short-lived radionuclides. As shown in Table 5-6, no 100-K waste sites exceed 15 mrem/yr at year 2075. At year 2075, the highest total radiation dose value is 13 mrem/yr at the 116-K-1 waste site.

A summary of the key exposure pathways and COPCs contributing 10% or more to present-day Yakama Resident radiation dose results at 100-K Decision Area sites with total dose above

15 mrem/yr is provided below. As described for Yakama Resident cancer risk, the use of shallow zone soil verification data to calculate local-area risks at those sites where short-lived radionuclides are the primary risk-drivers may contribute to a significant protective bias in the radiation dose results for these sites.

Site	COPC	Pathway	Fraction of Total Dose	Present-Day Total Dose (mrem/yr)
116-K-1	Strontium-90	Produce Ingestion	0.60	77
	Strontium-90	Beef Ingestion	0.20	
100-K-56:1	Europium-152	External Irradiation	0.35	38
	Strontium-90	Produce Ingestion	0.27	
	Cobalt-60	External Irradiation	0.10	
100-K-55:1	Strontium-90	Produce Ingestion	0.29	26
	Europium-152	External Irradiation	0.24	
	Europium-154	External Irradiation	0.12	
116-K-2	Strontium-90	Produce Ingestion	0.29	21
	Cesium-137	External Irradiation	0.19	
	Europium-152	External Irradiation	0.15	
116-KW-3	Europium-152	External Irradiation	0.45	17
	Strontium-90	Produce Ingestion	0.19	
116-KE-4	Strontium-90	Produce Ingestion	0.33	17
	Europium-152	External Irradiation	0.22	
	Strontium-90	Beef Ingestion	0.11	

5.4.5.3 Chemical Hazard. Table 5-80 shows the HI results for the Yakama Resident exposure scenario for all remediated waste sites in the 100-K Decision Area. The tabulated results are the sum of the risk values for the individual COPCs included in the risk calculation. As described in Section 3.4.4, chemical hazard is calculated separately for adult and child receptors. For the Yakama Resident scenario, child HIs are usually equal to or above those for adults. For this reason, only child HI values are presented here.

The range of child HI results for the Yakama Resident scenario is from approximately 0.0046 to 1,400. As shown in Table 5-5, 6 of the 13 waste sites in the 100-K Area had calculated total HI values above 10, but the remaining sites had HI values below 1.0.

A summary of the key exposure pathways and COPCs contributing 10% or more to the Yakama Resident HI at 100-K Decision Area sites with HI values above 1.0 is provided below.

Site	COPC	Pathway	Fraction of Child Hazard	Present-Day Child Hazard [a]
100-K-30	Mercury	Beef Ingestion	0.67	1,400

Site	COPC	Pathway	Fraction of Child Hazard	Present-Day Child Hazard [a]
	Mercury	Produce Ingestion	0.31	
100-K-31	Mercury	Beef Ingestion	0.65	425
	Mercury	Produce Ingestion	0.30	
100-K-32	Mercury	Beef Ingestion	0.62	204
	Mercury	Produce Ingestion	0.29	
100-K-33	Mercury	Beef Ingestion	0.62	221
	Mercury	Produce Ingestion	0.29	
116-KE-5	Ethylene glycol	Produce Ingestion	0.42	51
	Arsenic	Produce Ingestion	0.38	
	Mercury	Beef Ingestion	0.11	
116-KW-4	Arsenic	Produce Ingestion	0.95	14

[a] Values for mercury hazard at 100-K-30, 100-K-31, 100-K-32, and 100-K-33 are likely overestimated; see discussion in the text.

Exposure to mercury via beef and produce ingestion is the most important exposure pathway and analyte combination at each of these sites, with the exception of 116-KE-5 and 116-KW-4, where exposure to arsenic (and for 116-KE-5, ethylene glycol) via produce ingestion is the largest contributor to total risk. As discussed in the Uncertainty Analysis (Section 5.9), the linear models used to estimate tissue concentrations from soil data are prone to overestimation of plant and animal tissue concentrations at higher soil concentrations. The soil EPC values at these sites range from 2.4 mg/kg (100-K-32) to 17.5 mg/kg (100-K-30), which are much higher than the reference area 95% UCL mercury concentration of 0.022 mg/kg. Because mercury soil concentrations at these sites are approximately 100 to 1,000 times higher than background, the confidence in the 95% UCL mercury HI values based on modeled tissue concentrations is low. Additional discussion on this subject is provided in relation to the Subsistence Farmer HI in Section 5.4.3. As described in relation to Subsistence Farmer risks in Section 5.4.3, the relatively small surface area at the 100-K-30, 100-K-31, 100-K-32, and 100-K-33 waste sites may result in overestimated risks when exposures are related to food ingestion exposure pathways.

5.4.6 Risks Related to Lead in Soil for the 100-K Area

The highest calculated RME representative concentration for lead in shallow zone soil at any remediated waste site is approximately 210 mg/kg. This concentration applies to the 100-F-37 remediated waste site in the 100-F/100-IU-2/100-IU-6 Area. The highest calculated RME representative concentration for lead concentration in the 100-K Area is 63 mg/kg in shallow zone soil at the 100-K-29 waste site. No representative concentrations for lead at any waste site exceed EPA’s recommended residential screening level for lead in soil of 400 mg/kg, derived using the IEUBK model that has been associated with bare soil in a play area (40 CFR 745). No additional evaluation of lead is included in the HHRA because soil concentrations for lead are well below the most restrictive of EPA's soil screening criteria. More detailed information on lead toxicity and the basis of EPA's soil screening criteria is provided in Section 3.5.8.

5.5 100-N DECISION AREA RESULTS

Section 5.5 is the third of six sections of the local-area risk assessment (Sections 5.3 through 5.8) that present the results of the soil source term risk assessment for remediated waste sites in an individual ROD decision area. A summary of the results of the waste site soils risk assessment for all ROD decision areas is shown in Tables 5-2 through 5-7. Tables 5-2, 5-3, and 5-4 show the number of remediated waste sites with risk assessment results above or below decision thresholds for the Occupational scenarios. Table 5-2 shows results using present-day radionuclide soil concentrations, and the latter two tables show the results with radionuclide concentrations at years 2075 and 2150. Tables 5-5, 5-6, and 5-7 show these same summaries for the Residential exposure scenarios. The primary purpose of these summary tables is to allow the reader to quickly see how the waste sites risk assessment results for the different exposure scenarios compare to the thresholds used for risk management decisions. These tables also provide a perspective of how the waste site risk assessment results vary among the six ROD decision areas, and how cancer risk and radiation dose will change over time due to radioactive decay. There are only two remediated waste sites evaluated in this report in the 100-N Decision Area: 116-N-3 and 120-N-1.

As discussed in Section 5.2.3, the cleanup verification data sets include results for nitrate, nitrite, and the sum of both nitrate and nitrite. Because toxicity criteria are provided by EPA for nitrite and nitrate individually, these data are used preferentially in the risk calculations. At the 116-N-3 waste site, there is data for only the sum of nitrate and nitrite. A review of the data at the other waste site in the 100-N Area (120-N-1), as well as data for other waste sites, indicates that nitrate is far more commonly detected than nitrite, and that it is present at higher concentrations when both are detected. Therefore, the results for the sum of nitrate and nitrite at 116-N-3 are assigned to nitrate for the chemical hazard calculations.

As described in Section 3.3, broad-area EPCs in the upland environment are associated with the occupational portion of the Resident National Monument/Refuge scenario and are also used to supplement the local-area cleanup verification data to calculate EPCs for dust inhalation for all scenarios evaluated in Sections 5.3 through 5.8. The COPCs for the broad-area components of the waste site risk calculations are those related to upland surface soil, while the COPCs for the local-area components of the waste site risk calculations are those related to the cleanup verification data for a particular ROD decision area. Risk assessment results for the Resident National Monument/Refuge scenario are provided for the local-area and broad-area components separately. Broad-area representative concentrations for upland surface soil are calculated using the MIS soil data from all 20 upland sampling locations, using the same protocol described for MIS reference area data in Section 5.2.4.

5.5.1 Industrial/Commercial Scenario Results for the 100-N Area

The Industrial/Commercial exposure scenario encompasses potential occupational exposures related to shallow zone soils around remediated waste sites. Like the Resident National Monument/Refuge scenario, it focuses on direct contact exposure pathways and does not encompass the raising of produce or animals, nor the ingestion of Columbia River fish. It is similar to the Industrial/Commercial scenario used by EPA to calculate soil screening values using direct contact exposure pathways such as inadvertent soil ingestion, dermal contact with soil, and inhalation. Of the scenarios evaluated on the scale of an individual remediated waste site, the Industrial/Commercial scenario has the lowest exposure intensity. The scenario is more fully described in the CSM in Section 3.3 and is summarized in Table 3-12 and Figure 3-16.

Contaminants of potential concern evaluated in shallow zone soil in the 100-N Decision Area are shown in Table 5-26. The COPCs identified at the two 100-N Area remediated waste sites evaluated in this report, 116-N-3 and 120-N-1, are limited to cobalt-60, mercury, and nitrogen in nitrate and nitrite.

5.5.1.1 Cancer Risk. Table 5-81 shows the RME and CTE total cancer risk results for the Industrial/Commercial exposure scenario for the 116-N-3 remediated waste site; no carcinogenic COPCs were detected at the 120-N-1 remediated waste site. The tabulated results are the sum of the risk values for the individual COPCs included in the risk calculation. Total risk results use EPCs based on measured soil concentrations in the individual waste sites. Risk assessment results are shown for present-day radionuclide concentrations, and with radionuclide concentrations decayed to the years 2075 and 2150. Soil concentrations for these times were calculated using the RESRAD computer code, as described in Appendix D-1.4.1.

The present-day RME and CTE total cancer risk results for the Industrial/Commercial scenario (radionuclides + chemicals) at the 116-N-3 remediated waste site are 1×10^{-5} and 2×10^{-6}, respectively. No carcinogenic COPCs were identified at the 120-N-1 waste site. Per EPA guidance (EPA/540/1-89/002), cancer risk results are displayed with only one significant figure.

No waste site in the 100-N Area has an RME or CTE total cancer risk above 1×10^{-4} for the Industrial/Commercial exposure scenario. By year 2075, the RME total cancer risk at the 116-N-3 waste site is 6×10^{-9}, reflecting decay of the short-lived radionuclide cobalt-60.

5.5.1.2 Radiation Dose. Table 5-82 shows the RME and CTE total radiation dose results for the Industrial/Commercial exposure scenario for the 116-N-3 waste site. The tabulated results are the sum of the risk values for the individual COPCs included in the risk calculation. Total dose results use EPCs based on measured soil concentrations in the individual waste sites. Risk assessment results are shown for present-day radionuclide concentrations, and with radionuclide concentrations decayed to the years 2075 and 2150. Soil concentrations for these times were calculated using the RESRAD computer code, as described in Appendix D-1.4.1.

The 116-N-3 waste site in the 100-N Decision Area has a total RME radiation dose of 0.56 mrem/yr and a total CTE radiation dose of 0.39 mrem/yr. No radionuclide COPCs were identified at the 120-N-1 waste site.

There are no waste sites in the 100-N Area where the present-day radiation dose for the Resident National Monument/Refuge scenario approaches the threshold of 15 mrem/yr. By year 2075, the RME total radiation dose at the 116-N-3 waste site is 0.00012 mrem/yr, reflecting decay of the short-lived radionuclide cobalt-60.

5.5.1.3 Chemical Hazard. Table 5-83 shows the RME and CTE HI results for the Industrial/Commercial exposure scenario for the 120-N-1 and 116-N-3 remediated waste sites in the 100-N Decision Area. The tabulated results are the sum of the risk values for the individual COPCs included in the risk calculation.

The RME and CTE HI results for the Industrial/Commercial scenario at the 120-N-1 waste site are 0.00018 and 0.00013, respectively. The RME and CTE HI results at the 116-N-3 waste site are 0.00011 and 0.000099, respectively. There are no waste sites in the 100-N Area, where the HI for the Industrial/Commercial scenario approaches the threshold of 1.0.

5.5.2 Resident National Monument/Refuge Scenario Results for the 100-N Area

The Resident National Monument/Refuge exposure scenario encompasses potential residential exposures related to shallow zone soils around remediated waste sites, as well as potential occupational exposures to surface soil across the upland environment of the River Corridor. It focuses on direct contact exposure pathways and does not encompass the raising of produce or animals nor the ingestion of Columbia River fish. This scenario also includes exposures related to groundwater via a domestic water well. Risks related to groundwater use are addressed in Section 6.0. With respect to soil exposure pathways, this scenario is similar to the traditional urban/suburban residential scenario used by EPA to calculate soil screening values using direct contact exposure pathways such as inadvertent soil ingestion, dermal contact with soil, and inhalation. It may be viewed as akin to the Subsistence Farmer scenario for an adult receptor, minus the ingestion of foodstuffs and with an exposure model that averages exposure to both individual waste site soils (cleanup verification soil data) and surface soils over large areas (upland surface soil data). The scenario is more fully described in the CSM in Section 3.3 and is summarized in Table 3-12 and Figure 3-16.

COPCs evaluated in shallow zone soil in the 100-N Decision Area are shown in Table 5-26. The COPCs identified at the two 100-N Area remediated waste sites evaluated in this report, 116-N-3 and 120-N-1, are limited to cobalt-60, mercury, and nitrogen in nitrate and nitrite.

As described in the introductory text for Section 5.5, COPCs for the broad-area components of the waste site risk calculations are those related to upland surface soil, while the COPCs for the local-area components of the waste site risk calculations are those related to the cleanup verification data for a particular ROD decision area. Results for the local-area portion of the

Resident National Monument/Refuge exposure scenario are calculated for each individual remediated waste site where a residence might be situated. Risks related to the broad-area occupational component of the Resident National Monument/Refuge exposure scenario, which are additive to each of the local-area results, are shown in the first row of the summary tables for cancer risk, radiation dose, and chemical hazard. As discussed in Section 4.1.3, broad-area representative concentrations are calculated using surface soil data collected under DOE/RL-2005-42.

5.5.2.1 Cancer Risk. Table 5-84 shows the RME and CTE total cancer risk results for the Resident National Monument/Refuge exposure scenario for the 116-N-3 waste site; no carcinogenic COPCs were detected at the 120-N-1 waste site. The tabulated results are the sum of the risk values for the individual COPCs included in the risk calculation. Total risk results use EPCs based on measured soil concentrations in the individual remediated waste sites. Risk assessment results are shown for present-day radionuclide concentrations, and with radionuclide concentrations decayed to the years 2075 and 2150. Soil concentrations for these times were calculated using the RESRAD computer code, as described in Appendix D-1.4.1.

The present-day RME and CTE total cancer risk results for the local-area component of the Resident National Monument/Refuge scenario (radionuclides + chemicals) at the 116-N-3 waste site are 2×10^{-5} and 3×10^{-6}, respectively. Per EPA guidance (EPA/540/1-89/002), cancer risk results are displayed with only one significant figure.

The 116-N-3 waste site in the 100-N Decision Area had a combined present-day local and broad RME cancer risk of 4×10^{-5}. Approximately 95% of the local-area component of RME and CTE cancer risk is related to cobalt-60 via external irradiation. The half-life of cobalt-60 is approximately 5 years. About 80% to 90% of the CTE and RME broad-area cancer risk values are related to external exposure to europium-152, with a half-life of 13.5 years. Much of the local and broad-area cancer risk will decrease naturally over time as a function of radioactive decay. By year 2075, the combined local and broad RME cancer risk at the 116-N-3 waste site is 2×10^{-6}.

As noted in Table 2-8, the 116-N-3 waste site was excavated to a depth of approximately 6.5 m (21.3 ft). The shallow zone soil verification data used to calculate local-area risks at this site are from samples collected along the sidewalls of the excavation at depths between 0 and 4.6 m (15 ft). The use of these data to represent surface soil for current and future exposures may contribute to a significant protective bias in the local-area risk results for this site. Uncertainties in the use of shallow zone soil verification data from excavated waste sites are discussed in more detail in Section 5.9 and Appendix D-1.

5.5.2.2 Radiation Dose. Table 5-85 shows the RME and CTE total radiation dose results for the Resident National Monument/Refuge exposure scenario for the 116-N-3 waste site. No radionuclide COPCs were detected at the 120-N-1 waste site. The tabulated results are the sum of the risk values for the individual COPCs included in the risk calculation. Total dose results use EPCs based on measured soil concentrations in the individual waste sites. Risk assessment results are shown for present-day radionuclide concentrations, and with radionuclide concentrations decayed to the years 2075 and 2150. Soil concentrations for these times were calculated using the RESRAD computer code, as described in Appendix D-1.4.1.

The 116-N-3 waste site has present-day RME and CTE radiation dose results of 0.98 mrem/yr and 0.65 mrem/yr for the local-area component of exposure. The broad-area present-day RME and CTE radiation doses are 0.80 and 0.78 mrem/yr, respectively. No radionuclide COPCs were identified at the 120-N-1 waste site. There are no waste sites in the 100-N Area where the present-day radiation dose results for the Resident National Monument/Refuge scenario approach the threshold of 15 mrem/yr.

5.5.2.3 Chemical Hazard. Table 5-86 shows the RME and CTE HI results for the Resident National Monument/Refuge exposure scenario for the 120-N-1 remediated waste site in the 100-N Decision Area. The tabulated results are the sum of the risk values for the individual COPCs included in the risk calculation. No COPCs contributing to chemical hazard were identified at the 116-N-3 waste site.

The RME and CTE HI results for the local-area component of the Resident National Monument/Refuge scenario at the 120-N-1 waste site are approximately 0.002 and 0.0018, respectively. The RME and CTE HI results at the 116-N-3 waste site are 0.0019 and 0.0018, respectively. The broad-area RME and CTE HI results are 0.025 and 0.0079, respectively. There are no waste sites in the 100-N Area, where the HI for the Resident National Monument/Refuge scenario approaches the threshold of 1.0.

5.5.3 Subsistence Farmer Scenario Results for the 100-N Area

The Subsistence Farmer exposure scenario encompasses potential exposures related to shallow zone soils around remediated waste sites, including both direct contact exposure pathways and exposures related to raising produce and animals in the potentially affected area of the waste site. This scenario also includes exposures related to ingestion of Columbia River fish and exposure to groundwater via a domestic water well. Risks related to fish ingestion and groundwater use are addressed in Sections 4.4 and 6.0, respectively. The scenario is more fully described in the CSM in Section 3.3 and is summarized in Table 3-13 and Figure 3-17.

Contaminants of potential concern evaluated in shallow zone soil in the 100-N Decision Area are shown in Table 5-26. The COPCs identified at the two 100-N Area remediated waste sites evaluated in this report, 116-N-3 and 120-N-1, are limited to cobalt-60, mercury, and nitrogen in nitrate and nitrite.

5.5.3.1 Cancer Risk. Table 5-87 shows the RME and CTE total cancer risk results for the Subsistence Farmer exposure scenario for the 116-N-3 waste site; no carcinogenic COPCs were detected at the 120-N-1 remediated waste site. The tabulated results are the sum of the risk values for the individual COPCs included in the risk calculation. Total risk results use EPCs based on measured soil concentrations in the individual waste sites. Risk assessment results are shown for present-day radionuclide concentrations, and with radionuclide concentrations decayed to the years 2075 and 2150. Soil concentrations for these times were calculated using the RESRAD computer code, as described in Appendix D-1.4.1.

The present-day RME and CTE total cancer risk results for the Subsistence Farmer scenario (radionuclides + chemicals) at the 116-N-3 waste site are 4×10^{-5} and 8×10^{-6}, respectively. Per EPA guidance (EPA/540/1-89/002), cancer risk results are displayed with only one significant figure.

The 116-N-3 waste site had a calculated total RME cancer risk of 4×10^{-5}. Approximately 95% of RME and CTE cancer risk is related to cobalt-60 via external irradiation. The half-life of cobalt-60 is approximately 5 years. Therefore, much of the site-related cancer risk will decrease naturally over time as a function of radioactive decay. By year 2075, the total RME cancer risk at the 116-N-3 waste site is 1×10^{-8}. As described in relation to Resident National Monument/Refuge cancer risk results, the 116-N-3 waste site was excavated to a depth of approximately 6.5 m (21.3 ft). The use of shallow zone soil verification data to calculate risks at this site may contribute to a significant protective bias in the risk results.

5.5.3.2 Radiation Dose. Table 5-88 shows the RME and CTE total radiation dose results for the Subsistence Farmer exposure scenario for the 116-N-3 waste site; no radionuclide COPCs were detected at the 120-N-1 waste site. The tabulated results are the sum of the risk values for the individual COPCs included in the risk calculation. Total dose results use EPCs based on measured soil concentrations in the individual waste sites. Risk assessment results are shown for present-day radionuclide concentrations, and with radionuclide concentrations decayed to the years 2075 and 2150. Soil concentrations for these times were calculated using the RESRAD computer code, as described in Appendix D-1.4.1.

The 116-N-3 waste site has total RME and CTE radiation dose results of 1.7 and 1.1 mrem/yr, respectively. There are no waste sites in the 100-N Area where the present-day radiation dose for the Subsistence Farmer scenario approaches the threshold of 15 mrem/yr.

5.5.3.3 Chemical Hazard. Table 5-89 shows the RME and CTE HI results for the Subsistence Farmer exposure scenario for the 120-N-1 and 116-N-3 remediated waste sites in the 100-N Decision Area. The tabulated results are the sum of the risk values for the individual COPCs included in the risk calculation. As described in Section 3.4.4, chemical hazard is calculated separately for adult and child receptors. For the Subsistence Farmer scenario, RME and CTE child HIs are slightly higher those for adults. Because they are consistently higher, only child HI values are presented here.

The RME and CTE child HI results for the Subsistence Farmer scenario at the 120-N-1 waste site are 0.50 and 0.25. At the 116-N-3 waste site, these results are 0.00047 and 0.00042. Essentially all of the RME HI value of 0.50 at the 120-N-1 waste site is related to mercury, primarily via the beef ingestion and produce ingestion exposure pathways. As shown in Table 5-50, the RME and CTE Subsistence Farmer child HQs calculated using the upland reference area data for mercury are 0.28 and 0.14, indicating that approximately one-half of the child HI results at the 120-N-1 waste site are related to naturally occurring levels of mercury in soil. There is no information on excavation depth for the 120-N-1 waste site in Table 2-8.

5.5.4 CTUIR Resident Scenario Results for the 100-N Area

The CTUIR Resident exposure scenario encompasses potential exposures related to shallow zone soils around remediated waste sites, including both direct contact exposure pathways and exposures related to raising produce and animals in the potentially affected area of the waste site. This scenario also includes exposures related to ingestion of Columbia River fish (see Section 4.4) and exposure to groundwater via a domestic water well (see Section 6.0). It is similar to the Subsistence Farmer scenario, although exposure intensity is higher for many exposure pathways. The scenario is more fully described in the CSM in Section 3.3 and is summarized in Table 3-13 and Figure 3-17.

COPCs evaluated in shallow zone soil in the 100-N Decision Area are shown in Table 5-26. The COPCs identified at the two 100-N Area remediated waste sites evaluated in this report, 116-N-3 and 120-N-1, are limited to cobalt-60, mercury, and nitrogen in nitrate and nitrite.

5.5.4.1 Cancer Risk. Table 5-90 shows the total cancer risk results for the CTUIR Resident exposure scenario for the 116-N-3 waste site; no carcinogenic COPCs were detected at the 120-N-1 remediated waste site. The tabulated results are the sum of the risk values for the individual COPCs included in the risk calculation. Total risk results use EPCs based on measured soil concentrations in the individual waste sites. Risk assessment results are shown for present-day radionuclide concentrations, and with radionuclide concentrations decayed to the years 2075 and 2150. Soil concentrations for these times were calculated using the RESRAD computer code, as described in Appendix D-1.4.1.

The present-day total cancer risk result for the CTUIR Resident scenario (radionuclides + chemicals) at the 116-N-3 waste site is 1×10^{-4}. Per EPA guidance (EPA/540/1-89/002), cancer risk results are displayed with only one significant figure.

The 116-N-3 waste site had a calculated total cancer risk of 1×10^{-4}. Approximately 90% of total cancer risk is related to cobalt-60 via external irradiation. The half-life of cobalt-60 is approximately 5 years. Therefore, much of the site-related cancer risk will decrease naturally over time as a function of radioactive decay. By year 2075, the total cancer risk at the 116-N-3 waste site is 7×10^{-8}. As described in relation to the Subsistence Farmer results in Section 5.5.3, the 116-N-3 waste site was excavated to a depth of approximately 6.5 m (21.3 ft). The use of shallow zone soil verification data to calculate risks at this site may contribute to a significant protective bias in the risk results.

5.5.4.2 Radiation Dose. Table 5-91 shows the total radiation dose results for the CTUIR Resident exposure scenario for the 116-N-3 waste site; no radionuclide COPCs were detected at the 120-N-1 waste site. The tabulated results are the sum of the risk values for the individual COPCs included in the risk calculation. Total dose results use EPCs based on measured soil concentrations in the individual waste sites. Risk assessment results are shown for present-day radionuclide concentrations, and with radionuclide concentrations decayed to the years 2075 and 2150. Soil concentrations for these times were calculated using the RESRAD computer code, as described in Appendix D-1.4.1.

The 116-N-3 waste site has a present-day total radiation dose of 2.0 mrem/yr, related to cobalt-60 via external irradiation. There are no waste sites in the 100-N Area where the present-day radiation dose for the CTUIR Resident scenario approaches the threshold of 15 mrem/yr.

5.5.4.3 Chemical Hazard. Table 5-92 shows the HI results for the CTUIR Resident exposure scenario for the 120-N-1 and 116-N-3 remediated waste sites in the 100-N Decision Area. The tabulated results are the sum of the risk values for the individual COPCs included in the risk calculation. As described in Section 3.4.4, chemical hazard is calculated separately for adult and child receptors. For the CTUIR Resident scenario, child HIs are equal to or above those for adults. Therefore, only child HI values are presented here.

The child HI result for the CTUIR Resident scenario is 2.5 at 120-N-1. At the 116-N-3 waste site, child HI is 0.0005. A summary of the key exposure pathways and COPCs contributing 10% or more to the CTUIR Resident HI at 100-N Decision Area sites with HI values above 1.0 is provided below.

Site	COPC	Pathway	Fraction of Child Hazard	Present-Day Child Hazard
120-N-1	Mercury	Produce Ingestion	0.75	2.5
	Mercury	Beef Ingestion	0.24	

Essentially all of the HI value of 2.5 at the 120-N-1 waste site is related to mercury, primarily via the produce ingestion and beef ingestion exposure pathways. As shown in Table 5-50, the CTUIR Resident child HQ calculated using the upland reference area data for mercury is 1.4, indicating that approximately one-half of the child HI of 2.5 at the 120-N-1 waste site is related to naturally occurring levels of mercury in soil. There is no information on excavation depth for the 120-N-1 waste site in Table 2-8.

5.5.5 Yakama Resident Scenario Results for the 100-N Area

The Yakama Resident exposure scenario encompasses potential exposures related to shallow zone soils around remediated waste sites, including both direct contact exposure pathways and exposures related to raising produce and animals in the potentially affected area of the waste site. This scenario also includes exposures related to ingestion of Columbia River fish (see

Section 4.4) and exposure to groundwater via a domestic water well (see Section 6.0). It is similar to the Subsistence Farmer scenario, although exposure intensity is higher for many exposure pathways. The scenario is more fully described in the CSM in Section 3.3 and is summarized in Table 3-13 and Figure 3-17.

Contaminants of potential concern evaluated in shallow zone soil in the 100-N Decision Area are shown in Table 5-26. The COPCs identified at the two 100-N Area remediated waste sites evaluated in this report, 116-N-3 and 120-N-1, are limited to cobalt-60, mercury, and nitrogen in nitrate and nitrite.

5.5.5.1 Cancer Risk. Table 5-93 shows the total cancer risk results for the Yakama Resident exposure scenario for the 116-N-3 waste site; no carcinogenic COPCs were detected at the 120-N-1 remediated waste site. The tabulated results are the sum of the risk values for the individual COPCs included in the risk calculation. Total risk results use EPCs based on measured soil concentrations in the individual waste sites. Risk assessment results are shown for present-day radionuclide concentrations, and with radionuclide concentrations decayed to the years 2075 and 2150. Soil concentrations for these times were calculated using the RESRAD computer code, as described in Appendix D-1.4.1.

The present-day total cancer risk result for the Yakama Resident scenario (radionuclides + chemicals) at the 116-N-3 waste site is 1×10^{-4}. Per EPA guidance (EPA/540/1-89/002), cancer risk results are displayed with only one significant figure.

The 116-N-3 waste site had a calculated total cancer risk of 1×10^{-4}. Approximately 85% of total cancer risk is related to cobalt-60 via external irradiation. The half-life of cobalt-60 is approximately 5 years. Therefore, much of the site-related cancer risk will decrease naturally over time as a function of radioactive decay. By year 2075, the total cancer risk at the 116-N-3 waste site is 7×10^{-8}. As described in relation to the Subsistence Farmer results in Section 5.5.3, the 116-N-3 waste site was excavated to a depth of approximately 6.5 m (21.3 ft). The use of shallow zone soil verification data to calculate risks at this site may contribute to a significant protective bias in the risk results.

5.5.5.2 Radiation Dose. Table 5-94 shows the total radiation dose results for the Yakama Resident exposure scenario for the 116-N-3 waste site; no radionuclide COPCs were detected at the 120-N-1 waste site. The tabulated results are the sum of the risk values for the individual COPCs included in the risk calculation. Total dose results use EPCs based on measured soil concentrations in the individual waste sites. Risk assessment results are shown for present-day radionuclide concentrations, and with radionuclide concentrations decayed to the years 2075 and 2150. Soil concentrations for these times were calculated using the RESRAD computer code, as described in Appendix D-1.4.1.

The 116-N-3 waste site has a present-day total radiation dose of 2.1 mrem/yr, related to cobalt-60 via external irradiation. There are no waste sites in the 100-N Area where the present-day radiation dose for the Yakama Resident scenario exceeds the threshold of 15 mrem/yr.

5.5.5.3 Chemical Hazard. Table 5-95 shows the HI results for the Yakama Resident exposure scenario for the 120-N-1 and 116-N-3 remediated waste sites in the 100-N Decision Area. The tabulated results are the sum of the risk values for the individual COPCs included in the risk calculation. As described in Section 3.4.4, chemical hazard is calculated separately for adult and child receptors. For the Yakama Resident scenario, child HIs are equal to or above those for adults. Therefore, only child HI values are presented here.

The child HI result for the Yakama Resident scenario is 3.1 at 120-N-1. At the 116-N-3 waste site, child HI is 0.0005. A summary of the key exposure pathways and COPCs contributing 10% or more to the Yakama Resident HI at 100-N Decision Area sites with HI values above 1.0 is provided below.

Site	COPC	Pathway	Fraction of Child Hazard	Present-Day Child Hazard
120-N-1	Mercury	Beef Ingestion	0.67	3.1
	Mercury	Produce Ingestion	0.32	

Essentially all of the HI value of 3.1 at the 120-N-1 waste site is related to mercury, primarily via the beef ingestion and produce ingestion exposure pathways. As shown in Table 5-50, the Yakama Resident child HQ calculated using the upland reference area data for mercury is 1.7, indicating that approximately one-half of the child HI of 3.1 at the 120-N-1 waste site is related to naturally occurring levels of mercury in soil. There is no information on excavation depth for the 120-N-1 waste site in Table 2-8.

5.5.6 Risks Related to Lead in Soil for the 100-N Area

The highest calculated RME representative concentration for lead in shallow zone soil at any remediated waste site is approximately 210 mg/kg. This concentration applies to the 100-F-37 remediated waste site in the 100-F/100-IU-2/100-IU-6 Area. Lead was measured in two shallow soil samples at the 120-N-1 waste site. The highest value was 4.3 mg/kg. No representative concentrations for lead at any waste site exceed EPA's recommended residential screening level for lead in soil of 400 mg/kg, derived using the IEUBK model that has been associated with bare soil in a play area (40 CFR 745). Lead was not analyzed in cleanup verification soil samples at the 100-N Area remediated waste sites evaluated in this risk assessment.

5.6 100-D AND 100-H DECISION AREA RESULTS

Section 5.6 is the fourth of six sections of the local-area risk assessment (5.3 through 5.8) that present the results of the soil source term risk assessment for remediated waste sites in an individual ROD decision area. A summary of the results of the waste site soils risk assessment for all ROD decision areas is shown in Tables 5-2 through 5-7. Tables 5-2, 5-3, and 5-4 show the number of remediated waste sites with risk assessment results above or below decision thresholds for the Occupational scenarios. Table 5-2 shows results using present-day radionuclide soil concentrations, and the latter two tables show the results with radionuclide concentrations at years 2075 and 2150. Tables 5-5, 5-6, and 5-7 show these same summaries for the Residential exposure scenarios. The primary purpose of these summary tables is to allow the reader to quickly see how the waste sites risk assessment results for the different exposure scenarios compare to the thresholds used for risk management decisions. These tables also provide a perspective of how the waste site risk assessment results vary among the six ROD decision areas, and how cancer risk and radiation dose will change over time due to radioactive decay.

As discussed in Section 5.2.3, multiple types of data may exist for uranium metal. Uranium as a metal is evaluated in the risk assessment because of its potential toxicity to the kidney. The cleanup verification soil data include few results for inorganic uranium. Results for calculated total uranium, derived from the isotopic uranium results (as activity per mass of soil), are used in the risk calculations in lieu of data for the direct measurement of uranium (as mass of uranium per mass of soil). The single exception is at the 1607-D4 waste site in the 100-D/100-H Decision Area, where inorganic uranium data were obtained, but no isotopic uranium results are available.

As described in Section 3.3, broad-area EPCs in the upland environment are associated with the occupational portion of the Resident National Monument/Refuge scenario and are also used to supplement the local-area cleanup verification data to calculate EPCs for dust inhalation for all scenarios evaluated in Sections 5.3 through 5.8. The COPCs for the broad-area components of the waste site risk calculations are those related to upland surface soil, while the COPCs for the local-area components of the waste site risk calculations are those related to the cleanup verification data for a particular ROD decision area. Risk assessment results for the Resident National Monument/Refuge scenario are provided for the local-area and broad-area components separately. Broad-area representative concentrations for upland surface soil are calculated using the MIS soil data from all 20 upland sampling locations, using the same protocol described for MIS reference area data in Section 5.2.4.

5.6.1 Industrial/Commercial Scenario Results for the 100-D/100-H Area

The Industrial/Commercial exposure scenario encompasses potential occupational exposures related to shallow zone soils around remediated waste sites. Like the Resident National Monument/Refuge scenario, it focuses on direct contact exposure pathways and does not encompass the raising of produce or animals, nor the ingestion of Columbia River fish. It is similar to the Industrial/Commercial scenario used by EPA to calculate soil screening values using direct contact exposure pathways such as inadvertent soil ingestion, dermal contact with soil, and inhalation. Of the scenarios evaluated on the scale of an individual remediated waste site, the Industrial/Commercial scenario has the lowest exposure intensity. The scenario is more fully described in the CSM in Section 3.3 and is summarized in Table 3-12 and Figure 3-16. The COPCs evaluated in shallow zone soil in the 100-D/100-H Decision Area are shown in Table 5-26.

5.6.1.1 Cancer Risk. Table 5-96 shows the RME and CTE total cancer risk results for the Industrial/Commercial exposure scenario for all remediated waste sites in the 100-D/100-H Decision Area. The tabulated results are the sum of the risk values for the individual COPCs included in the risk calculation. Total risk results use EPCs based on measured soil concentrations in the individual waste sites. Risk assessment results are shown for present-day radionuclide concentrations, and with radionuclide concentrations decayed to the years 2075 and 2150. Soil concentrations for these times were calculated using the RESRAD computer code, as described in Appendix D-1.4.1.

The range of present-day RME and CTE total cancer risk results for the Industrial/Commercial scenario (radionuclides + chemicals) is from 4×10^{-7} to 2×10^{-4} and from 6×10^{-8} to 9×10^{-6}, respectively. No carcinogenic COPCs were detected at the 100-D-12 and 100-D-22 waste sites. Per EPA guidance (EPA/540/1-89/002), cancer risk results are displayed with only one significant figure. As shown in Table 5-2, only 1 of the 35 remediated waste sites evaluated in the 100-D/100-H Decision Area had calculated total RME cancer risks above 1×10^{-4}, and no waste site had a CTE total cancer risk above 1×10^{-4}.

A summary of the key exposure pathways and COPCs contributing 10% or more to present-day RME Industrial/Commercial cancer risk at 100-D/100-H Decision Area sites with total cancer risk above 1×10^{-4} is provided below.

Site	COPC	Pathway	Fraction of Total Risk	Present-Day Total Risk
116-DR-9	Cesium-137	External Irradiation	0.86	2×10^{-4}
	Europium-152	External Irradiation	0.12	

About 85% of the RME total cancer risk of 2×10^{-4} at the 116-DR-9 waste site is due to external irradiation from cesium-137 (half-life of approximately 30 years). No waste sites exceed 1×10^{-4} risk for the Industrial/Commercial scenario at year 2075. The highest RME total cancer risk at year 2075 is 3×10^{-5} at the 116-DR-9 waste site.

5.6.1.2 Radiation Dose. Table 5-97 shows the RME and CTE total radiation dose results for the Industrial/Commercial exposure scenario for all remediated waste sites in the 100-D/100-H Decision Area. The tabulated results are the sum of the dose values for the individual COPCs included in the risk calculation. Risk assessment results are shown for present-day radionuclide concentrations, and with radionuclide concentrations decayed to the years 2075 and 2150. Soil concentrations for these times were calculated using the RESRAD computer code, as described in Appendix D-1.4.1.

The range of present-day RME and CTE total radiation dose results for the Industrial/Commercial scenario is from 0.023 to 8.3 mrem/yr and from 0.023 to 2.4 mrem/yr, respectively. No radionuclide COPCs were detected at four waste sites.

There are no waste sites in the 100-D/100-H Area where the present-day radiation dose for the Industrial/Commercial scenario exceeds the threshold of 15 mrem/yr. The present-day total RME radiation dose of 8.3 mrem/yr at the 116-DR-9 waste site decreases to 1.5 mrem/yr by 2075, due primarily to radioactive decay of cesium-137.

5.6.1.3 Chemical Hazard. Table 5-98 shows the RME and CTE HI results for the Industrial/Commercial exposure scenario for all remediated waste sites in the 100-D/100-H Decision Area. The tabulated results are the sum of the risk values for the individual COPCs included in the risk calculation.

The range of RME and CTE HI results for the Industrial/Commercial scenario is from approximately 0.0002 to 0.026 and from 0.0001 to 0.0094, respectively. No COPCs contributing to chemical hazard were detected at three remediated waste sites. There are no waste sites in the 100-D/100-H Area, where HI for the Industrial/Commercial scenario approaches the threshold of 1.0.

5.6.2 Resident National Monument/Refuge Scenario Results for the 100-D/100-H Area

The Resident National Monument/Refuge exposure scenario encompasses potential residential exposures related to shallow zone soils around remediated waste sites, as well as potential occupational exposures to surface soil across the upland environment of the River Corridor. It focuses on direct contact exposure pathways and does not encompass the raising of produce or animals nor the ingestion of Columbia River fish. This scenario also includes exposures related to groundwater via a domestic water well. Risks related to groundwater use are addressed in Section 6.0. With respect to soil exposure pathways, this scenario is similar to the traditional urban/suburban residential scenario used by EPA to calculate soil screening values using direct contact exposure pathways such as inadvertent soil ingestion, dermal contact with soil, and inhalation. It may be viewed as akin to the Subsistence Farmer scenario for an adult receptor, minus the ingestion of foodstuffs and with an exposure model that averages exposure to both individual waste site soils (cleanup verification soil data) and surface soils over large areas (upland surface soil data). The scenario is more fully described in the CSM in Section 3.3 and is summarized in Table 3-12 and Figure 3-16. The COPCs evaluated in shallow zone soil in the 100-D/100-H Decision Area are shown in Table 5-26.

As described in the introductory text for Section 5.4, COPCs for the broad-area components of the waste site risk calculations are those related to upland surface soil, while the COPCs for the local-area components of the waste site risk calculations are those related to the cleanup verification data for a particular ROD decision area. Results for the local-area portion of the Resident National Monument/Refuge exposure scenario are calculated for each individual remediated waste site where a residence might be situated. Risks related to the broad-area occupational component of the Resident National Monument/Refuge exposure scenario are shown in the first row of the summary tables for cancer risk, radiation dose, and chemical hazard. As discussed in Section 4.1.3, broad-area representative concentrations are calculated using upland surface soil data collected under DOE/RL-2005-42.

5.6.2.1 Cancer Risk. Table 5-99 shows the RME and CTE total cancer risk results for the Resident National Monument/Refuge exposure scenario for all remediated waste sites in the 100-D/100-H Decision Area. The tabulated results are the sum of the risk values for the individual COPCs included in the risk calculation. Total risk results use EPCs based on measured soil concentrations in the individual waste sites (residential exposure) and River Corridor upland surface soil (occupational exposure). Risk assessment results are shown for present-day radionuclide concentrations, and with radionuclide concentrations decayed to the years 2075 and 2150. Soil concentrations for these times were calculated using the RESRAD computer code, as described in Appendix D-1.4.1.

The range of present-day RME and CTE total cancer risk results for the local-area component of the Resident National Monument/Refuge scenario (radionuclides + chemicals) is from 7×10^{-7} to 3×10^{-4} and from 9×10^{-8} to 2×10^{-5}, respectively. No carcinogenic COPCs were detected at the 100-D-12 and 100-D-22 waste sites. Per EPA guidance (EPA/540/1-89/002), cancer risk results are displayed with only one significant figure. As shown in Table 5-2, only 1 of the 35 remediated waste sites evaluated in the 100-D/100-H Decision Area had a combined local and broad total RME cancer risk above 1×10^{-4}, and no waste site had a combined local and broad CTE total cancer risk above 1×10^{-4}.

Because the broad-area component of cancer risk is the same across all calculations, the summary of the key exposure pathways and COPCs contributing 10% or more to present-day RME Resident National Monument/Refuge cancer risk at 100-D/100-H Decision Area sites focuses on the local-area component of the Resident National Monument/Refuge scenario. The local-area calculations with total cancer risk above 1×10^{-4} are summarized below.

Site	Analyte	Pathway	Fraction of Total Risk	Present-Day Total Risk
116-DR-9	Cesium-137	External Irradiation	0.86	3×10^{-4}
	Europium-152	External Irradiation	0.12	

About 85% of the local-area component of the RME total cancer risk of 3×10^{-4} at the 116-DR-9 waste site is due to external irradiation from cesium-137 (half-life of approximately 30 years), and the remainder is largely attributable to europium-152 (half-life of approximately 13.5 years). At the 100-D-48:2, 100-D-48:3 and 100-D-49:4 waste sites, where present-day RME cancer risks are 1×10^{-4}, risks are also due mostly to these short-lived radionuclides. At year 2075, RME cancer risks at these three sites range from 5×10^{-6} to 5×10^{-5} and, at year 2150, the RME risks are all below 1×10^{-5} .

The current condition of the 116-DR-9 remediated waste site is summarized in Table 2-8. The 116-DR-9 waste site was excavated to a depth of 4.75 m (15.6 ft) and then backfilled to grade following remediation. The shallow zone soil verification data used to calculate local-area risk are from samples collected along the sidewalls of the excavation at depths between 0 and 4.6 m (15 ft). The use of these data to represent surface soil for current and future exposures may contribute to a significant protective bias in the Resident National Monument/Refuge risk results for this site. Uncertainties in the use of shallow zone soil verification data from excavated waste sites are discussed in more detail in Section 5.9 and Appendix D-1.

5.6.2.2 Radiation Dose. Table 5-100 shows the RME and CTE total radiation dose results for the Resident National Monument/Refuge exposure scenario for all remediated waste sites in the 100-D/100-H Decision Area. The tabulated results are the sum of the risk values for the individual COPCs included in the risk calculation. Risk assessment results are shown for present-day radionuclide concentrations, and with radionuclide concentrations decayed to the years 2075 and 2150. Soil concentrations for these times were calculated using the RESRAD computer code, as described in Appendix D-1.4.1.

The range of present-day RME and CTE total radiation dose results for the local-area component of the Resident National Monument/Refuge scenario is from 0.097 to 15 mrem/yr and from 0.08 to 3.1 mrem/yr, respectively. No radionuclide COPCs were detected at four waste sites.

No remediated waste sites had RME or CTE total radiation dose values above 15 mrem/yr. The RME radiation dose of 15 mrem/yr at the 116-DR-9 waste site was mostly due to external irradiation from cesium-137, with a half-life of approximately 30 years. The highest RME total dose at year 2075 is 2.6 mrem/yr at the 116-DR-9 waste site.

5.6.2.3 Chemical Hazard. Table 5-101 shows the RME and CTE HI results for the Resident National Monument/Refuge exposure scenario for all remediated waste sites in the 100-D/100-H Decision Area. The tabulated results are the sum of the risk values for the individual COPCs included in the risk calculation.

The range of RME and CTE HI results for the local-area component of the Resident National Monument/Refuge scenario is from 0.0021 to 0.042 and from 0.0018 to 0.017, respectively. No chemical COPCs contributing to hazard were detected at three waste sites in the 100-D/100-H Area. There are no waste sites in the 100-D/100-H Area, where the HI for the Resident National Monument/Refuge scenario approaches the threshold of 1.0.

5.6.3 Subsistence Farmer Scenario Results for the 100-D/100-H Area

The Subsistence Farmer exposure scenario encompasses potential exposures related to shallow zone soils around remediated waste sites, including both direct contact exposure pathways and exposures related to raising produce and animals in the potentially affected area of the waste site. This scenario also includes exposures related to ingestion of Columbia River fish and exposure to groundwater via a domestic water well. Risks related to fish ingestion and groundwater use and are addressed in Sections 4.4 and 6.0, respectively. The scenario is more fully described in the CSM in Section 3.3 and is summarized in Table 3-13 and Figure 3-17. The COPCs evaluated in shallow zone soil in the 100-D/100-H Decision Area are shown in Table 5-26.

5.6.3.1 Cancer Risk. Table 5-102 shows the RME and CTE total cancer risk results for the Subsistence Farmer exposure scenario for all remediated waste sites in the 100-D/100-H Decision Area. The tabulated results are the sum of the risk values for the individual COPCs included in the risk calculation. Total risk results use EPCs based on measured soil concentrations in the individual waste sites. Risk assessment results are shown for present-day radionuclide concentrations, and with radionuclide concentrations decayed to the years 2075 and 2150. Soil concentrations for these times were calculated using the RESRAD computer code, as described in Appendix D-1.4.1.

The range of present-day RME and CTE total cancer risk results for the Subsistence Farmer scenario (radionuclides + chemicals) is from 7×10^{-6} to 2×10^{-3} and from 1×10^{-6} to 2×10^{-4}, respectively. No carcinogenic COPCs were detected at the 100-D-12 and 100-D-22 waste sites. Per EPA guidance (EPA/540/1-89/002), cancer risk results are displayed with only one significant figure. As shown in Table 5-5, 18 of the 35 waste sites evaluated in the 100-D/100-H Decision Area had calculated total RME cancer risks above 1×10^{-4}, and 1 waste site had a CTE total cancer risk above 1×10^{-4}.

A summary of the key exposure pathways and COPCs contributing 10% or more to present-day RME Subsistence Farmer cancer risk at 100-D/100-H Decision Area sites with total cancer risk above 1×10^{-4} is provided below.

Site	COPC	Pathway	Fraction of Total Risk	Present-Day Total Risk
100-H-21	Arsenic	Produce Ingestion	0.80	2×10^{-3}
116-H-1	Arsenic	Produce Ingestion	0.70	1×10^{-3}
1607-H2	Arsenic	Produce Ingestion	0.84	1×10^{-3}
116-H-7	Arsenic	Produce Ingestion	0.74	1×10^{-3}
118-DR-2:2	Arsenic	Produce Ingestion	0.51	8×10^{-4}
116-DR-9	Cesium-137	External Irradiation	0.63	7×10^{-4}
	Cesium-137	Milk Ingestion	0.12	
100-H-5	Arsenic	Produce Ingestion	0.82	7×10^{-4}
100-H-17	Arsenic	Produce Ingestion	0.69	7×10^{-4}
100-H-24	Arsenic	Produce Ingestion	0.85	7×10^{-4}
122-DR-1:2	Arsenic	Produce Ingestion	0.83	5×10^{-4}
100-D-48:3	Cesium-137	External Irradiation	0.53	4×10^{-4}
	Strontium-90	Milk Ingestion	0.10	
	Cesium-137	Milk Ingestion	0.10	
100-D-49:4	Europium-152	External Irradiation	0.69	2×10^{-4}
100-D-48:2	Cesium-137	External Irradiation	0.39	2×10^{-4}
	Europium-152	External Irradiation	0.26	
	Cobalt-60	External Irradiation	0.12	

Site	COPC	Pathway	Fraction of Total Risk	Present-Day Total Risk
1607-D4	Arsenic	Produce Ingestion	0.85	2×10^{-4}
1607-D2:1	Arsenic	Produce Ingestion	0.79	2×10^{-4}
100-D-20	Cesium-137	External Irradiation	0.38	2×10^{-4}
	Europium-152	External Irradiation	0.30	
116-D-1A	Cesium-137	External Irradiation	0.33	2×10^{-4}
	Europium-152	External Irradiation	0.23	
	Strontium-90	Milk Ingestion	0.12	
	Strontium-90	Produce Ingestion	0.10	
100-D-48:4	Cesium-137	External Irradiation	0.34	2×10^{-4}
	Europium-152	External Irradiation	0.16	
	Strontium-90	Milk Ingestion	0.12	

For the 10 remediated waste sites with the highest RME present-day cancer risks, exposure to arsenic via the produce ingestion exposure pathway was the main contributor at all of the sites except 116-DR-9. As shown in Table 5-48, the Subsistence Farmer RME arsenic cancer risk calculated using the upland reference area data is 5×10^{-4}. The range of RME arsenic concentrations at 100-H-21, 116-H-1, 1607-H2, 116-H-7, 118-DR-2:2, 100-H-5, 100-H-17, and 100-H-24, where arsenic-related risks exceed the reference areas level, range from 3.4 mg/kg (100-H-17) to 13.8 mg/kg (100-H-21). These RME arsenic soil concentrations are all above the upland reference area RME concentration of 3.2 mg/kg shown in Table 5-47. Reasonable maximum exposure arsenic concentrations at 116-H-7 (5.3 mg/kg), 116-H-1 (6.9 mg/kg), 1607-H2 (7.6 mg/kg) and 100-H-21 (13.8 mg/kg) are also above one or both of the Hanford Area (6.47 mg/kg) and Yakima Basin (5.13 mg/kg) 90th percentile soil background concentrations.

At the seven remediated waste sites with present-day RME cancer risk above 1×10^{-4} where arsenic is not the most important contributor, much of the cancer risk is related to short-lived radionuclides including cobalt-60, cesium-137, isotopic europium, and strontium-90. The half-lives of these radionuclides range from approximately 5 years (cobalt-60) to approximately 30 years (strontium-90 and cesium-137). Therefore, much of the site-related cancer risk at these waste sites will decrease naturally over time as a function of the decay of these radionuclides.

As shown in Table 5-6, there are only 11 waste sites where total RME cancer risk is above 1×10^{-4} at year 2075. These are the sites where risk is related primarily to arsenic. Because arsenic concentrations at the remediated waste sites where arsenic is a risk driver are at most about twice as high as commonly referenced background levels, risks for COPCs other than arsenic are also of interest for risk management at these sites. Eight of these eleven waste sites have RME cancer risks above 1×10^{-6} related to COPCs other than arsenic. A summary of the key exposure pathways and COPCs, other than arsenic, contributing 10% or more to present-day RME Subsistence Farmer cancer risk at these sites is provided below.

Site	COPC other than arsenic	Pathway	Fraction of Total Risk	Present-Day Total Risk (without arsenic)
118-DR-2:2	Cesium-137	External Irradiation	0.23	3×10^{-4}
	Technetium-99	Produce Ingestion	0.21	
	Technetium-99	Milk Ingestion	0.15	
	Europium-152	External Irradiation	0.15	
	Cobalt-60	External Irradiation	0.11	
116-H-1	Europium-152	External Irradiation	0.36	2×10^{-4}
	Strontium-90	Milk Ingestion	0.21	
	Strontium-90	Produce Ingestion	0.18	
	Cesium-137	External Irradiation	0.10	
100-H-21	Cesium-137	External Irradiation	0.28	1×10^{-4}
	Technetium-99	Produce Ingestion	0.20	
	Europium-152	External Irradiation	0.15	
	Technetium-99	Milk Ingestion	0.14	
116-H-7	Europium-152	External Irradiation	0.53	1×10^{-4}
	Technetium-99	Produce Ingestion	0.13	
100-H-17	Strontium-90	Milk Ingestion	0.35	1×10^{-4}
	Strontium-90	Produce Ingestion	0.30	
	Strontium-90	Beef Ingestion	0.16	
100-H-5	Strontium-90	Milk Ingestion	0.25	3×10^{-5}
	Strontium-90	Produce Ingestion	0.20	
	Cesium-137	External Irradiation	0.12	
	Strontium-90	Beef Ingestion	0.11	
1607-H2	Strontium-90	Milk Ingestion	0.16	2×10^{-5}
	Strontium-90	Produce Ingestion	0.14	
	Cesium-137	External Irradiation	0.13	
1607-D2:1	Cobalt-60	External Irradiation	0.24	1×10^{-5}
	Europium-152	External Irradiation	0.20	
	Uranium-238	External Irradiation	0.11	

In the absence of arsenic, RME cancer risks at the 100-H-21, 116-H-1, 1607-H2, 116-H-7, 118-DR-2:2, 100-H-5, 100-H-17, and 1607-D2:1 waste sites are related to the same set of short-lived radionuclides that were identified as risk drivers for those waste sites where arsenic was not a major contributor to risk. This indicates that RME Subsistence Farmer cancer risks from COPCs other than arsenic are likely to be relatively unimportant by year 2075 at all 100-D/100-H Area remediated waste sites evaluated in this risk assessment.

The range of RME arsenic soil concentrations at 100-D/100-H remediated waste sites where RME total cancer risk was highest due to arsenic is not much higher than might be expected of background soils. Although the arsenic risk estimates for the produce ingestion exposure pathway are based on plant uptake models that are prone to overestimation of plant concentrations when soil concentrations are high (see Section 5.9.2), this potential source of bias may not be very important for the arsenic risk estimates at the 100-D/100-H Area sites because site soil levels are not much greater than naturally occurring levels.

As described in relation to Resident National Monument/Refuge cancer risk results, the use of shallow zone soil verification data may contribute to a significant protective bias in the risk results for excavated sites. The 100-D/100-H Area remediated waste sites with Subsistence Farmer RME cancer risks above 1×10^{-4} due to the presence of either short-lived radionuclides or arsenic were generally excavated to significant depths. With the exception of 1607-D4 (no excavation noted), 100-D-20 (2.1 m), 100-H-17 (2.6 m), and 1607-D2:1 (3.4 m), excavation depths listed in Table 2-8 were 4.6 m (15 ft) or deeper at all sites where present-day RME Subsistence Farmer risks were above 1×10^{-4}. Uncertainties in the use of shallow zone soil verification data from excavated waste sites are discussed in more detail in Section 5.9 and Appendix D-1.

5.6.3.2 Radiation Dose. Table 5-103 shows the RME and CTE total radiation dose results for the Subsistence Farmer exposure scenario for all remediated waste sites in the 100-D/100-H Decision Area. The tabulated results are the sum of the dose values for the individual COPCs included in the dose calculation. Total dose results use EPCs based on measured soil concentrations in the individual waste sites. Risk assessment results are shown for present-day radionuclide concentrations, and with radionuclide concentrations decayed to the years 2075 and 2150. Soil concentrations for these times were calculated using the RESRAD computer code, as described in Appendix D-1.4.1.

The range of present-day RME and CTE total radiation dose results for the Subsistence Farmer scenario is from 0.46 to 34 mrem/yr and from 0.27 to 6.8 mrem/yr, respectively.
No radionuclide COPCs were detected at four waste sites. As shown in Table 5-5, 3 of the 35 waste sites evaluated in the 100-D/100-H Decision Area had calculated total RME radiation dose values above 15 mrem/yr, but no waste site had a CTE total radiation dose above 15 mrem/yr.

A summary of the key exposure pathways and COPCs contributing 10% or more to the present-day RME Subsistence Farmer radiation dose at 100-D/100-H Decision Area sites with total dose above 15 mrem/yr is provided below.

Site	COPC	Pathway	Fraction of Total Dose	Present-Day Total Dose (mrem/yr)
116-DR-9	Cesium-137	External Irradiation	0.63	34
100-D-48:3	Cesium-137	External Irradiation	0.47	21
	Strontium-90	Milk Ingestion	0.15	

Site	COPC	Pathway	Fraction of Total Dose	Present-Day Total Dose (mrem/yr)
	Strontium-90	Produce Ingestion	0.13	
118-DR-2:2	Strontium-90	Milk Ingestion	0.23	19
	Strontium-90	Produce Ingestion	0.19	
	Cesium-137	External Irradiation	0.15	
	Strontium-90	Beef Ingestion	0.10	

The present-day radiation dose at sites with calculated dose above the threshold is driven mostly by strontium-90 and cesium-137, radionuclides with half-lives of approximately 30 years. No 100-D/100-H waste site exceeds 15 mrem/yr at year 2075. At year 2075, the highest total RME radiation dose value is 6.2 mrem/yr at the 100-DR-9 waste site.
As described in relation to cancer risk, most remediated waste sites where short-lived radionuclides were important contributors to significant cancer risk (including 116-DR-9, 100-D-48:3, and 118-DR-2:2) were excavated to a depth of greater than 4.6 m (15 ft). The use of shallow zone soil verification data to calculate risks at these sites may contribute to a significant protective bias in the radiation dose results for these sites.

5.6.3.3 Chemical Hazard. Table 5-104 shows the RME and CTE HI results for the Subsistence Farmer exposure scenario for all remediated waste sites in the 100-D/100-H Decision Area. The tabulated results are the sum of the risk values for the individual COPCs included in the risk calculation. As described in Section 3.4.4, chemical hazard is calculated separately for adult and child receptors. For the Subsistence Farmer scenario, RME and CTE child HIs are slightly higher those for adults. Because they are consistently higher, only child HI values are presented here.

The range of RME and CTE child HI results for the Subsistence Farmer scenario is from 0.054 to 69 and from 0.018 to 6.1, respectively. No chemical COPCs contributing to hazard were detected at three of the remediated waste sites. As shown in Table 5-5, 2 of the 35 waste sites in the 100-D/100-H Decision Area had calculated RME HI values above 10, and another 9 had HI values between 1.0 and 10. Eight waste sites had CTE HI values between 1.0 and 10, but none had an HI above 10.

A summary of the key exposure pathways and COPCs contributing 10% or more to RME Subsistence Farmer HIs at 100-D/100-H Decision Area sites with HI values above 1.0 is provided below.

Site	COPC	Pathway	Fraction of Child Hazard	Present-Day Child Hazard [a]
1607-H2	Mercury	Beef Ingestion	0.63	69
	Mercury	Produce Ingestion	0.24	
100-H-21	Arsenic	Produce Ingestion	0.81	12
116-H-1	Arsenic	Produce Ingestion	0.82	5.9

Site	COPC	Pathway	Fraction of Child Hazard	Present-Day Child Hazard [a]
116-H-7	Arsenic	Produce Ingestion	0.78	4.7
100-H-5	Arsenic	Produce Ingestion	0.63	4.7
	Mercury	Beef Ingestion	0.14	
100-H-24	Arsenic	Produce Ingestion	0.81	3.6
100-H-17	Arsenic	Produce Ingestion	0.77	3.0
118-DR-2:2	Arsenic	Produce Ingestion	0.76	2.9
122-DR-1:2	Arsenic	Produce Ingestion	0.76	2.6
1607-D4	Arsenic	Produce Ingestion	0.73	1.4
1607-D2:1	Arsenic	Produce Ingestion	0.66	1.2

[a] Value for mercury hazard at 1607-H2 is likely overestimated; see discussion in the text.

Mercury exposure via beef ingestion is the most important exposure pathway contributing to HI values above 1.0 at the 1607-H2 remediated waste site, where the RME child HI is 69, and is also a contributor to HI values at 100-H-5. Arsenic exposure via produce ingestion is the most important contributor to HI at the other waste sites summarized above.

Critical effects underlying the oral RfD for mercury include effects on the immune and renal (kidney) system, whereas the critical effects underlying the oral RfD for arsenic relate to the vascular and integumentary (skin) system (IRIS 2009). Summing HQs for mercury and arsenic at the 100-H-5 waste site likely results in a protective bias for the tabulated HI values at this sites.

The RME soil mercury concentration at the 1607-H2 waste site (5.0 mg/kg, based on eight samples with a maximum detected concentration of 3.5 mg/kg) is more than 200 times higher than the reference area RME mercury concentration of 0.022 mg/kg. As discussed in the Uncertainty Analysis (Section 5.9), the linear models used in this risk assessment to estimate plant tissue concentrations from soil data are prone to overestimation of plant concentrations at higher soil concentrations. The mercury plant-soil regression model published by Bechtel (BJC/OR-133) predicts a mercury plant concentration of 0.89 mg/kg when the soil concentration is 5.0 mg/kg. By contrast, the dry-weight produce and fodder concentrations using the linear plant-soil ratios project plant concentration of approximately 13 and 5.0 mg/kg, respectively. Assuming the nonlinear model is more accurate at the higher soil mercury concentration of 5.0 mg/kg, this suggests mercury exposure at the 1607-H2 waste site may be overestimated by a factor of about 15 and 5.6 for produce ingestion and beef ingestion, respectively. In that case, the mercury HQ at 1607-H2 for produce ingestion would both be about 1.1, and for beef ingestion about 7.8. This results in an HI value of about 9 rather than 69.

As shown in Table 5-50, the Subsistence Farmer RME child HQ calculated using the upland reference area concentration of arsenic is 2.7. The range of RME arsenic concentrations at 100-H-21, 116-H-1, 116-H-7, 100-H-5, 100-H-24, 100-H-17, and 118-DR-2:2, where arsenic-related risks exceed the reference areas level, range from 3.4 mg/kg (100-H-17) to 13.8 mg/kg

(100-H-21). These RME arsenic soil concentrations are all above the upland reference area RME concentration of 3.2 mg/kg shown in Table 5-47. RME arsenic concentrations at 116-H-7 (5.3 mg/kg), 116-H-1 (6.9 mg/kg), and 100-H-21 (13.8 mg/kg) are also above one or both of the Hanford Area (6.47 mg/kg) and Yakima Basin (5.13 mg/kg) 90th percentile soil background concentrations.

Because arsenic concentrations at the remediated waste sites where arsenic is a risk driver are at most about twice as high as commonly referenced background levels, risks for COPCs other than arsenic are also of interest for risk management at these sites. Of the 10 remediated waste sites where RME child HI due to arsenic is above 1.0, there is only 1 site where child HI exceeds 1.0 without arsenic. A summary of the key exposure pathways and COPCs, other than arsenic, contributing 10% or more to present-day RME child HI at this site is provided below.

Site	COPC other than arsenic	Pathway	Fraction of Child Hazard	Present-Day Child Hazard (without arsenic)
100-H-5	Mercury	Beef Ingestion	0.58	1.1
	Mercury	Produce Ingestion	0.22	

As described in relation to cancer risk, with the exception of 1607-D4 (no excavation noted), 100-H-17 (2.6 m [8.5 ft]), and 1607-D2:1 (3.4 m [11.2 ft]), excavation depths for the waste sites where either arsenic and/or mercury drives an HI value of 1.0 were 4.6 m (15 ft) or deeper. The use of shallow zone soil verification data to calculate HI may contribute to a significant protective bias in the radiation dose results for these sites.

5.6.4 CTUIR Resident Scenario Results for the 100-D/100-H Area

The CTUIR Resident exposure scenario encompasses potential exposures related to shallow zone soils around remediated waste sites, including both direct contact exposure pathways and exposures related to raising produce and animals in the potentially affected area of the waste site. This scenario also includes exposures related to ingestion of Columbia River fish (see Section 4.4) and exposure to groundwater via a domestic water well (see Section 6.0). It is similar to the Subsistence Farmer scenario, although exposure intensity is higher for many exposure pathways. The scenario is more fully described in the CSM in Section 3.3 and is summarized in Table 3-13 and Figure 3-17. The COPCs evaluated in shallow zone soil in the 100-D/100-H Decision Area are shown in Table 5-26.

5.6.4.1 Cancer Risk. Table 5-105 shows the total cancer risk results for the CTUIR Resident exposure scenario for all remediated waste sites in the 100-D/100-H Decision Area. The tabulated results are the sum of the risk values for the individual COPCs included in the risk calculation. Total risk results use EPCs based on measured soil concentrations in the individual waste sites. Risk assessment results are shown for present-day radionuclide concentrations, and with radionuclide concentrations decayed to the years 2075 and 2150. Soil concentrations for these times were calculated using the RESRAD computer code, as described in Appendix D-1.4.1.

The present-day total cancer risk results for the CTUIR Resident scenario (radionuclides + chemicals) range from 2×10^{-5} to 4×10^{-2}. No carcinogenic COPCs were detected at waste sites 100-D-12 and 100-D-22. Per EPA guidance (EPA/540/1-89/002), cancer risk results are displayed with only one significant figure.

As shown in Table 5-5, 27 of the 35 waste sites evaluated in the 100-D/100-H Decision Area had calculated total cancer risks above 1×10^{-4}.

A summary of the key exposure pathways and COPCs contributing 10% or more to present-day CTUIR Resident cancer risk at 100-D/100-H Decision Area sites with total cancer risk above 1×10^{-4} is provided below.

Site	COPC	Pathway	Fraction of Total Risk	Present-Day Total Risk
100-H-21	Arsenic	Produce Ingestion	0.97	4×10^{-2}
1607-H2	Arsenic	Produce Ingestion	0.99	2×10^{-2}
116-H-1	Arsenic	Produce Ingestion	0.92	2×10^{-2}
116-H-7	Arsenic	Produce Ingestion	0.94	1×10^{-2}
100-H-5	Arsenic	Produce Ingestion	0.98	1×10^{-2}
100-H-24	Arsenic	Produce Ingestion	0.99	1×10^{-2}
118-DR-2:2	Arsenic	Produce Ingestion	0.79	1×10^{-2}
	Strontium-90	Produce Ingestion	0.10	
100-H-17	Arsenic	Produce Ingestion	0.90	9×10^{-3}
122-DR-1:2	Arsenic	Produce Ingestion	0.99	7×10^{-3}
1607-D4	Arsenic	Produce Ingestion	0.99	4×10^{-3}
1607-D2:1	Arsenic	Produce Ingestion	0.97	3×10^{-3}
116-DR-9	Cesium-137	External Irradiation	0.54	2×10^{-3}
	Cesium-137	Produce Ingestion	0.20	
	Strontium-90	Produce Ingestion	0.12	
100-D-48:3	Strontium-90	Produce Ingestion	0.42	2×10^{-3}
	Cesium-137	External Irradiation	0.36	
	Cesium-137	Produce Ingestion	0.14	
100-D-49:4	Europium-152	External Irradiation	0.56	9×10^{-4}
	Strontium-90	Produce Ingestion	0.15	
	Technetium-99	Produce Ingestion	0.10	

Site	COPC	Pathway	Fraction of Total Risk	Present-Day Total Risk
100-D-48:2	Cesium-137	External Irradiation	0.33	8×10^{-4}
	Europium-152	External Irradiation	0.22	
	Strontium-90	Produce Ingestion	0.17	
	Cesium-137	Produce Ingestion	0.12	
116-D-1A	Strontium-90	Produce Ingestion	0.48	7×10^{-4}
	Cesium-137	External Irradiation	0.22	
	Europium-152	External Irradiation	0.15	
100-D-48:4	Strontium-90	Produce Ingestion	0.47	7×10^{-4}
	Cesium-137	External Irradiation	0.22	
	Europium-152	External Irradiation	0.10	
100-D-20	Cesium-137	External Irradiation	0.33	6×10^{-4}
	Europium-152	External Irradiation	0.26	
	Strontium-90	Produce Ingestion	0.14	
	Cesium-137	Produce Ingestion	0.12	
116-DR-6	Strontium-90	Produce Ingestion	0.71	6×10^{-4}
	Europium-152	External Irradiation	0.20	
116-D-9	Strontium-90	Produce Ingestion	0.89	5×10^{-4}
116-D-7	Europium-152	External Irradiation	0.34	5×10^{-4}
	Strontium-90	Produce Ingestion	0.32	
	Cesium-137	External Irradiation	0.10	
116-DR-1&2	Strontium-90	Produce Ingestion	0.34	4×10^{-4}
	Cesium-137	External Irradiation	0.23	
	Europium-152	External Irradiation	0.18	
100-D-48:1	Strontium-90	Produce Ingestion	0.46	4×10^{-4}
	Europium-152	External Irradiation	0.20	
	Cesium-137	External Irradiation	0.19	
100-D-4	Strontium-90	Produce Ingestion	0.50	3×10^{-4}
	Europium-152	External Irradiation	0.32	
100-D-49:2	Strontium-90	Produce Ingestion	0.45	2×10^{-4}
	Europium-152	External Irradiation	0.29	
	Cesium-137	External Irradiation	0.12	
116-DR-4	Strontium-90	Produce Ingestion	0.80	2×10^{-4}
116-DR-7	Strontium-90	Produce Ingestion	0.47	2×10^{-4}
	Europium-152	External Irradiation	0.33	

For 11 of the 13 remediated waste sites that have total cancer risks above 1×10^{-3}, a significant fraction of site cancer risk is related to arsenic via the produce ingestion exposure pathway. As shown in Table 5-48, CTUIR Resident arsenic cancer risk calculated using the upland reference area data is 8×10^{-3}. As discussed for Subsistence Farmer cancer risk in Section 5.6.3, the range of site RME arsenic concentrations peaks at 13.8 mg/kg at 100-H-21. By comparison, the RME arsenic soil concentration for the upland reference area, shown in Table 5-47, is 3.2 mg/kg. The 90th percentile arsenic soil background concentrations for the Hanford Area and Yakima Basin are 6.47 mg/kg and 5.13 mg/kg, respectively.

At the remediated waste sites with present-day cancer risk above 1×10^{-4} where arsenic is not the most important contributor, much of the cancer risk is related to short-lived radionuclides including cesium-137, isotopic europium, and strontium-90. The half-lives of these radionuclides range from approximately 13.5 years (europium-152) to approximately 30 years (strontium-90 and cesium-137). Therefore, much of the site-related cancer risk at these waste sites will decrease naturally over time as a function of the decay of these radionuclides.

As shown in Table 5-6, by year 2075 only 13 of the 27 waste sites with present-day cancer risk above 1×10^{-4} still had a cancer risk above this threshold. These are the sites where risk is related primarily to arsenic. Because arsenic concentrations at the remediated waste sites where arsenic is a risk driver are at most about twice as high as commonly referenced background levels, risks for COPCs other than arsenic are also of interest for risk management at these sites. A summary of the key exposure pathways and COPCs, other than arsenic, contributing 10% or more to present-day RME Subsistence Farmer cancer risk at these sites is provided in Section 5.6.3. This summary is also applicable to the CTUIR Resident scenario.

As described in relation to Subsistence Farmer cancer risk results, 100-D/100-H Area remediated waste sites with cancer risks above 1×10^{-4} due to the presence of either short-lived radionuclides or arsenic were generally excavated to significant depths. The use of shallow zone soil verification data in these cases may contribute to a significant protective bias in the risk results for excavated sites. Uncertainties in the use of shallow zone soil verification data from excavated waste sites are discussed in more detail in Section 5.9 and Appendix D-1.

5.6.4.2 Radiation Dose. Table 5-106 shows the total radiation dose results for the CTUIR Resident exposure scenario for all remediated waste sites in the 100-D/100-H Decision Area. The tabulated results are the sum of the dose values for the individual COPCs included in the dose calculation. Total dose results use EPCs based on measured soil concentrations in the individual waste sites. Risk assessment results are shown for present-day radionuclide concentrations, and with radionuclide concentrations decayed to the years 2075 and 2150. Soil concentrations for these times were calculated using the RESRAD computer code, as described in Appendix D-1.4.1.

The present-day total radiation dose results for the CTUIR Resident scenario range from 0.47 to 42 mrem/yr. No radionuclide COPCs were detected at four waste sites. As shown in Table 5-5, 6 of the 35 waste sites evaluated in the 100-D/100-H Decision Area had calculated total radiation dose values above 15 mrem/yr.

A summary of the key exposure pathways and COPCs contributing 10% or more to present-day CTUIR Resident radiation dose at 100-D/100-H Decision Area sites with total dose above 15 mrem/yr is provided below.

Site	COPC	Pathway	Fraction of Total Dose	Present-Day Total Dose (mrem/yr)
116-DR-9	Cesium-137	External Irradiation	0.57	42
	Strontium-90	Produce Ingestion	0.16	
	Cesium-137	Produce Ingestion	0.14	
118-DR-2:2	Strontium-90	Produce Ingestion	0.60	39
	Technetium-99	Produce Ingestion	0.14	
100-D-48:3	Strontium-90	Produce Ingestion	0.50	33
	Cesium-137	External Irradiation	0.34	
116-H-1	Strontium-90	Produce Ingestion	0.72	28
	Europium-152	External Irradiation	0.16	
100-H-17	Strontium-90	Produce Ingestion	0.88	20
100-D-49:4	Europium-152	External Irradiation	0.55	16
	Strontium-90	Produce Ingestion	0.19	

The present-day radiation dose at sites with calculated dose above the threshold is driven mostly by strontium-90 and cesium-137, radionuclides with half-lives of approximately 30 years. No 100-D/100-H waste site exceeds 15 mrem/yr at year 2075. The highest total radiation dose values at years 2075 and 2150 are 8.6 and 2.4 mrem/yr, respectively, at the 118-DR-2:2 waste site. As described for CTUIR cancer risk, the use of shallow zone soil verification data to calculate risks may contribute to a significant protective bias in the radiation dose results for these sites.

5.6.4.3 Chemical Hazard. Table 5-107 shows the HI results for the CTUIR Resident exposure scenario for all remediated waste sites in the 100-D/100-H Decision Area. The tabulated results are the sum of the risk values for the individual COPCs included in the risk calculation. As described in Section 3.4.4, chemical hazard is calculated separately for adult and child receptors. For the CTUIR Resident scenario, child HIs are equal to or higher than those for adults. Therefore, only child HI values are presented here.

The child HI results for the CTUIR Resident scenario range from 0.048 to 380. No chemical COPCs contributing to hazard were detected at three of the remediated waste sites. As shown in Table 5-5, 11 of the 35 waste sites in the 100-D/100-H Decision Area had calculated HI values above 10, and another 2 had HI values between 1.0 and 10.

A summary of the key exposure pathways and COPCs contributing 10% or more to the CTUIR Resident child HI at 100-D/100-H Decision Area sites with HI values above 1.0 is provided below.

Site	COPC	Pathway	Fraction of Child Hazard	Present-Day Child Hazard [a]
1607-H2	Mercury	Produce Ingestion	0.60	380
	Mercury	Beef Ingestion	0.20	
	Arsenic	Produce Ingestion	0.19	
100-H-21	Arsenic	Produce Ingestion	0.98	140
116-H-1	Arsenic	Produce Ingestion	0.99	69
116-H-7	Arsenic	Produce Ingestion	0.98	53
100-H-5	Arsenic	Produce Ingestion	0.89	47
100-H-24	Arsenic	Produce Ingestion	0.98	41
100-H-17	Arsenic	Produce Ingestion	0.98	34
118-DR-2:2	Arsenic	Produce Ingestion	0.97	32
122-DR-1:2	Arsenic	Produce Ingestion	0.97	29
1607-D4	Arsenic	Produce Ingestion	0.96	15
1607-D2:1	Arsenic	Produce Ingestion	0.95	11
1607-D2:3	Mercury	Produce Ingestion	0.75	3.1
	Mercury	Beef Ingestion	0.24	
100-D-49:4	Mercury	Produce Ingestion	0.66	2.3
	Mercury	Beef Ingestion	0.21	

[a] Values for mercury hazard at 1607-H2 are likely overestimated; see discussion in the text.

Mercury exposure via produce ingestion is the most important exposure pathway contributing to HI values above 1.0 at the 1607-H2 remediated waste site, where the child HI is 380, and is also a contributor to HI values at 1607-D2:3 and 100-D-49:4. Arsenic exposure via produce ingestion is the most important contributor to HI at the other waste sites summarized above. As described for CTUIR cancer risk, the use of shallow zone soil verification data to assess risks may contribute to a significant protective bias in the HI results for these sites.

Critical effects underlying the oral RfD for mercury include effects on the immune and renal (kidney) system, whereas the critical effects underlying the oral RfD for arsenic relate to the vascular and integumentary (skin) system (IRIS 2009). Summing HQs for mercury and arsenic at the 1607-H2 waste site likely results in a protective bias for the tabulated HI values at this site.

The 95% UCL soil mercury concentration at the 1607-H2 waste site (5.0 mg/kg, based on eight samples with a maximum detected concentration of 3.5 mg/kg) is more than 200 times higher than the reference area mercury concentration of 0.022 mg/kg. As discussed in the Uncertainty Analysis (Section 5.9), the linear models used in this risk assessment to estimate plant tissue concentrations from soil data are prone to overestimation of plant concentrations at higher soil concentrations. The mercury plant-soil regression model published by Bechtel (BJC/OR-133) predicts a mercury plant concentration of 0.89 mg/kg when the soil concentration is 5.0 mg/kg. By contrast, the dry-weight produce and fodder concentrations using the linear plant-soil ratios project plant concentration are approximately 13 and 5.0 mg/kg, respectively. This suggests mercury exposure at the 1607-H2 waste site may be overestimated by a factor of about 15 and 5.6 for produce ingestion and beef ingestion, respectively. In that case, the mercury HQ at 1607-H2 for produce ingestion would both be about 18, and for beef ingestion about 14.

Because arsenic concentrations at the remediated waste sites where arsenic is a risk driver are at most about twice as high as commonly referenced background levels, risks for COPCs other than arsenic are also of interest for risk management at these sites. A summary of the key exposure pathways and COPCs, other than arsenic, contributing 10% or more to present-day RME Subsistence Farmer child HI at these sites is provided in Section 5.6.3. This summary is also applicable to the CTUIR Resident scenario.

5.6.5 Yakama Resident Scenario Results for the 100-D/100-H Area

The Yakama Resident exposure scenario encompasses potential exposures related to shallow zone soils around remediated waste sites, including both direct contact exposure pathways and exposures related to raising produce and animals in the potentially affected area of the waste site. This scenario also includes exposures related to ingestion of Columbia River fish (see Section 4.4) and exposure to groundwater via a domestic water well (see Section 6.0). It is similar to the Subsistence Farmer scenario, although exposure intensity is higher for many exposure pathways. The scenario is more fully described in the CSM in Section 3.3 and is summarized in Table 3-13 and Figure 3-17. The COPCs evaluated in shallow zone soil in the 100-D/100-H Decision Area are shown in Table 5-26.

5.6.5.1 Cancer Risk. Table 5-108 shows the total cancer risk results for the Yakama Resident exposure scenario for all remediated waste sites in the 100-D/100-H Decision Area. The tabulated results are the sum of the risk values for the individual COPCs included in the risk calculation. Total risk results use EPCs based on measured soil concentrations in the individual waste sites. Risk assessment results are shown for present-day radionuclide concentrations, and with radionuclide concentrations decayed to the years 2075 and 2150. Soil concentrations for these times were calculated using the RESRAD computer code, as described in Appendix D-1.4.1.

The present-day total cancer risk results for the Yakama Resident scenario (radionuclides + chemicals) range from 3×10^{-5} to 4×10^{-2}. No carcinogenic COPCs were detected at the 100-D-12 and 100-D-22 waste sites. Per EPA guidance (EPA/540/1-89/002), cancer risk results are displayed with only one significant figure. As shown in Table 5-5, 27 of the 35 waste sites evaluated in the 100-D/100-H Decision Area had calculated total cancer risks above 1×10^{-4}.

A summary of the key exposure pathways and COPCs contributing 10% or more to present-day Yakama Resident cancer risk at 100-D/100-H Decision Area sites with total cancer risk above 1×10^{-4} is provided below.

Site	COPC	Pathway	Fraction of Total Risk	Present-Day Total Risk
100-H-21	Arsenic	Produce Ingestion	0.94	4×10^{-2}
1607-H2	Arsenic	Produce Ingestion	0.97	2×10^{-2}
116-H-1	Arsenic	Produce Ingestion	0.89	2×10^{-2}
116-H-7	Arsenic	Produce Ingestion	0.92	1×10^{-2}
100-H-5	Arsenic	Produce Ingestion	0.95	1×10^{-2}
118-DR-2:2	Arsenic	Produce Ingestion	0.74	1×10^{-2}
100-H-24	Arsenic	Produce Ingestion	0.97	1×10^{-2}
100-H-17	Arsenic	Produce Ingestion	0.86	1×10^{-2}
122-DR-1:2	Arsenic	Produce Ingestion	0.97	7×10^{-3}
1607-D4	Arsenic	Produce Ingestion	0.97	4×10^{-3}
116-DR-9	Cesium-137	External Irradiation	0.42	3×10^{-3}
	Cesium-137	Produce Ingestion	0.16	
	Cesium-137	Beef Ingestion	0.14	
1607-D2:1	Arsenic	Produce Ingestion	0.95	3×10^{-3}
100-D-48:3	Strontium-90	Produce Ingestion	0.32	2×10^{-3}
	Cesium-137	External Irradiation	0.27	
	Cesium-137	Produce Ingestion	0.10	
	Strontium-90	Beef Ingestion	0.10	
100-D-48:2	Cesium-137	External Irradiation	0.27	9×10^{-4}
	Europium-152	External Irradiation	0.18	
	Strontium-90	Produce Ingestion	0.14	
	Cesium-137	Produce Ingestion	0.10	
100-D-49:4	Europium-152	External Irradiation	0.50	9×10^{-4}
	Strontium-90	Produce Ingestion	0.14	
116-D-1A	Strontium-90	Produce Ingestion	0.37	8×10^{-4}
	Cesium-137	External Irradiation	0.16	
	Strontium-90	Beef Ingestion	0.12	
	Europium-152	External Irradiation	0.12	
100-D-48:4	Strontium-90	Produce Ingestion	0.37	8×10^{-4}
	Cesium-137	External Irradiation	0.17	
	Strontium-90	Beef Ingestion	0.12	
116-DR-6	Strontium-90	Produce Ingestion	0.55	8×10^{-4}
	Strontium-90	Beef Ingestion	0.18	
	Europium-152	External Irradiation	0.15	

Site	COPC	Pathway	Fraction of Total Risk	Present-Day Total Risk
100-D-20	Cesium-137	External Irradiation	0.27	7×10^{-4}
	Europium-152	External Irradiation	0.22	
	Strontium-90	Produce Ingestion	0.12	
	Cesium-137	Produce Ingestion	0.10	
116-D-9	Strontium-90	Produce Ingestion	0.65	6×10^{-4}
	Strontium-90	Beef Ingestion	0.21	
116-D-7	Europium-152	External Irradiation	0.28	5×10^{-4}
	Strontium-90	Produce Ingestion	0.27	
116-DR-1&2	Strontium-90	Produce Ingestion	0.27	5×10^{-4}
	Cesium-137	External Irradiation	0.18	
	Europium-152	External Irradiation	0.14	
100-D-48:1	Strontium-90	Produce Ingestion	0.36	5×10^{-4}
	Europium-152	External Irradiation	0.15	
	Cesium-137	External Irradiation	0.15	
	Strontium-90	Beef Ingestion	0.12	
100-D-4	Strontium-90	Produce Ingestion	0.41	3×10^{-4}
	Europium-152	External Irradiation	0.26	
	Strontium-90	Beef Ingestion	0.13	
100-D-49:2	Strontium-90	Produce Ingestion	0.37	3×10^{-4}
	Europium-152	External Irradiation	0.23	
	Strontium-90	Beef Ingestion	0.12	
116-DR-4	Strontium-90	Produce Ingestion	0.59	3×10^{-4}
	Strontium-90	Beef Ingestion	0.19	
116-DR-7	Strontium-90	Produce Ingestion	0.39	2×10^{-4}
	Europium-152	External Irradiation	0.27	
	Strontium-90	Beef Ingestion	0.12	
116-D-2	Strontium-90	Produce Ingestion	0.48	1×10^{-4}
	Strontium-90	Beef Ingestion	0.15	
100-D-21	Strontium-90	Produce Ingestion	0.50	1×10^{-4}
	Europium-152	External Irradiation	0.17	
	Strontium-90	Beef Ingestion	0.16	

For 11 of the 13 remediated waste sites with total cancer risks above 1×10^{-3}, a significant fraction of site cancer risk is related to arsenic via the produce ingestion exposure pathway. As shown in Table 5-48, Yakama Resident arsenic cancer risk for the Yakama Resident scenario calculated using upland reference area data is 8×10^{-3}. As discussed for Subsistence Farmer cancer risk in Section 5.6.3, the range of site RME arsenic concentrations peaks at 13.8 mg/kg at 100-H-21. By comparison, the RME arsenic soil concentration for the upland reference area, shown in Table 5-47, is 3.2 mg/kg. The 90th percentile arsenic soil background concentrations for the Hanford Area and Yakima Basin are 6.47 mg/kg and 5.13 mg/kg, respectively.

At the remediated waste sites with present-day cancer risk above 1×10^{-4} where arsenic is not the most important contributor, much of the cancer risk is related to short-lived radionuclides including cesium-137, isotopic europium, and strontium-90. The half-lives of these radionuclides range from approximately 13.5 years (europium-152) to approximately 30 years (strontium-90 and cesium-137). Therefore, much of the site-related cancer risk at these waste sites will decrease naturally over time as a function of the decay of these radionuclides.

As shown in Table 5-6, by year 2075 only 16 of the 27 waste sites with present-day cancer risk above 1×10^{-4} still had a cancer risk above this threshold. These are the sites where risk is related primarily to arsenic. Because arsenic concentrations at the remediated waste sites where arsenic is a risk driver are at most about twice as high as commonly referenced background levels, risks for COPCs other than arsenic are also of interest for risk management at these sites. A summary of the key exposure pathways and COPCs, other than arsenic, contributing 10% or more to present-day RME Subsistence Farmer cancer risk at these sites is provided in Section 5.6.3. This summary is also applicable to the Yakama Resident scenario.

As described in relation to Subsistence Farmer cancer risk results, 100-D/100-H Area remediated waste sites with cancer risks above 1×10^{-4} due to the presence of either short-lived radionuclides or arsenic were generally excavated to significant depths. The use of shallow zone soil verification data in these cases may contribute to a significant protective bias in the risk results for excavated sites. Uncertainties in the use of shallow zone soil verification data from excavated waste sites are discussed in more detail in Section 5.9 and Appendix D-1.

5.6.5.2 Radiation Dose. Table 5-109 shows the total radiation dose results for the Yakama Resident exposure scenario for all remediated waste sites in the 100-D/100-H Decision Area. The tabulated results are the sum of the dose values for the individual COPCs included in the dose calculation. Total dose results use EPCs based on measured soil concentrations in the individual waste sites. Risk assessment results are shown for present-day radionuclide concentrations, and with radionuclide concentrations decayed to the years 2075 and 2150. Soil concentrations for these times were calculated using the RESRAD computer code, as described in Appendix D-1.4.1.

The present-day total radiation dose results for the Yakama Resident scenario range from 0.53 to 49 mrem/yr. No radionuclide COPCs were detected at four waste sites. As shown in Table 5-5, 10 of the 35 waste sites evaluated in the 100-D/100-H Decision Area had calculated total radiation dose values above 15 mrem/yr.

A summary of the key exposure pathways and COPCs contributing 10% or more to the present-day Yakama Resident radiation dose at 100-D/100-H Decision Area sites with total dose above 15 mrem/yr is provided below. Several sites have total radiation dose results of 16 or 17 mrem/yr; for brevity these sites are not included.

Site	COPC	Pathway	Fraction of Total Dose	Present-Day Total Dose (mrem/yr)
116-DR-9	Cesium-137	External Irradiation	0.48	49
	Strontium-90	Produce Ingestion	0.12	
	Cesium-137	Produce Ingestion	0.11	
	Cesium-137	Beef Ingestion	0.10	
118-DR-2:2	Strontium-90	Produce Ingestion	0.45	46
	Strontium-90	Beef Ingestion	0.15	
	Technetium-99	Produce Ingestion	0.11	
100-D-48:3	Strontium-90	Produce Ingestion	0.36	40
	Cesium-137	External Irradiation	0.28	
	Strontium-90	Beef Ingestion	0.12	
116-H-1	Strontium-90	Produce Ingestion	0.53	33
	Strontium-90	Beef Ingestion	0.18	
	Europium-152	External Irradiation	0.13	
100-H-17	Strontium-90	Produce Ingestion	0.63	25
	Strontium-90	Beef Ingestion	0.21	
	Strontium-90	Milk Ingestion	0.10	

The present-day radiation dose at sites with calculated dose above the threshold is driven mostly by strontium-90 and cesium-137, radionuclides with half-lives of approximately 30 years. Technetium-99 has a long half-life of more than 200,000 years, but it is very water soluble and, therefore, subject to leaching due to precipitation. No 100-D/100-H waste site exceeds 15 mrem/yr at year 2075. The highest total radiation dose values at years 2075 and 2150 are 9.9 and 2.7 mrem/yr, respectively, at the 118-DR-2:2 waste site. As described for Yakama Resident cancer risk, the use of shallow zone soil verification data to calculate risks may contribute to a significant protective bias in the radiation dose results for these sites.

5.6.5.3 Chemical Hazard. Table 5-110 shows the HI results for the Yakama Resident exposure scenario for all remediated waste sites in the 100-D/100-H Decision Area. The tabulated results are the sum of the risk values for the individual COPCs included in the risk calculation. As described in Section 3.4.4, chemical hazard is calculated separately for adult and child receptors. For the Yakama Resident scenario, child HIs are equal to or higher than those for adults. Therefore, only child HI values are presented here.

The child HI results for the Yakama Resident scenario range from 0.16 to 430. No chemical COPCs contributing to hazard were detected at three of the remediated waste sites. As shown in

Table 5-5, 9 of the 35 waste sites in the 100-D/100-H Decision Area had calculated HI values above 10, and another four had HI values between 1.0 and 10.

A summary of the key exposure pathways and COPCs contributing 10% or more to Yakama Resident child HIs at 100-D/100-H Decision Area sites with HI values above 1.0 is provided below.

Site	COPC	Pathway	Fraction of Child Hazard	Present-Day Child Hazard [a]
1607-H2	Mercury	Beef Ingestion	0.61	430
	Mercury	Produce Ingestion	0.29	
100-H-21	Arsenic	Produce Ingestion	0.95	76
116-H-1	Arsenic	Produce Ingestion	0.95	38
116-H-7	Arsenic	Produce Ingestion	0.93	30
100-H-5	Arsenic	Produce Ingestion	0.76	29
	Mercury	Beef Ingestion	0.13	
100-H-24	Arsenic	Produce Ingestion	0.95	23
100-H-17	Arsenic	Produce Ingestion	0.93	19
118-DR-2:2	Arsenic	Produce Ingestion	0.93	18
122-DR-1:2	Arsenic	Produce Ingestion	0.93	16
1607-D4	Arsenic	Produce Ingestion	0.92	8.6
1607-D2:1	Arsenic	Produce Ingestion	0.89	6.4
1607-D2:3	Mercury	Beef Ingestion	0.67	3.9
	Mercury	Produce Ingestion	0.32	
100-D-49:4	Mercury	Beef Ingestion	0.62	2.8
	Mercury	Produce Ingestion	0.29	

[a] Values for mercury hazard at 1607-H2 are likely overestimated; see discussion in the text.

Mercury exposure via beef ingestion is the most important exposure pathway contributing to HI values above 1.0 at the 1607-H2 remediated waste site, where the child HI is 430, and is also a contributor to HI values at 1607-D2:3 and 100-D-49:4. The higher beef ingestion rate, and associated higher HI value, is one of the characteristics differentiating the Yakama Resident scenario from the CTUIR scenario. Arsenic exposure via produce ingestion is the most important contributor to the HI at the other waste sites summarized above. As described for Yakama Resident cancer risk, the use of shallow zone soil verification data to assess risks may contribute to a significant protective bias in the HI results for these sites.

Critical effects underlying the oral RfD for mercury include effects on the immune and renal (kidney) system, whereas the critical effects underlying the oral RfD for arsenic relate to the vascular and integumentary (skin) system (IRIS 2009). Summing HQs for mercury and arsenic at the 100-H-5 waste site likely results in a protective bias for the tabulated HI values at this site.

As discussed in relation to CTUIR Resident child HI, the 95% UCL soil mercury concentration at the 1607-H2 waste site (5.0 mg/kg, based on eight samples with a maximum detected concentration of 3.5 mg/kg) is more than 200 times higher than the reference area mercury concentration of 0.022 mg/kg. Because the linear models used in this risk assessment to estimate plant tissue concentrations from soil data are prone to overestimation of plant concentrations at higher soil concentrations, mercury exposure at the 1607-H2 waste site may be overestimated. This is discussed further in the Uncertainty Analysis (Section 5.9).

Because arsenic concentrations at the remediated waste sites where arsenic is a risk driver are at most about twice as high as commonly referenced background levels, risks for COPCs other than arsenic are also of interest for risk management at these sites. A summary of the key exposure pathways and COPCs, other than arsenic, contributing 10% or more to present-day RME Subsistence Farmer child HI at these sites is provided in Section 5.6.3. This summary is also applicable to the Yakama Resident scenario.

5.6.6 Risks Related to Lead in Soil for the 100-D/100-H Area

The highest calculated RME representative concentration for lead in shallow zone soil at any remediated waste site is approximately 210 mg/kg. This concentration applies to the 100-F-37 remediated waste site in the 100-F/100-IU-2/100-IU-6 Area. The highest RME representative concentration for lead in the 100-D/100-H Area is 42 mg/kg at the 1607-H2 waste site. No representative concentrations for lead at any waste site exceed EPA’s recommended residential screening level for lead in soil of 400 mg/kg, derived using the IEUBK model that has been associated with bare soil in a play area (40 CFR 745). Because soil concentrations for lead are well below the most restrictive of EPA’s soil screening criteria, no additional evaluation of lead is included in the HHRA. More detailed information on lead toxicity and the basis of EPA’s soil screening criteria is provided in Section 3.5.8.

5.7 100-F, 100-IU-2, AND 100-IU-6 DECISION AREA RESULTS

Section 5.7 is the fifth of six sections of the local-area risk assessment (Sections 5.3 through 5.8) that present the results of the soil source term risk assessment for remediated waste sites in an individual ROD decision area. A summary of the results of the waste site soils risk assessment for all ROD decision areas is shown in Tables 5-2 through 5-7. Tables 5-2, 5-3, and 5-4 show the number of remediated waste sites with risk assessment results above or below decision thresholds for the Occupational scenarios. Table 5-2 shows results using present-day radionuclide soil concentrations, and the latter two tables show the results with radionuclide concentrations at years 2075 and 2150. Tables 5-5, 5-6, and 5-7 show these same summaries for the Residential exposure scenarios. The primary purpose of these summary tables is to allow the reader to quickly see how the waste sites risk assessment results for the different exposure scenarios compare to the thresholds used for risk management decisions. These tables also provide a perspective of how the waste site risk assessment results vary among the six ROD decision areas, and how cancer risk and radiation dose will change over time due to radioactive decay.

As described in Section 3.3, broad-area EPCs in the upland environment are associated with the occupational portion of the Resident National Monument/Refuge scenario and are also used to supplement the local-area cleanup verification data to calculate EPCs for dust inhalation for all scenarios evaluated in Sections 5.3 through 5.8. The COPCs for the broad-area components of the waste site risk calculations are those related to upland surface soil, while the COPCs for the local-area components of the waste site risk calculations are those related to the cleanup verification data for a particular ROD decision area. Risk assessment results for the Resident National Monument/Refuge scenario are provided for the local-area and broad-area components separately. Broad-area representative concentrations for upland surface soil are calculated using the MIS soil data from all 20 upland sampling locations, using the same protocol described for MIS reference area data in Section 5.2.4.

5.7.1 Industrial/Commercial Scenario Results for the 100-F/100-IU-2/100-IU-6 Area

The Industrial/Commercial exposure scenario encompasses potential occupational exposures related to shallow zone soils around remediated waste sites. Like the Resident National Monument/Refuge scenario, it focuses on direct contact exposure pathways and does not encompass the raising of produce or animals, nor the ingestion of Columbia River fish. It is similar to the Industrial/Commercial scenario used by EPA to calculate soil screening values using direct contact exposure pathways such as inadvertent soil ingestion, dermal contact with soil, and inhalation. Of the scenarios evaluated on the scale of an individual remediated waste site, the Industrial/Commercial scenario has the lowest exposure intensity. The scenario is more fully described in the CSM in Section 3.3 and is summarized in Table 3-12 and Figure 3-16. The COPCs evaluated in shallow zone soil in the 100-F/100-IU-2/100-IU-6 Decision Area are shown in Table 5-26.

5.7.1.1 Cancer Risk. Table 5-111 shows the RME and CTE total cancer risk results for the Industrial/Commercial exposure scenario for all remediated waste sites in the 100-F/100-IU-2/100-IU-6 Decision Area. The tabulated results are the sum of the risk values for the individual COPCs included in the risk calculation. Total risk results use EPCs based on measured soil concentrations in the individual waste sites. Industrial/Commercial risk assessment results are shown for present-day radionuclide concentrations, and with radionuclide concentrations decayed to the years 2075 and 2150. Soil concentrations for these times were calculated using the RESRAD computer code, as described in Appendix D-1.4.1.

The range of present-day RME and CTE total cancer risk results for the Industrial/Commercial scenario (radionuclides + chemicals) is from 5×10^{-9} to 2×10^{-4} and from 1×10^{-9} to 1×10^{-5}, respectively. No carcinogenic COPCs were detected at the 100-F-11 remediated waste site. Per EPA guidance (EPA/540/1-89/002), cancer risk results are displayed with only one significant figure. As shown in Table 5-2, there is only one remediated waste sites in the 100-F/100-IU-2/100-IU-6 Decision Area where total RME or CTE cancer risks were above 1×10^{-4} for the Industrial/Commercial scenario.

A summary of the key exposure pathways and COPCs contributing 10% or more to present-day RME Industrial/Commercial cancer risk at 100-F/100-IU-2/100-IU-6 Decision Area sites with total cancer risk above 1×10^{-4} is provided below.

Site	COPC	Pathway	Fraction of Total Risk	Present-Day Total Risk
116-F-14	Europium-152	External Irradiation	0.91	2×10^{-4}

The total RME cancer risk was 2×10^{-4} at the 116-F-14 waste site. About 90% of the RME cancer risk at the 116-F-14 waste site is due to external irradiation from the short-lived radionuclide europium-152. By year 2075, the highest RME total cancer risk is 7×10^{-6} at the 116-F-14 waste site.

5.7.1.2 Radiation Dose. Table 5-112 shows the RME and CTE total radiation dose results for the Industrial/Commercial exposure scenario for all remediated waste sites in the 100-F/100-IU-2/100-IU-6 Decision Area. The tabulated results are the sum of the dose values for the individual COPCs included in the dose calculation. Risk assessment results are shown for present-day radionuclide concentrations, and with radionuclide concentrations decayed to the years 2075 and 2150. Soil concentrations for these times were calculated using the RESRAD computer code, as described in Appendix D-1.4.1.

The range of present-day RME and CTE total radiation dose results for the Industrial/Commercial scenario is from 0.00057 to 8.6 mrem/yr and from 0.000053 to 2.0 mrem/yr, respectively. No radionuclide COPCs were detected at 22 remediated waste sites.

There are no waste sites in the 100-F/100-IU-2/100-IU-6 Area where the radiation dose for the Industrial/Commercial scenario exceeded the threshold of 15 mrem/yr. About 90% of the present-day total RME dose of 8.6 mrem/yr at the 116-F-14 waste site is related to external irradiation from the short-lived radionuclide europium-152. By year 2075, the highest RME total radiation dose is 0.36 mrem/yr at the 116-F-14 waste site.

5.7.1.3 Chemical Hazard. Table 5-113 shows the RME and CTE HI results for the Industrial/Commercial exposure scenario for all remediated waste sites in the 100-F/100-IU-2/100-IU-6 Decision Area. The tabulated results are the sum of the risk values for the individual COPCs included in the risk calculation.

The range of RME and CTE HI results for the Industrial/Commercial scenario is from 0.00011 to 0.050 and from 0.00010 to 0.0091, respectively. No chemical COPCs contributing to hazard were detected at eight remediated waste sites in the 100-F/100-IU-2/100-IU-6 Decision Area. There are no waste sites in the 100-F/100-IU-2/100-IU-6 Area, where the HI for the Industrial/Commercial scenario approaches the threshold of 1.0.

5.7.2 Resident National Monument/Refuge Scenario Results for the 100-F/100-IU-2/100-IU-6 Area

The Resident National Monument/Refuge exposure scenario encompasses potential residential exposures related to shallow zone soils around remediated waste sites, as well as potential occupational exposures to surface soil across the upland environment of the River Corridor. It focuses on direct contact exposure pathways and does not encompass the raising of produce or animals nor the ingestion of Columbia River fish. This scenario also includes exposures related to groundwater via a domestic water well. Risks related to groundwater use are addressed in Section 6.0. With respect to soil exposure pathways, this scenario is similar to the traditional urban/suburban residential scenario used by EPA to calculate soil screening values using direct contact exposure pathways such as inadvertent soil ingestion, dermal contact with soil, and inhalation. It may be viewed as akin to the Subsistence Farmer scenario for an adult receptor, minus the ingestion of foodstuffs and with an exposure model that averages exposure to both individual waste site soils (cleanup verification soil data) and surface soils over large areas (upland surface soil data). The scenario is more fully described in the CSM in Section 3.3 and is summarized in Table 3-12 and Figure 3-16. The COPCs evaluated in shallow zone soil in the 100-F/100-IU-2/100-IU-6 Decision Area are shown in Table 5-26.

As described in the introductory text for Section 5.4, COPCs for the broad-area components of the waste site risk calculations are those related to upland surface soil, while the COPCs for the local-area components of the waste site risk calculations are those related to the cleanup verification data for a particular ROD decision area. Results for the local-area portion of the Resident National Monument/Refuge exposure scenario are calculated for each individual remediated waste site where a residence might be situated. Risks related to the broad-area occupational component of the Resident National Monument/Refuge exposure scenario are shown in the first row of the summary tables for cancer risk, radiation dose, and chemical hazard. As discussed in Section 4.1.3, broad-area representative concentrations are calculated using upland surface soil data collected under DOE/RL-2005-42.

5.7.2.1 Cancer Risk. Table 5-114 shows the RME and CTE total cancer risk results for the Resident National Monument/Refuge exposure scenario for all remediated waste sites in the 100-F/100-IU-2/100-IU-6 Decision Area. The tabulated results are the sum of the risk values for the individual COPCs included in the risk calculation. Total risk results use EPCs based on measured soil concentrations in the individual remediated waste sites (residential exposure) and River Corridor upland surface soil (occupational exposure). Risk assessment results are shown for present-day radionuclide concentrations, and with radionuclide concentrations decayed to the years 2075 and 2150. Soil concentrations for these times were calculated using the RESRAD computer code, as described in Appendix D-1.4.1.

The range of present-day RME and CTE total cancer risk results for the local-area component of the Resident National Monument/Refuge scenario (radionuclides + chemicals) is from 1×10^{-8} to 3×10^{-4} and from 3×10^{-9} to 2×10^{-5}, respectively. No carcinogenic COPCs were detected at the 100-F-11 remediated waste site. Per EPA guidance (EPA/540/1-89/002), cancer risk results are displayed with only one significant figure. As shown in Table 5-2, only 1 of the 44 remediated waste sites evaluated in the 100-F/100-IU-2/100-IU-6 Decision Area had a combined local and broad present-day RME cancer risk above 1×10^{-4}, and no waste site had a CTE cancer risk above 1×10^{-4}. A present-day RME cancer risk of 1×10^{-4} was calculated for the UPR-100-F-2 waste site.

Because the broad-area component of cancer risk is the same across all calculations, the summary of the key exposure pathways and COPCs contributing 10% or more to present-day RME Resident National Monument/Refuge cancer risk at 100-F/100-IU-2/100-IU-6 Decision Area sites focuses on the local-area component of the Resident National Monument/Refuge scenario. The local-area calculations with total cancer risk above 1×10^{-4} are summarized below.

Site	COPC	Pathway	Fraction of Total Risk	Present-Day Total Risk
116-F-14	Europium-152	External Irradiation	0.90	3×10^{-4}

About 90% of present-day total RME cancer risk at the 116-F-14 remediated waste site is related to external irradiation from the short-lived radionuclide europium-152. There are no waste sites in the 100-F/100-IU-2/100-IU-6 Area where total RME cancer risk is above 1×10^{-4} at year 2075 for the Resident National Monument/Refuge exposure scenario. The highest local-area cancer risk value at year 2075 is 1×10^{-5} at the 116-F-14 waste site.

The current condition of the 116-F-14 remediated waste site is summarized in Table 2-8. The 116-F-14 waste site was excavated to a depth of 4.6 m (15 ft) and then backfilled to grade following remediation. The shallow zone soil verification data used to calculate local-area risk are from samples collected along the sidewalls of the excavation at depths between 0 and 4.6 m (15 ft). The use of these data to represent surface soil for current and future exposures may contribute to a significant protective bias in the Resident National Monument/Refuge risk results for this site. Uncertainties in the use of shallow zone soil verification data from excavated waste sites are discussed in more detail in Section 5.9 and Appendix D-1.

5.7.2.2 Radiation Dose. Table 5-115 shows the RME and CTE total radiation dose results for the Resident National Monument/Refuge exposure scenario for all remediated waste sites in the 100-F/100-IU-2/100-IU-6 Decision Area. The tabulated results are the sum of the dose values for the individual COPCs included in the dose calculation. Risk assessment results are shown for present-day radionuclide concentrations, and with radionuclide concentrations decayed to the years 2075 and 2150. Soil concentrations for these times were calculated using the RESRAD computer code, as described in Appendix D-1.4.1.

The range of present-day RME and CTE total radiation dose results for the local-area component of the Resident National Monument/Refuge scenario is from 0.0013 to 15 mrem/yr and from 0.00043 to 3.3 mrem/yr, respectively. The RME and CTE radiation dose results for the broad-area component were 0.80 and 0.78 mrem/yr, respectively. No radionuclide COPCs were detected at 22 remediated waste sites.

As shown in Table 5-2, only one of the remediated waste sites evaluated in the 100-F/100-IU-2/100-IU-6 Decision Area had a combined local-area and broad-area present-day RME radiation dose value above 15 mrem/yr. Approximately 90% of the 15 mrem/yr local-area component of the combined RME dose of 16 mrem/yr at the 116-F-14 remediated waste site is due to external irradiation from europium-152. By year 2075, the highest RME local-area dose is 0.62 mrem/yr at 116-F-14.

5.7.2.3 Chemical Hazard. Table 5-116 shows the RME and CTE HI results for the Resident National Monument/Refuge exposure scenario for all remediated waste sites in the 100-F/100-IU-2/100-IU-6 Decision Area. The tabulated results are the sum of the risk values for the individual COPCs included in the risk calculation.

The range of RME and CTE HI results for the local-area component of the Resident National Monument/Refuge scenario is from 0.0019 to 0.09 and from 0.0018 to 0.02, respectively. The RME and CTE HI results for the broad-area component were 0.025 and 0.0079, respectively. No chemical COPCs contributing to hazard were detected at eight of the waste sites. There are no waste sites in the 100-F/100-IU-2/100-IU-6 Area, where the HI for the Resident National Monument/Refuge scenario approaches the threshold of 1.0.

5.7.3 Subsistence Farmer Scenario Results for the 100-F/100-IU-2/100-IU-6 Area

The Subsistence Farmer exposure scenario encompasses potential exposures related to shallow zone soils around remediated waste sites, including both direct contact exposure pathways and exposures related to raising produce and animals in the potentially affected area of the waste site. This scenario also includes exposures related to ingestion of Columbia River fish and exposure to groundwater via a domestic water well. Risks related to fish ingestion and groundwater use are addressed in Sections 4.4 and 6.0, respectively. The scenario is more fully described in the CSM in Section 3.3 and is summarized in Table 3-13 and Figure 3-17. The COPCs evaluated in shallow zone soil in the 100-F/100-IU-2/100-IU-6 Decision Area are shown in Table 5-26.

5.7.3.1 Cancer Risk. Table 5-117 shows the RME and CTE total cancer risk results for the Subsistence Farmer exposure scenario for all remediated waste sites in the 100-F/100-IU-2/100-IU-6 Decision Area. The tabulated results are the sum of the risk values for the individual COPCs included in the risk calculation. Total risk results use EPCs based on measured soil concentrations in the individual waste sites. Risk assessment results are shown for present-day radionuclide concentrations, and with radionuclide concentrations decayed to the years 2075 and 2150. Soil concentrations for these times were calculated using the RESRAD computer code, as described in Appendix D-1.4.1.

The range of present-day RME and CTE total cancer risk results for the Subsistence Farmer scenario (radionuclides + chemicals) is from 2×10^{-8} to 3×10^{-3} and from 7×10^{-9} to 3×10^{-4}, respectively. No carcinogenic COPCs were detected at the 100-F-11 remediated waste site. Per EPA guidance (EPA/540/1-89/002), cancer risk results are displayed with only one significant figure. As shown in Table 5-5, 27 of the 44 remediated waste sites evaluated in the

100-F/100-IU-2/100-IU-6 Decision Area had calculated total RME cancer risks above 1×10^{-4}, but only 1 waste site had a CTE total cancer risk above 1×10^{-4}.

A summary of the key exposure pathways and COPCs contributing 10% or more to present-day RME Subsistence Farmer cancer risk at 100-F/100-IU-2/100-IU-6 Decision Area sites with total cancer risk above 1×10^{-4} is provided below.

Site	COPC	Pathway	Fraction of Total Risk	Present-Day Total Risk
100-F-37	Arsenic	Produce Ingestion	0.85	3×10^{-3}
116-F-1	Arsenic	Produce Ingestion	0.77	1×10^{-3}
116-F-14	Europium-152	External Irradiation	0.45	1×10^{-3}
	Arsenic	Produce Ingestion	0.32	
600-235	Arsenic	Produce Ingestion	0.85	7×10^{-4}
100-F-26:1	Arsenic	Produce Ingestion	0.83	5×10^{-4}
600-131	Arsenic	Produce Ingestion	0.85	5×10^{-4}
600-23	Arsenic	Produce Ingestion	0.80	5×10^{-4}
600-181	Arsenic	Produce Ingestion	0.85	5×10^{-4}
628-1	Arsenic	Produce Ingestion	0.85	5×10^{-4}
116-F-9	Strontium-90	Milk Ingestion	0.38	4×10^{-4}
	Strontium-90	Produce Ingestion	0.32	
	Strontium-90	Beef Ingestion	0.17	
	Carbon-14	Produce Ingestion	0.11	
600-204	Arsenic	Produce Ingestion	0.85	4×10^{-4}
600-190	Arsenic	Produce Ingestion	0.81	4×10^{-4}
600-132	Arsenic	Produce Ingestion	0.84	4×10^{-4}
100-F-12	Arsenic	Produce Ingestion	0.85	4×10^{-4}
100-F-7	Arsenic	Produce Ingestion	0.85	4×10^{-4}
100-F-9	Arsenic	Produce Ingestion	0.85	4×10^{-4}
100-F-38	Arsenic	Produce Ingestion	0.85	4×10^{-4}
100-F-26:7	Arsenic	Produce Ingestion	0.85	4×10^{-4}
600-128	Arsenic	Produce Ingestion	0.85	3×10^{-4}
100-F-26:2	Arsenic	Produce Ingestion	0.85	3×10^{-4}
100-F-26:5	Arsenic	Produce Ingestion	0.85	3×10^{-4}
116-F-6	Strontium-90	Milk Ingestion	0.31	3×10^{-4}
	Strontium-90	Produce Ingestion	0.26	
	Strontium-90	Beef Ingestion	0.14	
	Cesium-137	External Irradiation	0.12	
	Europium-152	External Irradiation	0.10	

Site	COPC	Pathway	Fraction of Total Risk	Present-Day Total Risk
116-F-5	Arsenic	Produce Ingestion	0.80	3×10^{-4}
100-F-14	Arsenic	Produce Ingestion	0.85	3×10^{-4}
100-F-18	Arsenic	Produce Ingestion	0.85	3×10^{-4}
UPR-100-F-2	Europium-152	External Irradiation	0.88	2×10^{-4}
116-F-10	Cesium-137	External Irradiation	0.45	2×10^{-4}
	Europium-152	External Irradiation	0.35	

With the exception of the 116-F-14, 116-F-9, 116-F-6, UPR-100-F-2, and 116-F-10 remediated waste sites, the cancer risk at waste sites with present-day RME cancer risk above 1×10^{-4} is related primarily to arsenic by ingestion of home-grown produce. At the 116-F-14, 116-F-9, 116-F-6, UPR-100-F-2, and 116-F-10 waste sites, risk is mostly related to radionuclides including cesium-137, europium-152, and strontium-90. The half-lives of these radionuclides range from approximately 13.5 years (europium-152) to approximately 30 years (strontium-90 and cesium-137). Therefore, much of the cancer risk at these five sites will decrease naturally over time as a function of the decay of these radionuclides.

For most of the remediated waste sites in the 100-F/100-IU-2/100-IU-6 Decision Area with RME present-day Subsistence Farmer cancer risks above 1×10^{-4}, arsenic is the primary contributor to risk. As shown in Table 5-48, the Subsistence Farmer RME arsenic cancer risk calculated using the upland reference area data is 5×10^{-4}. Hence, arsenic-related cancer risks only exceed reference area levels at the 100-F-37, 116-F-1, possibly at 116-F-14, and 600-235 waste sites. Excepting 116-F-14, RME arsenic concentrations at 100-F-37 (17.3 mg/kg), 116-F-1 (6.4 mg/kg), 116-F-14 (2.6 mg/kg), and 600-235 (4.5 mg/kg) are all above the upland reference area RME concentration of 3.2 mg/kg shown in Table 5-47. Reasonable maximum exposure arsenic concentrations at 100-F-37 (17.3 mg/kg) and 116-F-1 (6.4 mg/kg) are also near or above the Hanford Area (6.47 mg/kg) and Yakima Basin (5.13 mg/kg) 90th percentile soil background concentrations.

Arsenic concentrations at the remediated waste sites where arsenic drives RME risks above 1×10^{-4} are, with the exception of 100-F-37, consistent with commonly referenced background levels. Therefore, risks for COPCs other than arsenic at these sites are also of interest for risk management at these sites. Eleven of these 23 waste sites have RME cancer risks above 1×10^{-6} related to COPCs other than arsenic. A summary of the key exposure pathways and COPCs, other than arsenic, contributing 10% or more to present-day RME Subsistence Farmer cancer risk at these sites is provided below.

Site	COPC other than arsenic	Pathway	Fraction of Total Risk	Present-Day Total Risk (without arsenic)
116-F-14	Europium-152	External Irradiation	0.73	7×10^{-4}

Site	COPC other than arsenic	Pathway	Fraction of Total Risk	Present-Day Total Risk (without arsenic)
116-F-9	Strontium-90	Milk Ingestion	0.38	4×10^{-4}
	Strontium-90	Produce Ingestion	0.32	
	Strontium-90	Beef Ingestion	0.17	
	Carbon-14	Produce Ingestion	0.11	
116-F-6	Strontium-90	Milk Ingestion	0.31	3×10^{-4}
	Strontium-90	Produce Ingestion	0.26	
	Strontium-90	Beef Ingestion	0.14	
	Cesium-137	External Irradiation	0.12	
	Europium-152	External Irradiation	0.10	
UPR-100-F-2	Europium-152	External Irradiation	0.88	2×10^{-4}
116-F-10	Cesium-137	External Irradiation	0.45	2×10^{-4}
	Europium-152	External Irradiation	0.35	
116-F-1	Europium-152	External Irradiation	0.50	1×10^{-4}
	Carbon-14	Produce Ingestion	0.31	
	Cesium-137	External Irradiation	0.10	
600-23	Aroclor-1254	Milk Ingestion	0.39	3×10^{-5}
	Aroclor-1254	Beef Ingestion	0.21	
	Aroclor-1254	Soil Ingestion	0.13	
600-190	Aroclor-1254	Milk Ingestion	0.41	2×10^{-5}
	Aroclor-1254	Beef Ingestion	0.22	
	Aroclor-1254	Soil Ingestion	0.14	
	Aroclor-1254	Produce Ingestion	0.11	
116-F-5	Cesium-137	External Irradiation	0.58	2×10^{-5}
	Europium-152	External Irradiation	0.21	
	Cesium-137	Milk Ingestion	0.11	
100-F-26:1	Strontium-90	Milk Ingestion	0.43	2×10^{-5}
	Strontium-90	Produce Ingestion	0.36	
	Strontium-90	Beef Ingestion	0.19	
600-132	Aroclor-1254	Milk Ingestion	0.41	4×10^{-6}
	Aroclor-1254	Beef Ingestion	0.22	
	Aroclor-1254	Soil Ingestion	0.13	
	Aroclor-1254	Produce Ingestion	0.10	

In the absence of arsenic, RME cancer risks at sites where present-day RME risks are still at or above 1×10^{-4} are related to the same set of short-lived radionuclides that were described above. These radionuclides are also the primary RME risk drivers in the absence of arsenic at the 116-F-5 and 100-F-26:1 waste sites. At the 600-23, 600-190, and 600-132 waste sites, exposure to Aroclor-1254 from mostly food-related exposure pathways drives cancer risk in the absence of arsenic. This analysis indicates that RME Subsistence Farmer cancer risks from COPCs other than arsenic are likely to be relatively unimportant by year 2075 at all 100-F/100-IU-2/100-IU-6 Decision Area remediated waste sites evaluated in this risk assessment excepting 600-23, 600-190, and 600-132.

The range of RME arsenic soil concentrations at 100-F/100-IU-2/100-IU-6 remediated waste sites is not much higher than might be expected of background soils. Although the arsenic risk estimates for the produce ingestion exposure pathway are based on plant uptake models that are prone to overestimation of plant concentrations when soil concentrations are high (see Section 5.9.2), this potential source of bias may not be very important for the arsenic risk estimates at the 100-F/100-IU-2/100-IU-6 Area sites because site soil levels are not much greater than naturally occurring levels.

As described in relation to Resident National Monument/Refuge cancer risk results, the use of shallow zone soil verification data may contribute to a significant protective bias in the risk results for excavated sites. Information related to the excavation depth and area of the remediated waste sites is summarized in Table 2-8. The 100-F/100-IU-2/100-IU-6 Area sites with the designation “116-” are liquid waste disposal sites that were generally excavated to depths at or below 4.6 m (15 ft). The “600-” designated sites are mostly related to facilities in the White Bluffs and Hanford townsite area and were generally excavated to depths of less than 1 m (3.3 ft). An exception is the 600-23 waste site, which was excavated to a depth of 5 m (16.4 ft). Most of the “100-“ sites are related to drains and pipelines and were excavated to depths between 0 and 5 m (16.4 ft). Uncertainties in the use of shallow zone soil verification data from excavated waste sites are discussed in more detail in Section 5.9 and Appendix D-1.

Several of the 100-F/100-IU-2/100-IU-6 Area sites evaluated in this risk assessment have an estimated surface area of less than 10 m^2 (110 ft^2). The 100-F-37 waste site, where arsenic soil concentrations are highest, is an unexcavated french drain that has an estimated surface area of approximately 1 m^2 (10.8 ft^2). As described in Section 3.3, the risk assessment calculations do not use any weighting factors to apportion a fraction of exposure to a site. When the actual area of contamination may be relatively small, this may result in grossly overestimated risks. This is particularly true in residential exposures when risks are related to produce and livestock exposure pathways that require considerable amounts of arable land.

5.7.3.2 Radiation Dose. Table 5-118 shows the RME and CTE total radiation dose results for the Subsistence Farmer exposure scenario for all remediated waste sites in the 100-F/100-IU-2/100-IU-6 Decision Area. The tabulated results are the sum of the dose values for the individual COPCs included in the dose calculation. Risk assessment results are shown for present-day radionuclide concentrations, and with radionuclide concentrations decayed to the years 2075 and 2150. Soil concentrations for these times were calculated using the RESRAD computer code, as described in Appendix D-1.4.1.

The range of present-day RME and CTE total radiation dose results for the Subsistence Farmer scenario is from 0.018 to 32 mrem/yr and from 0.12 to 6.8 mrem/yr, respectively. No radionuclide COPCs were detected at 22 waste sites. As shown in Table 5-5, 3 of the 44 waste sites evaluated in the 100-F/100-IU-2/100-IU-6 Decision Area had calculated total RME radiation dose values above 15 mrem/yr, but no sites had CTE values above 15 mrem/yr.

A summary of the key exposure pathways and COPCs contributing 10% or more to the present-day RME Subsistence Farmer radiation dose at 100-F/100-IU-2/100-IU-6 Decision Area sites with total dose above 15 mrem/yr is provided below.

Site	COPC	Pathway	Fraction of Total Dose	Present-Day Total Dose (mrem/yr)
116-F-9	Strontium-90	Milk Ingestion	0.40	32
	Strontium-90	Produce Ingestion	0.33	
	Strontium-90	Beef Ingestion	0.18	
116-F-14	Europium-152	External Irradiation	0.71	32
116-F-6	Strontium-90	Milk Ingestion	0.35	22
	Strontium-90	Produce Ingestion	0.29	
	Strontium-90	Beef Ingestion	0.16	

Similar to the findings for cancer risk at remediated waste sites where risk was related to radionuclides, the present-day radiation dose at sites with calculated dose above the threshold is driven by relatively short-lived radionuclides. There are no waste sites in the 100-F/100-IU-2/100-IU-6 Area that exceed 15 mrem/yr at year 2075. At year 2075, the highest total RME radiation dose value is 5.6 mrem/yr at the 116-F-9 waste site.

As described in relation to cancer risk, most “116-“ designation remediated waste sites where short-lived radionuclides were important contributors to significant cancer risk (including 116-F-9, 116-F-14, and 116-F-6) were excavated to a depth of greater than 4.6 m (15 ft). The use of shallow zone soil verification data to calculate risks at these sites may contribute to a significant protective bias in the radiation dose results.

5.7.3.3 Chemical Hazard. Table 5-119 shows the RME and CTE HI results for the Subsistence Farmer exposure scenario for all remediated waste sites in the 100-F/100-IU-2/100-IU-6 Decision Area. The tabulated results are the sum of the risk values for the individual COPCs included in the risk calculation. As described in Section 3.4.4, chemical hazard is calculated separately for adult and child receptors. For the Subsistence Farmer scenario, RME and CTE child HIs are usually slightly higher those for adults. Because they are consistently higher, only child HI values are presented here.

The range of RME and CTE child HI results for the Subsistence Farmer scenario is from 0.0011 to 5.0 and from 0.00075 to 1.3, respectively. No chemical COPCs contributing to hazard were detected at eight remediated waste sites.

As shown in Table 5-5, only 1 of the 44 remediated waste sites in the 100-F/100-IU-2/100-IU-6 Decision Area had an RME HI value above 10 for the Subsistence Farmer scenario, but 24 sites had RME HIs between 1.0 and 10. No waste sites had a CTE HI value above 10. The majority of waste sites with RME HIs between 1 and 10 had values below 3.0.

A summary of the key exposure pathways and COPCs contributing 10% or more to the RME Subsistence Farmer HI at 100-F/100-IU-2/100-IU-6 Decision Area sites with HI values at or above 3.0 is provided below. For brevity, only sites with an RME HI of 3.0 or above, rather than the threshold of 1.0, are discussed.

Site	COPC	Pathway	Fraction of Total Child Hazard	Present-Day Child Hazard
100-F-37	Arsenic	Produce Ingestion	0.74	16
100-F-9	Mercury	Beef Ingestion	0.48	7.0
	Arsenic	Produce Ingestion	0.25	
	Mercury	Produce Ingestion	0.18	
116-F-1	Arsenic	Produce Ingestion	0.82	5.4
600-23	Arsenic	Produce Ingestion	0.39	5.0
	Aroclor-1254	Soil Ingestion	0.15	
	Aroclor-1254	Milk Ingestion	0.13	
600-235	Arsenic	Produce Ingestion	0.67	4.6
	Cadmium	Produce Ingestion	0.14	
600-190	Arsenic	Produce Ingestion	0.40	4.3
	Aroclor-1254	Soil Ingestion	0.15	
	Aroclor-1254	Milk Ingestion	0.13	
100-F-7	Arsenic	Produce Ingestion	0.44	3.9
	Mercury	Beef Ingestion	0.30	
	Mercury	Produce Ingestion	0.11	
100-F-26:1	Arsenic	Produce Ingestion	0.67	3.3
100-F-26:7	Arsenic	Produce Ingestion	0.56	3.0
	Mercury	Beef Ingestion	0.21	

Exposure to arsenic and/or mercury by produce or beef ingestion are the most important exposure pathways contributing to HI values above 3.0 at each of these waste sites. Exposure to Aroclor-1254 by soil and milk ingestion is also important at the 600-23 and 600-190 waste sites. Critical effects underlying the oral RfD for mercury include effects on the immune and renal (kidney) system, whereas the critical effects underlying the oral RfD for arsenic relate to the vascular and integumentary (skin) system (IRIS 2009). Summing HQs for mercury and arsenic likely results in a protective bias for the tabulated HI values at these sites.

As discussed in relation to cancer risk, the RME representative concentration of arsenic at the 100-F-37 remediated waste site is 17.3 mg/kg, and in upland reference area soil is 3.2 mg/kg. RME arsenic concentrations at the other sites where RME child HI exceeds 3.0 are near or below the Hanford Area (6.47 mg/kg) and Yakima Basin (5.13 mg/kg) 90th percentile soil background concentrations.

The 100-F-37 waste site, where RME child HI is highest, is an unexcavated french drain that has an estimated surface area of approximately 1 m^2 (10.8 ft^2). The 100-F-9 waste site, another french drain, has an estimated area of just 2.8 m^2 (30.1 ft^2). Because the risk assessment calculations do not use any weighting factors to apportion a fraction of exposure to a site, as discussed in relation to cancer risk, this may result in grossly overestimated risks at very small waste sites. Mercury was measured in only a single sample at the 100-F-9 waste site, so there is low confidence in the RME representative concentration of 0.38 mg/kg.

Arsenic concentrations at the remediated waste sites where arsenic is the main driver of RME child HI values above the threshold are consistent with commonly referenced background levels. Therefore, child HI results for COPCs other than arsenic are also of interest for risk management at these sites. As shown in Table 5-50, the RME Subsistence Farmer child HI for the upland reference area is 2.7. Of the nine remediated waste sites having RME child HI at or above 3.0, when arsenic is removed only five still have an RME child HI above 1.0. A summary of the key exposure pathways and COPCs, other than arsenic, contributing 10% or more to present-day RME Subsistence Farmer child HI at these sites is provided below.

Site	COPC other than arsenic	Pathway	Fraction of Child Hazard	Present-Day Child Hazard (without arsenic)
100-F-9	Mercury	Beef Ingestion	0.69	4.9
	Mercury	Produce Ingestion	0.26	
600-23	Aroclor-1254	Soil Ingestion	0.29	2.6
	Aroclor-1254	Milk Ingestion	0.26	
	Aroclor-1254	Beef Ingestion	0.14	
	Aroclor-1254	Soil Dermal	0.11	
600-190	Aroclor-1254	Soil Ingestion	0.29	2.2
	Aroclor-1254	Milk Ingestion	0.26	
	Aroclor-1254	Beef Ingestion	0.14	
	Aroclor-1254	Soil Dermal	0.11	
100-F-7	Mercury	Beef Ingestion	0.64	1.8
	Mercury	Produce Ingestion	0.24	

Site	COPC other than arsenic	Pathway	Fraction of Child Hazard	Present-Day Child Hazard (without arsenic)
100-F-37	Mercury	Beef Ingestion	0.36	1.7
	Cadmium	Produce Ingestion	0.32	
	Mercury	Produce Ingestion	0.13	

As described in relation to cancer risk, the use of shallow zone soil verification data may contribute to a significant protective bias in the risk results for excavated sites. Of the five sites evaluated for child HI in the absence of arsenic, only 600-235 has a significant excavation depth (5 m [16.4 ft]). The use of shallow zone soil verification data to calculate HI may contribute to a significant protective bias in the child HI results for these sites.

5.7.4 CTUIR Resident Scenario Results for the 100-F/100-IU-2/100-IU-6 Area

The CTUIR Resident exposure scenario encompasses potential exposures related to shallow zone soils around remediated waste sites, including both direct contact exposure pathways and exposures related to raising produce and animals in the potentially affected area of the waste site. This scenario also includes exposures related to ingestion of Columbia River fish (see Section 4.4) and exposure to groundwater via a domestic water well (see Section 6.0). It is similar to the Subsistence Farmer scenario, although exposure intensity is higher for many exposure pathways. The scenario is more fully described in the CSM in Section 3.3 and is summarized in Table 3-13 and Figure 3-17. The COPCs evaluated in shallow zone soil in the 100-F/100-IU-2/100-IU-6 Decision Area are shown in Table 5-26.

5.7.4.1 Cancer Risk. Table 5-120 shows the total cancer risk results for the CTUIR Resident exposure scenario for all remediated waste sites in the 100-F/100-IU-2/100-IU-6 Decision Area. The tabulated results are the sum of the risk values for the individual COPCs included in the risk calculation. Total risk results use EPCs based on measured soil concentrations in the individual waste sites. Risk assessment results are shown for present-day radionuclide concentrations, and with radionuclide concentrations decayed to the years 2075 and 2150. Soil concentrations for these times were calculated using the RESRAD computer code, as described in Appendix D-1.4.1.

The present-day total cancer risk results for the CTUIR Resident scenario (radionuclides + chemicals) range from 7×10^{-8} to 4×10^{-2}. No carcinogenic COPCs were detected at the 100-F-11 waste site. Per EPA guidance (EPA/540/1-89/002), cancer risk results are displayed with only one significant figure. As shown in Table 5-5, 38 of the 44 remediated waste sites evaluated in the 100-F/100-IU-2/100-IU-6 Decision Area had calculated total cancer risks above 1×10^{-4}.

A summary of the key exposure pathways and COPCs contributing 10% or more to present-day CTUIR Resident cancer risk at 100-F/100-IU-2/100-IU-6 Decision Area sites with total cancer risk above 1×10^{-4} is provided below.

Site	COPC	Pathway	Fraction of Total Risk	Present-Day Total Risk
100-F-37	Arsenic	Produce Ingestion	0.99	4×10^{-2}
116-F-1	Arsenic	Produce Ingestion	0.94	2×10^{-2}
600-235	Arsenic	Produce Ingestion	0.99	1×10^{-2}
116-F-14	Arsenic	Produce Ingestion	0.67	1×10^{-2}
	Europium-152	External Irradiation	0.15	
	Carbon-14	Produce Ingestion	0.12	
100-F-26:1	Arsenic	Produce Ingestion	0.98	8×10^{-3}
600-131	Arsenic	Produce Ingestion	0.99	8×10^{-3}
600-181	Arsenic	Produce Ingestion	0.99	8×10^{-3}
628-1	Arsenic	Produce Ingestion	0.99	8×10^{-3}
600-23	Arsenic	Produce Ingestion	0.98	7×10^{-3}
600-204	Arsenic	Produce Ingestion	0.99	7×10^{-3}
600-190	Arsenic	Produce Ingestion	0.98	6×10^{-3}
600-132	Arsenic	Produce Ingestion	0.99	6×10^{-3}
100-F-12	Arsenic	Produce Ingestion	0.99	6×10^{-3}
100-F-7	Arsenic	Produce Ingestion	0.99	6×10^{-3}
100-F-9	Arsenic	Produce Ingestion	0.99	6×10^{-3}
100-F-38	Arsenic	Produce Ingestion	0.99	6×10^{-3}
100-F-26:7	Arsenic	Produce Ingestion	0.99	6×10^{-3}
600-128	Arsenic	Produce Ingestion	0.99	6×10^{-3}
100-F-26:2	Arsenic	Produce Ingestion	0.99	5×10^{-3}
100-F-26:5	Arsenic	Produce Ingestion	0.99	5×10^{-3}
100-F-14	Arsenic	Produce Ingestion	0.99	5×10^{-3}
116-F-5	Arsenic	Produce Ingestion	0.98	5×10^{-3}
100-F-18	Arsenic	Produce Ingestion	0.99	4×10^{-3}
116-F-9	Strontium-90	Produce Ingestion	0.71	4×10^{-3}
	Carbon-14	Produce Ingestion	0.24	
116-F-6	Strontium-90	Produce Ingestion	0.79	2×10^{-3}
100-F-19:1	Carbon-14	Produce Ingestion	0.89	1×10^{-3}
100-F-19:2	Carbon-14	Produce Ingestion	0.53	9×10^{-4}
	Europium-152	External Irradiation	0.23	
	Strontium-90	Produce Ingestion	0.16	
100-F-23	Carbon-14	Produce Ingestion	0.91	8×10^{-4}
100-F-35	Strontium-90	Produce Ingestion	0.94	7×10^{-4}

Site	COPC	Pathway	Fraction of Total Risk	Present-Day Total Risk
116-F-2	Europium-152	External Irradiation	0.47	6×10^{-4}
	Carbon-14	Produce Ingestion	0.44	
118-F-8:1	Strontium-90	Produce Ingestion	0.33	6×10^{-4}
	Europium-152	External Irradiation	0.28	
	Cesium-137	External Irradiation	0.14	
	Carbon-14	Produce Ingestion	0.10	
UPR-100-F-2	Europium-152	External Irradiation	0.87	6×10^{-4}
116-F-10	Cesium-137	External Irradiation	0.43	5×10^{-4}
	Europium-152	External Irradiation	0.34	
	Cesium-137	Produce Ingestion	0.16	
1607-F6	Carbon-14	Produce Ingestion	0.71	5×10^{-4}
	Strontium-90	Produce Ingestion	0.10	
100-F-24	Carbon-14	Produce Ingestion	0.78	5×10^{-4}
	Strontium-90	Produce Ingestion	0.18	
116-F-4	Strontium-90	Produce Ingestion	0.75	3×10^{-4}
	Europium-152	External Irradiation	0.10	
100-F-2	Strontium-90	Produce Ingestion	0.75	2×10^{-4}
	Cesium-137	External Irradiation	0.14	
1607-F2	Europium-152	External Irradiation	0.82	2×10^{-4}
	Cesium-137	External Irradiation	0.12	

With the exception of the 116-F-14, 116-F-9, and 116-F-6 remediated waste sites, the cancer risk at waste sites with present-day cancer risk above 1×10^{-3} is related primarily to arsenic by ingestion of home-grown produce. At these three sites, and at sites with risks between 2×10^{-4} and 9×10^{-4}, risk is mostly related to radionuclides including cesium-137, europium-152, and strontium-90. The half-lives of these radionuclides range from approximately 13.5 years (europium-152) to approximately 30 years (strontium-90 and cesium-137). Therefore, much of the cancer risk at sites where risk is related to radionuclides will decrease naturally over time as a function of the decay of these radionuclides.

As shown in Table 5-48, CTUIR Resident arsenic cancer risk calculated using upland reference area data is 8×10^{-3}. Only four of the sites listed above have an arsenic-related cancer risk that exceed this value. The RME arsenic concentrations at 100-F-37 (17.3 mg/kg), 116-F-1 (6.4 mg/kg), and 600-235 (4.5 mg/kg) are all above the upland reference area RME concentration of 3.2 mg/kg, shown in Table 5-47. Reasonable maximum exposure arsenic concentrations at 100-F-37 (17.3 mg/kg) and 116-F-1 (6.4 mg/kg) are also near or above the Hanford Area (6.47 mg/kg) and Yakima Basin (5.13 mg/kg) 90th percentile soil background concentrations.

Because arsenic concentrations at the remediated waste sites where arsenic is a risk driver are similar to commonly referenced background levels, risks for COPCs other than arsenic are also of interest for risk management at these sites. A summary of the key exposure pathways and COPCs, other than arsenic, contributing 10% or more to present-day RME Subsistence Farmer cancer risk at these sites is provided in Section 5.7.3. This summary is also applicable to the CTUIR Resident scenario.

As described in relation to Subsistence Farmer cancer risk results, the use of shallow zone soil verification data may contribute to a significant protective bias in the risk results for excavated sites. The 100-F/100-IU-2/100-IU-6 Area sites with the designation "116-" are liquid waste disposal sites that were generally excavated to depths at or below 4.6 m (15 ft). The "600-" designated sites are mostly related to facilities in the White Bluffs and Hanford townsite area and were generally excavated to depths of less than 1 m (3.3 ft). An exception is the 600-23 waste site, which was excavated to a depth of 5 m (16.4 ft). Most of the "100-" sites are related to drains and pipelines and were excavated to depths between 0 and 5 m (16.4 ft). Uncertainties in the use of shallow zone soil verification data from excavated waste sites are discussed in more detail in Section 5.9 and Appendix D-1.

Several of the 100-F/100-IU-2/100-IU-6 Area sites evaluated in this risk assessment have an estimated surface area of less than 10 m^2 (110 ft^2). The 100-F-37 waste site, where arsenic soil concentrations are highest, is an unexcavated french drain that has an estimated surface area of approximately 1 m^2 (10.8 ft^2). As described in Section 3.3, the risk assessment calculations do not use any weighting factors to apportion a fraction of exposure to a site. When the actual area of contamination may be relatively small, this may result in grossly overestimated risks. This is particularly true in residential exposures when risks are related to produce and livestock exposure pathways that require considerable amounts of arable land.

5.7.4.2 Radiation Dose. Table 5-121 shows the total radiation dose results for the CTUIR Resident exposure scenario for all remediated waste sites in the 100-F/100-IU-2/100-IU-6 Decision Area. The tabulated results are the sum of the dose values for the individual COPCs included in the dose calculation. Total dose results use EPCs based on measured soil concentrations in the individual waste sites. Risk assessment results are shown for present-day radionuclide concentrations, and with radionuclide concentrations decayed to the years 2075 and 2150. Soil concentrations for these times were calculated using the RESRAD computer code, as described in Appendix D-1.4.1.

The present-day total radiation dose results for the CTUIR Resident scenario range from 0.18 to 86 mrem/yr. No radionuclide COPCs were detected at 22 waste sites. As shown in Table 5-5, 5 of the 44 remediated waste sites evaluated in the 100-F/100-IU-2/100-IU-6 Decision Area had calculated total radiation dose values above 15 mrem/yr.

A summary of the key exposure pathways and COPCs contributing 10% or more to the present-day CTUIR Resident radiation dose at 100-F/100-IU-2/100-IU-6 Decision Area sites with total dose above 15 mrem/yr is provided below.

Site	COPC	Pathway	Fraction of Total Dose	Present-Day Total Dose (mrem/yr)
116-F-9	Strontium-90	Produce Ingestion	0.79	86
	Carbon-14	Produce Ingestion	0.16	
116-F-14	Europium-152	External Irradiation	0.47	54
	Carbon-14	Produce Ingestion	0.32	
	Strontium-90	Produce Ingestion	0.13	
116-F-6	Strontium-90	Produce Ingestion	0.83	48
100-F-19:1	Carbon-14	Produce Ingestion	0.84	21
	Strontium-90	Produce Ingestion	0.14	
100-F-35	Strontium-90	Produce Ingestion	0.94	16

Similar to the findings for cancer risk at remediated waste sites where risk was related to radionuclides, the present-day radiation dose at sites with calculated dose above the threshold is driven by relatively short-lived radionuclides. There are no waste sites in the 100-F/100-IU-2/100-IU-6 Area that exceed 15 mrem/yr at year 2075. At year 2075, the highest total radiation dose value is 14 mrem/yr at the 116-F-9 waste site. As described for CTUIR cancer risk, the use of shallow zone soil verification data to calculate risks may contribute to a significant protective bias in the radiation dose results for excavated waste sites.

5.7.4.3 Chemical Hazard. Table 5-122 shows the HI results for the CTUIR Resident exposure scenario for all remediated waste sites in the 100-F/100-IU-2/100-IU-6 Decision Area. The tabulated results are the sum of the risk values for the individual COPCs included in the risk calculation. As described in Section 3.4.4, chemical hazard is calculated separately for adult and child receptors. For the CTUIR Resident scenario, child HIs are equal to or higher than those for adults. Therefore, only child HI values are presented here.

The child HI results for the CTUIR Resident scenario range from 0.0011 to 180. No chemical COPCs contributing to hazard were detected at eight of the remediated waste sites.

As shown in Table 5-5, 23 of the 44 remediated waste sites in the 100-F/100-IU-2/100-IU-6 Decision Area had an HI value above 10 for the CTUIR Resident scenario, and 3 sites had HIs between 1.0 and 10. The majority of waste sites with HIs between 1 and 10 had values below 3.0. For brevity, only sites with an HI of 10 or above are discussed.

A summary of the key exposure pathways and COPCs contributing 10% or more to CTUIR Resident child HIs at 100-F/100-IU-2/100-IU-6 Decision Area sites with HI values at or above 10 is provided below.

Site	COPC	Pathway	Fraction of Child Hazard	Present-Day Child Hazard
100-F-37	Arsenic	Produce Ingestion	0.92	180

Site	COPC	Pathway	Fraction of Child Hazard	Present-Day Child Hazard
116-F-1	Arsenic	Produce Ingestion	0.99	63
600-235	Arsenic	Produce Ingestion	0.82	53
	Cadmium	Produce Ingestion	0.17	
100-F-9	Arsenic	Produce Ingestion	0.50	49
	Mercury	Produce Ingestion	0.36	
	Mercury	Beef Ingestion	0.12	
600-23	Arsenic	Produce Ingestion	0.77	36
100-F-26:1	Arsenic	Produce Ingestion	0.88	36
100-F-7	Arsenic	Produce Ingestion	0.71	34
	Mercury	Produce Ingestion	0.18	
600-131	Arsenic	Produce Ingestion	0.96	32
600-181	Arsenic	Produce Ingestion	0.94	31
600-190	Arsenic	Produce Ingestion	0.80	31
600-132	Arsenic	Produce Ingestion	0.81	30
	Cadmium	Produce Ingestion	0.15	
628-1	Arsenic	Produce Ingestion	0.98	30
100-F-26:7	Arsenic	Produce Ingestion	0.82	28
	Mercury	Produce Ingestion	0.11	
600-204	Arsenic	Produce Ingestion	0.95	28
100-F-38	Arsenic	Produce Ingestion	0.87	27
116-F-14	Arsenic	Produce Ingestion	0.98	26
100-F-12	Arsenic	Produce Ingestion	0.95	26
100-F-14	Arsenic	Produce Ingestion	0.77	24
	Mercury	Produce Ingestion	0.16	
100-F-26:5	Arsenic	Produce Ingestion	0.86	24
	Cadmium	Produce Ingestion	0.12	
600-128	Arsenic	Produce Ingestion	0.91	24
100-F-18	Arsenic	Produce Ingestion	0.74	22
	Mercury	Produce Ingestion	0.18	
100-F-26:2	Arsenic	Produce Ingestion	0.98	21
116-F-5	Arsenic	Produce Ingestion	0.98	18

Exposure to arsenic, mercury, and cadmium by produce ingestion are the most important exposure pathways contributing to HI values above 10 at each of these waste sites except 600-23 and 600-190. Critical effects underlying the oral RfD for mercury include effects on the immune and renal (kidney) system, whereas the critical effects underlying the oral RfD for arsenic relate to the vascular and integumentary (skin) system (IRIS 2009). The critical target organ for chronic cadmium toxicity is also the renal system (IRIS 2009). Summing HQs for mercury and arsenic likely results in a protective bias for the tabulated HI values at these sites. There are no sites where mercury and cadmium are both significant contributors to an HI above 10. As described for CTUIR cancer risk, the use of shallow zone soil verification data to assess risks at excavated waste sites may contribute to a significant protective bias in the HI results for these sites.

Because arsenic concentrations at the remediated waste sites where arsenic is a risk driver are similar to commonly referenced background levels, as described in Section 5.7.3, risks for COPCs other than arsenic are also of interest for risk management at these sites. A summary of the key exposure pathways and COPCs, other than arsenic, contributing 10% or more to present-day RME Subsistence Farmer child HI at these sites is provided in Section 5.7.3. This summary is also applicable to the CTUIR Resident scenario.

Several of the 100-F/100-IU-2/100-IU-6 Area sites evaluated in this risk assessment have an estimated surface area of less than 10 m^2 (110 ft^2). The 100-F-37 and 100-F-9 waste sites, for example, are french drains with an estimated surface area of approximately 1 to 3 m^2 (10.8 to 32.3 ft^2). As described for CTUIR cancer risk, this may result in grossly overestimated risks because the risk assessment calculations do not use any weighting factors to apportion a fraction of exposure to a site. Mercury was measured in only a single sample at the 100-F-9 waste site, so there is low confidence in the representative concentration at this site.

5.7.5 Yakama Resident Scenario Results for the 100-F/100-IU-2/100-IU-6 Area

The Yakama Resident exposure scenario encompasses potential exposures related to shallow zone soils around remediated waste sites, including both direct contact exposure pathways and exposures related to raising produce and animals in the potentially affected area of the waste site. This scenario also includes exposures related to ingestion of Columbia River fish (see Section 4.4) and exposure to groundwater via a domestic water well (see Section 6.0). It is similar to the Subsistence Farmer scenario, although exposure intensity is higher for many exposure pathways. The scenario is more fully described in the CSM in Section 3.3 and is summarized in Table 3-13 and Figure 3-17. The COPCs evaluated in shallow zone soil in the 100-F/100-IU-2/100-IU-6 Decision Area are shown in Table 5-26.

5.7.5.1 Cancer Risk. Table 5-123 shows the total cancer risk results for the Yakama Resident exposure scenario for all remediated waste sites in the 100-F/100-IU-2/100-IU-6 Decision Area. The tabulated results are the sum of the risk values for the individual COPCs included in the risk calculation. Total risk results use EPCs based on measured soil concentrations in the individual waste sites. Risk assessment results are shown for present-day radionuclide concentrations, and with radionuclide concentrations decayed to the years 2075 and 2150. Soil concentrations for these times were calculated using the RESRAD computer code, as described in Appendix D-1.4.1.

The present-day total cancer risk results for the Yakama Resident scenario (radionuclides + chemicals) range from 6×10^{-8} to 4×10^{-2}. No carcinogenic COPCs were detected at the 100-F-11 waste site. Per EPA guidance (EPA/540/1-89/002), cancer risk results are displayed with only one significant figure. As shown in Table 5-5, 38 of the 44 remediated waste sites evaluated in the 100-F/100-IU-2/100-IU-6 Decision Area had calculated total cancer risks above 1×10^{-4}.

A summary of the key exposure pathways and COPCs contributing 10% or more to present-day Yakama Resident cancer risk at 100-F/100-IU-2/100-IU-6 Decision Area sites with total cancer risk above 1×10^{-4} is provided below.

Site	COPC	Pathway	Fraction of Total Risk	Present-Day Total Risk
100-F-37	Arsenic	Produce Ingestion	0.97	4×10^{-2}
116-F-1	Arsenic	Produce Ingestion	0.92	2×10^{-2}
600-235	Arsenic	Produce Ingestion	0.97	1×10^{-2}
116-F-14	Arsenic	Produce Ingestion	0.65	1×10^{-2}
	Europium-152	External Irradiation	0.14	
	Carbon-14	Produce Ingestion	0.12	
100-F-26:1	Arsenic	Produce Ingestion	0.95	8×10^{-3}
600-131	Arsenic	Produce Ingestion	0.97	8×10^{-3}
600-181	Arsenic	Produce Ingestion	0.97	7×10^{-3}
628-1	Arsenic	Produce Ingestion	0.97	7×10^{-3}
600-23	Arsenic	Produce Ingestion	0.95	7×10^{-3}
600-204	Arsenic	Produce Ingestion	0.97	7×10^{-3}
600-190	Arsenic	Produce Ingestion	0.95	6×10^{-3}
600-132	Arsenic	Produce Ingestion	0.97	6×10^{-3}
100-F-12	Arsenic	Produce Ingestion	0.97	6×10^{-3}
100-F-7	Arsenic	Produce Ingestion	0.97	6×10^{-3}
100-F-9	Arsenic	Produce Ingestion	0.97	6×10^{-3}
100-F-38	Arsenic	Produce Ingestion	0.97	6×10^{-3}
100-F-26:7	Arsenic	Produce Ingestion	0.97	6×10^{-3}
600-128	Arsenic	Produce Ingestion	0.97	5×10^{-3}
100-F-26:2	Arsenic	Produce Ingestion	0.97	5×10^{-3}
100-F-26:5	Arsenic	Produce Ingestion	0.97	5×10^{-3}
116-F-9	Strontium-90	Produce Ingestion	0.55	5×10^{-3}
	Carbon-14	Produce Ingestion	0.19	
	Strontium-90	Beef Ingestion	0.18	
100-F-14	Arsenic	Produce Ingestion	0.97	5×10^{-3}

Site	COPC	Pathway	Fraction of Total Risk	Present-Day Total Risk
116-F-5	Arsenic	Produce Ingestion	0.96	5×10^{-3}
100-F-18	Arsenic	Produce Ingestion	0.97	4×10^{-3}
116-F-6	Strontium-90	Produce Ingestion	0.59	3×10^{-3}
	Strontium-90	Beef Ingestion	0.19	
100-F-19:1	Carbon-14	Produce Ingestion	0.83	1×10^{-3}
100-F-19:2	Carbon-14	Produce Ingestion	0.48	9×10^{-4}
	Europium-152	External Irradiation	0.21	
	Strontium-90	Produce Ingestion	0.14	
100-F-35	Strontium-90	Produce Ingestion	0.68	9×10^{-4}
	Strontium-90	Beef Ingestion	0.22	
100-F-23	Carbon-14	Produce Ingestion	0.85	8×10^{-4}
118-F-8:1	Strontium-90	Produce Ingestion	0.28	7×10^{-4}
	Europium-152	External Irradiation	0.23	
	Cesium-137	External Irradiation	0.12	
116-F-2	Europium-152	External Irradiation	0.45	6×10^{-4}
	Carbon-14	Produce Ingestion	0.42	
116-F-10	Cesium-137	External Irradiation	0.36	6×10^{-4}
	Europium-152	External Irradiation	0.28	
	Cesium-137	Produce Ingestion	0.14	
	Cesium-137	Beef Ingestion	0.12	
UPR-100-F-2	Europium-152	External Irradiation	0.84	5×10^{-4}
1607-F6	Carbon-14	Produce Ingestion	0.62	5×10^{-4}
100-F-24	Carbon-14	Produce Ingestion	0.70	5×10^{-4}
	Strontium-90	Produce Ingestion	0.17	
116-F-4	Strontium-90	Produce Ingestion	0.57	3×10^{-4}
	Strontium-90	Beef Ingestion	0.18	
100-F-2	Strontium-90	Produce Ingestion	0.55	3×10^{-4}
	Strontium-90	Beef Ingestion	0.18	
1607-F2	Europium-152	External Irradiation	0.78	2×10^{-4}
	Cesium-137	External Irradiation	0.12	
100-F-25	Europium-152	External Irradiation	0.35	1×10^{-4}
	Strontium-90	Produce Ingestion	0.26	

With the exception of the 116-F-9, 116-F-6, and 100-F-19:1 remediated waste sites, cancer risk at remediated waste sites with present-day cancer risk at or above 1×10^{-3} is related primarily to arsenic by ingestion of home-grown produce. At these three sites, and at sites with risks between

1×10^{-4} and 9×10^{-4}, risk is mostly related to radionuclides including cesium-137, europium-152, and strontium-90. The half-lives of these radionuclides range from approximately 13.5 years (europium-152) to approximately 30 years (strontium-90 and cesium-137). Therefore, much of the cancer risk at sites where risk is related to radionuclides will decrease naturally over time as a function of the decay of these radionuclides.

As shown in Table 5-48, Yakama Resident arsenic cancer risk calculated using upland reference area data is 8×10^{-3}. Only four of the sites listed above have an arsenic-related cancer risk that exceeds this value. The RME arsenic concentrations at 100-F-37 (17.3 mg/kg), 116-F-1 (6.4 mg/kg), and 600-235 (4.5 mg/kg) are all above the upland reference area RME concentration of 3.2 mg/kg, shown in Table 5-47. The RME arsenic concentrations at 100-F-37 (17.3 mg/kg) and 116-F-1 (6.4 mg/kg) are also near or above the Hanford Area (6.47 mg/kg) and Yakima Basin (5.13 mg/kg) 90th percentile soil background concentrations.

Because arsenic concentrations at the remediated waste sites where arsenic is a risk driver are similar to commonly referenced background levels, risks for COPCs other than arsenic are also of interest for risk management at these sites. A summary of the key exposure pathways and COPCs, other than arsenic, contributing 10% or more to present-day RME Subsistence Farmer cancer risk at these sites is provided in Section 5.7.3. This summary is also applicable to the Yakama Resident scenario.

As described in relation to Subsistence Farmer cancer risk results, the use of shallow zone soil verification data may contribute to a significant protective bias in the risk results for excavated sites. The 100-F/100-IU-2/100-IU-6 Area sites with the designation “116-” are liquid waste disposal sites that were generally excavated to depths at or below 4.6 m (15 ft). The “600-” designated sites are mostly related to facilities in the White Bluffs and Hanford townsite area and were generally excavated to depths of less than 1 m (3.3 ft). An exception is the 600-23waste site, which was excavated to a depth of 5 m (16.4 ft). Most of the “100-“ sites are related to drains and pipelines and were excavated to depths between 0 and 5 m (16.4 ft). Uncertainties in the use of shallow zone soil verification data from excavated waste sites are discussed in more detail in Section 5.9 and Appendix D-1.

Several of the 100-F/100-IU-2/100-IU-6 Area sites evaluated in this risk assessment have an estimated surface area of less than 10 m^2 (110 ft^2). As described in Section 5.7.3, when the actual area of contamination may be relatively small, such as the 100-F-37 waste site with an estimated surface area of approximately 1 m^2 (10.8 ft^2), the risk assessment results may grossly overestimate risks.

5.7.5.2 Radiation Dose. Table 5-124 shows the total radiation dose results for the Yakama Resident exposure scenario for all remediated waste sites in the 100-F/100-IU-2/100-IU-6 Decision Area. The tabulated results are the sum of the dose values for the individual COPCs included in the dose calculation. Total dose results use EPCs based on measured soil concentrations in the individual waste sites. Risk assessment results are shown for present-day radionuclide concentrations, and with radionuclide concentrations decayed to the years 2075 and

2150. Soil concentrations for these times were calculated using the RESRAD computer code, as described in Appendix D-1.4.1.

The present-day total radiation dose results for the Yakama Resident scenario range from 0.21 to 100 mrem/yr. No radionuclide COPCs were detected at 22 waste sites. As shown in Table 5-5, 5 of the 44 remediated waste sites evaluated in the 100-F/100-IU-2/100-IU-6 Decision Area had calculated total radiation dose values above 15 mrem/yr.

A summary of the key exposure pathways and COPCs contributing 10% or more to the present-day Yakama Resident radiation dose at 100-F/100-IU-2/100-IU-6 Decision Area sites with total dose above 15 mrem/yr is provided below.

Site	COPC	Pathway	Fraction of Total Dose	Present-Day Total Dose (mrem/yr)
116-F-9	Strontium-90	Produce Ingestion	0.58	103
	Strontium-90	Beef Ingestion	0.20	
	Carbon-14	Produce Ingestion	0.12	
116-F-6	Strontium-90	Produce Ingestion	0.60	59
	Strontium-90	Beef Ingestion	0.20	
116-F-14	Europium-152	External Irradiation	0.46	55
	Carbon-14	Produce Ingestion	0.28	
	Strontium-90	Produce Ingestion	0.11	
100-F-19:1	Carbon-14	Produce Ingestion	0.76	21
	Strontium-90	Produce Ingestion	0.13	
100-F-35	Strontium-90	Produce Ingestion	0.67	20
	Strontium-90	Beef Ingestion	0.23	
	Strontium-90	Milk Ingestion	0.11	

Similar to the findings for cancer risk at remediated waste sites where risk was related to radionuclides, the present-day radiation dose at sites with calculated dose above the threshold is driven by relatively short-lived radionuclides. Carbon-14 has a relatively long half-life, but it has a short residence time in biologically active soils. The highest total radiation dose value at year 2075 is 17 mrem/yr at the 116-F-9 waste site. By year 2150, the total radiation dose is below 3 mrem/yr at all remediated waste sites. As described for Yakama Resident cancer risk, the use of shallow zone soil verification data to calculate risks may contribute to a significant protective bias in the radiation dose results for excavated waste sites.

5.7.5.3 Chemical Hazard. Table 5-125 shows the HI results for the Yakama Resident exposure scenario for all remediated waste sites in the 100-F/100-IU-2/100-IU-6 Decision Area. The tabulated results are the sum of the risk values for the individual COPCs included in the risk calculation. As described in Section 3.4.4, chemical hazard is calculated separately for adult and child receptors. For the Yakama Resident scenario, child HIs are equal to or higher than those for adults. Therefore, only child HI values are presented here.

The child HI results for the Yakama Resident scenario range from 0.0024 to 110. No chemical COPCs contributing to hazard were detected at eight of the remediated waste sites.

As shown in Table 5-5, 24 of the 44 remediated waste sites in the 100-F/100-IU-2/100-IU-6 Decision Area had an HI value above 10 for the Yakama Resident scenario, and 2 sites had HIs between 1.0 and 10. The waste sites with HIs between 1 and 10 had values of 2.5 and 4.9. For brevity, only sites with an HI of 10 or above are discussed.

A summary of the key exposure pathways and COPCs contributing 10% or more to Yakama Resident child HIs at 100-F/100-IU-2/100-IU-6 Decision Area sites with HI values at or above 10 is provided below.

Site	COPC	Pathway	Fraction of Child Hazard	Present-Day Child Hazard
100-F-37	Arsenic	Produce Ingestion	0.86	110
100-F-9	Mercury	Beef Ingestion	0.46	44
	Arsenic	Produce Ingestion	0.30	
	Mercury	Produce Ingestion	0.21	
116-F-1	Arsenic	Produce Ingestion	0.95	35
600-235	Arsenic	Produce Ingestion	0.78	30
	Cadmium	Produce Ingestion	0.16	
100-F-7	Arsenic	Produce Ingestion	0.52	25
	Mercury	Beef Ingestion	0.28	
	Mercury	Produce Ingestion	0.13	
600-23	Arsenic	Produce Ingestion	0.62	24
100-F-26:1	Arsenic	Produce Ingestion	0.78	21
600-190	Arsenic	Produce Ingestion	0.64	20
100-F-26:7	Arsenic	Produce Ingestion	0.66	19
100-F-26:7	Mercury	Beef Ingestion	0.20	19
600-131	Arsenic	Produce Ingestion	0.93	18
600-132	Arsenic	Produce Ingestion	0.75	17
	Cadmium	Produce Ingestion	0.14	
600-181	Arsenic	Produce Ingestion	0.90	17
100-F-14	Arsenic	Produce Ingestion	0.58	17
	Mercury	Beef Ingestion	0.26	
	Mercury	Produce Ingestion	0.12	
628-1	Arsenic	Produce Ingestion	0.95	17

Site	COPC	Pathway	Fraction of Child Hazard	Present-Day Child Hazard
100-F-18	Arsenic	Produce Ingestion	0.54	16
	Mercury	Beef Ingestion	0.29	
	Mercury	Produce Ingestion	0.13	
600-204	Arsenic	Produce Ingestion	0.91	16
100-F-38	Arsenic	Produce Ingestion	0.81	15
116-F-14	Arsenic	Produce Ingestion	0.94	15
100-F-12	Arsenic	Produce Ingestion	0.91	14
100-F-26:5	Arsenic	Produce Ingestion	0.82	13
	Cadmium	Produce Ingestion	0.12	
600-128	Arsenic	Produce Ingestion	0.87	13
100-F-26:2	Arsenic	Produce Ingestion	0.93	12
100-F-25	Mercury	Beef Ingestion	0.67	11
	Mercury	Produce Ingestion	0.32	
116-F-5	Arsenic	Produce Ingestion	0.94	10

Exposure to arsenic, mercury, and cadmium by produce ingestion are the most important exposure pathways contributing to HI values above 10 at each of these waste sites. Critical effects underlying the oral RfD for mercury include effects on the immune and renal (kidney) system, whereas the critical effects underlying the oral RfD for arsenic relate to the vascular and integumentary (skin) system (IRIS 2009). The critical target organ for chronic cadmium toxicity is also the renal system (IRIS 2009). Summing HQs for mercury and arsenic likely results in a protective bias for the tabulated HI values at these sites. There are no sites where mercury and cadmium are both significant contributors to an HI above 10. As described for Yakama Resident cancer risk, the use of shallow zone soil verification data to assess risks at excavated waste sites may contribute to a significant protective bias in the HI results for these sites.

Because arsenic concentrations at the remediated waste sites where arsenic is a risk driver are similar to commonly referenced background levels, as described in Section 5.7.3, risks for COPCs other than arsenic are also of interest for risk management at these sites. A summary of the key exposure pathways and COPCs, other than arsenic, contributing 10% or more to present-day RME Subsistence Farmer child HI at these sites is provided in Section 5.7.3. This summary is also applicable to the Yakama Resident scenario.

Several of the 100-F/100-IU-2/100-IU-6 Area sites evaluated in this risk assessment have an estimated surface area of less than 10 m^2 (110 ft^2). The 100-F-37 and 100-F-9 waste sites, for example, are french drains with an estimated surface area of approximately 1 to 3 m^2 (10.8 to 32.3 ft^2). As described for Yakama Resident cancer risk, this may result in grossly overestimated risks because the risk assessment calculations do not use any weighting factors to apportion a fraction of exposure to a site. Mercury was measured in only a single sample at the 100-F-9 waste site, so there is low confidence in the representative concentration at this site.

5.7.6 Risks Related to Lead in Soil for the 100-F/100-IU-2/100-IU-6 Area

The highest calculated RME representative concentration for lead in shallow zone soil at any remediated waste site is approximately 210 mg/kg. This concentration applies to the 100-F-37 remediated waste site in the 100-F/100-IU-2/100-IU-6 Area. No representative concentrations for lead at any waste site exceed EPA's recommended residential screening level for lead in soil of 400 mg/kg, derived using the IEUBK model that has been associated with bare soil in a play area (40 CFR 745). Because soil concentrations for lead are well below the most restrictive of EPA's soil screening criteria, no additional evaluation of lead is included in the HHRA. More detailed information on lead toxicity and the basis of EPA's soil screening criteria is provided in Section 3.5.8.

5.8 300 AREA RESULTS

Section 5.8 is the last of six sections of the local-area risk assessment (Sections 5.3 through 5.8) that present the results of the soil source term risk assessment for remediated waste sites in an individual ROD decision area. A summary of the results of the waste site soils risk assessment for all ROD decision areas is shown in Tables 5-2 through 5-7. Tables 5-2, 5-3, and 5-4 show the number of remediated waste sites with risk assessment results above or below decision thresholds for the Occupational scenarios. Table 5-2 shows results using present-day radionuclide soil concentrations, and the latter two tables show the results with radionuclide concentrations at years 2075 and 2150. Tables 5-5, 5-6, and 5-7 show these same summaries for the Residential exposure scenarios. The primary purpose of these summary tables is to allow the reader to quickly see how the waste sites risk assessment results for the different exposure scenarios compare to the thresholds used for risk management decisions. These tables also provide a perspective of how the waste site risk assessment results vary among the six ROD decision areas, and how cancer risk and radiation dose will change over time due to radioactive decay.

As described in Section 3.3, broad-area EPCs in the upland environment are associated with the occupational portion of the Resident National Monument/Refuge scenario and are also used to supplement the local-area cleanup verification data to calculate EPCs for dust inhalation for all scenarios evaluated in Sections 5.3 through 5.8. The COPCs for the broad-area components of the waste site risk calculations are those related to upland surface soil, while the COPCs for the local-area components of the waste site risk calculations are those related to the cleanup verification data for a particular ROD decision area. Risk assessment results for the Resident National Monument/Refuge scenario are provided for the local-area and broad-area components separately. Broad-area representative concentrations for upland surface soil are calculated using the MIS soil data from all 20 upland sampling locations, using the same protocol described for MIS reference area data in Section 5.2.4.

5.8.1 Industrial/Commercial Scenario Results for the 300 Area

The Industrial/Commercial exposure scenario encompasses potential occupational exposures related to shallow zone soils around remediated waste sites. Like the Resident National Monument/Refuge scenario, it focuses on direct contact exposure pathways and does not encompass the raising of produce or animals, nor the ingestion of Columbia River fish. It is similar to the Industrial/Commercial scenario used by EPA to calculate soil screening values using direct contact exposure pathways such as inadvertent soil ingestion, dermal contact with soil, and inhalation. Of the scenarios evaluated on the scale of an individual remediated waste site, the Industrial/Commercial scenario has the lowest exposure intensity. The scenario is more fully described in the CSM in Section 3.3 and is summarized in Table 3-12 and Figure 3-16. The COPCs evaluated in shallow zone soil in the 300 Decision Area are shown in Table 5-26.

5.8.1.1 Cancer Risk. Table 5-126 shows the RME and CTE total cancer risk results for the Industrial/Commercial exposure scenario for all remediated waste sites in the 300 Decision Area. The tabulated results are the sum of the risk values for the individual COPCs included in the risk calculation. Total risk results use EPCs based on measured soil concentrations in the individual waste sites. Risk assessment results are shown for present-day radionuclide concentrations, and with radionuclide concentrations decayed to the years 2075 and 2150. Soil concentrations for these times were calculated using the RESRAD computer code, as described in Appendix D-1.4.1.

The range of present-day RME and CTE total cancer risk results for the Industrial/Commercial scenario (radionuclides + chemicals) is from 4×10^{-7} to 1×10^{-4} and from 1×10^{-7} to 2×10^{-5}, respectively. No COPCs contributing to cancer risk were detected at the JA Jones waste site. Per EPA guidance (EPA/540/1-89/002), cancer risk results are displayed with only one significant figure.

As shown in Table 5-2, there were no remediated waste sites in the 300 Decision Area with calculated total RME cancer risks for the Industrial/Commercial scenario above 1×10^{-4}. The highest present-day RME cancer risk is 1×10^{-4} at the 316-1, 316-2, and 316-5. At the 316-1 waste site, RME cancer risk was mostly due to external irradiation from cobalt-60. The present-day RME cancer risk of 1×10^{-4} at the 316-2 and 316-5 waste sites was mostly due to external irradiation from uranium-238 and uranium-235. Cancer risk at the 316-1 waste site decreased from 1×10^{-4} to 2×10^{-5} at year 2075 due to decay of cobalt-60.

5.8.1.2 Radiation Dose. Table 5-127 shows the RME and CTE total radiation dose results for the Industrial/Commercial exposure scenario for all remediated waste sites in the 300 Decision Area. The tabulated results are the sum of the dose values for the individual COPCs included in the dose calculation. Risk assessment results are shown for present-day radionuclide concentrations, and with radionuclide concentrations decayed to the years 2075 and 2150. Soil concentrations for these times were calculated using the RESRAD computer code, as described in Appendix D-1.4.1.

The range of present-day RME and CTE total radiation dose results for the Industrial/Commercial scenario is from 0.021 to 7.4 mrem/yr and from 0.015 to 3.3 mrem/yr, respectively. No radionuclide COPCs were detected at the JA Jones waste site.

There are no waste sites in the 300 Area where the present-day radiation dose for the Industrial/Commercial scenario exceeded the threshold of 15 mrem/yr. At year 2075, the highest RME total radiation dose for the Industrial/Commercial exposure scenario is 5.5 mrem/yr at the 316-5 waste site.

5.8.1.3 Chemical Hazard. Table 5-128 shows the RME and CTE HI results for the Industrial/Commercial exposure scenario for all remediated waste sites in the 300 Decision Area. The tabulated results are the sum of the risk values for the individual COPCs included in the risk calculation.

The range of RME and CTE HI results for the Industrial/Commercial scenario is from 0.00012 to 0.21 and from 0.00010 to 0.060, respectively. No COPCs contributing to chemical hazard were detected at the 300 VTS and 600-259 waste sites.

There are no waste sites in the 300 Area where the HI for the Industrial/Commercial scenario approaches the threshold of 1.0.

5.8.2 Resident National Monument/Refuge Scenario Results for the 300 Area

The Resident National Monument/Refuge exposure scenario encompasses potential residential exposures related to shallow zone soils around remediated waste sites, as well as potential occupational exposures to surface soil across the upland environment of the River Corridor. It focuses on direct contact exposure pathways and does not encompass the raising of produce or animals nor the ingestion of Columbia River fish. This scenario also includes exposures related to groundwater via a domestic water well. Risks related to groundwater use are addressed in Section 6.0. With respect to soil exposure pathways, this scenario is similar to the traditional urban/suburban residential scenario used by EPA to calculate soil screening values using direct contact exposure pathways such as inadvertent soil ingestion, dermal contact with soil, and inhalation. It may be viewed as akin to the Subsistence Farmer scenario for an adult receptor, minus the ingestion of foodstuffs and with an exposure model that averages exposure to both individual waste site soils (cleanup verification soil data) and surface soils over large areas (upland surface soil data). The scenario is more fully described in the CSM in Section 3.3 and is summarized in Table 3-12 and Figure 3-16. The COPCs evaluated in shallow zone soil in the 300 Decision Area are shown in Table 5-26.

As described in the introductory text for Section 5.4, COPCs for the broad-area components of the waste site risk calculations are those related to upland surface soil, while the COPCs for the local-area components of the waste site risk calculations are those related to the cleanup verification data for a particular ROD decision area. Results for the local-area portion of the Resident National Monument/Refuge exposure scenario are calculated for each individual remediated waste site where a residence might be situated. Risks related to the broad-area occupational component of the Resident National Monument/Refuge exposure scenario are shown in the first row of the summary tables for cancer risk, radiation dose, and chemical hazard. As discussed in Section 4.1.3, broad-area representative concentrations are calculated using upland surface soil data collected under DOE/RL-2005-42.

5.8.2.1 Cancer Risk. Table 5-129 shows the RME and CTE total cancer risk results for the Resident National Monument/Refuge exposure scenario for all remediated waste sites in the 300 Decision Area. The tabulated results are the sum of the risk values for the individual COPCs included in the risk calculation. Total risk results use EPCs based on measured soil concentrations in the individual waste sites (residential exposure) and River Corridor upland surface soil (occupational exposure). Risk assessment results are shown for present-day radionuclide concentrations, and with radionuclide concentrations decayed to the years 2075 and 2150. Soil concentrations for these times were calculated using the RESRAD computer code, as described in Appendix D-1.4.1.

The range of present-day RME and CTE total cancer risk results for the local-area component of the Resident National Monument/Refuge scenario (radionuclides + chemicals) is from 8×10^{-7} to 3×10^{-4} and from 2×10^{-7} to 3×10^{-5}, respectively. No COPCs contributing to cancer risk were detected at the JA Jones waste site. Per EPA guidance (EPA/540/1-89/002), cancer risk results are displayed with only one significant figure. As shown in Table 5-2, 3 of the 17 waste sites in the 300 Decision Area had calculated total RME cancer risks above 1×10^{-4}, but no waste site had a CTE total cancer risk above 1×10^{-4}.

Because the broad-area component of cancer risk is the same across all calculations, the summary of the key exposure pathways and COPCs contributing 10% or more to present-day RME Resident National Monument/Refuge cancer risk at 300 Decision Area sites focuses on the local-area component of the Resident National Monument/Refuge scenario. The local-area calculations with total cancer risk above 1×10^{-4} are summarized below.

Site	COPC	Pathway	Fraction of Total Risk	Present-Day Total Risk
316-1	Cobalt-60	External Irradiation	0.81	3×10^{-4}
316-5	Uranium-238	External Irradiation	0.38	2×10^{-4}
	Uranium-235	External Irradiation	0.28	
	Cesium-137	External Irradiation	0.23	
316-2	Uranium-238	External Irradiation	0.38	2×10^{-4}
	Cobalt-60	External Irradiation	0.26	
	Uranium-235	External Irradiation	0.24	

With the exception of the 316-1 waste site, most of the contribution to RME local-area cancer risk is related to long-lived uranium isotopes. As shown in Table 5-129, cancer risks at the 316-5 and 316-2 waste sites do not change appreciably between the present-day and year 2150.

The local-area RME cancer risk at the 316-1 waste site has dropped to 4×10^{-5} at year 2075; at that time most of the RME cancer risk for the Resident National Monument/Refuge exposure scenario is related to uranium isotopes.

The current conditions of the 316-1, 316-2, and 316-5 remediated waste sites are summarized in Table 2-8. These waste sites were excavated to depths of 5.7 m (18.7 ft), 7.5 m (24.6 ft), and 6.3 m (20.7 ft), respectively. The sites were backfilled to grade following remediation. The shallow zone soil verification data used to calculate local-area risk are from samples collected along the sidewalls of the excavations at depths between 0 and 4.6 m (15 ft). The use of these data to represent surface soil for current and future exposures may contribute to a significant protective bias in the Resident National Monument/Refuge risk results for these sites. Uncertainties in the use of shallow zone soil verification data from excavated waste sites are discussed in more detail in Section 5.9 and Appendix D-1.

5.8.2.2 Radiation Dose. Table 5-130 shows the RME and CTE total radiation dose results for the Resident National Monument/Refuge exposure scenario for all remediated waste sites in the 300 Decision Area. The tabulated results are the sum of the risk values for the individual COPCs included in the risk calculation. Risk assessment results are shown for present-day radionuclide concentrations, and with radionuclide concentrations decayed to the years 2075 and 2150. Soil concentrations for these times were calculated using the RESRAD computer code, as described in Appendix D-1.4.1.

The range of present-day RME and CTE total radiation dose results for the local-area component of the Resident National Monument/Refuge scenario is from 0.036 to 13 mrem/yr and from 0.025 to 5.5 mrem/yr, respectively. The RME and CTE radiation dose results for the broad-area component were 0.80 and 0.78 mrem/yr, respectively. No radionuclide COPCs were detected at the JA Jones waste site.

As shown in Table 5-2, none of the 17 waste sites evaluated in the 300 Decision Area had a combined local-area and broad-area present-day RME radiation dose value above 15 mrem/yr. The local-area RME radiation dose of 13 mrem/yr at the 316-1 waste site was mostly due to external irradiation from cobalt-60. The highest RME dose at year 2075 is 9.4 mrem/yr at the 316-5 waste site.

5.8.2.3 Chemical Hazard. Table 5-131 shows the RME and CTE HI results for the Resident National Monument/Refuge exposure scenario for all remediated waste sites in the 300 Decision Area. The tabulated results are the sum of the risk values for the individual COPCs included in the risk calculation.

The range of RME and CTE HI results for the local-area component of the Resident National Monument/Refuge scenario is from 0.0019 to 0.32 and from 0.0018 to 0.14, respectively. The RME and CTE HI results for the broad-area component were 0.025 and 0.0079, respectively. There are no waste sites in the 300 Area where the HI for the Resident National Monument/Refuge scenario exceeds the threshold of 1.0.

5.8.3 Subsistence Farmer Scenario Results for the 300 Area

The Subsistence Farmer exposure scenario encompasses potential exposures related to shallow zone soils around remediated waste sites, including both direct contact exposure pathways and exposures related to raising produce and animals in the potentially affected area of the waste site. This scenario also includes exposures related to ingestion of Columbia River fish and exposure to groundwater via a domestic water well. Risks related to fish ingestion and groundwater use are addressed in Sections 4.4 and 6.0, respectively. The scenario is more fully described in the CSM in Section 3.3 and is summarized in Table 3-13 and Figure 3-17. The COPCs evaluated in shallow zone soil in the 300 Decision Area are shown in Table 5-26.

5.8.3.1 Cancer Risk. Table 5-132 shows the RME and CTE total cancer risk results for the Subsistence Farmer exposure scenario for all remediated waste sites in the 300 Decision Area. The tabulated results are the sum of the risk values for the individual COPCs included in the risk calculation. Total risk results use EPCs based on measured soil concentrations in the individual waste sites. Risk assessment results are shown for present-day radionuclide concentrations, and with radionuclide concentrations decayed to the years 2075 and 2150. Soil concentrations for these times were calculated using the RESRAD computer code, as described in Appendix D-1.4.1.

The range of present-day RME and CTE total cancer risk results for the Subsistence Farmer scenario (radionuclides + chemicals) is from 2×10^{-6} to 3×10^{-3} and from 5×10^{-7} to 3×10^{-4}, respectively. No COPCs contributing to cancer risk were detected at the JA Jones waste site. Per EPA guidance (EPA/540/1-89/002), cancer risk results are displayed with only one significant figure. As shown in Table 5-5, 12 of the 17 remediated waste sites in the 300 Decision Area had calculated Subsistence Farmer RME cancer risks above 1×10^{-4}, and 5 sites had CTE cancer risks above 1×10^{-4}.

A summary of the key exposure pathways and COPCs contributing 10% or more to present-day RME Subsistence Farmer cancer risk at 300 Decision Area sites with total cancer risk above 1×10^{-4} is provided below.

Site	COPC	Pathway	Fraction of Total Risk	Present-Day Total Risk
316-1	Arsenic	Produce Ingestion	0.64	3×10^{-3}
	Cobalt-60	External Irradiation	0.15	
300-10	Arsenic	Produce Ingestion	0.84	2×10^{-3}
316-2	Arsenic	Produce Ingestion	0.53	2×10^{-3}
618-12	Arsenic	Produce Ingestion	0.78	2×10^{-3}
316-5	Arsenic	Produce Ingestion	0.20	2×10^{-3}
	Uranium-238	External Irradiation	0.11	
300-50	Arsenic	Produce Ingestion	0.75	1×10^{-3}
300-49	Arsenic	Produce Ingestion	0.82	7×10^{-4}

Site	COPC	Pathway	Fraction of Total Risk	Present-Day Total Risk
618-5	Arsenic	Produce Ingestion	0.84	7×10^{-4}
628-4	Arsenic	Produce Ingestion	0.79	7×10^{-4}
618-4	Arsenic	Produce Ingestion	0.80	5×10^{-4}
600-47	Arsenic	Produce Ingestion	0.82	4×10^{-4}
300-18	Arsenic	Produce Ingestion	0.84	3×10^{-4}

Subsistence Farmer RME total cancer risk above 1×10^{-4} is primarily related arsenic from produce ingestion exposure. In addition to arsenic, cobalt-60 and uranium-238 contribute about 15% and 10% to the RME total cancer risks at the 316-1 and 316-5 waste sites, respectively. Because arsenic is the dominant contributor to RME cancer risk at these sites, risks do not change appreciably between present-day and year 2150.

Arsenic is a primary contributor to RME cancer risk for all remediated waste sites in the 300 Decision Area with RME present-day Subsistence Farmer cancer risks above 1×10^{-4}. As shown in Table 5-48, the Subsistence Farmer RME arsenic cancer risk calculated using the upland reference area data is 5×10^{-4}. Arsenic-related cancer risks exceed reference area levels at nine remediated waste sites in the 300 Area. At the 316-5 waste site, however, the RME arsenic soil concentration of 2.3 mg/kg is below the RME upland reference area value of 3.2 mg/kg. At this particular site, uranium isotopes contribute most to RME cancer risk, but do so by a variety of pathways with only one pathway-COPC combination (external irradiation for uranium-238) contributing more than 10%. The range of RME arsenic soil concentrations at 316-1, 300-10, 316-2, 618-12, 300-50, 300-49, 618-5, and 628-4, where arsenic-related risks exceed the reference areas level, range from 3.9 mg/kg (628-4) to 13.5 mg/kg (300-10). These RME arsenic soil concentrations are all above the upland reference area RME concentration of 3.2 mg/kg shown in Table 5-47. RME arsenic concentrations at 300-50 (6.3 mg/kg), 316-2 (8.0 mg/kg), 618-12 (9.4 mg/kg), 316-1 (13.1 mg/kg), and 300-10 (13.5 mg/kg) are also above one or both of the Hanford Area (6.47 mg/kg) and Yakima Basin (5.13 mg/kg) 90th percentile soil background concentrations.

Arsenic concentrations at the 12 remediated waste sites where arsenic drives RME risks above 1×10^{-4} are at most about a factor of 2 above commonly referenced background levels. Therefore, risks for COPCs other than arsenic are also of interest for risk management at these sites. All 12 of the sites have RME cancer risks above 1×10^{-6} related to COPCs other than arsenic. A summary of the key exposure pathways and COPCs, other than arsenic, contributing 10% or more to present-day RME Subsistence Farmer cancer risk at these sites is provided below.

Site	COPC other than arsenic	Pathway	Fraction of Total Risk	Present-Day Total Risk (without arsenic)
316-5	Uranium-238	External Irradiation	0.15	1×10^{-3}
	Uranium-235	External Irradiation	0.11	
316-2	Uranium-238	External Irradiation	0.16	8×10^{-4}
	Cobalt-60	External Irradiation	0.11	
316-1	Cobalt-60	External Irradiation	0.57	7×10^{-4}
300-50	Uranium-238	External Irradiation	0.21	1×10^{-4}
	Uranium-238	Chicken Ingestion	0.12	
	Uranium-238	Milk Ingestion	0.11	
618-12	Uranium-238	External Irradiation	0.21	1×10^{-4}
	Uranium-238	Chicken Ingestion	0.12	
	Uranium-238	Milk Ingestion	0.11	
628-4	Aroclor-1248	Milk Ingestion	0.28	5×10^{-5}
	Aroclor-1248	Beef Ingestion	0.15	
618-4	Uranium-238	External Irradiation	0.21	3×10^{-5}
	Uranium-238	Chicken Ingestion	0.12	
	Uranium-238	Milk Ingestion	0.11	
300-10	Cesium-137	External Irradiation	0.30	3×10^{-5}
	Uranium-238	External Irradiation	0.12	
300-49	Cesium-137	External Irradiation	0.30	2×10^{-5}
600-47	Uranium-238	External Irradiation	0.22	2×10^{-5}
	Uranium-238	Chicken Ingestion	0.12	
	Uranium-238	Milk Ingestion	0.11	
618-5	Uranium-238	External Irradiation	0.20	1×10^{-5}
	Uranium-238	Chicken Ingestion	0.12	
	Uranium-235	External Irradiation	0.12	
	Uranium-238	Milk Ingestion	0.10	
300-18	Uranium-238	External Irradiation	0.16	5×10^{-6}
	Uranium-233/234	Chicken Ingestion	0.14	
	Uranium-233/234	Milk Ingestion	0.12	

With the exception of the 316-1 waste site, where external radiation from cobalt-60 contributes about 60% of the present-day RME cancer risk, most individual pathway-COPC combinations contribute less than 20% of total risk in the absence of arsenic. The RME cancer risks without arsenic at these sites are mostly related to uranium isotopes, with the exception of Aroclor-1248 at the 628-4 waste site and cesium-137 at the 300-10 waste site.

The range of RME arsenic soil concentrations at 300 Area remediated waste sites where RME total cancer risk was highest due to arsenic is not much higher than might be expected of background soils. Although the arsenic risk estimates for the produce ingestion exposure pathway are based on plant uptake models that are prone to overestimation of plant concentrations when soil concentrations are high (see Section 5.9.2), this potential source of bias may not be very important for the arsenic risk estimates at the 300 Area sites because site soil levels are not much greater than naturally occurring levels.

As described in relation to Resident National Monument/Refuge cancer risk results, the use of shallow zone soil verification data may contribute to a significant protective bias in the risk results for excavated sites. The 300 Area remediated waste sites with Subsistence Farmer RME cancer risks above 1×10^{-4} due to the presence of arsenic or uranium isotopes were generally excavated to depths of 3 m (9.8 ft) or more, as shown in Table 2-8. The exceptions include 300-10 (1.8 m [5.9 ft]), 300-18 (1 m [3.3 ft]), and 600-47 (1.5 m [4.9 ft]). Uncertainties in the use of shallow zone soil verification data from excavated waste sites are discussed in more detail in Section 5.9 and Appendix D-1.

5.8.3.2 Radiation Dose. Table 5-133 shows the RME and CTE total radiation dose results for the Subsistence Farmer exposure scenario for all remediated waste sites in the 300 Decision Area. The tabulated results are the sum of the dose values for the individual COPCs included in the dose calculation. Risk assessment results are shown for present-day radionuclide concentrations, and with radionuclide concentrations decayed to the years 2075 and 2150. Soil concentrations for these times were calculated using the RESRAD computer code, as described in Appendix D-1.4.1.

The range of present-day RME and CTE total radiation dose results for the Subsistence Farmer scenario is from 0.086 to 81 mrem/yr, and from 0.074 to 29 mrem/yr, respectively. No radionuclide COPCs were detected at the JA Jones waste site. As shown in Table 5-5, 3 of the 17 waste sites in the 300 Decision Area had calculated total RME radiation dose values above 15 mrem/yr, and 2 waste sites had a CTE total radiation dose above 15 mrem/yr.

A summary of the key exposure pathways and COPCs contributing 10% or more to the present-day RME Subsistence Farmer radiation dose at 300 Decision Area sites with total dose above 15 mrem/yr is provided below.

Site	COPC	Pathway	Fraction of Total Dose	Present-Day Total Dose (mrem/yr)
316-5	Uranium-233/234	Poultry Ingestion	0.13	81
	Uranium-233/234	Milk Ingestion	0.12	
316-2	Uranium-233/234	Poultry Ingestion	0.12	51
	Uranium-238	External Irradiation	0.11	
	Uranium-233/234	Milk Ingestion	0.10	
	Uranium-238	Poultry Ingestion	0.10	

Site	COPC	Pathway	Fraction of Total Dose	Present-Day Total Dose (mrem/yr)
316-1	Cobalt-60	External Irradiation	0.54	33

Similar to the findings for the Resident National Monument/Refuge scenario, the Subsistence Farmer present-day RME radiation dose at the 316-5 and 316-2 waste sites is driven by long-lived radionuclides. Only at the 316-1 waste site does radiation dose change appreciably with time. In the Subsistence Farmer scenario, radiation dose is related to uranium exposure by uptake into agricultural products including poultry and milk. By year 2075, the total RME dose at the 316-1 waste sites is 13 mrem/yr and is related to uranium isotopes rather than cobalt-60.

As described in relation to cancer risk, most 300 Area remediated waste sites where significant risks were calculated (including 316-5, 316-2, and 316-1) were excavated to a depth of greater than 4.6 m (15 ft). The use of shallow zone soil verification data to calculate risks at these sites may contribute to a significant protective bias in the radiation dose results.

5.8.3.3 Chemical Hazard. Table 5-134 shows the RME and CTE HI results for the Subsistence Farmer exposure scenario for all remediated waste sites in the 300 Decision Area. The tabulated results are the sum of the risk values for the individual COPCs included in the risk calculation. As described in Section 3.4.4, chemical hazard is calculated separately for adult and child receptors. For the Subsistence Farmer scenario, RME and CTE child HIs are usually equal to or above those for adults. Therefore, only child HI values are presented here.

The range of RME and CTE child HI results for the Subsistence Farmer scenario is from 0.0014 to 77 and from 0.00076 to 23, respectively. No COPCs contributing to chemical hazard were detected at the 300 VTS and 600-259 waste sites. As shown in Table 5-5, 6 of the 17 waste sites in the 300 Decision Area had calculated RME HI values above 10, and 6 sites had an HI value between 1.0 and 10.

A summary of the key exposure pathways and COPCs contributing 10% or more to RME Subsistence Farmer HIs at 300 Decision Area sites with HI values above 1.0 is provided below.

Site	COPC	Pathway	Fraction of Child Hazard	Present-Day Child Hazard [a]
316-1	Copper	Milk Ingestion	0.17	77
	Copper	Produce Ingestion	0.16	
	Mercury	Beef Ingestion	0.13	
	Arsenic	Produce Ingestion	0.12	
316-5	Uranium	Poultry Ingestion	0.24	27
	Uranium	Milk Ingestion	0.21	
	Uranium	Soil Ingestion	0.17	
	Uranium	Egg Ingestion	0.16	

<table>
<tr><th>Site</th><th>COPC</th><th>Pathway</th><th>Fraction of Child Hazard</th><th>Present-Day Child Hazard [a]</th></tr>
<tr><td rowspan="5">316-2</td><td>Arsenic</td><td>Produce Ingestion</td><td>0.23</td><td rowspan="5">25</td></tr>
<tr><td>Uranium</td><td>Poultry Ingestion</td><td>0.17</td></tr>
<tr><td>Uranium</td><td>Milk Ingestion</td><td>0.15</td></tr>
<tr><td>Uranium</td><td>Soil Ingestion</td><td>0.13</td></tr>
<tr><td>Uranium</td><td>Egg Ingestion</td><td>0.11</td></tr>
<tr><td>300-10</td><td>Arsenic</td><td>Produce Ingestion</td><td>0.80</td><td>12</td></tr>
<tr><td>618-12</td><td>Arsenic</td><td>Produce Ingestion</td><td>0.56</td><td>12</td></tr>
<tr><td>300-50</td><td>Arsenic</td><td>Produce Ingestion</td><td>0.38</td><td>12</td></tr>
<tr><td rowspan="2">628-4</td><td>Arsenic</td><td>Produce Ingestion</td><td>0.40</td><td rowspan="2">6.7</td></tr>
<tr><td>Aroclor-1248</td><td>Milk Ingestion</td><td>0.12</td></tr>
<tr><td>300-49</td><td>Arsenic</td><td>Produce Ingestion</td><td>0.55</td><td>5.4</td></tr>
<tr><td>618-5</td><td>Arsenic</td><td>Produce Ingestion</td><td>0.76</td><td>3.9</td></tr>
<tr><td>618-4</td><td>Arsenic</td><td>Produce Ingestion</td><td>0.62</td><td>3.6</td></tr>
<tr><td>600-47</td><td>Arsenic</td><td>Produce Ingestion</td><td>0.66</td><td>2.4</td></tr>
<tr><td>300-18</td><td>Arsenic</td><td>Produce Ingestion</td><td>0.78</td><td>2.0</td></tr>
</table>

[a] Values for copper and mercury hazard at 316-1 and uranium hazard at 316-5 and 316-2 are likely overestimated; see discussion in the text.

Exposure to copper and mercury via produce, beef, and milk ingestion are the most important exposure pathways contributing to the RME child HI at the 316-1 waste site. Uranium exposure via a variety of pathways drives hazard at the 316-5 and 316-2 waste sites. Arsenic contributes about 25% of the child RME HI at the 316-2 waste site. The oral RfD for copper is related to gastrointestinal irritation, and is also protective of individuals with a rare medical condition that inhibits the regulation of copper absorption following ingestion (IRIS 2009). The HQ for copper is unlikely to be additive to that for mercury, which is based on immune system effects, leading to an overestimate of the HI value at the 316-1 waste site. Additive effects are also unlikely for uranium and arsenic at the 316-2 waste site. The critical target organ of uranium toxicity is the kidney, whereas arsenic toxicity relates to the vascular and integumentary (skin) systems (IRIS 2009). Only hazards related to uranium and mercury are potentially additive among the metals listed at these 12 sites because mercury, like uranium, is a known kidney toxicant (Klaassen 2001). Although they do not both contribute more than 10% to RME child HI for an individual exposure pathway at any remediated waste site, both mercury and uranium concentrations are elevated at the 316-1 waste site.

As discussed in the Uncertainty Analysis (Section 5.9), the linear models used to estimate tissue concentrations from soil data are prone to overestimation of plant and animal tissue concentrations at higher soil concentrations. For example, the RME soil concentrations of copper and mercury at the 316-1 waste site are approximately 1,420 and 1.1 mg/kg, respectively. The RME upland soil concentrations of copper and mercury in the reference area are 17 and 0.022 mg/kg, respectively. Because copper and mercury soil concentrations at this site are approximately 80 and 50 times higher than background, respectively, the confidence in the RME HI values based on modeled tissue concentrations is low. Both copper and mercury were measured in nine discrete soil samples at 316-1, so there is reasonably high confidence in the associated representative concentrations.

At a copper soil concentration of 1,420 mg/kg, the equivalent dry-weight concentration in produce is about 1,200 mg/kg, calculated using the copper produce-soil ratio (0.87) described in Appendix D-1.4.1. For animal fodder, the copper and mercury grass-soil ratios are 0.80 and 1.0, respectively, which result in dry-weight concentrations in grass of 1,100 mg/kg (copper) and 1.1 mg/kg (mercury). As discussed in Appendix D-1.4.1 and in the Uncertainty Analysis (Section 5.9), nonlinear regression models also exist for predicting plant concentrations of certain metals. The copper plant-soil regression model published by Bechtel (BJC/OR-133) predicts a copper plant concentration of 34 mg/kg when the soil concentration is 1,420 mg/kg. The mercury plant-soil regression model published by Bechtel (BJC/OR-133) predicts a mercury plant concentration of 0.39 mg/kg when the soil concentration is 1.1 mg/kg. Assuming the nonlinear model is more accurate at these higher soil concentrations, this suggests copper exposure at the 316-1 waste site may be overestimated by a factor of about 35 for produce ingestion (1,200 mg/kg versus 34 mg/kg) and about 32 for milk ingestion (1,100 mg/kg versus 34 mg/kg). For mercury, it suggests that beef ingestion exposure at the 316-1 waste site may be overestimated by factors of about three (1.1 mg/kg versus 0.39 mg/kg).

The RME representative concentrations of uranium at the 316-5 and 316-2 waste sites are approximately 240 mg/kg and 160 mg/kg, respectively. These values are far above the upland reference area RME concentration of 0.92 mg/kg calculated based on isotopic uranium measurements. The child HI values at these sites, which are based on food uptake, are also likely to be overestimated for the same reasons discussed for copper and mercury. Plant-soil concentration regression models are not available for uranium.

The 12 remediated waste sites where Subsistence Farmer RME child HI exceeds 1.0 are identical to the sites where RME cancer risk exceed 1×10^{-4}. As discussed in relation to cancer risk, arsenic concentrations at many of these sites were above the upland reference area value, and at five of the sites were also above one or both of the Hanford Area (6.47 mg/kg) and Yakima Basin (5.13 mg/kg) 90th percentile soil background concentrations. Because arsenic concentrations at the remediated waste sites where arsenic is a risk driver are at most about twice as high as commonly referenced background levels, risks for COPCs other than arsenic are also of interest for risk management at these sites. Of the eleven 300 Area remediated waste sites where arsenic contributes to an RME child HI above 1.0, there are six sites where child HI exceeds 1.0 without arsenic. A summary of the key exposure pathways and COPCs, other than arsenic, contributing 10% or more to present-day RME child HI at these sites is provided below.

Site	COPC other than arsenic	Pathway	Fraction of Child Hazard	Present-Day Child Hazard (without arsenic)
316-1	Copper	Milk Ingestion	0.20	66
	Copper	Produce Ingestion	0.18	
	Mercury	Beef Ingestion	0.15	
	Copper	Beef Ingestion	0.11	
316-2	Uranium	Chicken Ingestion	0.24	18
	Uranium	Milk Ingestion	0.21	
	Uranium	Soil Ingestion	0.17	
	Uranium	Egg Ingestion	0.16	
300-50	Uranium	Chicken Ingestion	0.16	6.3
	Uranium	Milk Ingestion	0.14	
	Uranium	Soil Ingestion	0.12	
	Uranium	Egg Ingestion	0.11	
618-12	Uranium	Chicken Ingestion	0.26	3.7
	Uranium	Milk Ingestion	0.24	
	Uranium	Soil Ingestion	0.20	
	Uranium	Egg Ingestion	0.18	
	Uranium	Produce Ingestion	0.11	
628-4	Aroclor-1248	Milk Ingestion	0.24	3.4
	Aroclor-1248	Soil Ingestion	0.19	
	Aroclor-1248	Beef Ingestion	0.13	
300-49	Nickel	Milk Ingestion	0.15	1.8
	Zinc	Beef Ingestion	0.13	
	Zinc	Milk Ingestion	0.12	
	Zinc	Produce Ingestion	0.10	

As described in relation to cancer risk, all of these sites were excavated to depths of 3 m (9.8 ft) or deeper. Therefore, the use of shallow zone soil verification data to calculate HI may contribute to a significant protective bias in the radiation dose results for these sites.

5.8.4 CTUIR Resident Scenario Results for the 300 Area

The CTUIR Resident exposure scenario encompasses potential exposures related to shallow zone soils around remediated waste sites, including both direct contact exposure pathways and exposures related to raising produce and animals in the potentially affected area of the waste site. This scenario also includes exposures related to ingestion of Columbia River fish (see Section 4.4) and exposure to groundwater via a domestic water well (see Section 6.0). It is similar to the Subsistence Farmer scenario, although exposure intensity is higher for many exposure pathways. The scenario is more fully described in the CSM in Section 3.3 and is summarized in Table 3-13 and Figure 3-17. The COPCs evaluated in shallow zone soil in the 100-F/100-IU-2/100-IU-6 Decision Area are shown in Table 5-26.

5.8.4.1 Cancer Risk. Table 5-135 shows the total cancer risk results for the CTUIR Resident exposure scenario for all remediated waste sites in the 300 Decision Area. The tabulated results are the sum of the risk values for the individual COPCs included in the risk calculation. Total risk results use EPCs based on measured soil concentrations in the individual waste sites. Risk assessment results are shown for present-day radionuclide concentrations, and with radionuclide concentrations decayed to the years 2075 and 2150. Soil concentrations for these times were calculated using the RESRAD computer code, as described in Appendix D-1.4.1.

The range of present-day total cancer risk results for the CTUIR Resident scenario (radionuclides + chemicals) is 6×10^{-6} and 3×10^{-2}. No COPCs contributing to cancer risk were detected at the JA Jones waste site. Per EPA guidance (EPA/540/1-89/002), cancer risk results are displayed with only one significant figure. As shown in Table 5-5, 12 of the 17 remediated waste sites in the 300 Decision Area had calculated total cancer risks above 1×10^{-4}.

A summary of the key exposure pathways and COPCs contributing 10% or more to present-day CTUIR Resident cancer risk at 300 Decision Area sites with total cancer risk above 1×10^{-4} is provided below.

Site	COPC	Pathway	Fraction of Total Risk	Present-Day Total Risk
316-1	Arsenic	Produce Ingestion	0.93	3×10^{-2}
300-10	Arsenic	Produce Ingestion	0.99	3×10^{-2}
618-12	Arsenic	Produce Ingestion	0.97	2×10^{-2}
316-2	Arsenic	Produce Ingestion	0.88	2×10^{-2}
300-50	Arsenic	Produce Ingestion	0.97	2×10^{-2}
618-5	Arsenic	Produce Ingestion	0.99	1×10^{-2}
300-49	Arsenic	Produce Ingestion	0.98	1×10^{-2}
628-4	Arsenic	Produce Ingestion	0.98	1×10^{-2}
316-5	Arsenic	Produce Ingestion	0.60	9×10^{-3}
	Uranium-233/234	Produce Ingestion	0.10	
618-4	Arsenic	Produce Ingestion	0.98	8×10^{-3}
600-47	Arsenic	Produce Ingestion	0.98	6×10^{-3}
300-18	Arsenic	Produce Ingestion	0.99	6×10^{-3}

Most of the total cancer risk above 1×10^{-4} is related arsenic exposure from produce ingestion. Uranium-234 contributes about 10% to the total cancer risk at the 316-5 waste site. Because arsenic is the dominant contributor to cancer risk at these sites, risks do not change appreciably between present-day and year 2150. As shown in Table 5-48, CTUIR Resident arsenic cancer risk calculated using the upland reference area data is 8×10^{-3}. The representative concentration of arsenic in upland reference area soil, shown in Table 5-47, is 3.2 mg/kg. The 90th percentile arsenic soil background concentrations for the Hanford Area and Yakima Basin are 6.47 mg/kg and 5.13 mg/kg, respectively. As discussed for Subsistence Farmer cancer risk in Section 5.8.3, the range of site RME arsenic concentrations at 316-1, 300-10, 618-12, 316-2, 300-50, 618-5, 300-49, and 628-4, where arsenic-related risks exceed the reference areas level, range from 3.9 mg/kg (628-4) to 13.5 mg/kg (300-10).

Because arsenic concentrations at the remediated waste sites where arsenic is a risk driver range up to about twice as high as commonly referenced background levels, risks for COPCs other than arsenic are also of interest for risk management at these sites. A summary of the key exposure pathways and COPCs, other than arsenic, contributing 10% or more to present-day RME Subsistence Farmer cancer risk at these sites is provided in Section 5.8.3. This summary is also applicable to the CTUIR Resident scenario.

As described in relation to Subsistence Farmer cancer risk results, 300 Area remediated waste sites with cancer risks above 1×10^{-4} were generally excavated to significant depths. The use of shallow zone soil verification data in these cases may contribute to a significant protective bias in the risk results for excavated sites. Uncertainties in the use of shallow zone soil verification data from excavated waste sites are discussed in more detail in Section 5.9 and Appendix D-1.

5.8.4.2 Radiation Dose. Table 5-136 shows the total radiation dose results for the CTUIR Resident exposure scenario for all remediated waste sites in the 300 Decision Area. The tabulated results are the sum of the dose values for the individual COPCs included in the dose calculation. Risk assessment results are shown for present-day radionuclide concentrations, and with radionuclide concentrations decayed to the years 2075 and 2150. Soil concentrations for these times were calculated using the RESRAD computer code, as described in Appendix D-1.4.

The range of present-day total radiation dose results for the CTUIR Resident scenario is 0.096 to 82 mrem/yr, respectively. No radionuclide COPCs were detected at the JA Jones waste site. As shown in Table 5-5, 3 of the 17 waste sites in the 300 Decision Area had calculated total radiation dose values above 15 mrem/yr.

A summary of the key exposure pathways and COPCs contributing 10% or more to the present-day CTUIR Resident radiation dose at 300 Decision Area sites with total dose above 15 mrem/yr is provided below.

Site	COPC	Pathway	Fraction of Total Dose	Present-Day Total Dose (mrem/yr)
316-5	Uranium-233/234	Produce Ingestion	0.33	82
	Uranium-238	Produce Ingestion	0.24	
	Uranium-238	External Irradiation	0.11	

Site	COPC	Pathway	Fraction of Total Dose	Present-Day Total Dose (mrem/yr)
316-2	Uranium-233/234	Produce Ingestion	0.30	52
	Uranium-238	Produce Ingestion	0.26	
	Uranium-238	External Irradiation	0.12	
316-1	Cobalt-60	External Irradiation	0.56	37
	Uranium-233/234	Produce Ingestion	0.12	
	Uranium-238	Produce Ingestion	0.11	

The CTUIR Resident present-day radiation dose at the 316-5 and 316-2 remediated waste sites is driven by long-lived radionuclides. At the 316-5 and 316-2 remediated waste sites, radiation dose is primarily related to uranium exposure by uptake into produce, and in small part to external irradiation from uranium-238. Only at the 316-1 waste site, where the short-lived radionuclide cobalt-60 drives the dose, does radiation dose change appreciably with time. By year 2075, the total dose at the 316-1 waste site is 13 mrem/yr and is related to uranium isotopes rather than cobalt-60. As described for CTUIR cancer risk, the use of shallow zone soil verification data to calculate risks may contribute to a significant protective bias in the radiation dose results for these sites.

5.8.4.3 Chemical Hazard. Table 5-137 shows the HI results for the CTUIR Resident exposure scenario for all remediated waste sites in the 300 Decision Area. The tabulated results are the sum of the risk values for the individual COPCs included in the risk calculation. As described in Section 3.4.4, chemical hazard is calculated separately for adult and child receptors. For the CTUIR Resident scenario, child HIs are usually equal to or above those for adults. Therefore, only child HI values are presented here.

The range of child HI results for the CTUIR Resident scenario is 0.0013 to 430. No COPCs contributing to chemical hazard were detected at the 300 VTS and 600-259 waste sites. As shown in Table 5-5, 12 of the 17 remediated waste sites in the 300 Decision Area had calculated HI values above 10. There were no sites with an HI value between 1.0 and 10.

A summary of the key exposure pathways and COPCs contributing 10% or more to CTUIR Resident child HIs at 300 Decision Area sites with HI values above 1.0 is provided below.

Site	COPC	Pathway	Fraction of Child Hazard	Present-Day Child Hazard [a]
316-1	Copper	Produce Ingestion	0.39	430
	Arsenic	Produce Ingestion	0.30	
	Mercury	Produce Ingestion	0.12	
300-10	Arsenic	Produce Ingestion	0.98	134
316-2	Arsenic	Produce Ingestion	0.68	115
	Uranium	Produce Ingestion	0.21	

Site	COPC	Pathway	Fraction of Child Hazard	Present-Day Child Hazard [a]
618-12	Arsenic	Produce Ingestion	0.91	100
300-50	Arsenic	Produce Ingestion	0.76	81
316-5	Uranium	Produce Ingestion	0.50	74
	Arsenic	Produce Ingestion	0.30	
	Uranium	Soil Ingestion	0.13	
300-49	Arsenic	Produce Ingestion	0.84	49
628-4	Arsenic	Produce Ingestion	0.83	45
618-5	Arsenic	Produce Ingestion	0.97	43
618-4	Arsenic	Produce Ingestion	0.93	33
600-47	Arsenic	Produce Ingestion	0.95	24
300-18	Arsenic	Produce Ingestion	0.98	22

[a] Values for copper and mercury hazard at 316-1 and uranium hazard at 316-5 and 316-2 are likely overestimated; see discussion in the text.

Exposure to copper, arsenic, and mercury via produce ingestion are the most important exposure pathways contributing to the child HI at the 316-1 waste site. Uranium and arsenic exposure via produce ingestion drives hazard at the 316-5 and 316-2 waste sites. At the remaining sites where the child HI exceeds 1.0, arsenic exposure by produce ingestion is the dominant factor. As described in relation to the Subsistence Farmer HI, the oral RfD for copper is related to gastrointestinal irritation, and is also protective of individuals with a rare medical condition that inhibits the regulation of copper absorption following ingestion. The HQ for mercury is based on immune system effects, although renal impacts are also important. The critical target organ of uranium toxicity is the kidney.

Uranium and arsenic are the main contributors to child HIs at all of these sites with the exception of 316-1, where copper and mercury are also important. The representative concentration of arsenic in upland reference area soil, shown in Table 5-47, is 3.2 mg/kg. The 90th percentile arsenic soil background concentrations for the Hanford Area and Yakima Basin are 6.47 mg/kg and 5.13 mg/kg, respectively. As described for CTUIR Resident cancer risk, the range of site RME arsenic concentrations at 316-1, 300-10, 316-2, 618-12, 300-50, 300-49, 628-4, and 618-5, where arsenic-related risks exceed the reference areas CTUIR Resident arsenic child HQ of 32 (see Table 5-50), range from 3.9 mg/kg (628-4) to 13.5 mg/kg (300-10). As described for CTUIR Resident cancer risk, the use of shallow zone soil verification data to assess risks may contribute to a significant protective bias in the HI results for these sites.

As discussed in the Uncertainty Analysis (Section 5.9), the linear models used to estimate tissue concentrations from soil data are prone to overestimation of plant and animal tissue concentrations at higher soil concentrations. A discussion of the potential overestimation of child HIs for these pathways in the 300 Area is provided in the risk assessment for the Subsistence Farmer scenario in the 300 Area (see Section 5.8.3). Child HI results related to uranium at the 316-5 and 316-2 waste sites are also likely to be overestimated due to nonlinearity in uranium uptake from soil into plants.

Because arsenic concentrations at the remediated waste sites where arsenic is a risk driver are up to about twice as high as commonly referenced background levels, risks for COPCs other than arsenic are also of interest for risk management at these sites. A summary of the key exposure pathways and COPCs, other than arsenic, contributing 10% or more to present-day RME Subsistence Farmer child HI at these 300 Area sites is provided in Section 5.8.3. This summary is also applicable to the CTUIR Resident scenario.

5.8.5 Yakama Resident Scenario Results for the 300 Area

The Yakama Resident exposure scenario encompasses potential exposures related to shallow zone soils around remediated waste sites, including both direct contact exposure pathways and exposures related to raising produce and animals in the potentially affected area of the waste site. This scenario also includes exposures related to ingestion of Columbia River fish (see Section 4.4) and exposure to groundwater via a domestic water well (see Section 6.0). It is similar to the Subsistence Farmer scenario, although exposure intensity is higher for many exposure pathways. The scenario is more fully described in the CSM in Section 3.3 and is summarized in Table 3-13 and Figure 3-17. The COPCs evaluated in shallow zone soil in the 100-F/100-IU-2/100-IU-6 Decision Area are shown in Table 5-26.

5.8.5.1 Cancer Risk. Table 5-138 shows the total cancer risk results for the Yakama Resident exposure scenario for all remediated waste sites in the 300 Decision Area. The tabulated results are the sum of the risk values for the individual COPCs included in the risk calculation. Total risk results use EPCs based on measured soil concentrations in the individual waste sites. Risk assessment results are shown for present-day radionuclide concentrations, and with radionuclide concentrations decayed to the years 2075 and 2150. Soil concentrations for these times were calculated using the RESRAD computer code, as described in Appendix D-1.4.1.

The range of present-day total cancer risk results for the Yakama Resident scenario (radionuclides + chemicals) is 7×10^{-6} and 4×10^{-2}. No COPCs contributing to cancer risk were detected at the JA Jones waste site. Per EPA guidance (EPA/540/1-89/002), cancer risk results are displayed with only one significant figure. As shown in Table 5-5, 12 of the 17 remediated waste sites in the 300 Decision Area had calculated total cancer risks above 1×10^{-4}.

A summary of the key exposure pathways and COPCs contributing 10% or more to present-day Yakama Resident cancer risk at 300 Decision Area sites with total cancer risk above 1×10^{-4} is provided below.

Site	COPC	Pathway	Fraction of Total Risk	Present-Day Total Risk
316-1	Arsenic	Produce Ingestion	0.90	4×10^{-2}
300-10	Arsenic	Produce Ingestion	0.97	3×10^{-2}

Site	COPC	Pathway	Fraction of Total Risk	Present-Day Total Risk
618-12	Arsenic	Produce Ingestion	0.95	2×10^{-2}
316-2	Arsenic	Produce Ingestion	0.86	2×10^{-2}
300-50	Arsenic	Produce Ingestion	0.94	2×10^{-2}
618-5	Arsenic	Produce Ingestion	0.97	1×10^{-2}
300-49	Arsenic	Produce Ingestion	0.96	1×10^{-2}
628-4	Arsenic	Produce Ingestion	0.94	1×10^{-2}
316-5	Arsenic	Produce Ingestion	0.56	1×10^{-2}
618-4	Arsenic	Produce Ingestion	0.96	8×10^{-3}
600-47	Arsenic	Produce Ingestion	0.96	6×10^{-3}
300-18	Arsenic	Produce Ingestion	0.97	6×10^{-3}

Most of the total cancer risk above 1×10^{-4} for the Yakama Resident scenario is related to arsenic due to produce ingestion. Because arsenic is the dominant contributor to cancer risk at these sites, risks do not change appreciably between present day and year 2150. As shown in Table 5-48, Yakama Resident arsenic cancer risk calculated using the upland reference area data is 8×10^{-3}. The representative concentration of arsenic in upland reference area soil, shown in Table 5-47, is 3.2 mg/kg. The 90th percentile arsenic soil background concentrations for the Hanford Area and Yakima Basin are 6.47 mg/kg and 5.13 mg/kg, respectively. As discussed for Subsistence Farmer cancer risk in Section 5.8.3, the range of site RME arsenic concentrations at 316-1, 300-10, 618-12, 316-2, 300-50, 618-5, 300-49, and 628-4, where arsenic-related risks exceed the reference areas level, range from 3.9 mg/kg (628-4) to 13.5 mg/kg (300-10).

Because arsenic concentrations at the remediated waste sites where arsenic is a risk driver range up to about twice as high as commonly referenced background levels, risks for COPCs other than arsenic are also of interest for risk management at these sites. A summary of the key exposure pathways and COPCs, other than arsenic, contributing 10% or more to present-day RME Subsistence Farmer cancer risk at these sites is provided in Section 5.8.3. This summary is also applicable to the Yakama Resident scenario.

As described in relation to Subsistence Farmer cancer risk results, 300 Area remediated waste sites with cancer risks above 1×10^{-4} were generally excavated to significant depths. The use of shallow zone soil verification data in these cases may contribute to a significant protective bias in the risk results for excavated sites. Uncertainties in the use of shallow zone soil verification data from excavated waste sites are discussed in more detail in Section 5.9 and Appendix D-1.

5.8.5.2 Radiation Dose. Table 5-139 shows the total radiation dose results for the Yakama Resident exposure scenario for all remediated waste sites in the 300 Decision Area. The tabulated results are the sum of the dose values for the individual COPCs included in the dose calculation. Risk assessment results are shown for present-day radionuclide concentrations, and with radionuclide concentrations decayed to the years 2075 and 2150. Soil concentrations for these times were calculated using the RESRAD computer code, as described in Appendix D-1.4.1.

The range of present-day total radiation dose results for the Yakama Resident scenario is 0.11 to 91 mrem/yr, respectively. No radionuclide COPCs were detected at the JA Jones waste site. As shown in Table 5-5, 3 of the 17 waste sites in the 300 Decision Area had calculated total radiation dose values above 15 mrem/yr.

COPC analytes contributing 10% or more to the present-day Yakama Resident radiation dose at 300 Decision Area sites with total dose above 15 mrem/yr is provided below.

Site	COPC	Pathway	Fraction of Total Dose	Present-Day Total Dose (mrem/yr)
316-5	Uranium-233/234	Produce Ingestion	0.26	91
	Uranium-238	Produce Ingestion	0.19	
316-2	Uranium-233/234	Produce Ingestion	0.24	57
	Uranium-238	Produce Ingestion	0.21	
	Uranium-238	External Irradiation	0.11	
316-1	Cobalt-60	External Irradiation	0.52	39

The Yakama Resident present-day radiation dose at the 316-5 and 316-2 remediated waste sites is driven by long-lived radionuclides. At the 316-5 and 316-2 remediated waste sites, radiation dose is related to uranium exposure by uptake into produce, and in small part to external irradiation from uranium-238 at the 316-2 waste site. Only at the 316-1 waste site, where the short-lived radionuclide cobalt-60 drives dose, does radiation dose change appreciably with time. By year 2075, the total dose at the 316-1 waste site is 15 mrem/yr and is related to uranium isotopes rather than cobalt-60. As described for Yakama Resident cancer risk, the use of shallow zone soil verification data to calculate risks may contribute to a significant protective bias in the radiation dose results for these sites.

5.8.5.3 Chemical Hazard. Table 5-140 shows the HI results for the Yakama Resident exposure scenario for all remediated waste sites in the 300 Decision Area. The tabulated results are the sum of the risk values for the individual COPCs included in the risk calculation. As described in Section 3.4.4, chemical hazard is calculated separately for adult and child receptors. For the Yakama Resident scenario, child HIs are usually equal to or above those for adults. Therefore, only child HI values are presented here.

The range of child HI results for the Yakama Resident scenario is 0.0032 to 370. No COPCs contributing to chemical hazard were detected at the 300 VTS and 600-259 waste sites. As shown in Table 5-5, 12 of the 17 remediated waste sites in the 300 Decision Area had calculated HI values above 10. There were no sites with an HI value between 1.0 and 10.

A summary of the key exposure pathways and COPCs contributing 10% or more to Yakama Resident child HIs at 300 Decision Area sites with HI values above 1.0 is provided below.

Site	COPC	Pathway	Fraction of Child Hazard	Present-Day Child Hazard [a]
316-1	Copper	Produce Ingestion	0.25	370
	Arsenic	Produce Ingestion	0.19	
	Mercury	Beef Ingestion	0.16	
	Copper	Beef Ingestion	0.12	
316-2	Arsenic	Produce Ingestion	0.54	77
	Uranium	Produce Ingestion	0.17	
300-10	Arsenic	Produce Ingestion	0.94	75
618-12	Arsenic	Produce Ingestion	0.84	58
316-5	Uranium	Produce Ingestion	0.34	57
	Arsenic	Produce Ingestion	0.21	
	Uranium	Soil Ingestion	0.18	
	Uranium	Milk Ingestion	0.16	
300-50	Arsenic	Produce Ingestion	0.64	51
628-4	Arsenic	Produce Ingestion	0.65	31
300-49	Arsenic	Produce Ingestion	0.73	30
618-5	Arsenic	Produce Ingestion	0.93	24
618-4	Arsenic	Produce Ingestion	0.87	19
600-47	Arsenic	Produce Ingestion	0.89	13
300-18	Arsenic	Produce Ingestion	0.94	12

[a] Values for copper and mercury hazard at 316-1 and uranium hazard at 316-5 and 316-2 are likely overestimated; see discussion in the text.

Exposure to copper, arsenic, and mercury via produce and beef ingestion are the most important exposure pathways contributing to the child HI at the 316-1 waste site. Uranium and arsenic exposure via various pathways drives hazard at the 316-5 and 316-2 waste sites. At the remaining sites where child HIs exceed 1.0 arsenic exposure by produce ingestion is the dominant factor. As described in relation to the Subsistence Farmer HI, the oral RfD for copper is related to gastrointestinal irritation, and is also protective of individuals with a rare medical condition that inhibits the regulation of copper absorption following ingestion. The HQ for mercury is based on immune system effects, although renal impacts are also important. The critical target organ of uranium toxicity is the kidney.

Uranium and arsenic are the main contributors to the child HI at all of these sites with the exception of 316-1, where copper, arsenic and mercury are the most important. The representative concentration of arsenic in upland reference area soil, shown in Table 5-47, is 3.2 mg/kg. The 90th percentile arsenic soil background concentrations for the Hanford Area and Yakima Basin are 6.47 mg/kg and 5.13 mg/kg, respectively. As described for Yakama Resident cancer risk, the range of site RME arsenic concentrations at 316-1, 316-2, 300-10, 618-12, 300-50, 628-4, 300-49, and 618-5, where arsenic-related risks exceed the reference areas Yakama Resident arsenic child HQ of 18 (see Table 5-50), range from 3.9 mg/kg (628-4) to 13.5 mg/kg (300-10). As described for Yakama Resident cancer risk, the use of shallow zone soil verification data to assess risks may contribute to a significant protective bias in the HI results for these sites.

As discussed in the Uncertainty Analysis (Section 5.9), the linear models used to estimate tissue concentrations from soil data are prone to overestimation of plant and animal tissue concentrations at higher soil concentrations. A discussion of the potential overestimation of child HIs for these pathways in the 300 Area is provided in the risk assessment for the Subsistence Farmer scenario in the 300 Area (see Section 5.8.3). Child HI results related to uranium at the 316-5 and 316-2 waste sites are also likely to be overestimated due to nonlinearity in uranium uptake from soil into plants.

Because arsenic concentrations at the remediated waste sites where arsenic is a risk driver are up to about twice as high as commonly referenced background levels, risks for COPCs other than arsenic are also of interest for risk management at these sites. A summary of the key exposure pathways and COPCs, other than arsenic, contributing 10% or more to present-day RME Subsistence Farmer child HI at these 300 Area sites is provided in Section 5.8.3. This summary is also applicable to the Yakama Resident scenario.

5.8.6 Risks Related to Lead in Soil for the 300 Area

The highest calculated RME representative concentration for lead in shallow zone soil at any remediated waste site is approximately 210 mg/kg. This concentration applies to the 100-F-37 remediated waste site in the 100-F/100-IU-2/100-IU-6 Area. The highest RME representative concentration for lead in the 300 Area is 99 mg/kg. No representative concentrations for lead at any waste site exceeds the IAROD cleanup level of 350 mg/kg or EPA’s recommended residential screening level for lead in soil of 400 mg/kg, derived using the IEUBK model that has been associated with bare soil in a play area (40 CFR 745). Because soil concentrations for lead are below the most soil screening criteria, no additional evaluation of lead is included in the HHRA. More detailed information on lead toxicity and the basis of EPA’s soil screening criteria is provided in Section 3.5.8.

5.9 UNCERTAINTY ANALYSIS FOR THE LOCAL-AREA RISK ASSESSMENT

The uncertainty analysis provides quantitative and qualitative information for evaluating the level of confidence, including the potential for over- or underestimating risk, in the local-area risk assessment results described in Sections 5.3 through 5.8. The main purpose of the uncertainty analysis is to put the numerical results in perspective with regard to the assumptions and uncertainties in the risk assessment process. Another purpose of the uncertainty analysis is to provide a basis for recommendations relating to additional information that could be of value for refining the risk estimates.

The discussion of uncertainties is organized according to the steps comprising a baseline risk assessment that lead to risk characterization. These steps include Data Collection and Evaluation Exposure Assessment, and Toxicity Assessment. The principal tools applied in the HHRA for quantifying uncertainties in the risk estimates are as follows:

- The use of RME and CTE exposure parameter values in the risk calculations
- The use of RME and CTE estimates of exposure concentrations in the risk calculations
- The use of multiple exposure scenarios to address a range of low- and high-intensity land uses.

The range of exposure parameter values related to behavioral and/or physiological characteristics (i.e., ingestion and inhalation rates, exposure frequency) provides a measure of uncertainty related to the attributes of individual receptors within a receptor population. The use of best estimate and upper-bound exposure concentrations in the CTE and RME calculations, respectively, provides a measure of the importance of uncertainty in the COPC concentrations in exposure media to the risk estimates. The range of the RME and CTE results across the various exposure scenarios provides information on the importance of the different exposure pathways and receptor characteristics on the risk estimates.

The quantitative measures of uncertainty evaluated via the CTE and RME calculations, and the use of multiple exposure scenarios as described here, can only address those aspects of uncertainty that relate to the choice of specific input parameter values and exposure pathways. A semiquantitative or qualitative assessment of uncertainty inherent to these types of assessments is provided for other aspects of the risk assessment that affect the final estimates. These include the following:

- Uncertainty in data collection and evaluation, including analytical data quality and data representativeness (Section 5.9.1)
- Uncertainty in the exposure assessment, particularly the various transport models used in developing the EPCs (Section 5.9.2)
- Uncertainty in the toxicity assessment, particularly the dose-response models underlying the chemical and radionuclide toxicity criteria and extrapolation from effects observed at high dose rates to effects at much lower environmental dose rates (Section 5.9.3).

Both the quantitative and qualitative assessments of uncertainty are directed towards identifying key assumptions and parameters that have the most potential to contribute to significant human health exposures and effects.

A summary of the key uncertainties in the local-area risk assessment results pertaining to soil exposures is provided in Table 5-141. More detailed discussions of each key uncertainty are provided in the following subsections. The "Potential Bias" for each uncertainty refers to the tendency for risk to be either underestimated or overestimated based on the methods used in the risk assessment.

5.9.1 Uncertainties Related to Data Collection and Evaluation

Uncertainty pertaining to data collection and evaluation encompasses sample collection activities, laboratory sample preparation and analysis, and data preparation and analysis. Uncertainty related to laboratory sample preparation and analysis is generally not a significant contributor to overall uncertainty in risk assessment results. A major reason for this is that quality control samples are used to ensure that analytical results are within acceptable levels of precision and accuracy. However, key uncertainties are associated with other aspects of sample collection and data analysis, and these uncertainties are discussed below.

5.9.1.1 Combining Environmental Data from Multiple Sampling Programs. The use of environmental data from a variety of sampling programs introduces uncertainty in the consistency of analytical results over time. This is due to differences among the programs in sample acquisition methods, sample preparation techniques, target analyte lists, and analytical methods from one study to another.

An example of how data have systematically changed with time relates to the cleanup verification data that are a major component of the local-area risk assessment. A condition of the cleanup verification soil data is that the analytical results are targeted to those specific contaminants identified as the likely risk drivers at a particular remediated waste site. Therefore, unlike the data collected in the RCBRA investigation, the particular analytes for which results are available for any individual waste site may be limited. This is particularly true for the liquid waste disposal sites, where there was usually good process knowledge relating to expected contaminants. The initial waste site cleanup verification sampling, such as that conducted in the 100-B/C Area, generally employed a wider range of analytical suites in order to ensure that all important contaminants were identified. As the CVP process was applied to similar sites in other reactor areas, the results from earlier sampling were used to focus analytical suites. Therefore, because of focused target analyte lists that were used for some waste sites, it is possible that some site-related contamination was not captured in the analytical results available for the HHRA and, therefore, total cumulative risks may be underestimated for those sites.

Because data were available from various sampling programs that had different objectives as noted above, all analytical data used in the human health and ecological risk assessments were subjected to a process for ascertaining their usability to support such assessments. All data were required to have, at a minimum, the following attributes in order to be considered usable:

- An analyte name or Chemical Abstracts Service identification number
- A numerical result without a rejected (R) qualifier
- Associated units for the results
- A media type
- Definitive locational information.

Even in cases where all 5 attributes were present, analytical data were at times labeled not usable for 1 or more of 29 reasons for which usability codes have been assigned in the database (see Section 4.4 of Appendix C-1). Some of these reasons include inappropriate analytical method; nonstandard units that cannot be converted; physically infeasible results; and mixed media type, such as paint chips or concrete.

Waste site soil cleanup verification data were subject to QA procedures under specific revisions of the remedial design/remedial action work plans and SAPs for that work, which were generally updated annually. Cleanup verification data received an “A” data usability level, indicating acceptable quality for unrestricted use in the risk assessment. A complete discussion of data usability is provided in Appendix C-2.

The application of the data usability protocol described above improves confidence in the analytical data used in the risk assessment by ensuring that all data used to quantify potential health risks have in common a shared set of attributes. The cleanup verification soil data have been used to support interim remedial decision making, and it has been assumed that the analytical data meet the performance criteria described in the remedial design/remedial action work plans and SAPs governing their collection. However, the cleanup verification data were not evaluated relative to the target practical quantitation limits provided in Section 2.0 of the RCBRA SAP (DOE/RL-2005-42).

5.9.1.2 Spatial Distribution of Contaminants.

Remediated Waste Site Soil Data. The waste site soil verification data were collected for the purpose of determining compliance with shallow zone and deep zone interim RAGs. While these data were often collected using a composite (statistical) sampling method, they have also been collected at many sites as grab (focused) samples specifically from locations where residual soil concentrations would likely be highest. In such situations, although the number of verification samples may be limited, there would be an expected high bias on the estimates of the mean concentrations of soil contaminants if the focused samples were used in the representative concentration calculations. A sampling protocol focused on locations where higher residual contamination might be located also may produce a data set with one or two outlying results. As described in Section 5.2.1, when analytical data from both statistical and focused soil samples were available at a waste site, only the statistical sample results were used to calculate representative concentrations for the risk assessment. This approach emphasizes the use of data that were collected for the purpose of calculating a statistical estimate of the average concentration of a COPC in soil at a remediated waste site.

In the individual CVP and RSVP reports, sample design maps are available that show the locations of the statistical and focused soil verification samples at a remediated waste site. However, the only locational identifiers associated with the verification data used in the risk assessment is whether they were collected in the shallow (0 to 4.6 m [15 ft]) or deep zone (>4.6 m [>15 ft]) of the waste site. The exposure assessment does not associate EPCs with any particular volume of soil. The project database records the sample type (statistical or focused), but not the spatial attributes of the verification data at a waste site (e.g., thickness of any overlying backfill, area over which the data were collected). However, these attributes have been summarized for the remediated waste sites included in the HHRA in Table 2-8.

As described above, when analytical data from both statistical and focused soil samples were available at a remediated waste site, only the statistical sample results for the analyte were used to calculate representative concentrations for the risk assessment. To determine whether this protocol had a significant impact on the results, a second set of supplemental representative concentrations for these waste sites was also calculated using both the focused and statistical samples. The differences between the two sets of representative concentrations for COPCs are presented in Table C3-11 in Appendix C-3. There are 13 remediated waste sites where shallow zone soil data for one or more analytes exist from both statistical and focused samples. In the majority of cases, the supplemental representative concentrations calculated using both focused and statistical samples are below the values from the statistical samples used in the risk assessment. The one exception where values are more than 50% higher than those used in the risk assessment is cesium-137 and lead at the 118-B-3 waste site in the 100-B/C Area. This analysis indicates that the decision to use only statistical samples to represent contaminant concentrations when both focused and statistical samples exist has a relatively minor influence on the risk assessment results for the remediated waste sites.

Upland Surface Soil MIS Data. These data are employed in the local-area risk assessment to represent average COPC concentrations in upland surface soil across the entire River Corridor for the occupational component of the Resident National Monument/Refuge exposure scenario. Uncertainties related to data collection and evaluation for these data are described in the uncertainty analysis for the broad-area risk assessment in Section 4.0 of this report. These data were obtained from 1-ha MIS soil samples at the 10 remediated native soil waste sites and 10 backfilled waste sites sampled for the RCBRA investigation. The principal uncertainty related to their use is the relationship of average soil concentrations from these 20 remediated waste sites to average soil concentrations across the entire upland environment of the River Corridor. Because soils at and adjacent to remediated waste sites are expected to have higher concentrations of Hanford Site-related COPCs than soils in other upland areas, COPC-related risks for the occupational component of the Resident National Monument/Refuge exposure scenario are likely to have a protective bias.

Reference Area Surface Soil MIS Data. The reference site data were used in the COPC identification process and for computing "above reference" cancer risk and radiation dose. The term "reference area" is used in the RCBRA in a manner consistent with its use in *Guidance for Comparing Background and Chemical Concentrations in Soil at CERCLA Sites* (EPA/540/R-01/003), where it is called a "background reference area." The selection of reference areas for the RCBRA was based on a number of factors that included ecological conditions in addition to factors such as soil chemistry and the absence of contamination. Although discrimination based on ecological characteristics is not relevant to the HHRA, the other reference site selection criteria ensure that these sites are applicable as reference areas for both the human health and ecological risk assessments.

MIS data from the 10 upland reference sites were used to compute COPC-specific risks for each of the local-area exposure scenarios evaluated in this section of the RCBRA. Five of the upland reference sites were located within "native" (mostly undisturbed) habitat areas, and the other five upland reference areas were located within uncontaminated borrow pits, which are disturbed areas used to acquire backfill material for remediated waste sites. The reference area samples from uncontaminated borrow pits may have lower concentrations of radionuclides such as cesium-137, strontium-90, and isotopic plutonium that are associated with radioactive atmospheric fallout from historical above-ground atomic weapons testing. These radionuclides are distributed within surface soils, but have near-zero background levels in deeper soils unaffected by atmospheric deposition.

As discussed in Section 5.2.4, MIS soil samples are a form of composite sample with 50 subsamples collected across approximately a 1-ha area. It is reasonable to assume that the variance in analyte concentrations observed among MIS samples would be biased low relative to what might be observed using discrete samples. This indicates a possible protective bias when identifying COPCs collected using other sampling methods. However, waste site soil verification data are also primarily based on composite samples, although generally from fewer subsamples, so this bias may in fact be minimal in this case. The upland reference area samples are also used to calculate representative concentrations. These values are used in calculating risks that are comparable to those for the individual remediated waste sites, but representative of areas that were not impacted by Hanford Site releases.

5.9.1.3 Calculation of Representative Concentrations. As described in Section 3.4, representative concentrations for the CTE and RME calculations are based on the mean, and the 95% UCL on the mean, when there were five or more detected values available to calculate a statistic. When there are between one and four detected values, then either the average or maximum value is used for the representative concentration. A summary of the number of representative concentrations based on one, two, three, or four and more than four detected values for the different environmental media used in the risk assessment is provided in Tables C3-1 through C3-5 in Appendix C-3. These tables can be used to quickly ascertain the number of samples supporting the representative concentrations used in the risk assessment. In general, there is an inverse relationship between the number of detected samples and the uncertainty associated with a representative concentration.

The methods used to calculate representative concentrations distinguish between cases where the number of detected values is below five or is equal to five or more. For small sample sizes, representative concentrations could be expected to differ if samples where an analyte was not detected were accounted for in the calculation. To evaluate this, supplemental risk assessment calculations were performed using the sample-specific reporting limit for chemicals and the method-specific minimum detectable activity for radionuclides for a "U"-qualified (nondetect) result. The primary purpose of this analysis is to determine possible bias in the risk assessment results based on small numbers of detected sample results when there are numerous nondetect data points.

Present-day RME cancer risk, and adult HI, were calculated for the Subsistence Farmer and Industrial/Commercial scenarios at the remediated waste sites using the supplemental representative concentrations described above. A comparison of these results with the results summarized in Sections 5.3 through 5.8 is provided in Table C3-6 in Appendix C-3. Among the 156 sites, there were 7 cases where a RME cancer risk that was at or above 1×10^{-4} in Sections 5.3 through 5.8 and decreased to below that threshold, and 1 case where a RME adult HI was at or above 1.0 in Sections 5.3 through 5.8 and decreased below that threshold, as shown below.

Site Name	Record of Decision Area	Scenario	Endpoint	Section 5.0 Result	Supplemental Representative Concentration Result
132-B-6	100-B/C	Industrial/Commercial	Cancer risk	1E-04	7×10^{-5}
100-K-56:1	100-K	Industrial/Commercial	Cancer risk	1E-04	7×10^{-5}
116-F-14	100-F/100-IU-2/ 100-IU-6	Industrial/Commercial	Cancer risk	2E-04	9×10^{-5}
100-B-8:1	100-B/C	Subsistence Farmer	Cancer risk	1E-04	5×10^{-5}
116-DR-1&2	100-D/100-H	Subsistence Farmer	Cancer risk	1E-04	8×10^{-5}
100-F-19:2	100-F/100-IU-2/ 100-IU-6	Subsistence Farmer	Cancer risk	1E-04	8×10^{-5}
UPR-100-F-2	100-F/100-IU-2/ 100-IU-6	Subsistence Farmer	Cancer risk	2E-04	7×10^{-5}
116-C-2A	100-B/C	Subsistence Farmer	Adult hazard	1.6	0.84

It is also of interest to establish the variability in risk results when using either the Section 5.0 or the supplemental representative concentrations. In terms of the magnitude of difference between the sets of results, the largest change was observed at remediated wste site 128-C-1. Subsistence Farmer RME cancer risk decreased from 1×10^{-5} to 2×10^{-6} at this waste site, a fivefold difference. Altogether, there were only seven waste sites where RME cancer risk or hazard changed by a factor of 2 or more (100-B-8:1, 100-D-48:3, 100-F-19:2, 100-K-55:1, 116-F-1, 128-C-1, and UPR-100-F-2). In all cases, the changes related to use of the supplemental representative concentrations were a decrease in the risk estimate. The treatment of nondetect values results in a protective bias in the risk assessment results at some waste sites, but the overall impact on the risk assessment conclusions is minor because the magnitude of the changes is relatively small.

When there are five or more detected values, the median of three separate estimates of the mean and UCL, based on different statistical methods, was used as the CTE and RME concentration, respectively. A summary of the number of representative concentrations calculated by the different methods is provided in Tables C3-7 and C3-8 in Appendix C-3. The variability between the results of the UCL methods and the relationship of the statistical UCL to the maximum detected concentration is provided in Tables C3-9 and C3-10 in Appendix C-3. There were no occurrences where the maximum UCL was more than five times the median. At 21 remediated waste sites, there was a COPC where the maximum UCL was between two and five times the median UCL. Europium-152 and cesium-137 were the main COPCs where median and maximum UCL values differed by this much. Table C3-10 shows that it is primarily polycyclic aromatic hydrocarbons (PAH) in upland surface soil where the maximum detected concentration exceeds the RME UCL value by more than a factor of 10. There were also two remediated waste sites in the 100-B/C Area (100-B-8:1 and 100-B-8:2) where some metals and radionuclides exceed the RME UCL value by more than a factor of 5. These analyses indicate that the selection of UCL method can have a moderate influence on the results of the risk assessment, but only at about 15% of the remediated waste sites evaluated in this report. Additionally, substitution of the maximum detected concentration for the UCL when there are five or more detected values would primarily impact the results based on upland surface soil, but would have minimal impact on the calculations for the remediated waste sites with the possible exceptions of 100-B-8:1 and 100-B-8:2.

Calculated Uranium Metal Results. In many of the cleanup verification soil samples, data were obtained for isotopic uranium but not for uranium as a metal. Therefore, the isotopic results (in units of activity per mass) were converted to total uranium data (in units of mass uranium per mass of sample). In this way, the effects of uranium metal as a kidney toxicant were assessed in addition to evaluation of radiation dose and cancer risk for isotopic uranium data. The results for calculated total uranium may vary slightly from measured results for inorganic uranium due to differences in the analytical methods, but these differences are not expected to significantly affect noncancer hazard results.

5.9.2 Uncertainties Related to the Exposure Assessment

There are inherent uncertainties in the application of hypothetical exposure scenarios to the as-yet unknown future conditions of the Hanford Site. To an extent, these uncertainties are addressed by the use of multiple exposure scenarios in the broad-area and local-area assessments to provide a range of potential exposure intensities in support of remedial decision making. In the context of the Uncertainty Analysis, the exposure scenarios are viewed in a relative sense and organized from low-intensity to high-intensity alternatives as shown below.

Exposure Scenario	Intensity of Exposure
Casual User, Industrial/Commercial	Low

Exposure Scenario	Intensity of Exposure
Resident National Monument/Refuge	Medium
Avid Hunting and Fishing, Nonresident Tribal	Medium
Residential	High

Two factors bear upon the likelihood of any particular exposure scenario being realized for future receptors in the River Corridor. One factor involves existing land-use planning decisions and laws governing future uses. Documents and laws relating to the reasonably assumed future land use in the River Corridor include the Hanford Site comprehensive land-use plan (DOE/EIS-0222-F, *Hanford Comprehensive Land-Use Plan Environmental Impact Statement)* and the Hanford Reach National Monument designation. A second factor related to the likelihood that a land-use scenario will be realized in the future, in the sense that all exposure pathways are complete over a chronic exposure period, is the number and complexity of exposure pathways. For example, the Subsistence Farmer scenario envisions receptors with a domestic water well, an active home garden and orchard, a poultry enclosure, cattle that are used for both meat and milk, and additional dietary contribution from fishing. By contrast, in the Industrial/Commercial scenario, the exposure pathways (soil contact, dust inhalation, and external radiation) are all complete if a receptor is present and are not dependent on any specific activities conducted by the receptor.

One aspect of the uncertainty in whether any particular exposure scenario may be realized in the future is a related question of *when* any scenario may be realized. This has special relevance in the assessment of cancer risk and radiation dose for short-lived radionuclides. At a number of remediated waste sites, there is a significant reduction in estimated cancer risk and dose over the next 70 to 140 years due to natural radioactive decay. These changes are described in detail in Sections 5.3 through 5.8.

5.9.2.1 Identification of Contaminants of Potential Concern. Section 3.2 of this assessment describes the process that was developed to identify COPCs in different environmental media that are related to Hanford Site operations. The results of this process for the cleanup verification data are described in Section 5.2. Key factors related to soil COPC identification include the magnitude and significance of the results of statistical comparisons with background data, process knowledge, and comparison of results among different media and for similar analytes. Because the COPC identification process was systematic, it is unlikely that Hanford Site-related analytes that could contribute to significant health risks were eliminated in this process. The use of data from MIS samples in the Upland Reference Area may contribute to a protective bias in COPC identification for the local-area risk assessment, as discussed in Section 5.9.1.

5.9.2.2 Values of Behavioral Variables. Within each of the exposure scenarios, there is uncertainty in the representative concentrations of contaminants in environmental media and in the exposure parameter values that is assessed by the calculation of CTE and RME risks. For any given exposure scenario, the CTE risks represent a hypothetical individual with approximately best-estimate levels of contact with contaminants in the exposure media across the various exposure pathways. The RME risk calculations represent a hypothetical individual with a RME condition of contact with contaminants in the exposure media.

Differences between CTE and RME risks vary as a function of exposure scenario and receptor depending on the magnitude of the differences between CTE and RME exposure parameter values and the type and number of such parameters included in the exposure equation. Differences between CTE and RME calculations are also related to the magnitude of differences between the CTE and RME EPC. The range between CTE and RME risk results was generally about a factor of 10 or less for calculations relating to the soil source term.

5.9.2.3 Applicability of Data From Excavated and Backfilled Waste Sites. At backfilled waste sites, shallow zone soil data reflect residual contamination in soils adjacent to, or below, a volume of excavated soil that has been replaced with clean backfill. The residual contamination on the sidewalls of such an excavation has been colloquially referred to as a "bathtub ring," reflecting an assumption that the cleanup verification data collected from these sidewalls represents a relatively thin layer of affected soils enclosing the clean backfill. Residual contamination is also generally expected in the vadose zone beneath a site, particularly if the site was associated with disposal of liquid wastes. In principle, residual contamination levels in the cleanup verification samples should be below interim cleanup levels that were applied to shallow zone (<4.6 m [<15 ft] bgs) soils and deep zone (>4.6 m [>15 ft]) soils.

Because shallow zone data from backfilled sites are related to subsurface soils with the configuration described above, these soils are not available for direct human contact unless they are first excavated and relocated to the ground surface. Two separate evaluations were performed to assess the possible degree of protective bias from using these data in the risk assessment. First, soil concentrations and risk assessment results were compared for the subset of remediated waste sites where both cleanup verification and RCBRA 0- to 15.2-cm (0- to 6-in.) surface soil (MIS) samples are available. Second, lateral migration of dissolved contaminants from standing water that may have been present during operations of a liquid waste disposal site was modeled to assess the possible extent of the "bathtub ring." These evaluations are documented in Appendix D1-6 and D1-7 of Volume II of the RCBRA.

Comparisons of representative concentrations for shallow zone and RCBRA surface soil show that there is relatively little difference at the seven remediated native soil waste sites where both data types are available. From these results one may conclude that differences in sampling and analysis methods between cleanup verification and MIS data sets had little influence on COPC concentrations and risk results. Unlike at the remediated native soil waste sites, significant differences in both soil concentrations and risk assessment results were observed at some of the 10 backfilled waste sites where both data types are available. The COPC concentrations, and especially concentrations of target analytes related to a particular waste site by process knowledge, were consistently higher in the cleanup verification data than in the MIS data. Depending on the magnitude of the concentrations and the inherent toxicity of the COPC, these concentration differences sometimes resulted in large differences in risk estimates.

The comparisons of shallow zone soil and RCBRA surface soil COPC concentrations and risk estimates confirm that there may be a strong conservative bias when using cleanup verification data to estimate EPCs in surface soil. The actual degree of bias at any particular excavated waste site will depend on several factors including the depth of the excavation, the lateral distance that contaminants may have migrated, assumptions regarding whether and how vertical soil mixing will occur during any future site development, and the applicability of the 1-ha surface soil exposure concentration to anticipated exposure activities.

In addition to the comparisons discussed above, vadose zone simulations were conducted to assess the potential lateral migration of dissolved contaminants from standing water that may have been present during operations of a liquid waste disposal site. These simulations address the question of the potential thickness of contaminated soil behind the sidewalls of an excavation. Simulations were conducted assuming, conservatively, that contaminant migration in soil favors lateral dispersion over vertical dispersion. The results indicate that, under conditions of constant standing water up to the top of a structure, contaminants in water could move laterally as far as 15 m (49.2 ft) near the top of the structure and as far as 20 to 25 m (65.6 to 82.0 ft) at a depth of about 5 m (16.4 ft) below the top of the structure. Because of the large volume of liquid moving through the soil, there would be little retardation of contaminants even for those with soil-water distribution coefficients of 50 mL/g or more.

These simulation results are meaningful for judging the plausibility of assuming chronic exposure to soils characterized by the cleanup verification shallow zone soil data at an excavated liquid waste disposal waste site. If the excavation depth was shallow, and if standing water existed in a structure and was near the ground surface, it is possible that contaminated soil exists relatively close to the ground surface adjacent to an excavated site.

5.9.2.4 Modeled Concentrations of Contaminants of Potential Concern in Foods. Another aspect of uncertainty related to the exposure assessment involves the models used to estimate EPC in unsampled exposure media. A variety of simple analytical models were employed in the risk assessment calculations. These include models to estimate EPCs in foodstuffs for the Residential scenarios (including plants, poultry and eggs, beef, and milk) and EPCs in ambient air modeled from measured concentrations in soil.

As discussed in Section 3.4, both plant-soil concentration ratios (K_{p-s}) and feed-to-tissue transfer factors for meat, milk, and eggs (B_a) reflect an assumption that there is a linear and unchanging relationship between soil (or feed) and tissue concentrations. Because this assumption ignores physiological mechanisms controlling the accumulation of toxic substances including metals and radionuclides, tissue concentrations of these COPCs are susceptible to overestimation when soil concentrations are elevated. At soil concentrations that are near naturally occurring levels, variability in plant uptake related to soil conditions, plant species and tissue type, harvest time, and other variables contribute to a high level of intrinsic variability in K_{p-s} values (Sheppard 2005). However, as soil concentrations increase, the influence of nonlinearity between plant and soil concentrations becomes more and more important (Sheppard 2005). Situations where risks may have been overestimated due to overprediction of plant tissue concentrations are identified in the text of Sections 5.3 through 5.8. The embedded tables in these report sections that show COPCs and pathways contributing greater than 10% to total site risk are also footnoted when risks by a plant-mediated pathway (plant, beef, or milk ingestion) may be overestimated.

To assess the magnitude of the likely overestimation of risk when soil concentrations are well above natural levels in soil, statistical regression models that predict plant tissue concentrations as a variable function of soil concentration were used in Sections 5.3 through 5.8 of the risk assessment. As an example, a Subsistence Farmer HI result of 220 was calculated at the 100-K-30 remediated waste site (see Section 5.4.3) using the linear K_{p-s} for produce and grasses. When the HI was recalculated based on the Bechtel regression for mercury (BJC/OR-133), an HI of 12 was estimated. Because the ratio of site (17.5 mg/kg) and reference area (0.022 mg/kg) concentrations are so large for mercury at 100-K-30, this is an example of a worst-case overestimation.

Uncertainty in plant-soil ratios can best be appreciated by comparing recommended values from a variety of references. Table 5-142 provides a compilation of K_{p-s} values for 26 metals from 7 publications, plus calculated ratios based on regression analysis of RCBRA upland environment soil and plant tissue data. The publications were selected because they are either published by a governmental or international agency or have been widely cited by the same. The K_{p-s} values used in the HHRA are those employed in Version 6.5 of the RESRAD computer code (http://web.ead.anl.gov/resrad/home2/). The RESRAD code has historically been used at the Hanford Site for performing radiation dose assessments and is the model that was employed to develop the residential IAROD radionuclide cleanup levels used to support interim closure of the remediated waste sites evaluated in this report. As described below, some paired data for plant and soil contaminant concentrations were collected under the RCBRA. However, these samples do not pertain to edible plant tissues. Also, the soil and plant sampling occurred over a 1-ha area and therefore provides only a spatially averaged measure of the relationship between concentrations in these two media. For these reasons, literature K_{p-s} values were used rather than values developed from these data pairs.

The published K_{p-s} values shown in Table 5-142 encompass different tissue types, and in two instances include both linear K_{p-s} values and a regression model for calculating a K_{p-s} value from a soil concentration. For the published regression models, an equivalent K_{p-s} value was calculated using the 90th percentile Hanford Site background value from *Hanford Site Background: Part 1, Soil Background for Nonradioactive Analytes* (DOE/RL-92-24). The ratio of the maximum and minimum K_{p-s} value among the values shown in Table 5-142 exceeds 1,000 for antimony, silver, technetium, and uranium. This same ratio is 100 or more for arsenic, cesium, chromium, mercury, and vanadium. The ratio of the maximum and minimum K_{p-s} value is below 10 only for europium and tin.

Arsenic and mercury were the most prominent risk drivers for plant-related residential scenario exposure pathways for cancer risk and chemical hazard, respectively. The high degree of uncertainty in the K_{p-s} values for these COPCs is therefore of great significance for the risk results in the three Residential scenarios. For arsenic, the K_{p-s} value of 0.53 from the RESRAD code was higher than all other values in Table 5-142 excepting the RCBRA paired data (1.1) and the value for root tissue from the *Development of Terrestrial Exposure and Bioaccumulation Information for the Army Risk Assessment Modeling System* (1.125) (USCHPPM 2004). For mercury, the K_{p-s} value of 2.5 from the RESRAD code is higher than any other tabulated value.

Because the plant ingestion exposure pathway is the key contributor to potential human health cancer risk and chemical hazard above threshold criteria for the Subsistence Farmer scenario, a sensitivity analysis was conducted for 10 remediated waste sites focusing on two key variables: K_{p-s} values and the ingestion rates of home-grown fruits and vegetables. The range of plant ingestion pathway cancer risk and chemical hazard were both approximately four orders of magnitude at most waste sites. Uncertainty in K_{p-s} values and home-grown fruit and vegetable ingestion rates contributed about equally to this range. Subsistence Farmer plant ingestion results described in Sections 5.3 through 5.8 are near the upper end of the range of high-end and low-end cases from the sensitivity analysis, especially with respect to cancer risk. Arsenic and isotopes of uranium are major contributors to high-end sensitivity analysis cancer risk results at the selected sites, and arsenic and mercury are major contributors for chemical hazard. The largest difference between remediated waste site and reference area soil concentrations for arsenic is about a factor of 5, which is small compared to a factor of 190 for published arsenic K_{p-s} values and factors of 50 to 100 for fruit and vegetable ingestion rates. However, differences of up to about a factor of 500 between remediated waste site and reference area soil concentrations of uranium isotopes (316-5) and mercury (100-K-30) are larger than the degree of variability in fruit and vegetable ingestion rates and sometimes larger than the range of published K_{p-s} values. This analysis indicates that Subsistence Farmer plant ingestion risks are equally sensitive to variability in potential plant concentrations and home-grown produce ingestion rates. In some cases, the magnitude of the increment of remediated waste site soil concentrations above reference area levels can be equal to or greater than the magnitude of this variability. This sensitivity analysis is documented in Section 1.4 of Appendix D-1.

The EPCs for plants were modeled from surface soil data rather than calculated from measured plant tissue values for two reasons. First, the RCBRA plant tissue data are related to shrubs and grasses that may have little relevance to the edible tissues of garden crops. Secondly, detection limits for certain COPCs are elevated in the available plant tissue data. Table 5-143 provides a comparison of measured and modeled EPCs for upland COPCs where there are one or more detected values for both upland surface soil and plants.

Among these COPCs, dieldrin would be the principal contributor to cancer risk and HI using the measured plant data in place of the modeled values. The organic chemical K_{p-s} values used to calculate plant EPCs in the HHRA were obtained from *Human Health Risk Assessment Protocol for Hazardous Waste Combustion Facilities* (EPA/530-R-05-006). For dieldrin, this value is 0.097 (dry weight). Other published values for a dieldrin K_{p-s} are 0.175 from *Review and Revision of Bioaccumulation Models Used to Calculate Ecological Screening Levels* (LA-UR-02-0487) and 0.41 from *Ecological Soil Screening Levels for Dieldrin: Interim Final* (OSWER 9285.7-56). Using the upland soil RME representative concentration of 0.0012 mg/kg and an assumed plant moisture content of 85%, the following range of predicted wet weight dieldrin plant concentrations are obtained: RCBRA model (1.7E-05 mg/kg), LA-UR-02-0487 (3.2E-05 mg/kg), and OSWER 9285.7-56 (7.4E-05 mg/kg). These modeled estimates of dieldrin in plants are relatively consistent, and are all well below the four detects in upland plants of 0.0094, 0.04, 0.16, and 0.34 mg/kg (wet weight). One reason for high detected pesticide values in RCBRA plants may be differences in the plant species sampled for the regression analyses underlying the $K_{p\text{-}s}$ values and those sampled in the RCBRA investigation. It is possible that the sample dilutions, as well as the unrealistically high measured values of pesticides in plants relative to the soil data, may be related to secondary plant chemicals that occur naturally in the sampled tissues that could co-elute with the pesticides during gas chromatography analysis for pesticides.

The models used to estimate organic chemical COPC concentrations in foodstuffs from soil concentrations are also subject to uncertainty. The octanol-water partition coefficients used to model $K_{p\text{-}s}$ and B_a values for organic chemicals are known to be imprecise, and the regression models in which they are employed may also over- or underestimate these values for particular organic chemicals. For all COPCs, the foodstuff exposure models assume that there is an adequate area of contaminated soil within which produce and forage crops can be raised. However, the models employed to calculate representative concentrations in soils at the remediated waste sites are not spatially explicit. The available verification data were used to calculate EPCs in foodstuffs regardless of the presence of backfill or the area within which the verification data were collected. With respect to surface area, the RESRAD computer code incorporates the assumption that a contaminated area of at least 20,000 m^2 (approximately 5 ac) is required in order to support the assumption that 100% of beef and milk production is associated with a particular site (ANL/EAD-4 2001, *User's Manual for RESRAD Version 6.0*, Appendix D.2.1.2). When the size of individual remediated waste sites is much smaller than this, risk estimates related to beef and milk ingestion will be overestimated.

Among the transfer factors used in this assessment, those for chicken and eggs are particularly susceptible to a high degree of uncertainty. As noted in Appendix D-1.4.1, many of the poultry transfer factors for metals and radionuclides published in *Residual Radioactive Contamination from Decommissioning – Technical Basis for Translating Contamination Levels to Annual Total Effective Dose Equivalent: Final Report* (NUREG/CR-5512-V1) that were used in this assessment derive from data published in *Transfer Coefficients for Assessing the Dose from Radionuclides in Meat and Eggs* (NUREG/CR-2976). However, transfer factors for various elements were not published in NUREG/CR-2976. In these cases, NUREG/CR-5512-V1 simply borrowed the transfer factors for chemically similar elements. The validity of applying the transfer factors across elements was not established. Even when there were element-specific data available, there were generally very few observations for chicken meat or eggs for each element (NUREG/CR-2976). A comparison of values from NUREG/CR-5512-V1 with the transfer factors for cadmium and zinc described in EPA/530-R-05-006 provides a measure of the degree of uncertainty in these values. The EPA/530-R-05-006 values of the chicken transfer factors for cadmium and zinc are smaller than those published in NUREG/CR-5512-V1 by factors of approximately 8 and 700, respectively.

The relative bioavailability of specific metals may be expected to vary between soil and feed, so that the application of the linear and unchanging feed-to-tissue factors to the soil-to-tissue pathway in this assessment is a source of still more uncertainty (and potentially, of protective bias) in the modeled tissue concentrations of contaminants. The relative bioavailability of COPCs would generally be expected to be lower from soil where they may be present in stable mineral complexes or strongly adsorbed to soil particulates. In Section 4.0 of this report, modeled tissue concentrations in beef and wild game are evaluated using the tissue data collected under the Surface Environmental Surveillance Project. Unfortunately, the available tissue data from the Surface Environmental Surveillance Project research are limited with respect to performing this validation, and the key COPCs contributing to risk by modeling uptake from soil to meat are not represented.

5.9.2.5 Lead Exposure Via Foods. Potential risks related to lead exposure were screened by comparing soil lead concentrations to EPA's 400 mg/kg soil lead criterion for bare soil in play areas (40 CFR 745). Exposure to lead may also be of concern due to uptake of soil lead by plants and animals that are used as sources of food. Site-related lead exposure from foods can be evaluated using the IEUBK blood lead model described in Section 3.5.8. Because of the problem of overestimating tissue concentrations when using linear plant-soil concentration ratios, this modeling is ideally conducted using measured lead tissue concentrations from produce and animals raised on affected soils, which are unavailable.

5.9.2.6 Modeling of Ambient Dust Concentrations in Air. The particle emission fraction model (OSWER Directive 9355.4-24, *Supplemental Guidance for Developing Soil Screening Levels for Superfund Sites*) was used to estimate dust concentrations in ambient air related to surface soils. This exposure pathway was rarely identified as a significant contributor to potential health effects. Being a protective screening-level model, it is unlikely that estimated dust concentrations calculated with this model would be biased low. One uncertainty with this model is the appropriate value for the vegetation fraction parameter. Because the model is applied to varying conditions (e.g., cattle pen, garden, residential landscaping), it is possible that the assumed value of 0.3 may underestimate dust emissions for specific areas. Decreasing the value of this parameter from 0.3 to 0.1 results in an increase in ambient dust concentrations of about 20%, which would not have an appreciable effect on the results for this exposure pathway.

5.9.2.7 Groundwater Contribution to Contamination in Agricultural Products. Although plant and animal tissue concentrations of COPCs are susceptible to overestimation when soil concentrations are elevated, exposure via these agricultural pathways may also be subject to underestimation because only uptake of soil COPCs is evaluated. The COPCs in groundwater may also impact agricultural exposures due to the use of groundwater for irrigation and for watering livestock. The relationship between soil and plant COPC concentrations and the use of contaminated groundwater is a complex one because irrigation with groundwater will also leach soluble radionuclides present in soil.

The RESRAD simulations used to evaluate breakthrough of soluble COPCs in soil to groundwater over time are documented in Appendix D. These simulations can also be used to assess the relative contributions of drinking water ingestion and agricultural uses to total exposure. The following relative concentrations in drinking water, vegetables, beef and milk were calculated from a uniform concentration of uranium in shallow zone soil following 500 years of irrigation.

Exposure Medium	Concentration
Drinking water	1 pCi/L
Nonleafy vegetable	1.3 pCi/kg
Leafy vegetable	6.3 pCi/kg
Beef	0.18 pCi/kg
Milk	0.32 pCi/kg

In this case, daily ingestion of 1 to 2 L of water will likely dominate uranium exposure relative to the agricultural pathways, although the latter will not be insignificant. The relative concentrations in these media will naturally vary among the different COPCs, and the importance of groundwater contributions to total exposure will also vary based on the relative concentrations of different COPCs in groundwater and soil. The RESRAD modeling for determining groundwater breakthrough due to soil leaching was based on the RESRAD models developed for the 100 Area and 300 Area and documented in Appendix B of the remedial design report/remedial action work plan documents (DOE/RL-96-17, *Remedial Design Report/Remedial Action Work Plan for the 100 Area*; DOE/RL-2001-47, *Remedial Design Report/Remedial Action Work Plan for the 300 Area*). This RESRAD model has a number of protective biases, such as assuming several centuries of continuous irrigation and assuming a uniformly contaminated volume of soil of 6 m (19.7 ft) thickness over an area of 10,000 m^2 (107,600 ft^2).

5.9.3 Uncertainties Related to the Toxicity Assessment

5.9.3.1 Updating of Toxicity Criteria. Toxicity criteria for chemical COPCs are continually revised by EPA and state agencies. Therefore, risk assessment calculations based on these criteria may become outdated over time as our understanding of chemical toxicity improves. The chemical toxicity criteria for this risk assessment were retrieved in the spring and summer of 2009. As this report was finalized, toxicity criteria published by EPA in December 2009 in support of their regional screening levels were reviewed (http://epa-prgs.ornl.gov/radionuclides/) to identify newly published values. For the cleanup verification soil COPCs listed in Table 5-26, revised toxicity criteria were discovered only for hexavalent chromium and cobalt. A new oral cancer slope factor (CSF) for hexavalent chromium is now published in the December 2009 Regional Screening Levels. The cobalt inhalation unit risk Provisional Peer Reviewed Toxicity Value has changed from 0.0028/$\mu g/m^3$ to 0.009/$\mu g/m^3$. Because cobalt risk via inhalation exposure is a negligible contributor to calculated risks in the local-area risk assessment, this threefold change in the Provisional Peer Reviewed Toxicity Value will not affect the results of this assessment.

The new oral CSF for hexavalent chromium could potentially have an impact on the results of the local-area risk assessment for soil at some remediated waste sites, but the largest impact is likely to be in the results of the groundwater risk assessment (Section 6.0) because hexavalent chromium is a major groundwater plume contaminant. An oral CSF for hexavalent chromium has been independently derived by the states of New Jersey and California in 2009 based on the results of a 2-year rodent study performed by the National Toxicology Program, which is an interagency program within the Public Health Service of the Department of Health and Human Services. If implemented, evaluations of hexavalent chromium would likely show an increased risk than presently esmitated.

5.9.3.2 Cancer Risk and Radiation Dose. General sources of uncertainty pertaining to the assessment of carcinogenicity include (1) high-to-low dose extrapolation, (2) uncertainty in the applicability of the no-threshold model of carcinogenicity, (3) the common use of a UCL (typically 95%) on the slope of the dose-response curve for chemical CSFs, and (4) uncertainty in whether a particular chemical is in fact a human carcinogen. As discussed in Section 3.5.3, most chemical CSFs are based on carcinogenic effects observed at relatively high dose rates in test animals that have been extrapolated to lower dose rates in humans. The human data on which radionuclide CSFs are based are also based on high-to-low-dose extrapolation. The underlying assumption for both chemicals and radionuclides is that even a very low level of exposure carries some risk of carcinogenesis.

Uncertainties in extrapolating carcinogenic response from high-to-low dose rates and in the no-threshold model of carcinogenicity are related. The human body has the capability to repair DNA damage that may lead to cancer, and such repair is constantly under way due to damage that is naturally incurred independent of any exposure to environmental contaminants. There are also a variety of cellular mechanisms for halting the proliferation of cells that, due to DNA damage, have developed characteristics of uncontrolled growth. At the low dose rates common to environmental exposures, it becomes impossible in either human or test animal populations to observe the slight potential increase in cancer incidence that may be related to exposure. As discussed in Section 3.5.3, there are some chemical carcinogens that do not directly act on DNA for which the no-threshold cancer model is generally understood to be inaccurate. At low dose rates, it is unlikely that a carcinogenic response would be invoked by such chemicals. Scientific positions on the validity of calculating cancer risks at low levels of exposure vary.

Cancer risks for each exposure route and COPC (radionuclides and chemicals) are summed to calculate cancer risk to an individual. This methodology is consistent with CERCLA guidance in *Establishment of Cleanup Levels for CERCLA Sites with Radioactive Contamination* (OSWER Directive 9200.4-18). The EPA reiterated their recommendation for summing chemical and radionuclide cancer risks in a 1999 memorandum titled “Distribution of OSWER Radiation Risk Assessment Q&A’s Final Guidance” (EPA 1999). Separate sums of cancer risk for radionuclides and chemicals are provided in the electronic results files in Appendix D-5, but summary tables in the main report provide cancer risk results for the sum of all COPCs.

An important distinction between chemical and radionuclide CSFs is that chemical CSFs are commonly calculated as the upper 95% confidence limit on the slope of the dose-response curve, and radionuclide CSFs reflect an average estimate of the lifetime risk of cancer. Although chemical and radionuclide cancer risk estimates have been summed in this assessment, the intentional bias associated with chemical CSFs does not strictly allow for simple summation of chemical and radionuclide cancer risks. Additionally, many chemical CSFs are based on animal studies and therefore incorporate uncertainties that do not pertain to radionuclide CSFs, which are based on human epidemiological studies. Ultimately, chemical and radiological CSFs both involve a similarly high degree of uncertainty related to extrapolation of effects at high doses to hypothetical effects at low (environmental) dose rates. Also, EPA is moving towards a mode-of-action approach for evaluating chemical carcinogenicity that stresses whether a chemical is carcinogenic by mutagenic or nonmutagenic processes (EPA/630/P-03/001F, *Guidelines for Carcinogen Risk Assessment*). From that perspective, additivity of cancer risk for mutagenic chemicals and radionuclides may be more defensible in principle than additivity of risk among chemicals that operate by mutagenic and nonmutagenic modes of action.

There are numerous instances where chemicals and radionuclides both contribute significantly to a cancer risk result. The most common occurrence is the summing of cancer risks for arsenic and radionuclides. Although the different bases of the chemical and radionuclide slope factors makes the general summation of these cancer risks uncertain, arsenic is (like ionizing radiation) a known human carcinogen and has a slope factor based on human epidemiological data. Therefore, uncertainty introduced by the addition of chemical (arsenic) and radionuclide cancer risks is not as large in this assessment as might more generally be the case for other chemicals.

As discussed in Section 3.5.3, *Supplemental Guidance for Assessing Susceptibility from Early-Life Exposure to Carcinogens* (EPA/630/R-03/003F) contains recently published guidance for adjusting cancer potency estimates for childhood exposures to mutagenic carcinogens. Among the 12 chemicals listed in Table 1b of EPA/630/R-03/003F that EPA calls out as having been identified as mutagens, the 4 PAHs (benzo(a)pyrene, dibenzanthracene, dimethylbenz(a)anthracene, and 3-methylcholanthrene) are of particular relevance in this assessment because carcinogenic PAHs are among the COPCs addressed in the HHRA. A quantitative adjustment of the CSFs for carcinogenic PAHs was performed for child receptors in this assessment. The adjustment factors for CSFs described in EPA/630/R-03/003F are 10 for ages 0 through 2 years, and 3 for ages 2 through 16 years. Over a 30 year residential RME exposure duration, these changes amount to an increase in estimated lifetime cancer risk of approximately 2.5 times.

The 2005 EPA guidance related to childhood exposure to mutagens (EPA/630/R-03/003F) is contained in a companion document to *Guidelines for Carcinogen Risk Assessment* (EPA/630/P-03/001F), also published in 2005. In these guidelines, EPA distinguishes between mutagenic and nonmutagenic carcinogens. Although CSFs for both types of carcinogens are currently developed using a no-threshold model of dose-response (see Section 3.5.3), EPA/630/P-03/001F stresses the importance of differentiating these mechanistically-different types of carcinogens because there is likely to be a threshold dose for the latter. Hence, a complete implementation of EPA's 2005 cancer risk assessment guidelines may have the effect of reducing the estimated cancer incidence risk for chemicals with laboratory evidence of carcinogenicity that are not mutagens.

With respect to radionuclides, uncertainties may relate to the estimation of radiation dose as well as to the assessment of carcinogenic risk associated with any particular dose. As described in Section 3.5.5, age-dependent internal dose coefficients published in *Age-Dependent Doses to Members of the Public from Intake of Radionuclides: Part 5, Compilation of Ingestion and Inhalation Dose Coefficients* (ICRP 72) were used for the radiation dose calculations. For the

Residential, exposure scenarios that include both child and adult receptors, age-dependent dose coefficients that apply to a 30-year exposure period were calculated because the dose threshold (15 mrem/yr) used in the local-area risk assessment also pertains to a 30-year period. However, when calculating radiation dose for a child receptor of ages 1 to 6 years, these dose conversion factors (DCFs) will tend to underestimate the actual dose, because effective dose is usually lower for an adult than a child. A discussion of the calculation of age-dependent DCFs, including a comparison of DCFs for specific age classes to the 30-year exposure DCFs used in this assessment, is provided in Appendix D-3.

5.9.3.3 Systemic Toxicity. General sources of uncertainty pertaining to assessment of systemic toxicity include (1) the application of uncertainty factors on the dose-response data, (2) relying on a single critical effect to measure toxicity, and (3) toxicological interactions among the various COPCs. Uncertainty factors are used to account for several possible sources of uncertainty in developing an RfD including extrapolating from the no observable adverse effect level (NOAEL) or lowest observable adverse effect level (LOAEL) to a chronic RfD, variability in sensitivity in the human population, interspecies variability between humans and test animals, and inadequate dosing periods in a critical study. These uncertainty factors (and related modifying factors) are designed to introduce a protective bias in the toxicity criteria such that the potential for adverse effects in sensitive human subpopulations will not be underestimated. There is also considerable uncertainty, and usually a protective bias, in the screening-level practice adopted in this assessment of summing HQs across chemicals to estimate an HI.

Some of the metal COPCs for which chemical hazard is quantified in the risk assessment (i.e., copper, total chromium, and zinc) are also essential micronutrients required by the body for normal functioning. The body normally exerts a degree of homeostatic control over the body burdens of these metals following ingestion exposures, meaning that uptake is regulated depending on need. In order for uptake followed by toxicity to occur, the body’s control mechanisms on uptake must be overwhelmed or incapacitated. Thus, chronic toxicity calculated as HQs below or only a few times higher than 1.0 for exposure to these metals is probably unrealistic. Specific examples are discussed in relation to the risk assessment results presented in Sections 5.3 through 5.8.

As described in Section 3.4.4, each chemical RfD value has associated uncertainty and modifying factors, used as protective multipliers on the observed dose-response data to account for various uncertainties when applying these data to the human population. The extent to which uncertainty and modifying factors in the chemical RfD values impact the results of the risk assessment is chemical-specific. Uncertainty and modifying factors in oral RfDs for some of the different chemicals for which significant chemical hazard was observed are as follows:

Contaminant of Potential Concern	Uncertainty Factor	Modifying Factor
Arsenic	3	1
Copper	1	1
Mercury	1,000	1
PCB Aroclors	300	1

Contaminant of Potential Concern	Uncertainty Factor	Modifying Factor
Uranium	100	1

Calculated HQ values for mercury, uranium, and PCB Aroclors are, therefore, associated with a much higher degree of protective bias than those for copper and arsenic. In this assessment, HQs for individual COPCs were protectively summed to generate an HI for all chemicals (see Section 3.6.2). There are some instances where two or more chemicals contributed in a roughly equal manner to an HI substantially above 1.0. In these circumstances, the potential additivity of the HQ values for each chemical is discussed in Sections 5.3 through 5.8. In some instances, chemical hazard has likely been overestimated by perhaps a factor of 2 to 3 by the summation of chemical-specific HQ values for chemicals whose toxic effects are manifested in different organ systems.

5.10 REFERENCES

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6.0 SCREENING-LEVEL GROUNDWATER RISK ASSESSMENT

6.1 INTRODUCTION

This groundwater screening-level risk assessment presents an initial evaluation of potential risks associated with groundwater exposures for each of the groundwater operable units (OUs) within the River Corridor. Potential risks are characterized in the risk assessment by the use of groundwater monitoring well data. The data were obtained from a selection of wells representative of groundwater conditions between 1998 and 2008. This assessment is considered to be screening level because the representative concentrations used in the calculations may not adequately represent present-day exposure concentrations in particular subareas of a groundwater OU.

The overall purpose of the groundwater screening-level risk assessment is to provide risk managers with baseline risk information that can be used in conjunction with additional groundwater evaluations that will be completed as part of the remedial investigation/feasibility study (RI/FS) process to support final remedial action decisions for groundwater for the River Corridor record of decision (ROD) decision areas. To accomplish this overall purpose, the groundwater screening-level assessment provided in this section of the human health risk assessment (HHRA) has the following objectives:

- Provide risk managers with estimates of baseline risks for each groundwater OU and exposure scenario. These risks, with consideration of the associated uncertainties, may be compared to the results of the risk assessment related to soil exposures at the individual remediated waste sites (Section 5.0) in order to establish the relative magnitude of risks for these two exposure media. This comparison is described in the Executive Summary and Section 7.0 of the HHRA.

- Identify the contaminants of potential concern (COPCs) with the highest potential groundwater risks at the scale of both an entire OU and at the individual monitoring wells where risks are highest.

- Identify detected analytes for which existing groundwater data are inadequate to determine whether their presence may be related to Hanford Site operations.

Exposure to groundwater is evaluated for the three residential scenarios (Subsistence Farmer, Confederated Tribes of the Umatilla Indian Reservation [CTUIR] Resident, and Yakama Resident scenarios) and the Resident National Monument/Refuge exposure scenario. Direct exposure to contaminants in groundwater is evaluated for household uses of groundwater in each of these scenarios (e.g., drinking and cooking [ingestion], and bathing [dermal absorption]). The inhalation pathway for volatile organic compounds (VOCs) associated with household use of groundwater is evaluated for VOCs that are identified as COPCs in groundwater. The main difference between the residential scenarios and the Resident National Monument/Refuge exposure scenario is that the latter does not include children as potential receptors.

Ingestion, inhalation, and dermal exposures to COPCs in groundwater used in a sweat lodge are evaluated in the CTUIR Resident and Yakama Resident scenarios. As discussed in Section 3.4.3, exposure-point concentrations of VOCs and semivolatile organic compounds (SVOCs) in sweat lodge air are calculated for both the CTUIR and Yakama Resident scenarios according to the models described in *Exposure Scenario for the CTUIR Traditional Subsistence Lifeways* (Harris and Harper 2004). These models provide air concentrations for groundwater COPCs from water that is poured over heated rocks to create steam. Appendix 4 of the CTUIR exposure scenario report (Harris and Harper 2004) also provides equations for estimating air-phase contaminant concentrations for nonvolatile chemicals including metals and most radionuclides. Although the physical model underlying these equations is not clearly identified, they are used to calculate CTUIR sweat-lodge risks for consistency with Harris and Harper (2004). Based on correspondence between the U.S. Department of Energy (DOE) and the Yakama Nation discussing these technical concerns, inhalation risks for nonvolatile COPCs in the sweat lodge are not calculated for the Yakama Resident scenario.

Indirect exposures to contaminants in groundwater may also occur if contaminated groundwater is used to irrigate a garden or provide water for livestock or by migration of vapors from groundwater upwards through the vadose zone and into a building. The vapor intrusion pathway and irrigation pathway are not evaluated in the risk assessment calculations, although the potential significance of these pathways is discussed in the Uncertainty Analysis of the local-area risk assessment in Section 5.0 and Appendix D-1.

Summaries of potentially complete exposure pathways for the Resident National Monument/Refuge exposure scenario and the three residential scenarios are provided in Tables 3-11 and 3-12, respectively. Exposure parameter values are summarized in Tables 3-18 and 3-19. The equations and exposure parameters for the CTUIR Resident and Yakama Resident scenarios were taken without modification from the scenario reports provided by each tribe (Harris and Harper 2004; Ridolfi 2007, *Yakama Nation Exposure Scenario for the Hanford Site Risk Assessment, Richland, Washington*).

Groundwater risk calculations were performed separately for the seven groundwater OUs in the River Corridor. A crosswalk of the groundwater OUs and the ROD decision areas forming the basis of the HHRA is as follows:

- The 100-B/C Decision Area includes the 100-BC-5 Groundwater OU
- The 100-K Decision Area includes the 100-KR-4 Groundwater OU
- The 100-N Decision Area includes the 100-NR-2 Groundwater OU
- The 100-D/100-H Decision Area includes the 100-HR-3 Groundwater OU

- The 100-F/100-IU-2/100-IU-6 Decision Area includes the 100-FR-3 Groundwater OU and the 100-F/100-IU-2/100-IU-6 Groundwater OU
- The 300 Decision Area includes the 300-FF-5 Groundwater OU.

Risk assessment results for each OU are intended to provide approximate measures of the central and upper-bound risks at most locations within the OU. The 50^{th} and 90^{th} percentile concentrations of a COPC from all wells in an OU were used to represent central and upper-bound concentrations, respectively. Because there may be one or more groundwater contaminant plume in an OU, there are sometimes large differences in COPC concentrations in different locations within an OU. Therefore, the initial results were evaluated to identify individual wells where potential risks could exceed those associated with the 90^{th} percentile concentrations in each OU. Supplemental risk calculations were performed for these wells. In addition, risks for the OU were recalculated using all wells except those where potential risks could exceed those associated with the 90^{th} percentile concentrations, to give a second estimate of overall exposure within the OU without the contributions from wells with highest COPC concentrations.

Activities that may help reduce uncertainties, update the conclusions of the screening-level groundwater risk assessment presented in this section, and ensure that no contaminants were inadvertently overlooked based on use of the existing data set are identified in the 100 Area RI/FS work plan. These activities include the following.

- Identify existing and/or install new monitoring wells that are spatially representative of the ROD decision area. This set of monitoring wells will represent locations where a receptor potentially could contact groundwater.
- Conduct multiple rounds of sampling to obtain temporal representation of COPC concentrations in the unconfined aquifer. Additional rounds of sampling at spatially representative monitoring wells will represent current groundwater conditions and capture the influence of river fluctuations on COPC concentrations.
- Analyze all spatially representative monitoring wells for a focused list of groundwater COPCs identified for the ROD decision area for each round of sampling. Analyzing each of the monitoring wells for COPCs will provide a data set that is representative of potential releases to the groundwater.
- Evaluate sample results from characterization activities to support final remedial-action decisions for groundwater.

Additional groundwater evaluations will be presented in the RI/FS reports for the River Corridor ROD decision areas based on the additional data collection activities described above.

The remainder of Section 6.0 is organized as follows: The data set used in the groundwater screening-level risk assessment, results of the COPC selection process, and calculation of representative concentrations are described in Section 6.3. Risk assessment results for each OU are presented in Sections 6.2 through 6.8. Details of the uncertainties associated with the screening-level risk assessment are presented in Section 6.9.

6.1.1 Summary of Groundwater Screening-Level Risk Assessment Results

The results of this groundwater screening-level risk assessment indicate potential risks above EPA thresholds within each groundwater OU. Reasonable maximum exposure (RME) cancer risks were above the upper end of the EPA target risk range of 1×10^{-6} to 1×10^{-4} for each exposure scenario in all of the groundwater OUs, with the exception of Resident National Monument/Refuge and Subsistence Farmer risks in the 100-HR-3 and 100-FR-3 OUs. With the exception of the Resident National Monument/Refuge scenario in the 100-B/C-5 OU, hazard indices were above EPA's threshold value of one for each exposure scenario and each groundwater OU.

There are several key uncertainties for the groundwater results that contribute to qualification of these calculations as "screening level." These uncertainties are mostly related to the ability of the existing groundwater data set to represent current baseline conditions for possible exposures within each groundwater OU. Analytical data used for the screening-level groundwater risk assessment are obtained from several groundwater-monitoring programs including the *Atomic Energy Act of 1954* (AEA) surveillance program, the *Resource Conservation and Recovery Act of 1976* (RCRA) compliance program, and the *Comprehensive Environmental Response, Compensation, and Liability Act of 1980* (CERCLA) program. Sampling and analysis data from these programs comprehensively define the suite of contaminants associated with existing and potential groundwater contamination sources. However, differences in sampling frequencies (monthly, annually, or tri-annually), analytes analyzed at each monitoring well (radiological and chemical), and method detection limits create uncertainties associated with the spatial, chemical, and temporal representative qualities of the data set used for this risk assessment.

6.2 SITE INVESTIGATION AND DATA ANALYSIS

6.2.1 Groundwater Data and Monitoring Well Selection

As described in Section 3.2.1, the data used in the HHRA to characterize groundwater exposure within the separate ROD decision areas were obtained from the Hanford Environmental Information System (HEIS) database. Groundwater data from unfiltered samples collected during the past 10 years (1998 through 2008) were used in the HHRA. These data were not collected specifically to support HHRA, but rather for groundwater monitoring programs that fulfill a variety of state and federal regulations including the AEA, RCRA, CERCLA, and Title 173 of the *Washington Administrative Code* (WAC). The following explains the primary objectives of the different programs as explained in the *Hanford Site Groundwater Monitoring for Fiscal Year 2007* (DOE/RL-2008-01).

6.2.1.1 AEA Monitoring Requirements. DOE O 450.1, *Environmental Protection Program*, implements requirements of the AEA. This order requires environmental monitoring to detect, characterize, and respond to releases from DOE activities; assess impacts; and characterize exposure pathways. The order recommends implementing a site-wide approach for groundwater protection and requires compliance with other applicable environmental protection requirements (DOE/RL-2008-01).

6.2.1.2 CERCLA Monitoring Requirements. Monitoring wells are located and sampled in accordance with RI/FS work plans to define the nature and extent of the contaminant plumes. Groundwater also is monitored under CERCLA to assess the effectiveness of groundwater remediation (DOE/RL-2008-01).

6.2.1.3 RCRA and WAC Monitoring Requirements. The groundwater monitoring requirements for Hanford Site RCRA units fall into one of two categories: interim status or final status. A permitted RCRA unit requires final status monitoring as specified in WAC 173-303-645. RCRA units that have not yet been incorporated into permits require interim-status monitoring as specified in WAC 173-303-400, which invokes 40 CFR 265. RCRA groundwater monitoring is conducted under one of three possible phases: (1) indicator parameter, or final status detection; (2) assessment, or final status compliance; or (3) corrective action, via administrative order for interim status sites or in accordance with the Hanford Sitewide RCRA Permit for final status (DOE/RL-2008-01).

More information about the contaminants monitored, sampling frequencies, method detection limits, quality assurance/quality control (QA/QC), and other sampling and analysis requirements can be found in the following documents for each groundwater OU:

- DOE/RL-2003-38, Rev. 1, 2004, *100-BC-5 Operable Unit Sampling and Analysis Plan*
- DOE/RL-2006-52, Rev. 0, 2006, *The KW Pump-and-Treat System Remedial Design and Remedial Action Work Plan, Supplement to the 100-KR-4 Groundwater Operable Unit Interim Action*
- PNNL-15798, 2006, *100-N Shoreline Groundwater Monitoring Plan*
- DOE/RL-2003-04, Rev. 1, 2005, *Sampling and Analysis Plan for the 200-PO-1 Groundwater Operable Unit*
- DOE/RL-2002-11, Rev. 2, 2008, *300-FF-5 Operable Unit Sampling and Analysis Plan.*

A similar process was used to select the data set for characterizing exposure at each groundwater OU. Two primary activities were conducted to develop the final groundwater data set that includes the identification of spatially representative monitoring wells and the selection of temporally representative analytical data. In addition, the type, time, quantity, and quality of analytical data were evaluated for each monitoring well to determine if it was usable for characterizing exposure within its assigned OU.

The selection of monitoring wells that spatially represent the exposure area is the first step in developing the final data set. For the purpose of characterizing exposure, only monitoring wells located within the unconfined aquifer were used in the risk assessment; injection wells, extraction wells, or wells that are dedicated to the in situ REDOX manipulation treatability testing are not included.

The following steps were taken to identify spatially representative monitoring wells for the final data set.

6.2.1.3.1 Step 1. All existing monitoring wells located within the boundaries of each respective groundwater OU were initially considered. Monitoring wells were identified by the review of Hanford Site groundwater monitoring well maps and the use of the Hanford Site Geospatial Map Portal. Monitoring wells were not distinguished between those used for AEA, RCRA, or CERCLA monitoring requirements.

6.2.1.3.2 Step 2. Analytical data from the period 2004 through 2008 were initially extracted from the HEIS database for those monitoring wells identified in Step 1. The purpose of this step is to identify those monitoring wells that contain contaminant data that are appropriate for inclusion in the risk assessment. Only those monitoring wells that are specifically sampled and analyzed for contaminants (e.g., cations, anions, radionuclides, or organic constituents) were selected for inclusion in the final data set. Only unfiltered sample results were used to characterize exposure because it represents total concentrations of contaminants prior to any remedial action (e.g., filtering or treatment). If monitoring wells are only monitored for field parameters (e.g., temperature, pH, or water level) then they were excluded from the final data set at this step of the process. Field parameters are not typically used to characterize exposure.

The selection of the appropriate temporal range of data is important for determining a "snapshot" of concentrations that represent current groundwater conditions. In addition to representing current groundwater conditions, the temporal range selected should have a sufficient number of sampling rounds to calculate the exposure point concentration (EPC). As a result of the different monitoring programs, it is difficult to identify one time frame that can adequately represent current concentrations for all monitoring wells and provide a sufficient number of sampling rounds. Sampling frequencies can vary from a quarterly basis to an annual basis.

For the purposes of calculating the representative concentrations (see Section 3.4.2), 10 or more sample results are preferred for computing percentiles. Therefore, a maximum 10-year time frame was selected to obtain the minimum data requirement. It should be noted that if a monitoring well is sampled on a frequent basis, all sampling results reported during the entire 10-year time frame are included in the EPC. At a minimum, all contaminants have a minimum of four sampling rounds.

The number of monitoring wells evaluated in each of the seven groundwater OUs is as follows:

- 100-BC-5 24 wells
- 100-KR-4 36 wells
- 100-NR-2 46 wells
- 100-HR-3 97 wells
- 100-FR-3 20 wells
- 100-IU-2/100-IU-6 25 wells
- 300-FF-5 83 wells.

6.2.2 Results of COPC Identification for Groundwater

There are seven groundwater OUs evaluated in this risk assessment, which correspond to six ROD decision areas for other environmental media. These OUs include the 100-BC-5 OU (100-B/C Area), the 100-KR-4 OU (100-K Area), the 100-NR-2 OU (100-N Area), the 100-HR-3 OU (100-D/100-H Area), and the 300-FF-5 OU (300 Area). There are two groundwater OUs corresponding to the 100-F/100-IU-2/100-IU-6 ROD Area. These are the 100-FR-3 OU and the groundwater underlying the 100-IU-2 and 100-IU-6 decision area (assigned to the 200-PO-1 OU). As described in Section 3.2, COPCs for this risk assessment were determined using comparisons of analyte concentrations among the groundwater OUs, identification as a COPC in the cleanup verification soil data for a ROD area, and identification as a groundwater inclusion list analyte are among the tools used to identify groundwater COPCs in this assessment.

Groundwater COPCs have also been identified as a part of the RI/FS planning for each ROD decision area. The RI/FS planning used a similar method for identifying COPCs with some notable differences. For the RI/FS planning, COPCs were identified using a comparison of groundwater concentrations to applicable or relevant and appropriate requirements (ARARs) and risk-based screening levels. The COPC lists from the RI/FS planning were then used to identify general analytical methods (e.g., VOCs, metals, radionuclides) for the additional groundwater sampling that will be conducted as part of the RI activities for the River Corridor. By using method-based analyses, data will be available for analytes included in the analytical method.

In general, the COPC lists from the RI/FS planning contain more COPCs than were identified for this groundwater screening-level assessment. However, there are some analytes on the RCBRA COPC lists that are not on the RI/FS COPC lists (i.e., several metals, VOCs, and radionuclides). Because method-based analyses are being used for the additional sampling being conducted for the RI activities, data will be available for the analytes that are COPCs for the RCBRA, but not on the RI COPC list. Final remedial decisions regarding groundwater in the River Corridor will be based on the evaluations conducted in the RI report using data from the method-based analyses.

6.2.2.1 Identification of Groundwater Inclusion List Analytes. The inclusion list of groundwater analytes (Table 3-8) reflects those contaminants that the Tri-Parties agree need to be addressed in this risk assessment in order to prepare a meaningful and effective regulatory document for the Hanford Site. The inclusion list of groundwater analytes is based on the list of the key groundwater plume contaminants compiled for the 100 Area and 300 Area. These key groundwater plume contaminants are based on contaminants featured in the *Summary of Hanford Site Groundwater Monitoring for Fiscal Year 2006* (PNNL-16346). The inclusion list of COPCs for the groundwater OUs is a subset of the key plume contaminants, with consideration of the magnitude and frequency of analyte detections, and on the relevance of these detections to Hanford Site operational history and published groundwater standards. The list of contaminants evaluated in the 2006 annual groundwater monitoring reports (PNNL-16346) is summarized in the following paragraphs.

100-BC-5. Tritium, strontium-90, nitrate, and chromium are identified as key contaminants in PNNL-16346. *The 100-B/C Pilot Project Risk Assessment* (DOE/RL-2005-40) identified five contaminants of concern for groundwater: strontium-90, tritium, nitrate, hexavalent chromium, and antimony. Antimony was identified based on two apparent detections in well 199-B2-12 in 1997 and 1999. Both of these results were associated with duplicate samples with no detectable antimony; however, the antimony detection limit is greater than the 6-µg/L drinking water standard.

100-KR-4. Chromium, carbon-14, nitrate, strontium-90, technetium-99, trichloroethene, and tritium are identified as key contaminants in PNNL-16346. Concentrations of these analytes exceed drinking water standards. Other VOCs that may be found along with trichloroethene (e.g., tetrachloroethene and cis-1,2-dichloroethene) are not detected at the wells where trichloroethene is detected (199-K-106A and 199-K-132). In the past, several metals have been measured in filtered samples at concentrations above the secondary drinking water standards (e.g., aluminum, iron, and manganese). These occurrences have not been positively connected to waste sites or waste streams. They are not considered COPCs because of their limited areal extent, sporadic occurrence, and possibility that their occurrence may be related to well construction and, therefore, not representative of groundwater conditions. However, their concentrations continue to be monitored as part of basic water quality analyses (e.g., collective analyses for major cations and anions).

100-NR-2. Strontium-90, tritium, nitrate, sulfate, total petroleum hydrocarbons (TPHs), manganese, chromium, and iron are identified as key contaminants in PNNL-16346. TPHs are related to a 1960s diesel fuel leak (DOE/RL-95-111). Of the effected wells, 199-N-18 is closest to the former leak site and had the highest levels of groundwater contamination. The maximum fiscal year (FY) 2006 result for TPHs in the diesel range was 23 mg/L, the lowest value since 2000. Manganese continued to exceed its secondary drinking water standard (50 µg/L) in two wells affected by petroleum contamination: 199-N-16 (809 µg/L) and 199-N-18 (3,420 µg/L). Iron also exceeded its secondary drinking water standard (300 µg/L) in well 199-N-18 (17,600 µg/L). Biodegradation of the hydrocarbons creates reducing conditions, which increases the solubility of metals such as manganese and iron from the well casing or aquifer sediment.

Only one well in the 100-N Area (199-N-80) has chromium concentrations (~163 µg/L) above the drinking water standard (100 µg/L).

100-HR-3-D and 100-HR-3-H. The 100-HR-3 OU is divided into east (100-HR-3-H) and west (100-HR-3-D) areas. Chromium, tritium, nitrate, sulfate, technetium-99, uranium, and strontium-90 are identified as key contaminants in PNNL-16346 in one or both of these areas. Concentrations of technetium-99 and tritium did not exceed drinking water standards in the 100-HR-3-D Area in 2006.

100-FR-3. Nitrate, strontium-90, trichloroethene, chromium, tritium, and uranium are identified as key contaminants in PNNL-16346. Concentrations of tritium, uranium, and chromium did not exceed drinking water standards in 2006. Iron (751 µg/L), manganese (89.6 µg/L), and zinc (5,020 µg/L) were elevated in well 699-58-24, south of the 100-F Area. These metals were even higher when the well was sampled in December 2003. The elevated metals are likely caused by casing corrosion.

300-FF-5. Uranium, trichloroethene, cis-1,2-dichloroethene, tetrachloroethene, strontium-90, tritium (618-11 Burial Ground), and tributyl phosphate (316-4 Crib and 618-10 Burial Ground) are identified as key contaminants in PNNL-16346. Technetium-99, tritium, and nitrate have migrated into the 300-FF-5 North subregions from upgradient sources in the 200 East Area. Trichloroethene in 300-FF-5 groundwater has sources both within and outside of the 300 Area. Tributyl phosphate tends to bind to soil in the vadose zone, where it slowly degrades with time. It is not very soluble in water and therefore is not widely dispersed via water transport mechanisms. A drinking water standard for tributyl phosphate has not been established.

Strontium-90 has been detected at relatively low levels and as an isolated occurrence at well 399-3-11 in previous years (PNNL-13788). Results from well 399-3-11 during FY 2006 were 3.3 and 2.7 pCi/L for samples collected during January and July 2006, respectively (the drinking water standard is 8 pCi/L). The source for the strontium-90 is not clearly evident, but one candidate is a historical long-term leak from transfer lines associated with the 307 Retention Basins.

1100-EM-1. Trichloroethene, tritium, nitrate, uranium, ammonia, and fluoride are identified as key contaminants in PNNL-16346. Potential breakdown products of trichloroethene, including vinyl chloride and 1,1-dichloroethene, continued to show levels less than their respective minimum detection limits during FY 2006. Nitrate contamination in this area is likely the result of industrial and agricultural uses off the Hanford Site. Agricultural uses include application of fertilizers onto irrigation circles in the central portion of the 1100-EM-1 groundwater interest area.

6.2.2.2 Data Summary for Groundwater COPC Identification. Tables 6-1 through 6-7 show the number of sample results, detection information, range of detected values, and range of nondetect values for all analytes measured in groundwater samples for each groundwater OU. The range of groundwater concentrations for detected analytes in each of the seven groundwater OUs is also shown in Appendix C-11.0.

6.2.2.3 Analyte-Specific Evaluation for Groundwater COPC Identification. As discussed in Section 3.2.2, neither background nor reference site data are available for groundwater; therefore, such comparisons could not be performed. For detected analytes in groundwater, a number of semiquantitative measures are used to differentiate COPCs from analytes not identified as COPCs. Graphical analyses comparing the range and percentiles of analyte concentrations among the seven groundwater OUs is a major factor in determining COPC status. These analyses are predicated on the observation that operational releases to groundwater occur at specific locations; therefore, significant releases should be associated with significantly elevated concentrations at a subset of approximately 330 wells included in this assessment. By contrast, analyte concentrations consistent with naturally occurring levels or nonoperational sources such as global fallout should show comparatively little spatial variability. Boxplots showing groundwater concentrations for detected analytes in each of the seven groundwater OUs are provided in Appendix C-11.0.

These graphical comparisons may be informed by the range of the detected concentrations relative to detection limits, detection frequency, status of an analyte as a COPC in the cleanup verification soil data in the ROD area, and status as a groundwater inclusion list analyte. For sparsely detected organic chemicals in particular, the range of the detected concentrations relative to detection limits and the detection frequency are used to determine COPC status. In cases where there is particularly high uncertainty in assigning COPC status, an analyte is assigned to the category "uncertain" in Tables 6-8 through 6-14. These analytes are summarized and discussed in the groundwater uncertainty analysis in Section 6.9.2.

The analyte-specific evaluations for detected analytes in groundwater are provided for each OU in Tables 6-8 through 6-14. Contaminants of potential concern in groundwater for each groundwater OU are summarized in Table 6-15.

6.2.3 Representative Concentrations in Groundwater

Methods for the computation of representative concentrations in groundwater were described in Section 3.4.3. Groundwater representative concentrations for each OU are calculated using groundwater data from unfiltered samples collected during the past 5 to 10 years (1998 through 2008). Groundwater representative concentrations for each OU are calculated as the 50th and 90th percentiles of the monitoring well data within the OU boundaries, respectively. The 50th percentile value is used to evaluate central tendency groundwater concentrations across the entire OU. The 90th percentile values are intended to provide a protective estimate of groundwater COPC concentrations at most locations within an OU. At the Hanford Site, the use of percentiles for estimating groundwater concentrations was initially performed for the baseline risk assessment for the 200-ZP-1 OU (DOE/RL-2007-28, *Feasibility Study Report for the 200-ZP-1 Groundwater Operable Unit*). That baseline risk assessment was used to support a ROD for the 200-ZP-1 OU (EPA 2008b). Based on this precedent, this approach for estimating groundwater exposure concentrations has also been used in this risk assessment.

The 50th percentile (used to represent CTE conditions) and the 90th percentile (used to represent RME conditions) groundwater concentrations in a groundwater OU are also the specific concentrations of a COPC at a particular well and at a particular time. In those wells where

COPCs are present at concentrations greater than the overall 90th percentile for the OU, exposures to these COPCs (at these locations) will be underestimated by the 90th percentile for the OU. If such COPCs are also major contributors to risk at the OU, then risks calculated using the 90th percentile for the entire OU may not represent RME conditions for that OU. Under such circumstances, supplemental well-specific risk calculations are warranted. Combinations of individual monitoring wells and COPCs have been identified for supplemental risk analyses based on the results of the process described below.

Individual wells where calculated health risks could significantly exceed those based on the 90th percentile for an entire OU were identified using a process summarized in the following six steps.

1. Identify all COPCs that contribute 10% or more to an RME cancer risk greater than or equal to 1×10^{-4}, or hazard index (HI) equal to or greater than 10, in each groundwater OU. The COPC cancer risk or hazard quotient (HQ) for the COPC must, therefore, be at least 1×10^{-5} or 1.0, respectively. This step ensures that only COPCs contributing to significant risks are further evaluated.

2. Identify individual wells where the maximum detected concentration of a COPC from Step 1 exceeds the OU RME value.

3. Calculate the ratios of the mean COPC well concentration to the OU RME, and the 95% upper confidence limit (UCL) well concentration to the OU RME.

4. Identify those wells where there is a high degree of confidence that well concentrations of the risk-driving COPC exceed the 90th percentile for the OU. This is accomplished by selecting wells where the ratio (well mean COPC concentration: OU RME COPC concentration) exceeds 1.5 for one or more COPCs.

5. Supplement the wells identified in Step 4 with wells where the ratio (well 95% UCL COPC concentration: OU RME COPC concentration) is significantly higher than the same ratio calculated in Step 4. The purpose of this step is to ensure that wells are considered even when there are just a few high outlying values of risk-driving COPCs.

6. Prepare separate risk assessment calculations for the individual wells identified in Steps 4 and 5, including all COPCs analyzed and detected in each monitoring well.

After the six-step process was implemented, the CTE (50th percentile) and RME (90th percentile) OU groundwater concentrations were then recomputed without the data for the wells identified in the process. These recomputed values were used to represent CTE and RME conditions in the OU without the influence of wells where concentrations of risk-driving COPCs were highest. The recomputed values for the 50th and 90th percentiles were used to conduct the CTE and RME risk assessment calculations for an entire groundwater OU, where these values represent general conditions both within and outside groundwater contamination plumes. The CTE and RME risk calculations for the individual wells represent conditions at the specific locations in a groundwater OU where risks are highest.

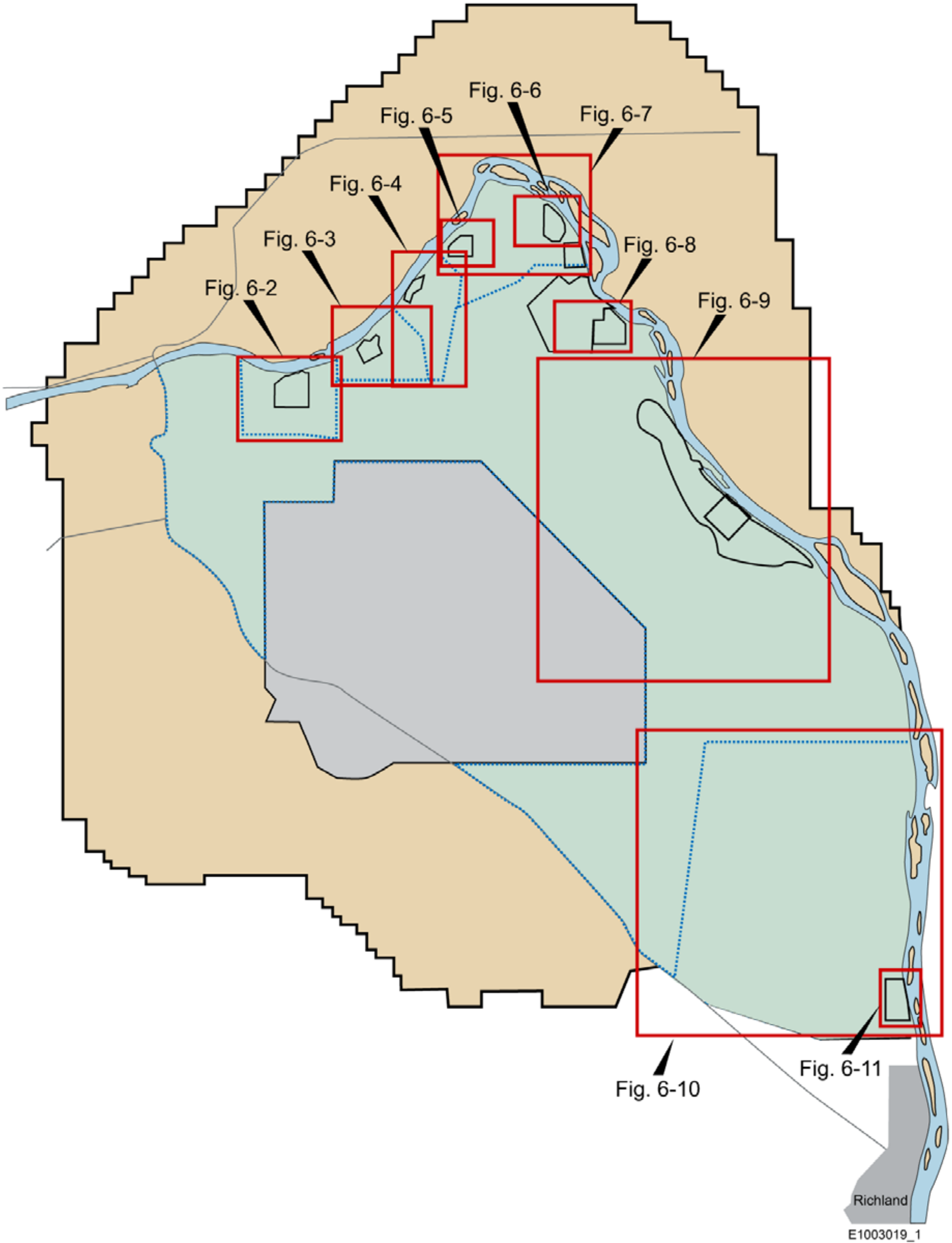
Table 6-16 shows the COPCs, which include inorganic chemicals and radionuclides, identified in Step 1 for consideration for well-specific supplemental assessments for each OU. Uranium in the 300 Area was included in Table 6-16 because the HI of 9.2 is very close to 10. The Subsistence Farmer scenario, which includes a child receptor, was used to identify the COPCs in Step 1. Although groundwater exposure intensity and receptors vary among the scenarios, the exposure routes (e.g., water ingestion, dermal absorption, and tritium inhalation) are the same across all scenarios.

Table 6-17 shows the mean, 95% UCL, and maximum detected concentration in a monitoring well of the COPC and OU combinations from Table 6-16. Table 6-17 includes information for all monitoring wells where the maximum detected concentration exceeds the 90th percentile for the OU. The COPC concentrations in an individual monitoring well may be compared to the 50th and 90th percentile COPC concentrations in the OU. The well-analyte combinations shaded in Table 6-17 are evaluated separately in the groundwater risk assessment, according to the protocol described in Steps 4 and 5. The recomputed CTE and RME OU groundwater concentrations, without the wells identified for supplemental analysis, are also shown in Table 6-17. The locations of the monitoring wells within the groundwater OUs and their relation to groundwater contaminant plumes are shown in a series of figures cited in Sections 6.3 through 6.8. The orientation of the groundwater OUs and the regions shown in these figures are provided in Figure 6-1.

Files of the representative concentrations used in the groundwater risk assessment are provided as an electronic attachment to Appendix C-3. The EPCs and the results of the risk assessment calculations for each individual COPC and exposure pathway contributing to the sums shown in Section 6.0 risk result tables are provided in electronic format through a web-based interface at http://rcbra.gisdt.org and are also tabulated as an electronic attachment to this report. Exposure point concentrations, which include both representative concentrations in groundwater and modeled concentrations in indoor and sweat lodge air, are provided in electronic format in Appendix D-5. The GiSdT interface for the RCBRA was developed as a repository and interface for analytical data and risk assessment results for this report.

The results of the Resident National Monument/Refuge and Subsistence Farmer scenario groundwater risk assessments for each of the seven groundwater OUs are summarized in Table 6-18. Groundwater risk assessment results for the CTUIR Resident and Yakama Resident scenarios for each of the seven groundwater OUs are summarized in Table 6-19. These tables of results present the risks that have been summed across the individual exposure pathways and COPCs. The principal COPCs and exposure pathways contributing to risks above the threshold criteria described in Section 3.6 are discussed in the text of the risk assessment. The results of the groundwater risk assessment for individual monitoring wells where risks potentially exceed those characterized by the 90th percentile value for an OU are shown in subsequent tables.

Figure 6-1. Key to Groundwater Well Locations and Plume Maps.



6.3 100-B/C DECISION AREA RESULTS

The location of the groundwater monitoring wells evaluated in the 100-B/C Area and the boundaries of key groundwater plumes in this area are shown in Figure 6-2. A summary of the cancer risk, radiation dose, and chemical hazard results for the 100-B/C Area is provided in this section. Results pertaining to the Resident National Monument/Refuge and Subsistence Farmer exposure scenarios, which include domestic exposures to groundwater, are presented in Section 6.3.1. Results for the CTUIR Resident and Yakama Resident exposure scenarios include both domestic groundwater exposures and exposure related to use of groundwater in a sweat lodge. The results of these calculations, differentiating between domestic and sweat lodge exposures, are presented in Section 6.3.2.

Figure 6-2. Groundwater Well Locations and Key Plumes for the 100-B/C Area.

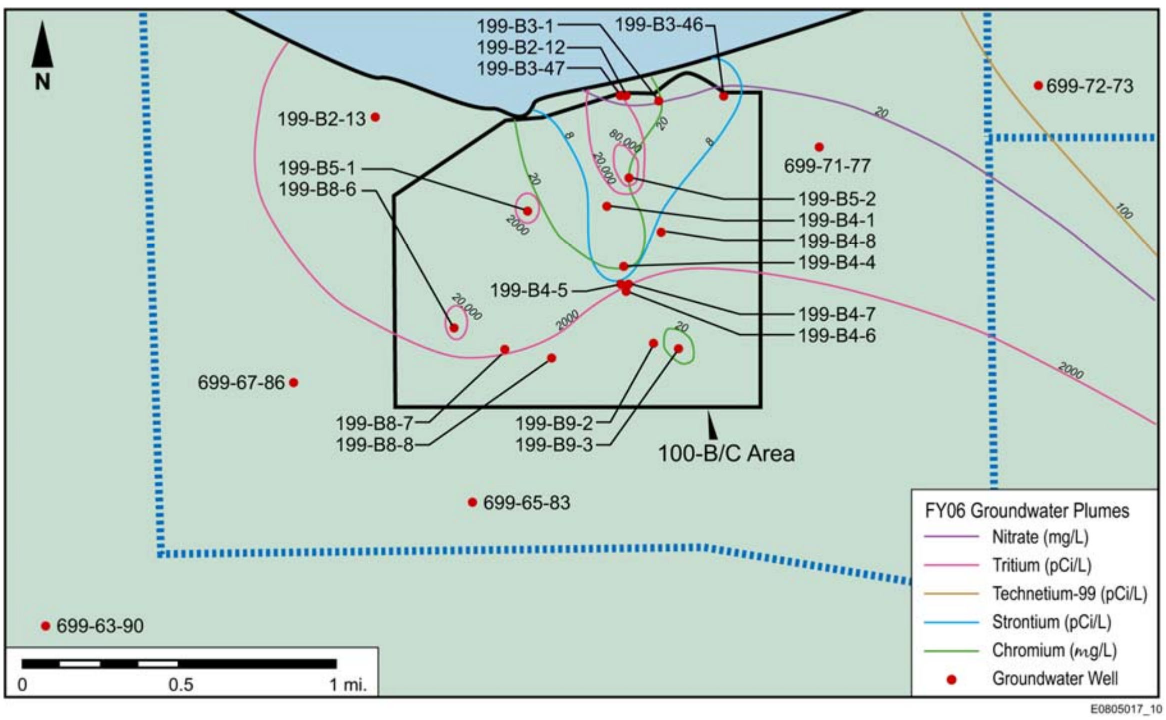


Table 6-16 shows that tritium and strontium-90 concentrations are associated with potential cancer risks that are above the 1×10^{-4} threshold in the 100-B/C Area. Table 6-17 shows that the 95% UCL value of tritium at one well (199-B5-2) is more than fourfold higher than the 90th percentile for the OU, indicating that there is relatively high confidence that RME risks at this well are higher than those associated with the entire OU. The 50th and 90th percentiles for COPC concentrations in the 100-B/C Area were recalculated without the data for well 199-B5-2. A supplemental set of risk calculations are provided for this well. Cancer risk, radiation dose, and chemical hazard results for the 100-B/C Area shown in Tables 6-18 and 6-19 do not include the results for well 199-B5-2.

6.3.1 Resident National Monument/Refuge and Subsistence Farmer Results in the 100-B/C Area

6.3.1.1 Cancer Risk. The RME and CTE groundwater pathways risk results for the Resident National Monument/Refuge scenario (radionuclides + chemicals) in the 100-B/C Area are 2×10^{-4} and 6×10^{-6}, respectively. The RME and CTE groundwater pathways risk results for the Subsistence Farmer scenario (radionuclides + chemicals) in the 100-B/C Area are 2×10^{-4} and 4×10^{-6}, respectively. Groundwater cancer risk results for these scenarios are presented in Table 6-18.

A summary of the key exposure pathways and analytes contributing 10% or more to RME groundwater cancer risk above 1×10^{-4} in the 100-B/C Area for the Resident National Monument/Refuge and Subsistence Farmer scenarios is provided below.

Scenario	Analyte	Pathway	Fraction of OU Cancer Risk	OU Cancer Risk
Resident National Monument/Refuge	Tritium	groundwaterIngestion	0.51	2×10^{-4}
	Strontium-90	groundwaterIngestion	0.45	
Subsistence Farmer	Tritium	groundwaterIngestion	0.50	2×10^{-4}
	Strontium-90	groundwaterIngestion	0.45	

The RME risks for both scenarios are above the 1×10^{-4} upper end of EPA's target cancer risk range, while the CTE risks are within the lower portion of the range. About 90% or more of the RME and CTE cancer risks are associated with tritium and strontium-90 via the water ingestion exposure route. Although future trends in groundwater concentrations have not been quantified in this assessment, natural radioactive decay of tritium (12.3-year half-life) and strontium-90 (28.8-year half-life) will result in a decrease of risk from these COPCs over time compared to present-day groundwater conditions.

Cancer risks for the Resident National Monument/Refuge and Subsistence Farmer scenarios calculated using the data from monitoring well 199-B5-2 are shown in Table 6-20. The RME and CTE groundwater pathways risk results for the Resident National Monument/Refuge scenario (radionuclides + chemicals) at well 199-B5-2 are 4×10^{-4} and 2×10^{-5}, respectively. The RME and CTE groundwater pathways risk results for the Subsistence Farmer scenario (radionuclides + chemicals) at well 199-B5-2 are 5×10^{-4} and 2×10^{-5}, respectively. More than 90% of the RME risks at this well for both scenarios are related to tritium by groundwater ingestion. As shown in Table 6-17, this well was identified for supplemental analysis based on tritium concentrations.

A summary of the key exposure pathways and analytes contributing 10% or more to RME groundwater cancer risk above 1×10^{-4} in well 199-B5-2 for the Resident National Monument/Refuge and Subsistence Farmer scenarios are provided below.

Scenario	Analyte	Pathway	Fraction of Well Cancer Risk	Well Cancer Risk
Resident National Monument/Refuge	Tritium	groundwaterIngestion	0.93	4×10^{-4}
Subsistence Farmer	Tritium	groundwaterIngestion	0.92	5×10^{-4}

6.3.1.2 Radiation Dose. The RME and CTE groundwater pathways radiation dose results for the Resident National Monument/Refuge scenario in the 100-B/C Area are 10 mrem/yr and 1.4 mrem/yr, respectively. The RME and CTE groundwater pathways radiation dose results for the Subsistence Farmer scenario in the 100-B/C Area are 12 mrem/yr and 1.5 mrem/yr, respectively. Groundwater radiation dose results for these scenarios are presented in Table 6-18.

A summary of the key exposure pathways and analytes contributing 10% or more to RME groundwater radiation dose above the 4 mrem/yr threshold in the 100-B/C Area for the Resident National Monument/Refuge and Subsistence Farmer scenarios is provided below. The basis of the 4 mrem/yr threshold is presented in Section 3.6.4.

Scenario	Analyte	Pathway	Fraction of OU Dose	OU Radiation Dose (mrem/yr)
Resident National Monument/Refuge	Tritium	groundwaterIngestion	0.48	10
	Strontium-90	groundwaterIngestion	0.47	
Subsistence Farmer	Strontium-90	groundwaterIngestion	0.55	12
	Tritium	groundwaterIngestion	0.39	

Reasonable maximum exposure radiation dose is associated primarily with tritium and strontium-90 via the water ingestion exposure route. As described above, natural radioactive decay of tritium (12.3-year half-life) and strontium-90 (28.8-year half-life) will result in decreased concentrations of these COPCs over time. The RME dose results are about two to three times higher than the 4 mrem/yr threshold used to assess the significance of the groundwater dose results. The results of the CTE dose calculations are about a factor of three below the 4 mrem/yr threshold.

Radiation dose results calculated using the data from monitoring well 199-B5-2 are shown in Table 6-20. The RME and CTE groundwater pathways radiation dose results for the Resident National Monument/Refuge scenario at well 199-B5-2 are 25 mrem/yr and 5.3 mrem/yr, respectively. The RME and CTE groundwater pathways radiation dose results for the Subsistence Farmer scenario at well 199-B5-2 are 25 mrem/yr and 4.7 mrem/yr, respectively. Approximately 80% to 90% of the RME radiation dose at this well for both scenarios is related to tritium by groundwater ingestion.

A summary of the key exposure pathways and analytes contributing 10% or more to RME groundwater radiation dose above the 4.0 mrem/yr threshold at well 199-B5-2 for the Resident National Monument/Refuge and Subsistence Farmer scenarios is provided below.

Scenario	Analyte	Pathway	Fraction of Well Dose	Well Radiation Dose (mrem/yr)
Resident National Monument/Refuge	Tritium	groundwaterIngestion	0.89	25
Subsistence Farmer	Tritium	groundwaterIngestion	0.84	25

6.3.1.3 Chemical Hazard. As described in Section 3.6.2, chemical HI is calculated separately for adult and child receptors in the Residential scenarios. Reasonable maximum exposure child HI values tend to exceed those for adults; for the CTE calculations, adult and child HIs tend to be approximately equivalent to adult values for water ingestion slightly higher than those for young children due to the higher adult ingestion rate. The higher of the adult and child HI values are shown in Table 6-18 and described here.

The RME and CTE chemical HI results for the Resident National Monument/Refuge scenario in the 100-B/C Area are 0.80 and 0.18, respectively. The RME and CTE child chemical HI results for the Subsistence Farmer scenario in the 100-B/C Area are 1.5 and 0.19, respectively.

A summary of the key exposure pathways and analytes contributing 10% or more to an RME groundwater HI above 1.0 in the 100-B/C Area for the Subsistence Farmer scenario is provided below.

Scenario	Analyte	Pathway	Fraction of OU HI	OU Hazard Index
Subsistence Farmer	Hexavalent chromium	groundwaterIngestionChild	0.52	1.5
	Hexavalent chromium	groundwaterDermalChild	0.30	
	Nitrogen in nitrate	groundwaterIngestionChild	0.14	

The RME HI value for the Subsistence Farmer scenario is above the significance threshold of 1.0. Approximately 80% of the Subsistence Farmer child HI of 1.5 is due to hexavalent chromium by drinking water ingestion and dermal absorption while bathing. Another 14% is due to nitrate exposure via groundwater ingestion. The oral reference dose (RfD) for chromium is not related to any specific adverse effect, but the effects of nitrate are very specific and relate to inhibition of oxygen binding with hemoglobin in the blood. It is unlikely that additive effects would be observed for these COPCs.

Chemical hazard calculated using the data from monitoring well 199-B5-2 is shown in Table 6-20. Only the Subsistence Farmer RME HI value (1.1) is above the threshold of 1.0. Risk drivers for HI at well 199-B5-2 are the same as for the entire OU. Approximately 85% of the Subsistence Farmer child HI of 1.1 is due to hexavalent chromium by drinking water ingestion and dermal absorption while bathing. Another 12% is due to nitrate exposure via groundwater ingestion.

A summary of the key exposure pathways and analytes contributing 10% or more to an RME groundwater HI above 1.0 at well 199-B5-2 for the Subsistence Farmer scenario is provided below.

Scenario	Analyte	Pathway	Fraction of Well Hazard Index	Well Hazard Index
Subsistence Farmer	Hexavalent chromium	groundwaterIngestionChild	0.54	1.1
	Hexavalent chromium	groundwaterDermalChild	0.31	
	Nitrogen in nitrate	groundwaterIngestionChild	0.12	

6.3.2 CTUIR Resident and Yakama Resident Results in the 100-B/C Area

The risk assessment results presented in this section apply the parameter values and models documented in Harris and Harper (2004) and Ridolfi (2007). As discussed in Section 6.1, risks related to inhalation of volatile and semivolatile COPCs in sweat lodge air are calculated for both the CTUIR and Yakama Resident scenarios; however, inhalation risks for nonvolatile COPCs are only calculated for the CTUIR Resident. Uncertainty and bias in the sweat lodge exposure models are discussed in Section 6.9.

6.3.2.1 Cancer Risk. The groundwater pathways risk results for the CTUIR Resident and Yakama Resident scenarios (radionuclides + chemicals) in the 100-B/C Area are 4×10^{-3} and 1×10^{-3}, respectively. Domestic exposures contribute less than sweat lodge exposures in the CTUIR Resident scenario, but are of equal importance in the Yakama Resident scenario. Groundwater cancer risk results for these scenarios are presented in Table 6-19.

A summary of the key exposure pathways and analytes contributing 10% or more to groundwater cancer risk above 1×10^{-4} in the 100-B/C Area for the CTUIR Resident and Yakama Resident scenarios is provided below.

Scenario	Analyte	Pathway	Fraction of OU Cancer Risk	OU Cancer Risk
CTUIR Resident	Hexavalent chromium	waterInhalationSweatLodge	0.77	4×10^{-3}
Yakama Resident	Tritium	waterInhalationSweatLodge	0.35	1×10^{-3}
	Tritium	groundwaterIngestion	0.25	
	Strontium-90	groundwaterIngestion	0.22	

Inhalation of hexavalent chromium in the sweat lodge is the primary contributor to cancer risk in the CTUIR Resident scenario. In the Yakama Resident scenario, where inhalation of gas-phase nonvolatile COPCs is not modeled, sweat lodge inhalation of tritium and domestic ingestion of tritium and strontium-90 in drinking water are the main contributors to cancer risk. Ingestion of tritium and strontium-90 in drinking water are also the main contributors to domestic cancer risk in the Yakama Resident scenario.

Cancer risks for the CTUIR Resident and Yakama Resident scenarios calculated using the data from monitoring well 199-B5-2 are shown in Table 6-21. The total groundwater pathway cancer risk results for the CTUIR Resident and Yakama Resident scenarios at well 199-B5-2 are 5×10^{-3} and 4×10^{-3}, respectively. Similar to the results for the entire OU, inhalation exposure to hexavalent chromium in the sweat lodge is the main contributor to the CTUIR Resident cancer risk. In the Yakama Resident scenario, and for CTUIR Resident domestic exposures, tritium is the main contributor to cancer risk at monitoring well 199-B5-2.

A summary of the key exposure pathways and analytes contributing 10% or more to groundwater cancer risk above 1×10^{-4} in well 199-B5-2 for the CTUIR Resident and Yakama Resident scenarios is provided below. The higher tritium sweat lodge risk in the Yakama Resident scenario is because of the higher daily use cited for the Yakama (7 hr/day; Ridolfi 2007) than for the CTUIR (1 hr/day; Harris and Harper 2004).

Scenario	Analyte	Pathway	Fraction of Well Cancer Risk	Well Cancer Risk
CTUIR Resident	Hexavalent chromium	waterInhalationSweatLodge	0.50	5×10^{-3}
	Tritium	groundwaterIngestion	0.30	
Yakama Resident	Tritium	waterInhalationSweatLodge	0.50	4×10^{-3}
	Tritium	groundwaterIngestion	0.35	
	Tritium	waterIngestionSweatLodge	0.11	

6.3.2.2 Radiation Dose. The groundwater pathways radiation dose results for the CTUIR Resident and Yakama Resident scenarios in the 100-B/C Area are 37 mrem/yr and 79 mrem/yr, respectively. The higher Yakama scenario dose is due to longer daily exposure in the sweat lodge. Groundwater radiation dose results for these scenarios are presented in Table 6-19.

A summary of the key exposure pathways and analytes contributing 10% or more to groundwater radiation dose above the 4.0 mrem/yr threshold in the 100-B/C Area for the CTUIR Resident and Yakama Resident scenarios is provided below.

Scenario	Analyte	Pathway	Fraction of OU Dose	OU Radiation Dose (mrem/yr)
CTUIR Resident	Strontium-90	groundwaterIngestion	0.26	37
	Tritium	waterInhalationSweatLodge	0.24	
	Tritium	groundwaterIngestion	0.18	
	Strontium-90	waterIngestionSweatLodge	0.11	

Scenario	Analyte	Pathway	Fraction of OU Dose	OU Radiation Dose (mrem/yr)
Yakama Resident	Strontium-90	waterIngestionSweatLodge	0.13	79
	Tritium	waterInhalationSweatLodge	0.68	
	Strontium-90	groundwaterIngestion	0.12	

Reasonable maximum exposure radiation dose is associated primarily with strontium-90 via the water ingestion exposure route and tritium by ingestion and vapor phase inhalation. Although future trends in groundwater concentrations have not been quantified in this assessment, natural radioactive decay of tritium (12.3-year half-life) and strontium-90 (28.8-year half-life) will result in decreased concentrations of these COPCs over time.

Radiation dose for the CTUIR Resident and Yakama Resident scenarios calculated using the data from monitoring well 199-B5-2 is shown in Table 6-21. Radiation dose values were 93 mrem/yr for the CTUIR Resident and 300 mrem/yr for the Yakama Resident. Approximately 80% of the Yakama Resident dose is due to inhalation of tritium during sweat lodge exposure. Inhalation of tritium in the sweat lodge contributes 44% to the CTUIR Resident radiation dose of 93 mrem/yr.

A summary of the key exposure pathways and analytes contributing 10% or more to groundwater radiation dose above the 4.0 mrem/yr threshold in well 199-B5-2 for the CTUIR Resident and Yakama Resident scenarios is provided below.

Scenario	Analyte	Pathway	Fraction of Well Cancer Risk	Well Cancer Risk
CTUIR Resident	Tritium	waterInhalationSweatLodge	0.44	93
	Tritium	groundwaterIngestion	0.33	
	Tritium	waterIngestionSweatLodge	0.15	
Yakama Resident	Tritium	waterInhalationSweatLodge	0.81	300
	Tritium	groundwaterIngestion	0.10	

6.3.2.3 Chemical Hazard. The chemical child HI results for the CTUIR Resident and Yakama Resident scenarios in the 100-B/C Area are 2.3 and 2.4, respectively. The adult HI results are 6.8 and 3.5, respectively. The adult HI includes contribution from sweat lodge exposure beginning at the age of 6 years old.

A summary of the key exposure pathways and analytes contributing 10% or more to an adult groundwater HI above 1.0 in the 100-B/C Area for the CTUIR Resident and Yakama Resident scenarios is provided below.

Scenario	Analyte	Pathway	Fraction of OU HI	OU Hazard Index
CTUIR Resident	Hexavalent chromium	waterInhalationSweatLodge	0.46	6.8
	Manganese	waterInhalationSweatLodge	0.21	

Scenario	Analyte	Pathway	Fraction of OU HI	OU Hazard Index
Yakama Resident	Hexavalent chromium	waterDermalSweatLodge	0.62	3.5
	Hexavalent chromium	groundwaterIngestion	0.18	

Approximately 50% of the CTUIR Resident adult HI is due to inhalation of hexavalent chromium in the sweat lodge. Another 20% to 25% is due to manganese exposure via inhalation in the sweat lodge. In CTUIR domestic exposures, and in the Yakama Resident scenario where sweat lodge inhalation of gas-phase nonvolatiles is not evaluated, hexavalent chromium is also a main contributor to HI, but by ingestion and dermal absorption exposure pathways.

Chemical hazard for the CTUIR Resident and Yakama Resident scenarios calculated using the data from monitoring well 199-B5-2 is shown in Table 6-21. The child HI results (below an HI of 2.0) were lower than the adult HI results, which include sweat lodge exposures. Contaminants of potential concern and exposure pathways contributing to HI at well 199-B5-2 are similar to those identified for the entire OU.

A summary of the key exposure pathways and analytes contributing 10% or more to a groundwater HI above 1.0 in well 199-B5-2 for the CTUIR Resident and Yakama Resident scenarios is provided below.

Scenario	Analyte	Pathway	Fraction of Well HI	Well Hazard Index
CTUIR Resident	Hexavalent chromium	waterInhalationSweatLodge	0.51	4.8
	Manganese	waterInhalationSweatLodge	0.16	
	Hexavalent chromium	groundwaterIngestion	0.10	
Yakama Resident	Hexavalent chromium	waterDermalSweatLodge	0.63	2.7
	Hexavalent chromium	groundwaterIngestion	0.19	

6.4 100-K DECISION AREA RESULTS

The location of the groundwater monitoring wells evaluated in the 100-K Area and the boundaries of key groundwater plumes in this area are shown in Figure 6-3. A summary of the cancer risk, radiation dose, and chemical hazard results for the 100-K Area is provided in this section. Results pertaining to the Resident National Monument/Refuge and Subsistence Farmer exposure scenarios that include domestic exposures to groundwater are presented in Section 6.4.1. Results for the CTUIR Resident and Yakama Resident scenarios include both domestic groundwater exposures and exposure related to use of groundwater in a sweat lodge. The results of these calculations, differentiating domestic and sweat lodge exposures, are shown in Section 6.4.2.

Figure 6-3. Groundwater Well Locations and Key Plumes for the 100-K Area.

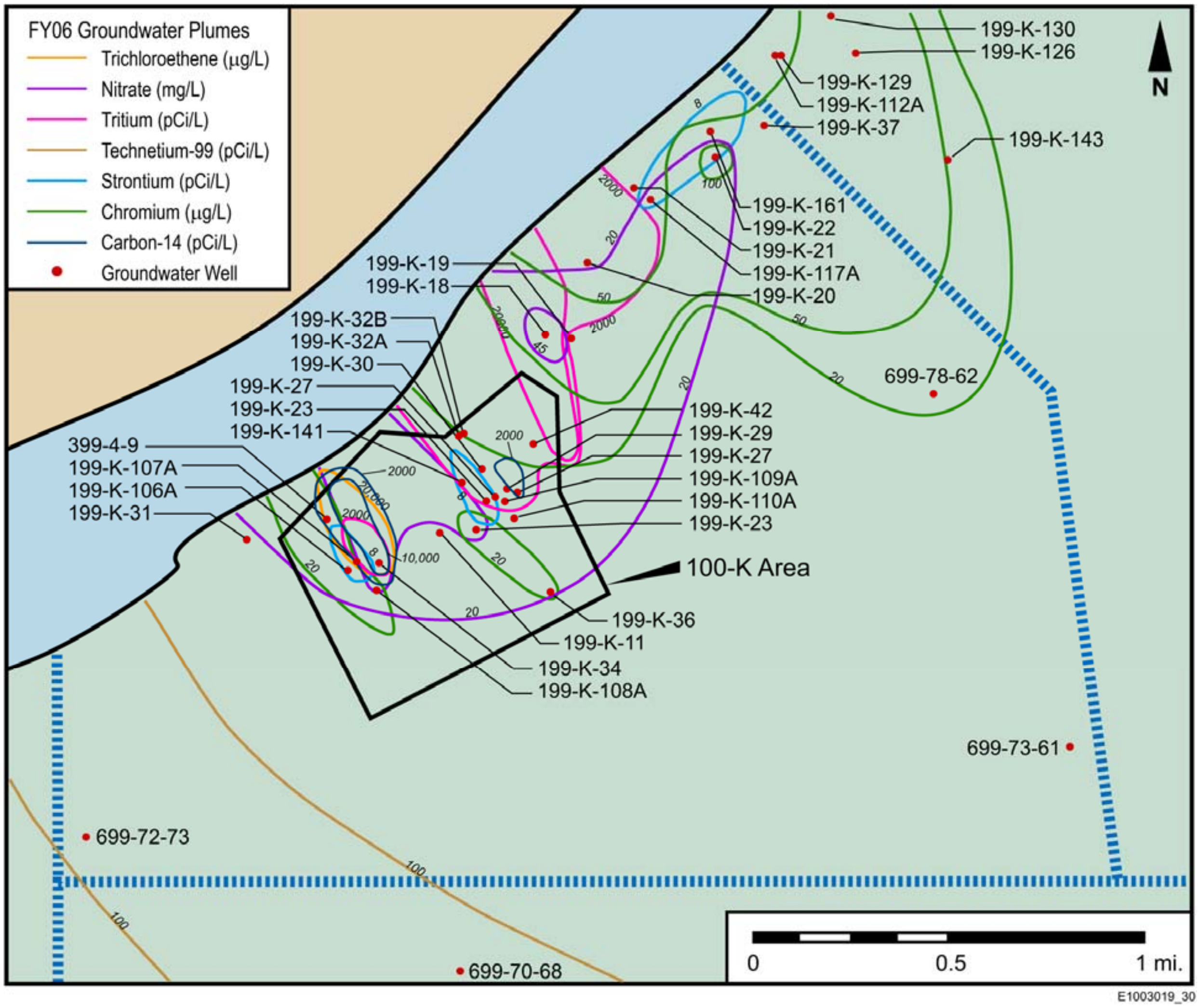


Table 6-16 shows that strontium-90 concentrations are associated with potential cancer risks that are above the 1×10^{-4} threshold in the 100-K Area. Table 6-17 shows that the average strontium-90 concentration in well 199-K-109A is more than 50% higher than the 90th percentile value for the entire OU. This indicates that there is relatively high confidence that risks at this well are higher than those associated with the entire OU. The 50th and 90th percentiles for COPC concentrations in the 100-K Area were recalculated without the data for well 199-K-109A, and a supplemental set of risk calculations are provided for this well. Cancer risk, radiation dose, and chemical hazard results for the 100-K Area do not include the results for well 199-K-109A.

6.4.1 Resident National Monument/Refuge and Subsistence Farmer Results in the 100-K Area

6.4.1.1 Cancer Risk. The RME and CTE groundwater pathways risk results for the Resident National Monument/Refuge scenario (radionuclides + chemicals) in the 100-K Area are 8×10^{-4} and 6×10^{-6}, respectively. The RME and CTE groundwater pathways risk results for the Subsistence Farmer scenario (radionuclides + chemicals) in the 100-K Area are 9×10^{-4} and 4×10^{-6}, respectively. Operable unit groundwater cancer risk results for these scenarios are presented in Table 6-18.

A summary of the key exposure pathways and analytes contributing 10% or more to RME groundwater cancer risk above 1×10^{-4} in the 100-K Area for the Resident National Monument/Refuge and Subsistence Farmer scenarios is provided below.

Scenario	Analyte	Pathway	Fraction of OU Cancer Risk	OU Cancer Risk
Resident National Monument/Refuge	Tritium	groundwaterIngestion	0.48	8×10^{-4}
	Carbon-14	groundwaterIngestion	0.43	
Subsistence Farmer	Tritium	groundwaterIngestion	0.48	9×10^{-4}
	Carbon-14	groundwaterIngestion	0.42	

The OU RME risks for both scenarios are above the 1×10^{-4} upper end of EPA's target cancer risk range, while the CTE risks are within the lower portion of EPA's target cancer risk range. About 50% of the RME cancer risks are associated with tritium via the water ingestion exposure route. About another 40% of the RME risk is related to carbon-14, also via water ingestion. Central tendency exposure OU cancer risks are also related to tritium and carbon-14 via water ingestion. Although future trends in groundwater concentrations have not been quantified in this assessment, natural radioactive decay of tritium (12.3-year half-life) will result in a decrease of risk from this COPC over time compared to present-day groundwater conditions.

Cancer risks for the Resident National Monument/Refuge and Subsistence Farmer scenarios calculated using the data from monitoring well 199-K-109A are shown in Table 6-20. The RME and CTE risk results at well 199-K-109A for the Resident National Monument/Refuge scenario are 8×10^{-3} and 7×10^{-4}, respectively. Subsistence Farmer RME and CTE cancer risks are 8×10^{-3} and 5×10^{-4}, respectively. Almost 100% of the RME and CTE risks for both scenarios at well 199-K-109A are related to strontium-90 by drinking water ingestion. As shown in Table 6-17, this well was identified for supplemental analysis based on strontium-90 concentrations. The mean (2,960 pCi/L) and 95% UCL (5,140 pCi/L) values of strontium-90 in this well differ by less than a factor of two. The difference between the CTE and RME risk results is related more to the difference between CTE and RME values for drinking water ingestion and exposure duration than to uncertainty in the average groundwater concentration at the well.

A summary of the key exposure pathways and analytes contributing 10% or more to RME groundwater cancer risks above 1×10^{-4} in well 199-K-109A for the Resident National Monument/Refuge and Subsistence Farmer scenarios is provided below.

Scenario	Analyte	Pathway	Fraction of Well Cancer Risk	Well Cancer Risk
Resident National Monument/Refuge	Strontium-90	groundwaterIngestion	0.99	8×10^{-3}
Subsistence Farmer	Strontium-90	groundwaterIngestion	0.99	8×10^{-3}

6.4.1.2 Radiation Dose. The RME and CTE groundwater pathways radiation dose results for the Resident National Monument/Refuge scenario in the 100-K Area are 47 and 1.2 mrem/yr, respectively. The RME and CTE groundwater pathways radiation dose results for the Subsistence Farmer scenario in the 100-K Area are 45 and 1.0 mrem/yr, respectively. Groundwater radiation dose results for these scenarios are presented in Table 6-18.

A summary of the key exposure pathways and analytes contributing 10% or more to RME groundwater radiation dose above the 4.0 mrem/yr threshold in the 100-K Area for the Resident National Monument/Refuge and Subsistence Farmer scenarios is provided below.

Scenario	Analyte	Pathway	Fraction of OU Dose	OU Radiation Dose (mrem/yr)
Resident National Monument/Refuge	Tritium	groundwaterIngestion	0.48	47
	Carbon-14	groundwaterIngestion	0.42	
Subsistence Farmer	Tritium	groundwaterIngestion	0.46	45
	Carbon-14	groundwaterIngestion	0.40	

The OU RME radiation dose for both scenarios is more than 10 times higher than the threshold criterion of 4 mrem/yr. As for cancer risk, about 50% of the RME cancer risks are associated with tritium via the water ingestion exposure route and another 40% are related to carbon-14. The large differences between RME and CTE OU dose is mostly due to the large differences between the 50^{th} and 90^{th} percentiles for carbon-14 (540 and 5,400 pCi/L, respectively) and tritium (11,300 and 177,000 pCi/L, respectively) in groundwater at the 100-K Area.

Radiation doses calculated using the data from monitoring well 199-K-109A are shown in Table 6-20. The RME and CTE groundwater pathways radiation dose results for the Resident National Monument/Refuge scenario are 480 and 170 mrem/yr, respectively. The RME and CTE groundwater pathways radiation dose results for the Subsistence Farmer scenario are 630 and 190 mrem/yr, respectively. Almost 100% of the RME and CTE dose for both scenarios at well 199-K-109A is related to strontium-90 by drinking water ingestion.

A summary of the key exposure pathways and analytes contributing 10% or more to RME groundwater radiation dose above the 4.0 mrem/yr threshold in well 199-K-109A for the Resident National Monument/Refuge and Subsistence Farmer scenarios is provided below.

Scenario	Analyte	Pathway	Fraction of Well Dose	Well Radiation Dose (mrem/yr)
Resident National Monument/Refuge	Strontium-90	groundwaterIngestion	0.99	480
Subsistence Farmer	Strontium-90	groundwaterIngestion	0.99	630

6.4.1.3 Chemical Hazard. As described in Section 3.6.2, chemical HI is calculated separately for adult and child receptors in the residential scenarios. Reasonable maximum exposure child HI values tend to exceed those for adults, while for the CTE calculations adult and child HIs tend to be approximately equivalent with adult values for water ingestion, slightly higher than those for young children due to the higher adult ingestion rate. The higher of the adult and child HI values are shown in Table 6-18 and described here.

The RME and CTE chemical HI results for the Resident National Monument/Refuge scenario in the 100-K Area are 3.2 and 0.62, respectively. The RME and CTE child chemical HI results for the Subsistence Farmer scenario in the 100-K Area are 5.6 and 0.64, respectively.

A summary of the key exposure pathways and analytes contributing 10% or more to an RME groundwater HI above 1.0 in the 100-K Area for the Resident National Monument/Refuge and Subsistence Farmer scenarios is provided below.

Scenario	Analyte	Pathway	Fraction of OU HI	OU Hazard Index
Resident National Monument/Refuge	Uranium	groundwaterIngestion	0.34	3.2
	Hexavalent chromium	groundwaterIngestion	0.32	
	Hexavalent chromium	groundwaterDermal	0.12	
	Nitrogen in nitrate	groundwaterIngestion	0.11	
Subsistence Farmer	Uranium	groundwaterIngestionChild	0.32	5.6
	Hexavalent chromium	groundwaterIngestionChild	0.30	
	Hexavalent chromium	groundwaterDermalChild	0.18	
	Nitrogen in nitrate	groundwaterIngestionChild	0.11	

About 40% to 50% of the RME HI for the Resident National Monument/Refuge and Subsistence Farmer scenarios in the 100-K Area is due to exposure to hexavalent chromium by ingestion and dermal absorption. Another 30% is due to exposure to uranium via drinking water ingestion. About 10% of HI is due to ingestion of nitrate in drinking water. The oral RfD for chromium is not related to any specific adverse effect, but the effects of nitrate are very specific and relate to inhibition of oxygen binding with hemoglobin in the blood. The uranium oral RfD is based on toxic effects in the kidney. It is unlikely that additive effects would be observed for these three COPCs. The HI results probably overstate hazard for this reason.

Chemical hazard calculated using the data from monitoring well 199-K-109A is shown in Table 6-20. The RME and CTE HI results at well 199-K-109A for the Resident National

Monument/Refuge scenario are 1.5 and 0.37, respectively. Subsistence Farmer RME and CTE cancer risk are 2.5 and 0.37, respectively. For both scenarios, RME HI is related to nickel (about 30%), uranium (about 25%), and iron (about 20%) by drinking water ingestion. The oral reference dose values for chronic exposure to nickel and iron are not specific, but uranium toxicity is specific to the kidney. Additive effects related to the toxicity criteria would not be anticipated, and the HI is likely biased high as a result.

A summary of the key exposure pathways and analytes contributing 10% or more to an RME groundwater HI above 1.0 in well 199-K-109A for the Resident National Monument/Refuge and Subsistence Farmer scenarios is provided below.

Scenario	Analyte	Pathway	Fraction of Well HI	Well Hazard Index
Resident National Monument/Refuge	Nickel	groundwaterIngestion	0.33	1.5
	Uranium (inorganic)	groundwaterIngestion	0.24	
	Iron	groundwaterIngestion	0.17	
Subsistence Farmer	Nickel	groundwaterIngestionChild	0.32	2.5
	Uranium (inorganic)	groundwaterIngestionChild	0.24	
	Iron	groundwaterIngestionChild	0.17	

6.4.2 CTUIR Resident and Yakama Resident Results in the 100-K Area

The risk assessment results presented in this section apply the parameter values and models documented in Harris and Harper (2004) and Ridolfi (2007). As discussed in Section 6.1, risks related to inhalation of volatile and semivolatile COPCs in sweat lodge air are calculated for both the CTUIR and Yakama Resident scenarios, but inhalation risks for nonvolatile COPCs are only calculated for the CTUIR Resident. Uncertainty and bias in the sweat lodge exposure models are discussed in Section 6.9.

6.4.2.1 Cancer Risk. The groundwater pathways risk results for the CTUIR Resident and Yakama Resident scenarios (radionuclides + chemicals) in the 100-K Area are 1×10^{-2} and greater than 1×10^{-2}, respectively. Sweat lodge exposure dominates total cancer risk in both the CTUIR Resident and Yakama Resident scenarios. Groundwater cancer risk results for these scenarios are presented in Table 6-19.

A summary of the key exposure pathways and analytes contributing 10% or more to groundwater cancer risk above 1×10^{-4} in the 100-K Area for the CTUIR Resident and Yakama Resident scenarios is provided below.

Scenario	Analyte	Pathway	Fraction of OU Cancer Risk	OU Cancer Risk
CTUIR Resident	Hexavalent chromium	waterInhalationSweatLodge	0.50	1×10^{-2}
	Carbon-14	waterInhalationSweatLodge	0.16	
	Tritium	groundwaterIngestion	0.11	
Yakama Resident	Carbon-14	waterInhalationSweatLodge	0.69	$>1 \times 10^{-2}$
	Tritium	waterInhalationSweatLodge	0.11	

Inhalation of hexavalent chromium in the sweat lodge is the primary contributor to cancer risk in the CTUIR Resident scenario. In CTUIR domestic exposures, and in the Yakama Resident scenario where sweat lodge inhalation of gas-phase nonvolatiles is not evaluated, inhalation and/or ingestion of carbon-14 and tritium are the main contributors to cancer risk in these scenarios. Carbon-14 is protectively assumed to exist as dissolved carbon dioxide gas in groundwater. The higher risk related to sweat lodge exposure to carbon-14 and tritium in the Yakama Resident scenario is because of the higher daily use cited for the Yakama (7 hr/day; Ridolfi 2007) than for the CTUIR (1 hr/day; Harris and Harper 2004).

Cancer risks for the CTUIR Resident and Yakama Resident scenarios calculated using the data from monitoring well 199-K-109A are shown in Table 6-21. The groundwater pathways risk results for the CTUIR Resident and Yakama Resident scenarios at well 199-K-109A are both greater than 1×10^{-2}. Almost 100% of the cancer risk at this well for both scenarios is related to strontium-90, with water ingestion being the dominant exposure pathway. As shown in Table 6-17, this well was identified for supplemental analysis based on strontium-90 concentrations. Although future trends in groundwater concentrations have not been quantified in this assessment, natural radioactive decay of strontium-90 (28.8-year half-life) will result in a decrease of risk from this COPC over time compared to present-day groundwater concentrations.

A summary of the key exposure pathways and analytes contributing 10% or more to groundwater cancer risk above 1×10^{-4} at well 199-K-109A for the CTUIR Resident and Yakama Resident scenarios is provided below.

Scenario	Analyte	Pathway	Fraction of Well Cancer Risk	Well Cancer Risk
CTUIR Resident	Strontium-90	groundwaterIngestion	0.69	$>1 \times 10^{-2}$
	Strontium-90	waterIngestionSweatLodge	0.22	
Yakama Resident	Strontium-90	groundwaterIngestion	0.74	$>1 \times 10^{-2}$
	Strontium-90	waterIngestionSweatLodge	0.23	

6.4.2.2 Radiation Dose. The groundwater pathways radiation dose results for the CTUIR Resident and Yakama Resident scenarios in the 100-K Area are 290 mrem/yr and 1,200 mrem/yr, respectively. The higher Yakama scenario dose is due to longer daily exposure in the sweat lodge. Groundwater radiation dose results for these scenarios are presented in Table 6-19.

A summary of the key exposure pathways and analytes contributing 10% or more to groundwater radiation dose above the 4.0 mrem/yr threshold in the 100-K Area for the CTUIR Resident and Yakama Resident scenarios is provided below.

Scenario	Analyte	Pathway	Fraction of OU Dose	OU Radiation Dose (mrem/yr)
CTUIR Resident	Carbon-14	waterInhalationSweatLodge	0.49	290
	Tritium	waterInhalationSweatLodge	0.13	
	Tritium	groundwaterIngestion	0.10	
Yakama Resident	Carbon-14	waterInhalationSweatLodge	0.72	1,200
	Tritium	waterInhalationSweatLodge	0.20	

Reasonable maximum exposure radiation dose is associated primarily with carbon-14 (as carbon dioxide gas) and tritium via vapor phase inhalation. Natural radioactive decay of tritium (12.3-year half-life) will result in decreased concentrations of this COPC over time.

Radiation dose for the CTUIR Resident and Yakama Resident scenarios calculated using the data from monitoring well 199-K-109A is shown in Table 6-21. Radiation dose values were 1,700 and 1,400 mrem/yr for the CTUIR Resident and Yakama Resident scenarios, respectively.

A summary of the key exposure pathways and analytes contributing 10% or more to groundwater radiation dose above the 4.0 mrem/yr threshold at well 199-K-109A for the CTUIR Resident and Yakama Resident scenarios is provided below.

Scenario	Analyte	Pathway	Fraction of Well Dose	Well Radiation Dose (mrem/yr)
CTUIR Resident	Strontium-90	groundwaterIngestion	0.55	1,700
	Strontium-90	waterIngestionSweatLodge	0.25	
	Strontium-90	waterInhalationSweatLodge	0.18	
Yakama Resident	Strontium-90	groundwaterIngestion	0.65	1,400
	Strontium-90	waterIngestionSweatLodge	0.29	

6.4.2.3 Chemical Hazard. The chemical HI results for adults in the CTUIR Resident and Yakama Resident scenarios in the 100-K Area are 27 and 10, respectively. The adult HI results are higher than the child HI values because they include contribution from sweat lodge exposure beginning at the age of 6 years.

A summary of the key exposure pathways and analytes contributing 10% or more to a groundwater HI above 1.0 in the 100-K Area for the CTUIR Resident and Yakama Resident scenarios is provided below.

Scenario	Analyte	Pathway	Fraction of OU HI	OU Hazard Index
CTUIR Resident	Manganese	waterInhalationSweatLodge	0.47	27
	Hexavalent chromium	waterInhalationSweatLodge	0.25	
Yakama Resident	Hexavalent chromium	waterDermalSweatLodge	0.45	10
	Uranium	groundwaterIngestion	0.14	
	Hexavalent chromium	groundwaterIngestion	0.13	

Approximately 50% to 60% of the CTUIR Resident adult HI of 27 is due to inhalation exposure to manganese in the sweat lodge. Another 25% to 30% is related to hexavalent chromium by this same pathway. In domestic exposures, and in the Yakama Resident scenario where sweat lodge inhalation of gas-phase nonvolatiles is not evaluated, drinking water ingestion is the dominant exposure pathway, and risks are related mostly to hexavalent chromium and uranium.

Chemical hazard for the CTUIR Resident and Yakama Resident scenarios calculated using the data from monitoring well 199-K-109A is shown in Table 6-21. The adult HI values are much higher than the child values for the CTUIR scenario due to the influence of inhalation of metals in the sweat lodge. The adult HIs for the CTUIR Resident and Yakama Resident scenarios are 39 and 3.2, respectively. The child HI value for the Yakama Resident scenario is 4.5.

A summary of the key exposure pathways and analytes contributing 10% or more to a groundwater HI above 1.0 at well 199-K-109A for the CTUIR Resident (adult) and Yakama Resident (child) scenarios is provided below. The Yakama Resident child HI is related to nickel (about 30%), uranium (about 25%), and iron (about 20%) by drinking water ingestion. As described in Section 6.4.1, it is unlikely that additive effects would be observed for these three COPCs and these HI results probably overstate hazard for this reason.

Scenario	Analyte	Pathway	Fraction of Well HI	Well Hazard Index
CTUIR Resident	Manganese	waterInhalationSweatLodge	0.76	39
	Aluminum	waterInhalationSweatLodge	0.15	
Yakama Resident	Nickel	groundwaterIngestionChild	0.33	4.5
	Uranium	groundwaterIngestionChild	0.24	
	Iron	groundwaterIngestionChild	0.17	

6.5 100-N DECISION AREA RESULTS

The location of the groundwater monitoring wells evaluated in the 100-N Area and the boundaries of key groundwater plumes are shown in Figure 6-4. A summary of the cancer risk, radiation dose, and chemical hazard results for the 100-N Area is provided in this section. Results pertaining to the Resident National Monument/Refuge and Subsistence Farmer scenarios, which include domestic exposures to groundwater, are presented in Section 6.5.1.

Results for the CTUIR Resident and Yakama Resident scenarios include both domestic groundwater exposures and exposure related to use of groundwater in a sweat lodge. The results of these calculations, differentiating between domestic and sweat lodge exposures, are shown in Section 6.5.2.

Table 6-16 shows strontium-90 concentrations are associated with potential cancer risks that are above the 1×10^{-4} threshold in the 100-N Area. Table 6-17 shows that the average strontium-90 concentrations in wells 199-N-46 and 199-N-67 are more than 50% higher than the 90th percentile value for the entire OU. This indicates that there is relatively high confidence that risks at these wells are higher than those associated with the entire OU. The 50th and 90th percentiles for COPC concentrations in the 100-N Area were recalculated without the data for these wells, and a supplemental set of risk calculations are provided for each well. Cancer risk, radiation dose, and chemical hazard results for the 100-N Area do not include the results for wells 199-N-46 and 199-N-67.

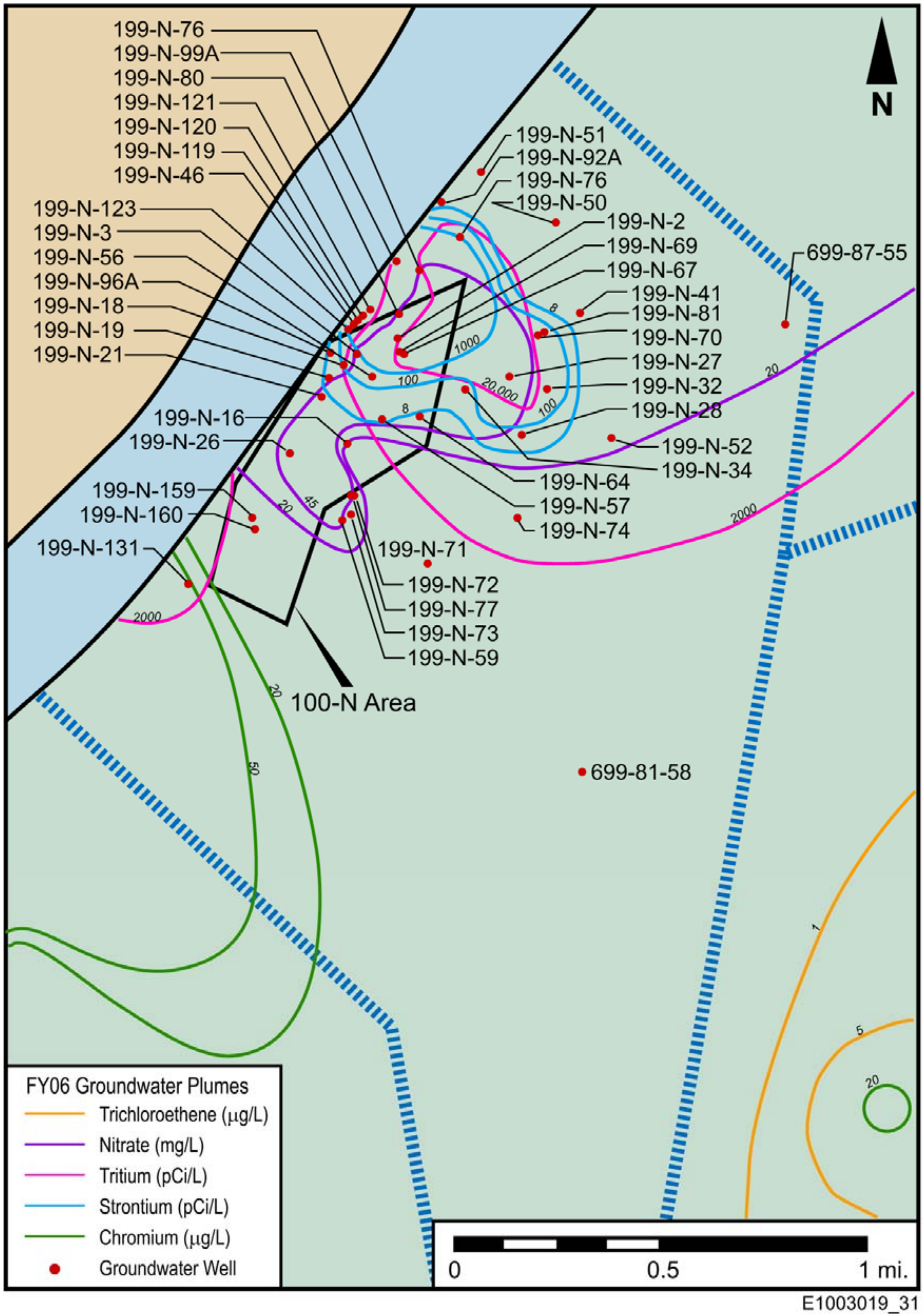
6.5.1 Resident National Monument/Refuge and Subsistence Farmer Results in the 100-N Area

6.5.1.1 Cancer Risk. The RME and CTE groundwater pathways risk results for the Resident National Monument/Refuge scenario (radionuclides + chemicals) in the 100-N Area are 2×10^{-3} and 1×10^{-5} , respectively. The RME and CTE groundwater pathways risk results for the Subsistence Farmer scenario (radionuclides + chemicals) in the 100-N Area are 2×10^{-3} and 9×10^{-6} , respectively. Operable unit groundwater cancer risk results for these scenarios are presented in Table 6-18.

A summary of the key exposure pathways and analytes contributing 10% or more to RME groundwater cancer risk above 1×10^{-4} in the 100-N Area for the Resident National Monument/ Refuge and Subsistence Farmer scenarios is provided below.

Scenario	Analyte	Pathway	Fraction of OU Cancer Risk	OU Cancer Risk
Resident National Monument/Refuge	Strontium-90	groundwaterIngestion	0.97	2×10^{-3}
Subsistence Farmer	Strontium-90	groundwaterIngestion	0.97	2×10^{-3}

Figure 6-4. Groundwater Well Locations and Key Plumes for the 100-N Area.



The OU RME risks for both scenarios are more than a factor of 10 above the 10^{-4} upper end of EPA's target cancer risk range, while the CTE risks are within the lower portion of EPA's target cancer risk range of 10^{-6} to 10^{-4}. More than 95% of the RME cancer risk is associated with strontium-90 via the water ingestion exposure route. There is approximately a 200-fold difference between the RME and CTE OU cancer risks. This is due to a 30-fold difference between the 90th percentile (1,220 pCi/L) and 50th percentile (41.5 pCi/L) groundwater concentrations of this COPC in the 100-N Area and about a 7-fold difference in exposure intensity between the CTE and RME calculations. Although future trends in groundwater concentrations have not been quantified in this assessment, natural radioactive decay of strontium-90 (28.8-year half-life) will result in a decrease of risk from this COPC over time compared to present-day groundwater concentrations.

Cancer risks calculated using the data from monitoring wells 199-N-46 and 199-N-67 are shown in Table 6-20. At wells 199-N-46 and 199-N-67, Resident National Monument/Refuge and Subsistence Farmer RME cancer risks for strontium-90 are 1×10^{-2} and greater than 1×10^{-2}, respectively. The Resident National Monument/Refuge CTE risks are 1×10^{-3} and 2×10^{-3} at wells 199-N-46 and 199-N-67, respectively. The equivalent Subsistence Farmer CTE scenario risks are 9×10^{-4} and 2×10^{-3}, respectively. Both RME and CTE cancer risk for strontium-90 at wells 199-N-46 and 199-N-67 are above the 10^{-4} upper end of EPA's target cancer risk range. One hundred percent of the CTE and RME cancer risks at both wells are due to strontium-90 by drinking water ingestion.

A summary of the key exposure pathways and analytes contributing 10% or more to RME groundwater cancer risk above 1×10^{-4} in wells 199-N-46 and 199-N-67 for the Resident National Monument/Refuge and Subsistence Farmer scenarios is provided below.

Scenario	Well	Analyte	Pathway	Fraction of Well Cancer Risk	Well Cancer Risk
Resident National Monument/Refuge	199-N-46	Strontium-90	groundwaterIngestion	1.0	$>1 \times 10^{-2}$
	199-N-67	Strontium-90	groundwaterIngestion	1.0	
Subsistence Farmer	199-N-46	Strontium-90	groundwaterIngestion	1.0	1×10^{-2}
	199-N-67	Strontium-90	groundwaterIngestion	1.0	$>1 \times 10^{-2}$

6.5.1.2 Radiation Dose. The RME and CTE groundwater pathways radiation dose results for the Resident National Monument/Refuge scenario in the 100-N Area are 120 mrem/yr and 3.1 mrem/yr, respectively. The RME and CTE groundwater pathways radiation dose results for the Subsistence Farmer scenario in the 100-N Area are 150 and 3.4 mrem/yr, respectively. Operable unit groundwater radiation dose results for these scenarios are tabulated in Table 6-18.

A summary of the key exposure pathways and analytes contributing 10% or more to groundwater radiation dose above the 4.0 mrem/yr threshold in the 100-N Area for the Resident National Monument/Refuge and Subsistence Farmer scenarios is provided below.

Scenario	Analyte	Pathway	Fraction of OU Dose	OU Radiation Dose (mrem/yr)
Resident National Monument/Refuge	Strontium-90	groundwaterIngestion	0.97	120
Subsistence Farmer	Strontium-90	groundwaterIngestion	0.98	150

Central tendency exposure dose results for the 100-N OU are below the 4 mrem/yr threshold, but the RME results are far above this threshold. Operable unit RME radiation dose is almost entirely associated with strontium-90 via the water ingestion exposure route. The difference between RME and CTE OU radiation dose is smaller than for cancer risk because the dose calculation does not incorporate differences in the RME and CTE assumptions of exposure duration. Natural radioactive decay of strontium-90 (28.8-year-half-life) will result in a decrease in dose from this COPC over time compared to present-day groundwater conditions.

Radiation doses calculated using the data from monitoring wells 199-N-46 and 199-N-67 are shown in Table 6-20. Reasonable maximum exposure and CTE radiation dose results at wells 199-N-46 and 199-N-67 for the Resident National Monument/Refuge scenario are 820 mrem/yr and 300 mrem/yr and 1,100 mrem/yr and 570 mrem/yr, respectively. Subsistence Farmer RME and CTE radiation doses at these wells are 1,100 mrem/yr and 350 mrem/yr and 1,500 mrem/yr and 660 mrem/yr, respectively. One hundred percent of the CTE and RME radiation dose at both wells is due to strontium-90 by drinking water ingestion. The range of RME and CTE results is less than a factor of three, reflecting the relatively small differences between the mean and 95% UCL values for strontium-90 at these wells. The difference between the CTE and RME annual dose is mostly due to the difference in CTE and RME assumptions for annual drinking water consumption.

A summary of the key exposure pathways and analytes contributing 10% or more to groundwater radiation dose above the 4.0 mrem/yr threshold in wells 199-N-46 and 199-N-67 for the Resident National Monument/Refuge and Subsistence Farmer scenarios is provided below.

Scenario	Well	Analyte	Pathway	Fraction of Well Dose	Well Radiation Dose (mrem/yr)
Resident National Monument/Refuge	199-N-46	Strontium-90	groundwaterIngestion	1.0	820
	199-N-67	Strontium-90	groundwaterIngestion	1.0	1,100
Subsistence Farmer	199-N-46	Strontium-90	groundwaterIngestion	1.0	1,100
	199-N-67	Strontium-90	groundwaterIngestion	1.0	1,500

6.5.1.3 Chemical Hazard. As described in Section 3.6.2, chemical HI is calculated separately for adult and child receptors in the residential scenarios. Reasonable maximum exposure child HI values tend to exceed those for adults, for the CTE calculations for adult and child HIs tend to be approximately equivalent to adult values for water ingestion, slightly higher than those for young children due to the higher adult ingestion rate. The higher of the adult and child HI values are shown in Table 6-18 and described here.

The RME and CTE chemical HI results for the Resident National Monument/Refuge scenario in the 100-N Area are 1.6 and 0.21, respectively. The RME and CTE child chemical HI results for the Subsistence Farmer scenario in the 100-N Area are 2.7 and 0.21, respectively.

A summary of the key exposure pathways and analytes contributing 10% or more to a groundwater HI above 1.0 in the 100-N Area for the Resident National Monument/Refuge and Subsistence Farmer scenarios is provided below.

Scenario	Analyte	Pathway	Fraction of OU HI	OU Hazard Index
Resident National Monument/Refuge	Fluoride	groundwaterIngestionChild	0.24	2.7
	Hexavalent chromium	groundwaterIngestionChild	0.21	
	Nitrogen in nitrate	groundwaterIngestionChild	0.17	
	Hexavalent chromium	groundwaterDermalChild	0.12	
	Uranium	groundwaterIngestionChild	0.11	
Subsistence Farmer	Fluoride	groundwaterIngestion	0.25	1.6
	Hexavalent chromium	groundwaterIngestion	0.22	
	Nitrogen in nitrate	groundwaterIngestion	0.18	
	Uranium	groundwaterIngestion	0.12	

Hazard index for the Resident National Monument/Refuge and Subsistence Farmer scenarios is based on exposures to fluoride, hexavalent chromium, nitrate, and uranium, mostly by ingestion of drinking water. The critical effect underlying the oral RfD for fluoride is discoloration of the teeth, with higher doses having effects in the skeletal system. Nitrate toxicity relates to inhibition of oxygen binding to hemoglobin in the blood. No specific adverse effects were noted in the Integrated Risk Information System (EPA 2008a) studies from which the oral RfD for chromium was derived. Uranium toxicity is manifested in the kidney. Because the toxic effects of nitrate, uranium, and fluoride are quite specific and involve different targets, it is unlikely that additive effects would exist for these COPCs. The HI results are probably an overestimate of actual chemical hazards.

Chemical hazard calculated using the data from monitoring wells 199-N-46 and 199-N-67 are shown in Table 6-20. RME and CTE HI results at wells 199-N-46 and 199-N-67 for the Resident National Monument/Refuge scenario are 0.96 and 0.52 and 1.5 and 0.70, respectively. Subsistence Farmer RME and CTE HI values at these wells are 1.6 and 0.52 and 2.5 and 0.70, respectively. Nitrate is the main contributor to HI values above 1.0 at well 199-N-67, but manganese is the main contributor at well 199-N-46. The oral RfD for iron is not related to any specific adverse effect, but the effects of nitrate are very specific and relate to inhibition of oxygen binding with hemoglobin in the blood. Additive effects related to the toxicity criteria for these COPCs are unlikely. Neurological effects are the critical toxic results of manganese exposure. Additivity of effects from manganese and nitrate are also unlikely.

A summary of the key exposure pathways and analytes contributing 10% or more to a groundwater HI above 1.0 in wells 199-N-46 and 199-N-67 for the Resident National Monument/Refuge and Subsistence Farmer scenarios is provided below.

Scenario	Well	Analyte	Pathway	Fraction of Well HI	Well Hazard Index
Resident National Monument/Refuge	199-N-67	Nitrogen in nitrate	groundwaterIngestion	0.71	1.5
		Iron	groundwaterIngestion	0.11	
Subsistence Farmer	199-N-46	Manganese	groundwaterIngestionChild	0.54	1.6
		Nitrogen in nitrite	groundwaterIngestionChild	0.12	
	199-N-67	Nitrogen in nitrate	groundwaterIngestionChild	0.71	2.5
		Iron	groundwaterIngestionChild	0.11	

6.5.2 CTUIR Resident and Yakama Resident Results in the 100-N Area

The risk assessment results presented in this section apply the parameter values and models documented in Harris and Harper (2004) and Ridolfi (2007). As discussed in Section 6.1, risks related to inhalation of volatile and semivolatile COPCs in sweat lodge air are calculated for both the CTUIR and Yakama Resident scenarios, but inhalation risks for nonvolatile COPCs are only calculated for the CTUIR Resident. Uncertainty and bias in the sweat lodge exposure models are discussed in Section 6.9.

6.5.2.1 Cancer Risk. The groundwater pathways risk results for the CTUIR Resident and Yakama Resident scenarios (radionuclides + chemicals) in the 100-N Area are 1×10^{-2} and 9×10^{-3}, respectively. Domestic and sweat lodge exposures are of approximately equal importance in the CTUIR Resident scenario, but domestic exposure is more dominant in the Yakama Resident scenario. Groundwater cancer risk results for these scenarios are presented in Table 6-19.

A summary of the key exposure pathways and analytes contributing 10% or more to groundwater cancer risk above 1×10^{-4} in the 100-N Area for the CTUIR Resident and Yakama Resident scenarios is provided below.

Scenario	Analyte	Pathway	Fraction of OU Cancer Risk	OU Cancer Risk
CTUIR Resident	Strontium-90	groundwaterIngestion	0.54	1×10^{-2}
	Hexavalent chromium	waterInhalationSweatLodge	0.19	
	Strontium-90	waterIngestionSweatLodge	0.17	

Scenario	Analyte	Pathway	Fraction of OU Cancer Risk	OU Cancer Risk
Yakama Resident	Strontium-90	groundwaterIngestion	0.71	9×10^{-3}
	Strontium-90	waterIngestionSweatLodge	0.23	

Strontium-90 is a major contributor to cancer risk in both the CTUIR Resident and Yakama Resident scenarios. Although future trends in groundwater concentrations have not been quantified in this assessment, natural radioactive decay of strontium-90 (28.8-year half-life) will result in a decrease of risk from this COPC over time compared to present-day groundwater conditions.

Cancer risks for the CTUIR Resident and Yakama Resident scenarios calculated using the data from monitoring wells 199-N-46 and 199-N-67 are shown in Table 6-21. The groundwater pathways risk results for the CTUIR Resident and Yakama Resident scenarios at wells 199-N-46 and 199-N-67 are both greater than 1×10^{-2}. Almost 100% of the cancer risk at both wells for both scenarios is related to strontium-90, with drinking water ingestion being the dominant exposure pathway. A discussion of the range of applicability of the cancer slope factors is provided in the uncertainty analysis in Section 6.9.3.

A summary of the key exposure pathways and analytes contributing 10% or more to groundwater cancer risk above 1×10^{-4} in wells 199-N-46 and 199-N-67 for the CTUIR Resident and Yakama Resident scenarios is provided below.

Scenario	Well	Analyte	Pathway	Fraction of Well Risk	Well Cancer Risk
CTUIR Resident	199-N-46	Strontium-90	groundwaterIngestion	0.70	$>1 \times 10^{-2}$
		Strontium-90	waterIngestionSweatLodge	0.22	
	199-N-67	Strontium-90	groundwaterIngestion	0.70	$>1 \times 10^{-2}$
		Strontium-90	waterIngestionSweatLodge	0.22	
Yakama Resident	199-N-46	Strontium-90	groundwaterIngestion	0.76	$>1 \times 10^{-2}$
		Strontium-90	waterIngestionSweatLodge	0.24	
	199-N-67	Strontium-90	groundwaterIngestion	0.75	$>1 \times 10^{-2}$
		Strontium-90	waterIngestionSweatLodge	0.24	

6.5.2.2 Radiation Dose. The groundwater pathways radiation dose results for the CTUIR Resident and Yakama Resident scenarios in the 100-N Area are 410 mrem/yr and 360 mrem/yr, respectively. Groundwater radiation dose results for these scenarios are presented in Table 6-19.

A summary of the key exposure pathways and analytes contributing 10% or more to groundwater radiation dose above the 4.0 mrem/yr threshold in the 100-N Area for the CTUIR Resident and Yakama Resident scenarios is provided below.

Scenario	Analyte	Pathway	Fraction of OU Dose	OU Radiation Dose (mrem/yr)
CTUIR Resident	Strontium-90	groundwaterIngestion	0.54	410
	Strontium-90	waterIngestionSweatLodge	0.24	
	Strontium-90	waterInhalationSweatLodge	0.17	
Yakama Resident	Strontium-90	groundwaterIngestion	0.61	360
	Strontium-90	waterIngestionSweatLodge	0.27	

Strontium-90 is the main contributor to radiation dose in both the CTUIR Resident and Yakama Resident scenarios. As described above, natural radioactive decay of strontium-90 (28.8-year half-life) will result in decreased concentrations of this COPC over time.

Table 6-21 shows radiation dose for the CTUIR Resident and Yakama Resident scenarios calculated using the data from monitoring wells 199-N-46 and 199-N-67. Radiation dose values for the CTUIR Resident were 2,800 mrem/yr and 3,800 mrem/yr in wells 199-N-46 and 199-N-67, respectively. For the Yakama Resident, the analogous doses were 2,300 mrem/yr and 3,200 mrem/yr. Domestic exposures were larger contributors to radiation dose than sweat lodge exposures in both the CTUIR Resident and Yakama Resident scenarios.

A summary of the key exposure pathways and analytes contributing 10% or more to groundwater radiation dose above the 4.0 mrem/yr threshold in wells 199-N-46 and 199-N-67 for the CTUIR Resident and Yakama Resident scenarios is provided below.

Scenario	Well	Analyte	Pathway	Fraction of Well Dose	Well Radiation Dose (mrem/yr)
CTUIR Resident	199-N-46	Strontium-90	groundwaterIngestion	0.57	2,800
		Strontium-90	waterIngestionSweatLodge	0.25	
		Strontium-90	waterInhalationSweatLodge	0.18	
	199-N-67	Strontium-90	groundwaterIngestion	0.57	3,800
		Strontium-90	waterIngestionSweatLodge	0.25	
		Strontium-90	waterInhalationSweatLodge	0.18	
Yakama Resident	199-N-46	Strontium-90	groundwaterIngestion	0.69	2,300
		Strontium-90	waterIngestionSweatLodge	0.31	
	199-N-67	Strontium-90	groundwaterIngestion	0.68	3,200
		Strontium-90	waterIngestionSweatLodge	0.30	

6.5.2.3 Chemical Hazard. The child chemical HI results for the CTUIR Resident and Yakama Resident scenarios in the 100-N Area are 4.5 and 4.6, respectively. The adult HI result for the CTUIR Resident, which includes contribution from sweat lodge exposure beginning at the age of 6 years, is higher. Because inhalation risks for nonvolatile COPCs are not calculated for the

Yakama Resident scenario, the contribution of sweat lodge exposure to total adult HI is not as large as in the CTUIR Resident scenario.

A summary of the key exposure pathways and analytes contributing 10% or more to a groundwater adult HI above 1.0 in the 100-N Area for the CTUIR Resident (adult) and Yakama Resident (child) scenarios is provided below.

Scenario	Analyte	Pathway	Fraction of OU HI	OU Hazard Index
CTUIR Resident	Manganese	waterInhalationSweatLodge	0.66	19
	Hexavalent chromium	waterInhalationSweatLodge	0.12	
Yakama Resident	Fluoride	groundwaterIngestionChild	0.26	4.6
	Hexavalent chromium	groundwaterIngestionChild	0.22	
	Nitrogen in nitrate	groundwaterIngestionChild	0.18	
	Uranium	groundwaterIngestionChild	0.12	

Approximately 65 of the CTUIR Resident adult HI values are due to inhalation exposure to manganese in the sweat lodge. Exposure to hexavalent chromium by inhalation contributes approximately another 10 to the adult HI. HI for the Yakama Resident scenario is based on drinking water exposure to fluoride, hexavalent chromium, nitrate, and uranium. As discussed in relation to HI in Section 6.5.1, the critical effects underlying the oral RfD values for these COPCs are not similar. Therefore, the HI results are probably an overestimate of actual chemical hazard.

Chemical hazard calculated using the data from monitoring wells 199-N-46 and 199-N-67 are shown in Table 6-21. Adult HI results at wells 199-N-46 and 199-N-67 for the CTUIR Resident scenario are 110 and 14, respectively. In the Yakama Resident scenario, where inhalation risks for nonvolatile COPCs are not calculated, the adult HI results are less than the child HI values. Child HI results for the Yakama Resident scenario at wells 199-N-46 and 199-N-67 are 2.8 and 4.5, respectively.

A summary of the key exposure pathways and analytes contributing 10% or more to a groundwater HI above 1.0 in wells 199-N-46 and 199-N-67 for the CTUIR Resident and Yakama Resident scenarios is provided below. Manganese inhalation exposure in the sweat lodge is the main contributor to CTUIR Resident adult HI values above 1.0 at wells 199-N-46 and 199-N-67.

Scenario	Well	Analyte	Pathway	Fraction of Well HI	Well Hazard Index
CTUIR Resident	199-N-46	Manganese	waterInhalationSweatLodge	0.97	110
	199-N-67	Manganese	waterInhalationSweatLodge	0.67	14
		Barium	waterInhalationSweatLodge	0.13	
		Nitrogen in nitrate	groundwaterIngestion	0.11	
Yakama Resident	199-N-46	Manganese	groundwaterIngestionChild	0.56	2.8
		Nitrogen in nitrite	groundwaterIngestionChild	0.13	
	199-N-67	Nitrogen in nitrate	groundwaterIngestionChild	0.71	4.5
		Iron	groundwaterIngestionChild	0.11	

6.6 100-D/100-H DECISION AREA RESULTS

The location of the groundwater monitoring wells evaluated in the 100-D Area, 100-H Area, and the combined areas, as well as the boundaries of key groundwater plumes in these areas, are shown in Figures 6-5, 6-6, and 6-7. A summary of the cancer risk, radiation dose, and chemical hazard results for the 100-D/100-H Area is provided in this section. Results pertaining to the Resident National Monument/Refuge and Subsistence Farmer scenarios, which include domestic exposures to groundwater, are presented in Section 6.6.1. Results for the CTUIR Resident and Yakama Resident scenarios include both domestic groundwater exposures and exposure related to use of groundwater in a sweat lodge. The results of these calculations, differentiating between domestic and sweat lodge exposures, are discussed in Section 6.6.2.

Table 6-16 shows that hexavalent chromium concentrations are associated with potential hazard quotients in the 100-D/100-H Area that are more than 10 times the threshold of 1.0. Table 6-17 shows that the average hexavalent chromium concentrations in wells 199-D5-104 and 199-D5-99 are more than 50% higher than the 90th percentile value for the entire OU. The 95% UCL hexavalent chromium concentration in well 199-D5-41 is more than twice the 90th percentile value for the entire OU. This indicates that there is relatively high confidence that risks at these wells are higher than those associated with the entire OU. The 50th and 90th percentiles for COPC concentrations in the 100-D/100-H Area were recalculated without the data for these wells and a supplemental set of risk calculations are provided for each well. Cancer risk, radiation dose, and chemical hazard results for the 100-D/100-H Area do not include the results for wells 199-D5-104, 199-D5-41, and 199-D5-99.

Figure 6-5. Groundwater Well Locations and Key Plumes for the 100-D Area.

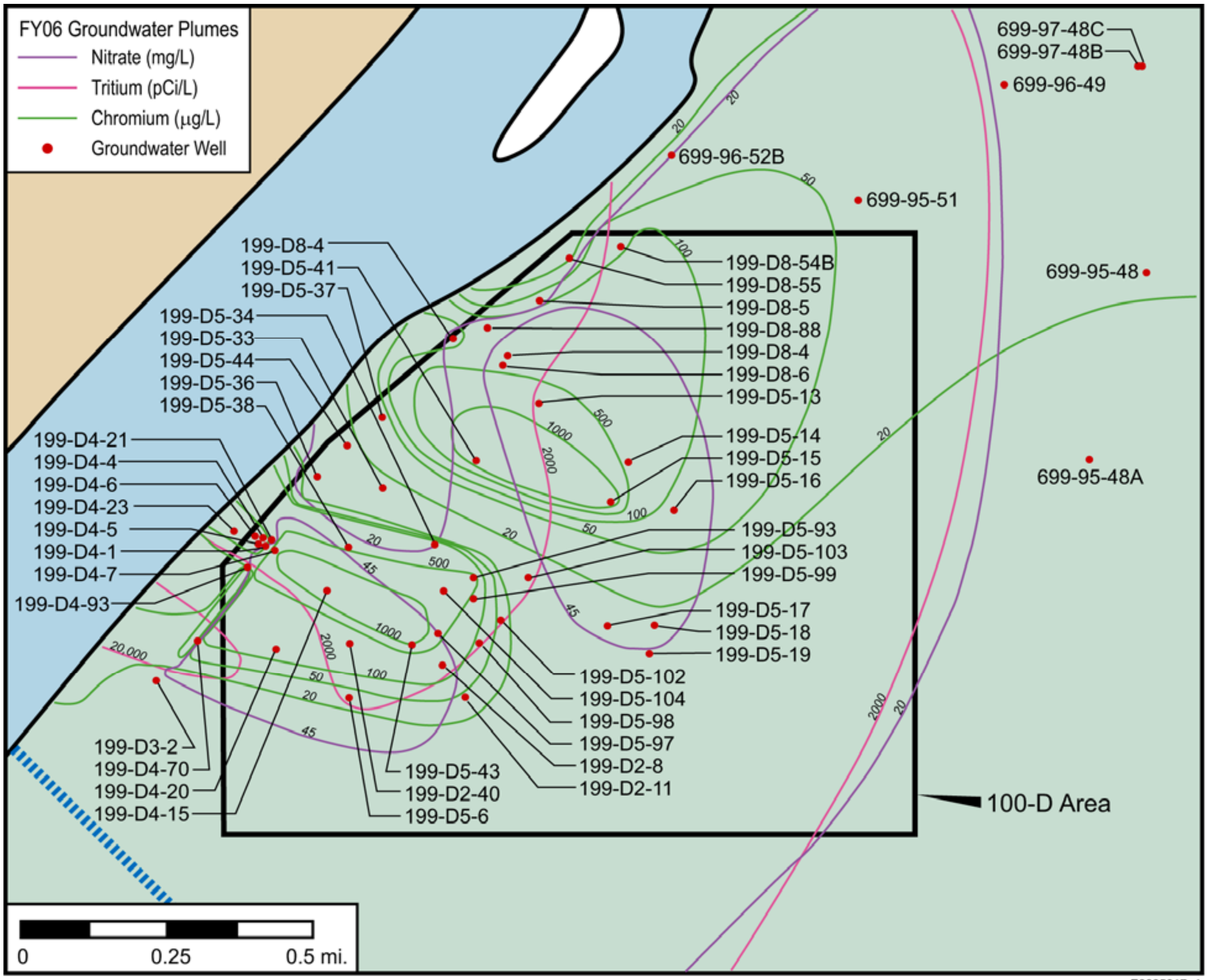


Figure 6-6. Groundwater Well Locations and Key Plumes for the 100-H Area.

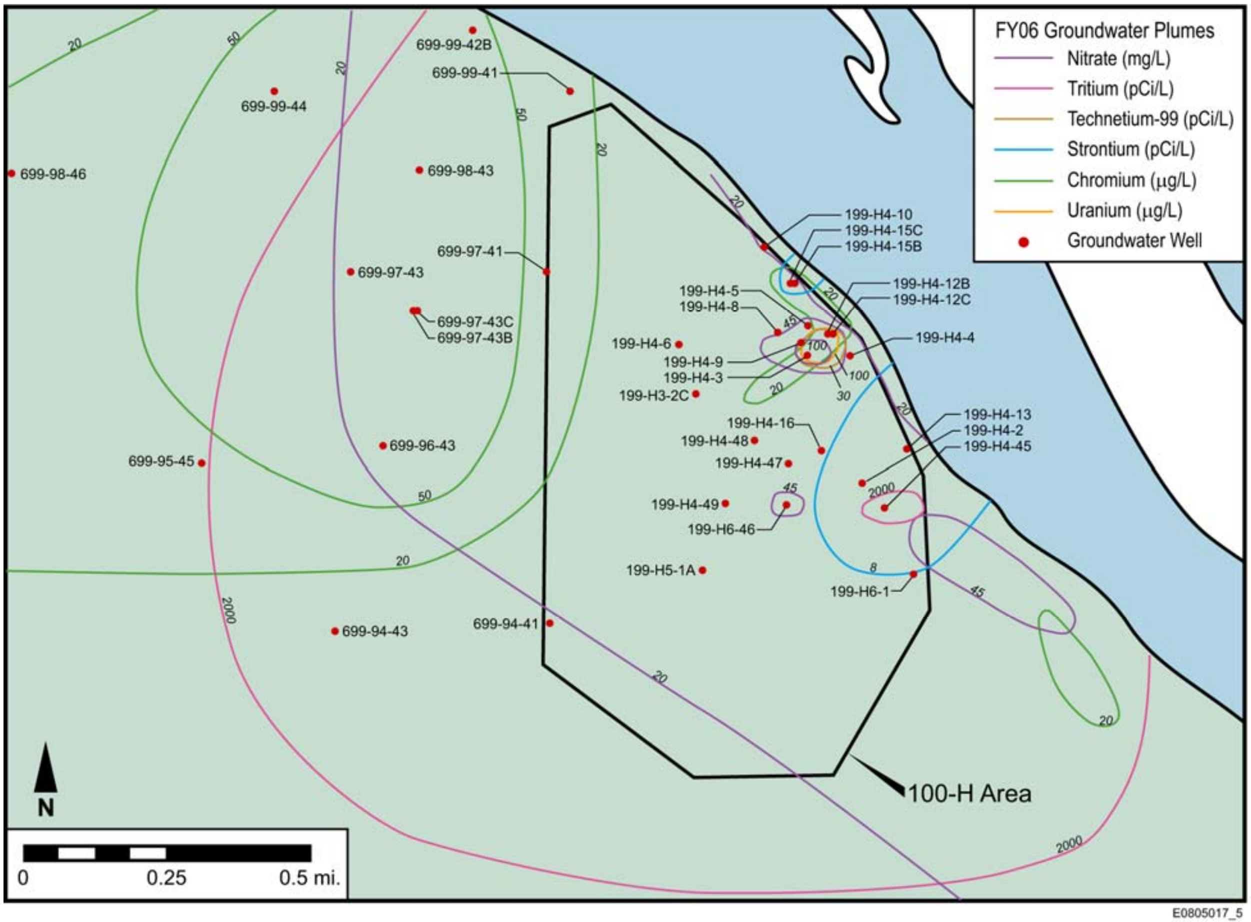
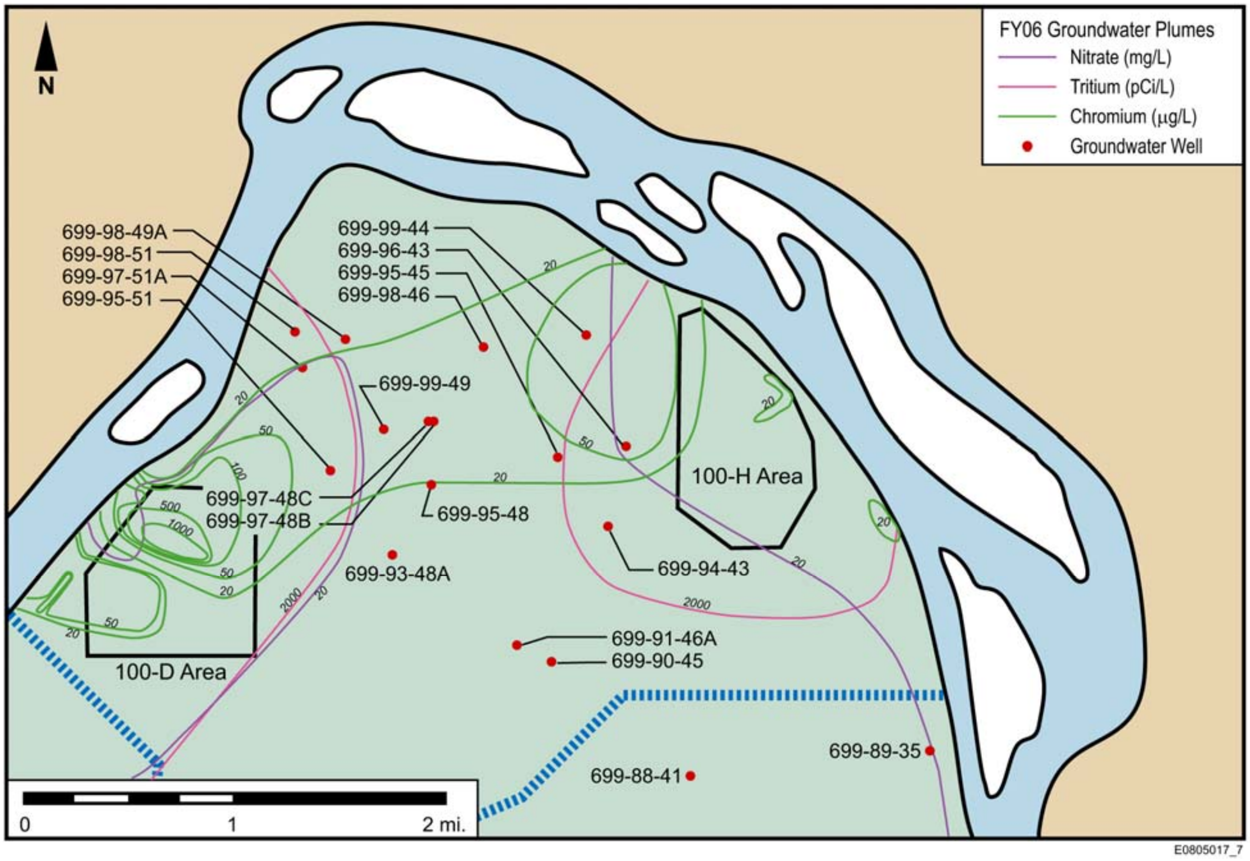


Figure 6-7. Groundwater Well Locations and Key Plumes for the 100-D and 100-H Areas.



6.6.1 Resident National Monument/Refuge and Subsistence Farmer Results in the 100-D/100-H Area

6.6.1.1 Cancer Risk. The RME and CTE groundwater pathways risk results for the Resident National Monument/Refuge scenario (radionuclides + chemicals) in the 100-D/100-H Area are 6×10^{-5} and 2×10^{-6}, respectively. The RME and CTE groundwater pathways risk results for the Subsistence Farmer scenario (radionuclides + chemicals) in the 100-D/100-H Area are 6×10^{-5} and 1×10^{-6}, respectively. Groundwater cancer risk results for these scenarios are presented in Table 6-18. RME and CTE cancer risks for both scenarios are within EPA's risk management range of 1×10^{-6} to 1×10^{-4}. About 65% of the RME cancer risk is related in equal parts to technetium-99 and tritium via drinking water ingestion, with another 25% from strontium-90 via drinking water ingestion.

Cancer risks for the Resident National Monument/Refuge and Subsistence Farmer scenarios calculated using the data from monitoring wells 199-D5-104, 199-D5-41, and 199-D5-99 are shown in Table 6-22. These wells were identified for the supplemental analysis based on concentrations of hexavalent chromium. Carcinogens were only measured at well 199-D5-104.

The RME cancer risk result for both scenarios at this well was 2×10^{-6}. Cancer risk is related to tritium by groundwater ingestion.

6.6.1.2 Radiation Dose. The RME and CTE groundwater pathways radiation dose results for the Resident National Monument/Refuge scenario in the 100-D/100-H Area are 3.2 mrem/yr and 0.39 mrem/yr, respectively. The RME and CTE groundwater pathways radiation dose results for the Subsistence Farmer scenario in the 100-D/100-H Area are 3.8 mrem/yr and 0.36 mrem/yr, respectively. Radiation dose risk results for these scenarios are presented in Table 6-18. Both RME and CTE dose results are near or below the 4 mrem/yr threshold used to assess the significance of the dose results for drinking water.

Radiation dose results for the Resident National Monument/Refuge and Subsistence Farmer scenarios calculated using the data from monitoring wells 199-D5-104, 199-D5-41, and 199-D5-99 are shown in Table 6-22. These wells were identified for the supplemental analysis based on concentrations of hexavalent chromium. Radionuclides were only measured at well 199-D5-104. The RME radiation dose result for both scenarios at this well was 0.12 mrem/yr.

6.6.1.3 Chemical Hazard. As described in Section 3.6.2, chemical HI is calculated separately for adult and child receptors in the residential scenarios. RME child HI values tend to exceed those for adults; for the CTE calculations adult and child HIs tend to be approximately equivalent to adult values for water ingestion, slightly higher than those for young children due to the higher adult ingestion rate. The higher of the adult and child HI values is shown in Table 6-18 and described here.

The RME and CTE chemical HI results for the Resident National Monument/Refuge scenario in the 100-D/100-H Area are 9.1 and 0.50, respectively. The RME and CTE child chemical HI results for the Subsistence Farmer scenario in the 100-D/100-H Area are 17 and 0.52, respectively.

A summary of the key exposure pathways and analytes contributing 10% or more to a groundwater RME HI above 1.0 in the 100-D/100-H Area for the Resident National Monument/ Refuge and Subsistence Farmer scenarios is provided below.

Scenario	Analyte	Pathway	Fraction of OU HI	OU Hazard Index
Resident National Monument/Refuge	Hexavalent chromium	groundwaterIngestion	0.60	9.1
	Hexavalent chromium	groundwaterDermal	0.22	
	Uranium	groundwaterIngestion	0.12	
Subsistence Farmer	Hexavalent chromium	groundwaterIngestionChild	0.53	17
	Hexavalent chromium	groundwaterDermalChild	0.31	
	Uranium	groundwaterIngestionChild	0.10	

The OU RME HI values for both scenarios are above the significance threshold of 1.0, but the results of the CTE calculations are below the threshold. Approximately 80% of the OU RME hazard for both scenarios is related to hexavalent chromium, with exposure by drinking water ingestion being two to three times as important as exposure via dermal absorption from water while bathing. Exposure to uranium in drinking water contributes approximately 10% of the hazard in both scenarios. The oral RfD for chromium is not related to any specific adverse effect, but the uranium oral RfD is based on toxic effects in the kidney. It is unlikely that effects related to the toxicity criteria would be additive for these COPCs.

The RME and CTE HI results for monitoring wells 199-D5-104, 199-D5-41, and 199-D5-99 are shown in Table 6-22. Both CTE and RME results are well above the threshold of 1.0 in each of these wells with the highest values (Subsistence Farmer CTE and RME HIs of 160 and 800) at well 199-D5-99. Essentially, 100% of the OU RME and CTE hazard for both scenarios is related to hexavalent chromium, with exposure by drinking water ingestion being about three times as important as exposure via dermal absorption from water while bathing.

A summary of the key exposure pathways and analytes contributing 10% or more to a groundwater RME HI above 1.0 in wells 199-D5-104, 199-D5-41, and 199-D5-99 for the Resident National Monument/Refuge and Subsistence Farmer scenarios is provided below.

Scenario	Well	Analyte	Pathway	Fraction of Well HI	Well Hazard Index
Resident National Monument/ Refuge	199-D5-104	Hexavalent chromium	groundwaterIngestion	0.73	120
		Hexavalent chromium	groundwaterDermal	0.26	
	199-D5-41	Hexavalent chromium	groundwaterIngestion	0.73	40
		Hexavalent chromium	groundwaterDermal	0.26	
	199-D5-99	Hexavalent chromium	groundwaterIngestion	0.73	420
		Hexavalent chromium	groundwaterDermal	0.27	

Scenario	Well	Analyte	Pathway	Fraction of Well HI	Well Hazard Index
Subsistence Farmer	199-D5-104	Hexavalent chromium	groundwaterIngestionChild	0.63	230
		Hexavalent chromium	groundwaterDermalChild	0.37	
	199-D5-41	Hexavalent chromium	groundwaterIngestionChild	0.63	77
		Hexavalent chromium	groundwaterDermalChild	0.37	
	199-D5-99	Hexavalent chromium	groundwaterIngestionChild	0.63	800
		Hexavalent chromium	groundwaterDermalChild	0.37	

6.6.2 CTUIR Resident and Yakama Resident Results in the 100-D/100-H Area

The risk assessment results presented in this section apply the parameter values and models documented in Harris and Harper (2004) and Ridolfi (2007). As discussed in Section 6.1, risks related to inhalation of volatile and semivolatile COPCs in sweat lodge air are calculated for both the CTUIR and Yakama Resident scenarios, but inhalation risks for nonvolatile COPCs are only calculated for the CTUIR Resident. Uncertainty and bias in the sweat lodge exposure models are discussed in Section 6.9.

6.6.2.1 Cancer Risk. The groundwater pathways risk results (radionuclides + chemicals) in the 100-D/100-H Area are greater than 1×10^{-2} for the CTUIR Resident scenario and 4×10^{-4} for the Yakama Resident scenario. Sweat lodge inhalation exposure to hexavalent chromium dominates the high cancer risk for the CTUIR Resident scenario. Groundwater cancer risk results for these scenarios are presented in Table 6-19.

A summary of the key exposure pathways and analytes contributing 10% or more to groundwater cancer risk above 1×10^{-4} in the 100-D/100-H Area for the CTUIR Resident and Yakama Resident scenarios is provided below.

Scenario	Analyte	Pathway	Fraction of OU Cancer Risk	OU Cancer Risk
CTUIR Resident	Hexavalent chromium	waterInhalationSweatLodge	0.98	$>1 \times 10^{-2}$
Yakama Resident	Tritium	waterInhalationSweatLodge	0.27	4×10^{-4}
	Tritium	groundwaterIngestion	0.19	
	Technetium-99	groundwaterIngestion	0.18	
	Strontium-90	groundwaterIngestion	0.13	

Inhalation exposure to hexavalent chromium in the sweat lodge accounts for effectively 100% of the hypothetical cancer risk for the CTUIR Resident scenario. The cancer risk results for the Yakama Resident scenario are different because inhalation risks for nonvolatile COPCs like hexavalent chromium are not calculated. About 50% of the Yakama Resident cancer risk is due to tritium exposure. Although future trends in groundwater concentrations have not been quantified in this assessment, natural radioactive decay of tritium (12.3-year half-life) will result in a decrease of risk from this COPC over time compared to present-day groundwater concentrations.

Cancer risks for the CTUIR Resident and Yakama Resident scenarios calculated using the data from monitoring wells 199-D5-104, 199-D5-41, and 199-D5-99 are shown in Table 6-23. These wells were identified for the supplemental analysis based on concentrations of hexavalent chromium. The reason Table 6-23 only shows CTUIR Resident cancer risk results for sweat lodge exposure at wells 199-D5-104 and 199-D5-99 is that hexavalent chromium is the only carcinogenic COPC at these wells. Hexavalent chromium is carcinogenic by the inhalation exposure route, and inhalation of nonvolatile COPCs from water is only calculated for sweat lodge exposures for the CTUIR Resident scenario. The groundwater pathways risk results for the CTUIR Resident scenario at all three wells are greater than 1×10^{-2}.

A summary of the key exposure pathways and analytes contributing 10% or more to a groundwater HI above 1.0 in wells 199-D5-104, 199-D5-41, and 199-D5-99 for the CTUIR Resident scenario is provided below. No carcinogenic COPCs other than hexavalent chromium were identified in wells 199-D5-104 and 199-D5-99. The Yakama Resident cancer risk at well 199-D5-41 was 2×10^{-5} and is related to tritium exposure.

Scenario	Well	Analyte	Pathway	Fraction of OU Cancer Risk	OU Cancer Risk
CTUIR Resident	199-D5-104	Hexavalent chromium	waterInhalationSweatLodge	1.0	$>1 \times 10^{-2}$
	199-D5-41	Hexavalent chromium	waterInhalationSweatLodge	1.0	$>1 \times 10^{-2}$
	199-D5-99	Hexavalent chromium	waterInhalationSweatLodge	1.0	$>1 \times 10^{-2}$

6.6.2.2 Radiation Dose. The groundwater pathways radiation dose results for the CTUIR Resident and Yakama Resident scenarios in the 100-D/100-H Area are 26 mrem/yr and 20 mrem/yr, respectively. Groundwater radiation dose results for these scenarios are presented in Table 6-19. Exposures in the sweat lodge were greater than domestic exposures in both scenarios.

A summary of the key exposure pathways and analytes contributing 10% or more to groundwater radiation dose above the 4.0 mrem/yr threshold in the 100-D/100-H Area for the CTUIR Resident and Yakama Resident scenarios is provided below.

Scenario	Analyte	Pathway	Fraction of OU Dose	OU Radiation Dose (mrem/yr)
CTUIR Resident	Uranium-233/234	waterInhalationSweatLodge	0.34	26
	Uranium-238	waterInhalationSweatLodge	0.18	
Yakama Resident	Tritium	waterInhalationSweatLodge	0.60	20

Exposure to uranium isotopes by inhalation in the sweat lodge is the main contributor to radiation dose in the CTUIR Resident scenario. In the Yakama Resident scenario, where inhalation of nonvolatile uranium isotopes is not modeled, inhalation exposure to tritiated water vapor lodge is the main contributor to radiation dose.

Radiation dose for the CTUIR Resident and Yakama Resident scenarios calculated using the data from monitoring wells 199-D5-104, 199-D5-41, and 199-D5-99 are shown in Table 6-23. These wells were identified for the supplemental analysis based on concentrations of hexavalent chromium. Radionuclides were only measured at well 199-D5-104. Radiation dose values for the CTUIR Resident and Yakama Resident were 0.45 mrem/yr and 1.5 mrem/yr at well 199-D5-104.

6.6.2.3 Chemical Hazard. The chemical HI results for a child in the CTUIR Resident and Yakama Resident scenarios in the 100-D/100-H Area are 26 and 27, respectively. The adult HI, which includes exposure beginning at the age of 6 years in the sweat lodge, is higher in both scenarios.

A summary of the key exposure pathways and analytes contributing 10% or more to an adult groundwater HI above 1.0 in the 100-D/100-H Area for the CTUIR Resident and Yakama Resident scenarios is provided below.

Scenario	Analyte	Pathway	Fraction of OU HI	OU Hazard Index
CTUIR Resident	Hexavalent chromium	waterInhalationSweatLodge	0.58	62
	Hexavalent chromium	groundwaterIngestion	0.12	
Yakama Resident	Hexavalent chromium	waterDermalSweatLodge	0.62	40
	Hexavalent chromium	groundwaterIngestion	0.19	

Approximately 70% of the CTUIR Resident adult HI result is due to hexavalent chromium exposure, primarily by inhalation exposure in the sweat lodge. Sweat lodge risks are also the major contributor to adult HI in the Yakama Resident scenario, with dermal exposure to groundwater the dominant exposure pathway. The calculation of dermal sweat lodge exposure to nonvolatile COPCs in water is based on summation of dissolved and vapor-phase COPC concentrations (Harris and Harper 2004, Appendix 4). In the Yakama Resident scenario, vapor-phase concentrations of nonvolatile COPCs were not calculated and only COPC concentrations in water were used in the dermal risk calculation. It is presumed that exposure may occur if water is directly poured over the body during the intervals when new heated rocks are brought into the lodge.

Chemical hazard calculated using the data from monitoring wells 199-D5-104, 199-D5-41, and 199-D5-99 is shown in Table 6-23. Adult HI results at wells 199-D5-104, 199-D5-41, and 199-D5-99 for the CTUIR Resident scenario are 820, 280, and 2,900, respectively. Yakama Resident adult HI results at these wells are 590, 200, and 2,100, respectively. Adult HI results are higher due to the influence of exposures in the sweat lodge.

A summary of the key exposure pathways and analytes contributing 10% or more to a groundwater HI above 1.0 in wells 199-D5-104, 199-D5-41, and 199-D5-99 for the CTUIR Resident and Yakama Resident scenarios is provided below.

Scenario	Well	Analyte	Pathway	Fraction of Well HI	Well Hazard Index
CTUIR Resident	199-D5-104	Hexavalent chromium	waterInhalationSweatLodge	0.70	820
		Hexavalent chromium	groundwaterIngestion	0.14	
	199-D5-41	Hexavalent chromium	waterInhalationSweatLodge	0.69	280
		Hexavalent chromium	groundwaterIngestion	0.14	
	199-D5-99	Hexavalent chromium	waterInhalationSweatLodge	0.70	2,900
		Hexavalent chromium	groundwaterIngestion	0.14	
Yakama Resident	199-D5-104	Hexavalent chromium	waterDermalSweatLodge	0.67	590
		Hexavalent chromium	groundwaterIngestion	0.20	
	199-D5-41	Hexavalent chromium	waterDermalSweatLodge	0.67	200
		Hexavalent chromium	groundwaterIngestion	0.20	
	199-D5-99	Hexavalent chromium	waterDermalSweatLodge	0.68	2,100
		Hexavalent chromium	groundwaterIngestion	0.20	

6.7 100-F, 100-IU-2, AND 100-IU-6 DECISION AREA RESULTS

As described in Section 6.1, the 100-F/100-IU-2/100-IU-6 ROD Decision Area includes two groundwater OUs: the 100-FR-3 groundwater OU and 100-F/100-IU-2/100-IU-6 groundwater OU. The locations of the groundwater monitoring wells evaluated in the 100-F Area, the 100-IU-2/100-IU-6 Area, as well as the boundaries of key groundwater plumes in these areas, are shown in Figures 6-8 and 6-9. A summary of the cancer risk, radiation dose, and chemical hazard results for the two groundwater OUs within the 100-F/100-IU-2/100-IU-6 Decision Area is provided in this section. Results pertaining to the Resident National Monument/Refuge and Subsistence Farmer scenarios, which include domestic exposures to groundwater, are presented in Section 6.7.1. Results for the CTUIR Resident and Yakama Resident scenarios include both domestic groundwater exposures and exposure related to use of groundwater in a sweat lodge. The results of these calculations, differentiating between domestic and sweat lodge exposures, are discussed in Section 6.7.2.

Figure 6-8. Groundwater Well Locations and Key Plumes for the 100-F Area.

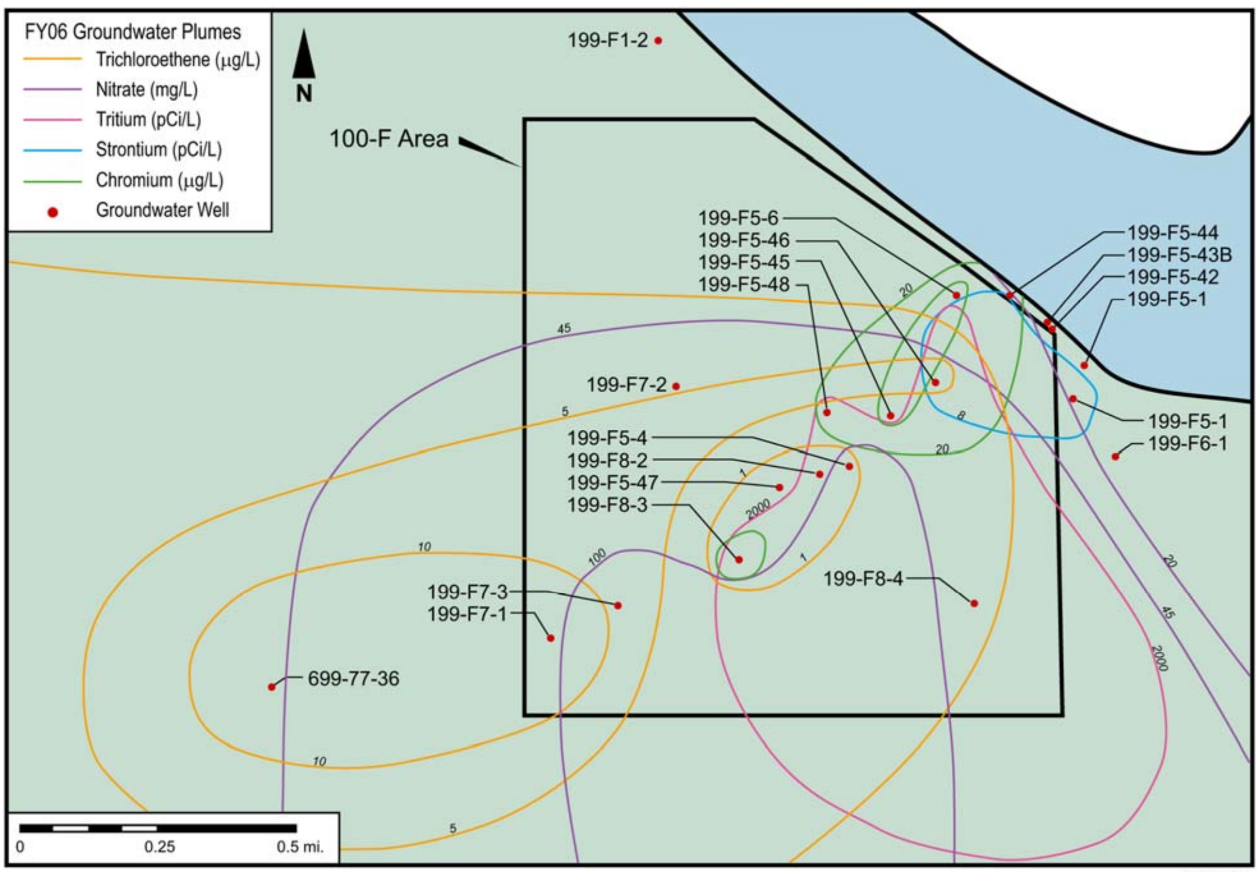


Figure 6-9. Groundwater Well Locations and Key Plumes for the 100-IU-2/100-IU-6 Operable Unit Area.

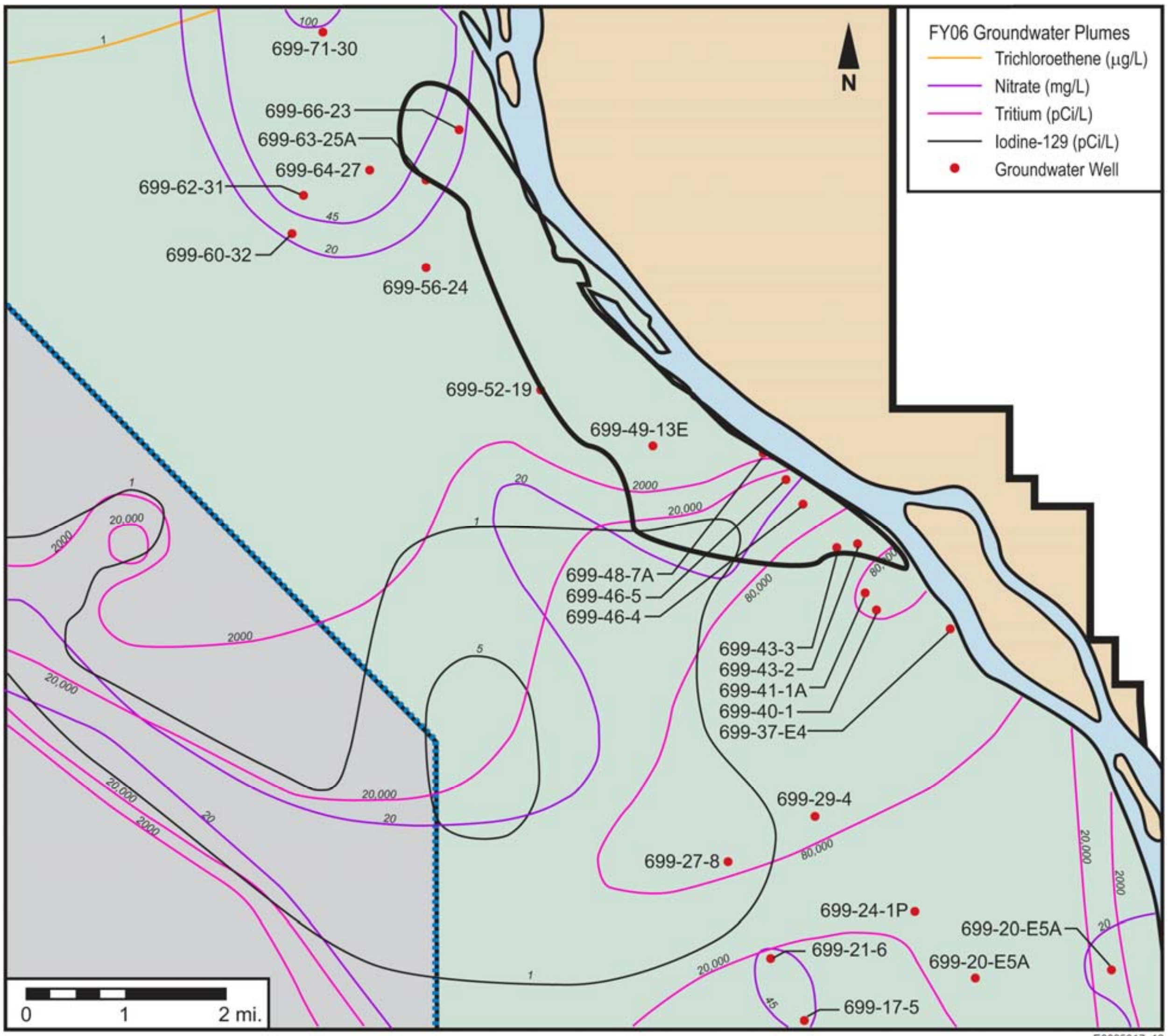


Table 6-16 shows that strontium-90 and tritium concentrations are associated with potential cancer risks that are above the 1×10^{-4} threshold in the 100-F/100-IU-2/100-IU-6 Area. Table 6-17 shows that the average tritium concentration in well 199-F8-3 is more than 50% higher than the 90th percentile value for the entire OU. This indicates that there is relatively high confidence that risks at this well are higher than those associated with the entire OU. The 50th and 90th percentiles for COPC concentrations in the 100-F/100-IU-2/100-IU-6 Area were recalculated without the data for well 199-F8-3 and a supplemental set of risk calculations are provided for this well. Cancer risk, radiation dose, and chemical hazard results for the 100-F/100-IU-2/100-IU6 Area do not include the results for well 199-F8-3.

6.7.1 Resident National Monument/Refuge and Subsistence Farmer Results in the 100-F/100-IU-2/100-IU-6 Area

6.7.1.1 Cancer Risk. The RME groundwater pathways risk results for the Resident National Monument/Refuge scenario (radionuclides + chemicals) in the 100-F/100-IU-2/100-IU-6 Area are 8 x 10^{-5} and 2 x 10^{-4} for the 100-FR-3 and 100-F/100-IU-2/100-IU-6 OUs, respectively. The corresponding CTE groundwater pathways risk results for the Resident National Monument/Refuge scenario are 2 x 10^{-6} and 1 x 10^{-5}. The RME groundwater pathways risk results for the Subsistence Farmer scenario (radionuclides + chemicals) in the 100-F/100-IU-2/100-IU-6 Area are 9 x 10^{-5} and 2 x 10^{-4} for the 100-FR-3 and 100-F/100-IU-2/100-IU-6 OUs, respectively. The corresponding CTE groundwater pathways risk results for the Subsistence Farmer scenario are 2 x 10^{-6} and 7 x 10^{-6}. Groundwater cancer risk results for these scenarios are presented in Table 6-18.

A summary of the key exposure pathways and analytes contributing 10% or more to groundwater RME cancer risk above 1 × 10^{-4} in the 100-F/100-IU-2/100-IU-6 Area for the Resident National Monument/Refuge and Subsistence Farmer scenarios is provided below.

Scenario	Groundwater OU	Analyte	Pathway	Fraction of OU Cancer Risk	OU Cancer Risk
Resident National Monument/Refuge	100-F/ 100-IU-2/ 100-IU-6	Tritium	groundwaterIngestion	0.96	2 × 10^{-4}
Subsistence Farmer	100-F/ 100-IU-2/ 100-IU-6	Tritium	groundwaterIngestion	0.96	2 × 10^{-4}

The RME risks are slightly above the 10^{-4} upper end of EPA's target cancer risk range. The CTE risks are within the lower portion of EPA's target cancer risk range. In the 100-F/100-IU-2/100-IU-6 OU, about 95% of the RME cancer risk is associated with tritium via the groundwater ingestion exposure route. In the CTE risk calculation, tritium is responsible for about 90% of cancer risk, with the remainder being related to technetium-99. In the 100-FR-3 OU, about 60% of RME cancer risk is related to ingestion of isotopic uranium in drinking water, with about 30% related to ingestion of strontium-90 in drinking water.

Cancer risks for the Resident National Monument/Refuge and Subsistence Farmer scenarios, calculated using the data from monitoring well 199-F8-3, are shown in Table 6-20. The RME and CTE risk results at well 199-F8-3 for the Resident National Monument/Refuge scenario are 8 × 10^{-5} and 9 × 10^{-6}, respectively. Subsistence Farmer RME and CTE cancer risk are 8 × 10^{-5} and 6 × 10^{-6}, respectively. These values are within EPA's risk management range of 1 × 10^{-6} to 1 × 10^{-4}. About 90% of these risks are due to ingestion of tritium in drinking water.

6.7.1.2 Radiation Dose. The RME groundwater pathways radiation dose results for the Resident National Monument/Refuge scenario in the 100-F/100-IU-2/100-IU-6 Area are 4.9 mrem/yr and 13 mrem/yr for the 100-FR-3 and 100-F/100-IU-2/100-IU-6 OUs, respectively.

The corresponding CTE groundwater pathways radiation dose results for the Resident National Monument/Refuge scenario are 0.62 and 2.1 mrem/yr. The RME groundwater pathways radiation dose results for the Subsistence Farmer scenario in the 100-F/100-IU-2/100-IU-6 Area are 6.0 mrem/yr and 13 mrem/yr for the 100-FR-3 and 100-F/100-IU-2/100-IU-6 OUs, respectively. The corresponding CTE groundwater pathways radiation dose results for the Subsistence Farmer scenario are 0.60 mrem/yr and 1.8 mrem/yr. Radiation dose risk results for these scenarios are tabulated in Table 6-18.

A summary of the key exposure pathways and analytes contributing 10% or more to groundwater radiation dose above the 4.0 mrem/yr threshold in the 100-F/100-IU-2/100-IU-6 Area for the Resident National Monument/Refuge and Subsistence Farmer scenarios is provided below.

Scenario	Groundwater OU	Analyte	Pathway	Fraction of OU Dose	OU Radiation Dose (mrem/yr)
Resident National Monument/Refuge	100-FR-3	Strontium-90	groundwaterIngestion	0.70	4.9
		Uranium-233/234	groundwaterIngestion	0.11	
	100-F/ 100-IU-2/ 100-IU-6	Tritium	groundwaterIngestion	0.94	13
Subsistence Farmer	100-FR-3	Strontium-90	groundwaterIngestion	0.75	6.0
	100-F/ 100-IU-2/ 100-IU-6	Tritium	groundwaterIngestion	0.90	13

The RME dose assessment results for the 100-F/100-IU-2/100-IU-6 OU are about a factor of three above the 4 mrem/yr threshold for drinking water. The RME dose results for the 100-FR-3 OU are up to 50% higher than the 4 mrem/yr threshold. The CTE dose results are below the 4 mrem/yr threshold for both scenarios and both OUs. Strontium-90 by drinking water ingestion is the primary exposure pathway in the 100-FR-3 OU, contributing about 70% to 75% of the dose. Tritium by drinking water ingestion contributes 90% or more of the dose in the 100-F/100-IU-2/100-IU-6 OU.

Radiation doses for the Resident National Monument/Refuge and Subsistence Farmer scenarios calculated using the data from monitoring well 199-F8-3 are shown in Table 6-20. The RME and CTE dose results at well 199-F8-3 for the Resident National Monument/Refuge scenario are 4.7 mrem/yr and 2.1 mrem/yr, respectively. Subsistence Farmer RME and CTE dose results are 4.5 mrem/yr and 1.7 mrem/yr, respectively. The RME values are only slightly above the 4.0 mrem/yr threshold. About 70% to 80% of these risks are due to ingestion of tritium in drinking water.

A summary of the key exposure pathways and analytes contributing 10% or more to groundwater radiation dose above the 4.0 mrem/yr threshold in well 199-F8-3 for the Resident National Monument/Refuge and Subsistence Farmer scenarios is provided below.

Scenario	Analyte	Pathway	Fraction of Well Dose	Well Radiation Dose (mrem/yr)
Resident National Monument/Refuge	Tritium	groundwaterIngestion	0.81	4.7
Subsistence Farmer	Tritium	groundwaterIngestion	0.77	4.5

6.7.1.3 Chemical Hazard. As described in Section 3.6.2, chemical HI is calculated separately for adult and child receptors in the residential scenarios. Reasonable maximum exposure child HI values tend to exceed those for adults. The CTE calculations for adult and child HIs tend to be approximately equivalent to adult values for water ingestion and slightly higher than those for young children due to the higher adult ingestion rate. The higher of the adult and child HI values are shown in Table 6-18 and described here.

The RME groundwater pathways HI results for the Resident National Monument/Refuge scenario in the 100-F/100-IU-2/100-IU-6 Area are 2.8 and 1.4 for the 100-FR-3 and 100-F/100-IU-2/100-IU-6 OUs, respectively. The corresponding CTE groundwater pathways HI results for the Resident National Monument/Refuge scenario are 0.57 and 0.14. The RME groundwater pathways child HI results for the Subsistence Farmer scenario in the 100-F/100-IU-2/100-IU-6 Area are 4.8 and 2.4 for the 100-FR-3 and 100-F/100-IU-2/100-IU-6 OUs, respectively. The corresponding CTE groundwater pathways child HI results for the Subsistence Farmer scenario are 0.57 and 0.14. HI results for these scenarios are presented in Table 6-18.

A summary of the key exposure pathways and analytes contributing 10% or more to a groundwater HI above 1.0 in the 100-F/100-IU-2/100-IU-6 Area for the Resident National Monument/Refuge and Subsistence Farmer scenarios is provided below.

Scenario	Groundwater OU	Analyte	Pathway	Fraction of OU HI	OU Hazard Index
Resident National Monument/Refuge	100-FR-3	Uranium	groundwaterIngestion	0.42	2.8
		Hexavalent chromium	groundwaterIngestion	0.27	
		Nitrogen in nitrate	groundwaterIngestion	0.17	
	100-F/ 100-IU-2/ 100-IU-6	Iron	groundwaterIngestion	0.29	1.4
		Zinc	groundwaterIngestion	0.24	
		Nitrogen in nitrate	groundwaterIngestion	0.23	
		Manganese	groundwaterIngestion	0.18	

<table>
<tr><th>Scenario</th><th>Groundwater OU</th><th>Analyte</th><th>Pathway</th><th>Fraction of OU HI</th><th>OU Hazard Index</th></tr>
<tr><td rowspan="8">Subsistence Farmer</td><td rowspan="4">100-FR-3</td><td>Uranium</td><td>groundwaterIngestionChild</td><td>0.40</td><td rowspan="4">4.8</td></tr>
<tr><td>Hexavalent chromium</td><td>groundwaterIngestionChild</td><td>0.26</td></tr>
<tr><td>Nitrogen in nitrate</td><td>groundwaterIngestionChild</td><td>0.16</td></tr>
<tr><td>Hexavalent chromium</td><td>groundwaterDermalChild</td><td>0.15</td></tr>
<tr><td rowspan="4">100-F/ 100-IU-2/ 100-IU-6</td><td>Iron</td><td>groundwaterIngestionChild</td><td>0.28</td><td rowspan="4">2.4</td></tr>
<tr><td>Zinc</td><td>groundwaterIngestionChild</td><td>0.24</td></tr>
<tr><td>Nitrogen in nitrate</td><td>groundwaterIngestionChild</td><td>0.22</td></tr>
<tr><td>Manganese</td><td>groundwaterIngestionChild</td><td>0.18</td></tr>
</table>

About 40% to 45% of the RME HI values of 2.4 and 4.8 in the 100-FR-3 OU are related to uranium by groundwater ingestion. An approximately equal percentage is contributed by chromium and nitrate combined, also by groundwater ingestion. The toxicity of nitrate is very specific; it inhibits oxygen binding to hemoglobin in the blood. The oral RfD for chromium is not related to any particular adverse effect, and the uranium oral RfD is based on toxic effects in the kidney. It is unlikely that additive effects would be observed for these three COPCs. The HI results probably overstate hazard for this reason.

Between about 20% and 30% of the RME HI values of 1.4 and 2.4 in the 100-F/100-IU-2/ 100-IU-6 groundwater OU is contributed by iron, zinc, nitrate, and manganese. All except nitrate are essential nutrients for which there exists homeostatic regulation of absorption in the gastrointestinal tract. As mentioned above, the oral RfD for nitrate is based on a very specific effect related to blood-oxygen content. It is unlikely that additive effects would be observed among nitrate and the other COPCs.

Chemical hazard for the Resident National Monument/Refuge and Subsistence Farmer scenarios calculated using the data from monitoring well 199-F8-3 are shown in Table 6-20. The RME and CTE HI values at well 199-F8-3 for the Resident National Monument/Refuge scenario are 1.3 and 0.67, respectively. Subsistence Farmer RME and CTE HI values are 2.1 and 0.67, respectively. Uranium contributes 50% of the HI for both scenarios, with nitrate and chromium contributing about 30% and 10%, respectively. As discussed above, it is unlikely that additive effects would be observed for these three COPCs.

A summary of the key exposure pathways and analytes contributing 10% or more to a groundwater HI above 1.0 in well 199-F8-3 for the Resident National Monument/Refuge and Subsistence Farmer scenarios is provided below.

Scenario	Analyte	Pathway	Fraction of Well HI	Well Hazard Index
Resident National Monument/Refuge	Uranium	groundwaterIngestion	0.51	1.3
	Nitrogen in nitrate	groundwaterIngestion	0.28	
	Hexavalent chromium	groundwaterIngestion	0.11	
Subsistence Farmer	Uranium	groundwaterIngestionChild	0.50	2.1
	Nitrogen in nitrate	groundwaterIngestionChild	0.27	
	Hexavalent chromium	groundwaterIngestionChild	0.11	

6.7.2 CTUIR Resident and Yakama Resident Results in the 100-F/100-IU-2/100-IU-6 Area

The risk assessment results presented in this section apply the parameter values and models documented in Harris and Harper (2004) and Ridolfi (2007). As discussed in Section 6.1, risks related to inhalation of volatile and semivolatile COPCs in sweat lodge air are calculated for both the CTUIR and Yakama Resident scenarios, but inhalation risks for nonvolatile COPCs are only calculated for the CTUIR Resident. Uncertainty and bias in the sweat lodge exposure models are discussed in Section 6.9.

6.7.2.1 Cancer Risk. The groundwater pathways risk results for the CTUIR Resident scenario (radionuclides + chemicals) in the 100-F/100-IU-2/100-IU-6 Area are 6×10^{-3} and 1×10^{-3} for the 100-FR-3 and 100-F/100-IU-2/100-IU-6 OUs, respectively. Domestic exposures contribute more than sweat lodge exposure to the risk results in the 100-F/100-IU-2/100-IU-6 OU, but the reverse is true in the 100-FR-3 OU. The groundwater pathways risk results for the Yakama Resident scenario (radionuclides + chemicals) in the 100-F/100-IU-2/100-IU-6 Area are 5×10^{-4} and 2×10^{-3} for the 100-FR-3 and 100-F/100-IU-2/100-IU-6 OUs, respectively. Domestic and sweat lodge exposures contribute equally in the 100-F/100-IU-2/100-IU-6 OU, but sweat lodge exposure dominates in the 100-FR-3 OU. Groundwater cancer risk results for these scenarios are presented in Table 6-19.

A summary of the key exposure pathways and analytes contributing 10% or more to groundwater cancer risk above 1×10^{-4} in the 100-F/100-IU-2/100-IU-6 Area for the CTUIR Resident and Yakama Resident scenarios is provided below.

Scenario	Groundwater OU	Analyte	Pathway	Fraction of OU Cancer Risk	OU Cancer Risk
CTUIR Resident	100-FR-3	Hexavalent chromium	waterInhalationSweatLodge	0.87	6×10^{-3}
	100-F/ 100-IU-2/ 100-IU-6	Tritium	groundwaterIngestion	0.62	1×10^{-3}
		Tritium	waterIngestionSweatLodge	0.20	
		Tritium	waterInhalationSweatLodge	0.15	
Yakama Resident	100-FR-3	Strontium-90	groundwaterIngestion	0.45	5×10^{-4}
		Strontium-90	waterIngestionSweatLodge	0.14	
	100-F/ 100-IU-2/ 100-IU-6	Tritium	waterInhalationSweatLodge	0.50	2×10^{-3}
		Tritium	groundwaterIngestion	0.35	
		Tritium	waterIngestionSweatLodge	0.11	

In the 100-F/100-IU-2/100-IU-6 OU, greater than 95% of the cancer risk is associated with tritium via the groundwater ingestion or sweat lodge inhalation pathways. The larger contribution of sweat lodge exposure in the Yakama Resident scenario is because of the higher daily use cited for the Yakama (7 hr/day; Ridolfi 2007) than for the CTUIR (1 hr/day; Harris and Harper 2004). In the 100-FR-3 OU, about 90% of the CTUIR Resident cancer risk is related to inhalation of hexavalent chromium in the sweat lodge. In the Yakama Resident scenario, where inhalation of gas-phase nonvolatile COPCs is not modeled, strontium-90 is the main contributor to cancer risk in the 100-FR-3 OU.

Cancer risks for the CTUIR Resident and Yakama Resident scenarios calculated using the data from monitoring well 199-F8-3 are shown in Table 6-21. The risk results at well 199-F8-3 for the CTUIR Resident and Yakama Resident scenarios are 2×10^{-3} and 7×10^{-4}, respectively. Approximately 60% of the CTUIR Resident risk at well 199-F8-3 is due to inhalation of hexavalent chromium in the sweat lodge. Domestic and sweat lodge exposures to tritium at well 199-F8-3 are the most important exposure pathways for the Yakama Resident scenario.

A summary of the key exposure pathways and analytes contributing 10% or more to groundwater cancer risk above 1×10^{-4} in well 199-F8-3 for the CTUIR Resident and Yakama Resident scenarios is provided below.

Scenario	Analyte	Pathway	Fraction of Well Cancer Risk	Well Cancer Risk
CTUIR Resident	Hexavalent chromium	waterInhalationSweatLodge	0.58	2×10^{-3}
	Tritium	groundwaterIngestion	0.15	
Yakama Resident	Tritium	waterInhalationSweatLodge	0.49	7×10^{-4}
	Tritium	groundwaterIngestion	0.34	
	Tritium	waterIngestionSweatLodge	0.11	

6.7.2.2 Radiation Dose. The groundwater pathways radiation dose results for the CTUIR Resident scenario in the 100-F/100-IU-2/100-IU-6 Area are 39 mrem/yr and 50 mrem/yr for the 100-FR-3 and 100-F/100-IU-2/100-IU-6 OUs, respectively. Sweat lodge exposures contribute more than domestic exposures to the risk results in both OUs. The groundwater pathways radiation dose results for the Yakama Resident scenario in the 100-F/100-IU-2/100-IU-6 Area are 18 mrem/yr and 160 mrem/yr for the 100-FR-3 and 100-F/100-IU-2/100-IU-6 OUs, respectively. Sweat lodge exposures again contribute more than domestic exposures. Groundwater radiation dose results for these scenarios are presented in Table 6-19.

A summary of the key exposure pathways and analytes contributing 10% or more to groundwater radiation dose above the 4.0 mrem/yr threshold in the 100-F/100-IU-2/100-IU-6 Area for the CTUIR Resident and Yakama Resident scenarios is provided below.

Scenario	Groundwater OU	Analyte	Pathway	Fraction of OU Dose	OU Radiation Dose (mrem/yr)
CTUIR Resident	100-FR-3	Uranium-233/234	waterInhalationSweatLodge	0.35	39
		Uranium-238	waterInhalationSweatLodge	0.25	
		Strontium-90	groundwaterIngestion	0.17	
	100-F/ 100-IU-2/ 100-IU-6	Tritium	waterInhalationSweatLodge	0.45	50
		Tritium	groundwaterIngestion	0.34	
		Tritium	waterIngestionSweatLodge	0.15	
Yakama Resident	100-FR-3	Strontium-90	groundwaterIngestion	0.37	18
		Tritium	waterInhalationSweatLodge	0.29	
		Strontium-90	waterIngestionSweatLodge	0.16	
	100-F/ 100-IU-2/ 100-IU-6	Tritium	waterInhalationSweatLodge	0.83	160
		Tritium	groundwaterIngestion	0.10	

In the 100-F/100-IU-2/100-IU-6 OU, cancer risk is associated with tritium via the groundwater ingestion or sweat lodge inhalation exposure pathways. The larger contribution of sweat lodge inhalation exposure in the Yakama Resident scenario is because of the higher daily use cited for the Yakama (7 hr/day; Ridolfi 2007) than for the CTUIR (1 hr/day; Harris and Harper 2004). In the 100-FR-3 OU, about 60% of CTUIR Resident cancer risk is related to inhalation exposure to uranium isotopes in the sweat lodge. In the Yakama Resident scenario, where inhalation of gas-phase nonvolatile COPCs is not modeled, strontium-90 and tritium are the main contributors to cancer risk in the 100-FR-3 OU.

Radiation dose for the CTUIR Resident and Yakama Resident scenarios calculated using the data from monitoring well 199-F8-3 are shown in Table 6-21. The dose results at well 199-F8-3 for the CTUIR Resident and Yakama Resident scenarios are 33 and 50 mrem/yr, respectively. Again, a large proportion of the CTUIR Resident radiation dose is due to inhalation exposure to uranium isotopes in the sweat lodge, whereas tritium is the main contributor to Yakama Resident scenario dose.

A summary of the key exposure pathways and analytes contributing 10% or more to groundwater radiation dose above the 4.0 mrem/yr threshold in well 199-F8-3 for the CTUIR Resident and Yakama Resident scenarios is provided below.

Scenario	Analyte	Pathway	Fraction of Well Dose	Well Radiation Dose (mrem/yr)
CTUIR Resident	Uranium-233/234	waterInhalationSweatLodge	0.31	33
	Uranium-238	waterInhalationSweatLodge	0.21	
	Tritium	waterInhalationSweatLodge	0.20	
	Tritium	groundwaterIngestion	0.15	
Yakama Resident	Tritium	waterInhalationSweatLodge	0.81	50
	Tritium	groundwaterIngestion	0.10	

6.7.2.3 Chemical Hazard. The chemical HI results for children in the CTUIR Resident scenario in the 100-F/100-IU-2/100-IU-6 Area are 7.9 and 4.1 for the 100-FR-3 and 100-F/100-IU-2/100-IU-6 OUs, respectively. The child HI results for the Yakama Resident scenario in the 100-F/100-IU-2/100-IU-6 Area are 8.4 and 4.3. The adult HI results, which include contribution from sweat lodge exposure beginning at the age of 6 years, are higher for the CTUIR Resident. This is not true for the Yakama Resident scenario because inhalation of gas-phase nonvolatiles is not modeled in the Yakama Resident scenario. Adult HI results for these scenarios are presented in Table 6-19.

A summary of the key exposure pathways and analytes contributing 10% or more to a groundwater HI above 1.0 in the 100-F/100-IU-2/100-IU-6 Area for the CTUIR Resident (adult) and Yakama Resident (child) scenarios is provided below.

Scenario	Groundwater OU	Analyte	Pathway	Fraction of OU HI	OU Hazard Index
CTUIR Resident	100-FR-3	Hexavalent chromium	waterInhalationSweatLodge	0.33	15
		Manganese	waterInhalationSweatLodge	0.26	
		Uranium	groundwaterIngestion	0.10	
	100-F/ 100-IU-2/ 100-IU-6	Manganese	waterInhalationSweatLodge	0.93	58

Scenario	Groundwater OU	Analyte	Pathway	Fraction of OU HI	OU Hazard Index
Yakama Resident	100-FR-3	Hexavalent chromium	waterDermalSweatLodge	0.41	8.4
		Uranium	groundwaterIngestion	0.19	
		Hexavalent chromium	groundwaterIngestion	0.12	
	100-F/ 100-IU-2/ 100-IU-6	Iron	groundwaterIngestionChild	0.29	4.3
		Zinc	groundwaterIngestionChild	0.24	
		Nitrogen in nitrate	groundwaterIngestionChild	0.23	
		Manganese	groundwaterIngestionChild	0.18	

Hexavalent chromium and uranium, with the addition of manganese for the CTUIR Resident scenario, are the key COPCs in the 100-FR-3 OU. Inhalation exposure in the sweat lodge is the main exposure pathway for hexavalent chromium and manganese, and groundwater ingestion is the most important for uranium. The higher CTUIR Resident HI in the 100-FR-3 OU reflects the contribution of inhalation exposure for the nonvolatile COPCs uranium and hexavalent chromium. The toxicity of nitrate is very specific, it inhibits oxygen binding to hemoglobin in the blood. The oral RfD for chromium is not related to any particular adverse effect, and the uranium oral RfD is based on toxic effects in the kidney. It is unlikely that additive effects would be observed for these three COPCs.

As discussed in Section 6.6.2, the calculation of dermal sweat lodge exposure to nonvolatile COPCs in water is based on summation of dissolved and vapor-phase COPC concentrations (Harris and Harper 2004, Appendix 4). In the Yakama Resident scenario, vapor-phase concentrations of nonvolatile COPCs were not calculated. Therefore, only COPC concentrations in water contribute to the dermal exposure pathway HI results in the 100-FR-3 OU. It is presumed that exposure may occur if water is directly poured over the body during the intervals when new heated rocks are brought into the lodge.

Exposure to manganese by inhalation exposure in the sweat lodge is responsible for more than 90% of the CTUIR Resident adult HI in the 100-F/100-IU-2/100-IU-6 OU. In the Yakama Resident scenario, child HI is related to domestic groundwater ingestion with significant contributions from four COPCs. With the exception of nitrate, these four COPCs are essential nutrients for which there exists homeostatic regulation of absorption in the gastrointestinal tract. The oral RfD for nitrate is based on a very specific effect related to blood-oxygen content. It is unlikely that additive toxic effects would be observed among nitrate and the other COPCs.

Chemical hazard for the CTUIR Resident and Yakama Resident scenarios calculated using the data from monitoring well 199-F8-3 is shown in Table 6-21. The CTUIR Resident and Yakama Resident child HI values are 3.6 and 3.8, respectively. The CTUIR Resident adult HI result is higher because of adult exposure in the sweat lodge, but this is not the case for the

Yakama Resident. Exposure to various metals via inhalation in the sweat lodge dominates the adult HI in the CTUIR Resident scenario. Domestic groundwater ingestion of uranium is the most important contributor to child HI for the Yakama Resident scenario at monitoring well 199-F8-3.

A summary of the key exposure pathways and analytes contributing 10% or more to a groundwater HI above 1.0 in well 199-F8-3 for the CTUIR Resident (adult) and Yakama Resident (child) scenarios is provided below.

Scenario	Analyte	Pathway	Fraction of Well HI	Well Hazard Index
CTUIR Resident	Barium	waterInhalationSweatLodge	0.27	5.6
	Hexavalent chromium	waterInhalationSweatLodge	0.16	
	Uranium	groundwaterIngestion	0.16	
	Manganese	waterInhalationSweatLodge	0.14	
Yakama Resident	Uranium	groundwaterIngestionChild	0.51	3.8
	Nitrogen in nitrate	groundwaterIngestionChild	0.28	
	Hexavalent chromium	groundwaterIngestionChild	0.11	

6.8 300 DECISION AREA RESULTS

The locations of the groundwater monitoring wells evaluated in the southern portion of the 300 and 600 Areas, and within the 300 Area, as well as the boundaries of key groundwater plumes in these areas, are shown in Figures 6-10 and 6-11. A summary of the cancer risk, radiation dose, and chemical hazard results for the 300 Area is provided in this section. Results pertaining to the Resident National Monument/Refuge and Subsistence Farmer scenarios, which include domestic exposures to groundwater, are presented in Section 6.8.1. Results for the CTUIR Resident and Yakama Resident scenarios include both domestic groundwater exposures and exposure related to use of groundwater in a sweat lodge. The results of these calculations, differentiating between domestic and sweat lodge exposures, are discussed in Section 6.8.2.

Figure 6-10. Groundwater Well Locations and Key Plumes for the 300 and South 600 Areas.

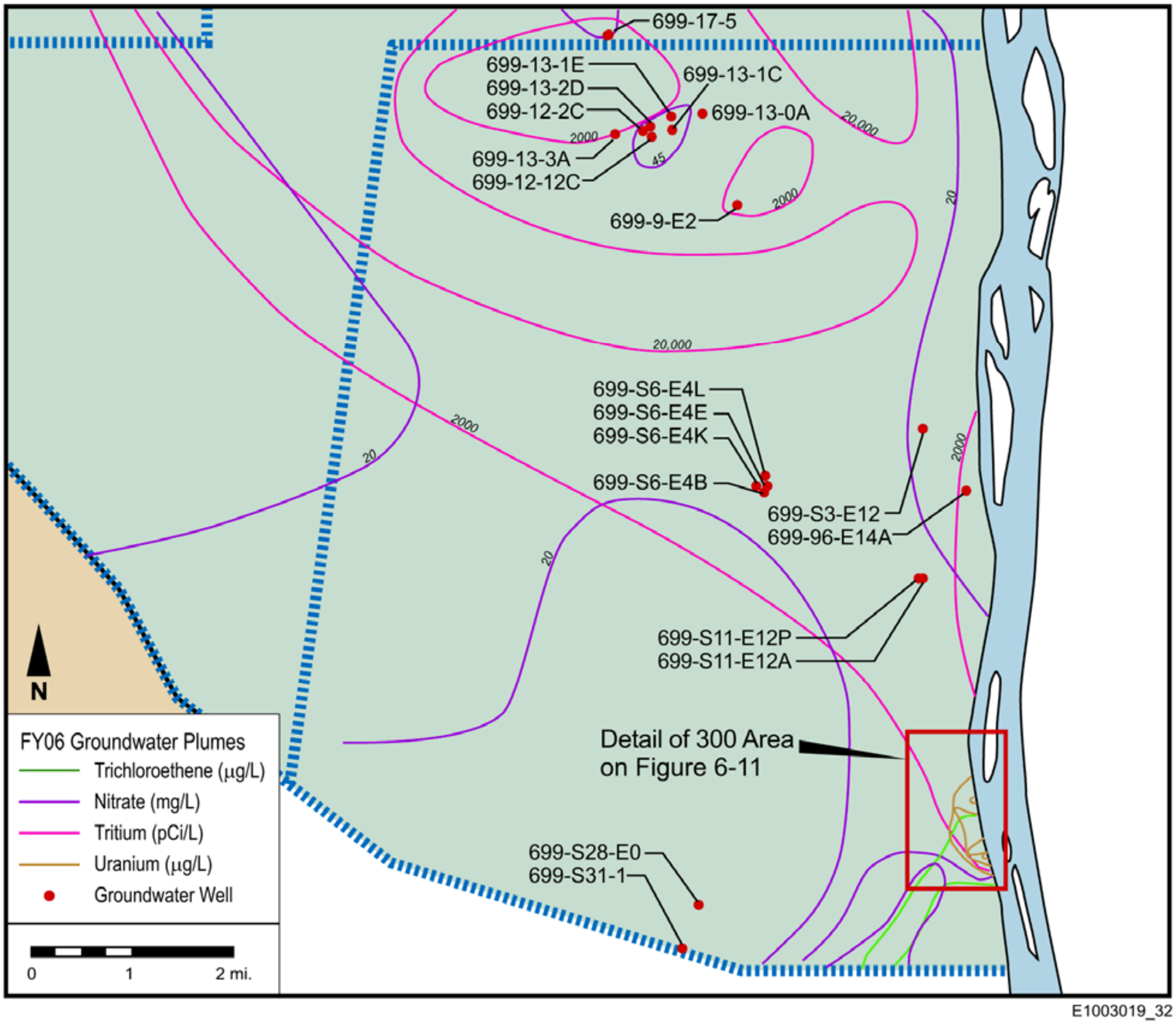


Figure 6-11. Groundwater Well Locations and Key Plumes for the 300 Area.

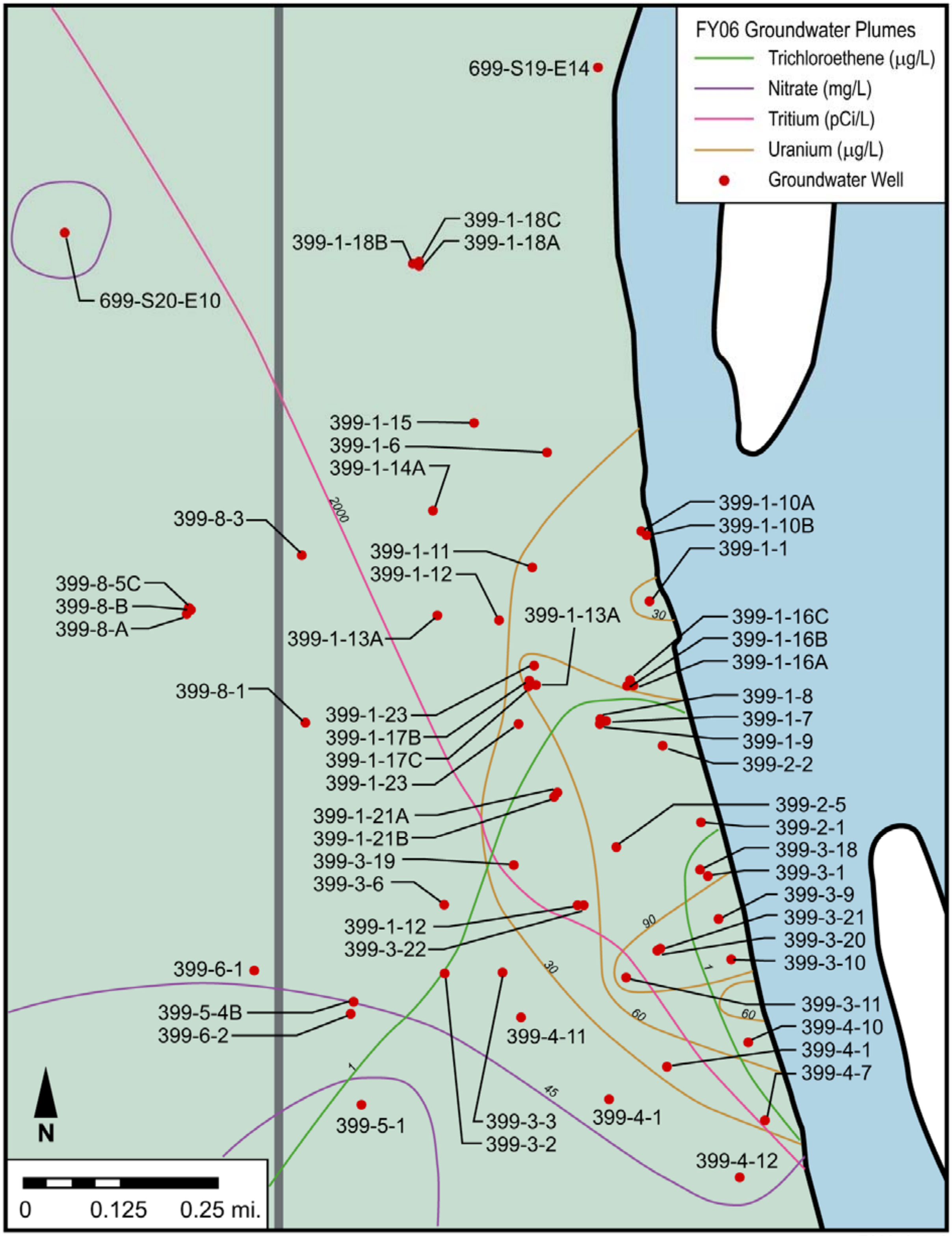


Table 6-16 shows that tritium concentrations are associated with potential cancer risks that are above the 1×10^{-4} threshold in the 300 Area. Uranium concentrations are associated with potential hazard quotients in the 300 Area that are approximately 10 times the threshold of 1.0. Table 6-17 shows that the average tritium concentration in well 699-13-3A and the average uranium concentration in well 399-3-1 are more than 50% higher than the 90th percentile values for the entire OU. The 95% UCL uranium concentration in well 399-2-2 is almost twice the 90th percentile value for the entire OU. This indicates that there is relatively high confidence that risks at these wells are higher than those associated with the entire OU. The 50th and 90th percentiles for COPC concentrations in the 300 Area were recalculated without the data for these wells and a supplemental set of risk calculations are provided for each well. Cancer risk, radiation dose, and chemical hazard results for the 300 Area do not include the results for wells 699-13-3A, 399-3-1, and 399-2-2.

6.8.1 Resident National Monument/Refuge and Subsistence Farmer Results in the 300 Area

6.8.1.1 Cancer Risk. The RME and CTE groundwater pathways risk results for the Resident National Monument/Refuge scenario (radionuclides + chemicals) in the 300 Area are 9×10^{-4} and 1×10^{-5} , respectively. The RME and CTE groundwater pathways risk results for the Subsistence Farmer scenario (radionuclides + chemicals) in the 300 Area are 9×10^{-4} and 7×10^{-6}, respectively. Operable unit groundwater cancer risk results for these scenarios are tabulated in Table 6-18.

A summary of the key exposure pathways and analytes contributing 10% or more to RME groundwater cancer risk above 1×10^{-4} in the 300 Area for the Resident National Monument/Refuge and Subsistence Farmer scenarios is provided below.

Scenario	Analyte	Pathway	Fraction of OU Cancer Risk	OU Cancer Risk
Resident National Monument/Refuge	Tritium	groundwaterIngestion	0.80	9 × 10
Subsistence Farmer	Tritium	groundwaterIngestion	0.79	9 × 10

The OU RME risks for both scenarios are about a factor of 10 above the 10^{-4} upper end of EPA's target cancer risk range, while the CTE risks are in the lower portion of EPA's 10^{-6} to 10^{-4} target cancer risk range. About 80% of the RME cancer risks are associated with tritium via the water ingestion exposure route. For the CTE risk calculations, about 70% to 75% of the 300 Area cancer risks are related to uranium-238 and uranium-233/234 via the water ingestion exposure pathway. The difference between the COPCs driving the RME and CTE cancer risks is due to the relatively large difference between the 90th (307,000 pCi/L) and 50th percentile (4,400 pCi/L) groundwater concentrations of tritium in the 300 Area. Although future trends in groundwater concentrations have not been quantified in this assessment, natural radioactive decay of tritium (12.3-year half-life) will result in a relatively quick decrease of risk from this COPC over time compared to present-day groundwater concentrations.

Cancer risks for the Resident National Monument/Refuge and Subsistence Farmer scenarios calculated using the data from monitoring wells 699-13-3A, 399-3-1, and 399-2-2 are shown in Table 6-22. Both CTE and RME cancer risks exceed the 1×10^{-4} threshold at well 699-13-3A. Reasonable maximum exposure cancer risks exceed the threshold at well 399-2-2. Both CTE and RME risks are below 1×10^{-6} at well 399-3-1. Tritium by drinking water ingestion is responsible for almost 100% of the RME risk at well 699-13-3A. The mean (3,570,000 pCi/L) and 95% UCL (4,770,000 pCi/L) values of tritium in this well are very close. The difference between the CTE and RME risk results is related more to the difference between CTE and RME values for drinking water ingestion and exposure duration than to uncertainty in the average groundwater concentration at the well. Well 399-2-2 was identified for supplemental analysis based on uranium concentrations. Approximately 50% of the RME cancer risk of 4×10^{-4} is related to tetrachloroethene by ingestion and dermal absorption while bathing. Isotopic uranium contributes 30% or more to cancer risk by drinking water ingestion at well 399-2-2.

A summary of the key exposure pathways and analytes contributing 10% or more to groundwater cancer risk above 1×10^{-4} in wells 699-13-3A, 399-3-1, and 399-2-2 for the Resident National Monument/Refuge and Subsistence Farmer scenarios is provided below.

Scenario	Well	Analyte	Pathway	Fraction of OU HI	OU Hazard Index
Resident National Monument/Refuge	399-2-2	Tetrachloroethene	groundwaterIngestion	0.31	4×10^{-4}
		Uranium-238	groundwaterIngestion	0.22	
		Uranium-233/234	groundwaterIngestion	0.19	
		Tetrachloroethene	groundwaterDermal	0.16	
	699-13-3A	Tritium	groundwaterIngestion	0.99	1×10^{-2}
Subsistence Farmer	399-2-2	Tetrachloroethene	groundwaterIngestion	0.34	4×10^{-4}
		Uranium-238	groundwaterIngestion	0.19	
		Tetrachloroethene	groundwaterDermal	0.18	
		Uranium-233/234	groundwaterIngestion	0.16	
	699-13-3A	Tritium	groundwaterIngestion	0.98	1×10^{-2}

6.8.1.2 Radiation Dose. The RME and CTE groundwater pathways radiation dose results for the Resident National Monument/Refuge scenario in the 300 Area are 52 mrem/yr and 3.1 mrem/yr, respectively. The RME and CTE groundwater pathways radiation dose results for the Subsistence Farmer scenario in the 300 Area are 51 and 2.8 mrem/yr, respectively. Operable unit groundwater radiation dose risk results for these scenarios are tabulated in Table 6-18.

A summary of the key exposure pathways and analytes contributing 10% or more to groundwater radiation dose above the 4.0 mrem/yr threshold in the 300 Area for the Resident National Monument/Refuge and Subsistence Farmer scenarios is provided below.

Scenario	Analyte	Pathway	Fraction of OU Dose	OU Radiation Dose (mrem/yr)
Resident National Monument/Refuge	Tritium	groundwaterIngestion	0.73	52
	Uranium-233/234	groundwaterIngestion	0.11	
	Uranium-238	groundwaterIngestion	0.10	
Subsistence Farmer	Tritium	groundwaterIngestion	0.70	51
	Uranium-233/234	groundwaterIngestion	0.12	
	Uranium-238	groundwaterIngestion	0.11	

Central tendency exposure dose results for the 300 Area are below the 4 mrem/yr threshold, but the RME results are more than a factor of 10 above this threshold. About 70% of the RME radiation dose is associated with tritium by the water ingestion exposure route. Approximately another 20% is related to uranium-238 and uranium-233/234 by water ingestion. As described for cancer risk, natural radioactive decay of tritium (12.3-year half-life) will result in a decrease in dose from this COPC over time compared to present-day groundwater conditions.

Radiation doses for the Resident National Monument/Refuge and Subsistence Farmer scenarios calculated using the data from monitoring wells 699-13-3A, 399-3-1, and 399-2-2 are shown in Table 6-22. At well 699-13-3A, RME and CTE radiation doses are far above the 4 mrem/yr threshold. The RME and CTE radiation dose results at well 399-2-2 are between two and four times the threshold value of 4 mrem/yr. Radioactive COPCs were not measured at well 399-3-1. The contributions of COPCs and exposure routes are similar to that described for cancer risks.

A summary of the key exposure pathways and analytes contributing 10% or more to groundwater radiation dose above the 4.0 mrem/yr threshold in wells 699-13-3A, 399-3-1, and 399-2-2 for the Resident National Monument/Refuge and Subsistence Farmer scenarios is provided below.

Scenario	Well	Analyte	Pathway	Fraction of OU Dose	OU Radiation Dose (mrem/yr)
Resident National Monument/Refuge	399-2-2	Uranium-233/234	groundwaterIngestion	0.46	15
		Uranium-238	groundwaterIngestion	0.43	
	699-13-3A	Tritium	groundwaterIngestion	0.95	630
Subsistence Farmer	399-2-2	Uranium-233/234	groundwaterIngestion	0.46	15
		Uranium-238	groundwaterIngestion	0.44	
	699-13-3A	Tritium	groundwaterIngestion	0.93	590

6.8.1.3 Chemical Hazard. As described in Section 3.6.2, chemical HI is calculated separately for adult and child receptors in the residential scenarios. Reasonable maximum exposure child HI values tend to exceed those for adults; for the CTE calculations, adult and child HIs tend to be approximately equivalent to adult values for water ingestion, slightly higher than those for young children due to the higher adult ingestion rate. The higher of the adult and child HI values are shown in Table 6-18 and described here.

A summary of the key exposure pathways and analytes contributing 10% or more to a groundwater HI above 1.0 in the 300 Area for the Resident National Monument/Refuge and Subsistence Farmer scenarios is provided below.

Scenario	Analyte	Pathway	Fraction of OU HI	OU Hazard Index
Resident National Monument/Refuge	Uranium	groundwaterIngestion	0.87	5.3
Subsistence Farmer	Uranium	groundwaterIngestion	0.87	8.8

The RME and CTE chemical HI results for the Resident National Monument/Refuge scenario in the 300 Area are 5.3 and 0.63, respectively. The RME and CTE child chemical HI results for the Subsistence Farmer scenario in the 300 Area are 8.8 and 0.63, respectively.

About 90% of the RME HI values for both exposure scenarios are related to uranium exposure via ingestion of drinking water.

Chemical HI for the Resident National Monument/Refuge and Subsistence Farmer scenarios calculated using the data from monitoring wells 699-13-3A, 399-3-1, and 399-2-2 are shown in Table 6-22. Reasonable maximum exposure HI at wells 399-3-1 and 399-2-2 range between approximately 10 and 20, and CTE HIs are 4.3 (399-2-2) and 5.1 (399-3-1). At well 699-13-3A, RME HI values are between 1.0 and 2.0. Ninety-five percent or more of the HI at wells 399-2-2 and 399-3-1 is related to ingestion of uranium in drinking water. At well 699-13-3A, about 50% of HI is related to uranium, and about 45% is related to nitrate and fluoride combined. The kidney is the key organ where the toxicity of uranium is expressed, but nitrate toxicity relates to inhibition of oxygen binding to hemoglobin in the blood. Fluoride toxicity is related to skeletal effects. It is unlikely that chemical hazards from these COPCs is additive.

A summary of the key exposure pathways and analytes contributing 10% or more to a groundwater HI above 1.0 in wells 699-13-3A, 399-3-1, and 399-2-2 for the Resident National Monument/Refuge and Subsistence Farmer scenarios is provided below.

<table>
<tr><th>Scenario</th><th>Well</th><th>Analyte</th><th>Pathway</th><th>Fraction of OU HI</th><th>OU Hazard Index</th></tr>
<tr><td rowspan="5">Resident National Monument/Refuge</td><td>399-2-2</td><td>Uranium</td><td>groundwaterIngestion</td><td>0.95</td><td>9.5</td></tr>
<tr><td>399-3-1</td><td>Uranium</td><td>groundwaterIngestion</td><td>0.96</td><td>12</td></tr>
<tr><td rowspan="3">699-13-3A</td><td>Uranium</td><td>groundwaterIngestion</td><td>0.48</td><td rowspan="3">1.1</td></tr>
<tr><td>Nitrogen in nitrate</td><td>groundwaterIngestion</td><td>0.35</td></tr>
<tr><td>Fluoride</td><td>groundwaterIngestion</td><td>0.11</td></tr>
</table>

Scenario	Well	Analyte	Pathway	Fraction of OU HI	OU Hazard Index
Subsistence Farmer	399-2-2	Uranium	groundwaterIngestionChild	0.95	16
	399-3-1	Uranium	groundwaterIngestionChild	0.96	20
	699-13-3A	Uranium	groundwaterIngestionChild	0.48	1.8
		Nitrogen in nitrate	groundwaterIngestionChild	0.35	
		Fluoride	groundwaterIngestionChild	0.11	

6.8.2 CTUIR Resident and Yakama Resident Results in the 300 Area

The risk assessment results presented in this section apply the parameter values and models documented in Harris and Harper (2004) and Ridolfi (2007). As discussed in Section 6.1, risks related to inhalation of volatile and semivolatile COPCs in sweat lodge air are calculated for both the CTUIR and Yakama Resident scenarios, but inhalation risks for nonvolatile COPCs are only calculated for the CTUIR Resident. Uncertainty and bias in the sweat lodge exposure models are discussed in Section 6.9.

6.8.2.1 Cancer Risk. The groundwater pathways risk results for the CTUIR Resident and Yakama Resident scenarios (radionuclides + chemicals) in the 300 Area are 9×10^{-3} and 8×10^{-3}, respectively. Sweat lodge exposures are more important than domestic exposures in both scenarios. Groundwater cancer risk results for these scenarios are presented in Table 6-19.

A summary of the key exposure pathways and analytes contributing 10% or more to groundwater cancer risk above 1×10^{-4} in the 300 Area for the CTUIR Resident and Yakama Resident scenarios is provided below.

Scenario	Analyte	Pathway	Fraction of OU Cancer Risk	OU Cancer Risk
CTUIR Resident	Tritium	groundwaterIngestion	0.29	9×10^{-3}
	Uranium-233/234	waterInhalationSweatLodge	0.25	
	Uranium-238	waterInhalationSweatLodge	0.19	
Yakama Resident	Tritium	waterInhalationSweatLodge	0.46	8×10^{-3}
	Tritium	groundwaterIngestion	0.33	
	Tritium	waterIngestionSweatLodge	0.10	

Tritium by domestic drinking water ingestion and inhalation of uranium isotopes in the sweat lodge are the main contributors to cancer risk in the CTUIR Resident scenario. Inhalation exposure to uranium isotopes in the sweat lodge does not occur in the Yakama Resident scenario because inhalation of gas-phase nonvolatile COPCs is only modeled for the CTUIR Resident scenario. The higher tritium sweat lodge risk in the Yakama Resident scenario is because of the higher daily use cited for the Yakama (7 hr/day; Ridolfi 2007) than for the CTUIR (1 hr/day; Harris and Harper 2004). Although future trends in groundwater concentrations have not been quantified in this assessment, natural radioactive decay of tritium (12.3-year half-life) will result in a decrease of risk from this COPC over time compared to present-day groundwater conditions.

Cancer risks for the CTUIR Resident and Yakama Resident scenarios calculated using the data from monitoring wells 699-13-3A, 399-3-1, and 399-2-2 are shown in Table 6-23. Cancer risks at well 399-3-1 were near 1×10^{-6} for both scenarios. The groundwater pathways risk results for wells 699-13-3A and 399-2-2 are much higher than at well 399-3-1. Wells 699-13-3A and 399-2-2 have elevated levels of tritium, while well 399-3-1 was identified for supplemental analysis based on uranium concentrations.

Almost 100% of the cancer risk at well 699-13-3A for both scenarios is related to tritium, with drinking water ingestion and sweat lodge inhalation being the dominant exposure pathways. Inhalation exposure to isotopes of uranium in the sweat lodge is the basis of the CTUIR Resident cancer risk results at well 399-2-2. Ingestion of tetrachloroethene and uranium isotopes in domestic drinking water are the main contributors to cancer risk at well 399-2-2 for the Yakama Resident scenario.

A summary of the key exposure pathways and analytes contributing 10% or more to groundwater cancer risk above 1×10^{-4} in wells 699-13-3A, 399-3-1, and 399-2-2 for the CTUIR Resident and Yakama Resident scenarios is provided below.

Scenario	Well	Analyte	Pathway	Fraction of Well Risk	Well Cancer Risk
CTUIR Resident	399-2-2	Uranium-233/234	waterInhalationSweatLodge	0.41	7×10^{-3}
		Uranium-238	waterInhalationSweatLodge	0.31	
	699-13-3A	Tritium	groundwaterIngestion	0.63	$>1 \times 10^{-2}$
		Tritium	waterIngestionSweatLodge	0.20	
		Tritium	waterInhalationSweatLodge	0.15	
Yakama Resident	399-2-2	Tetrachloroethene	groundwaterIngestion	0.26	2×10^{-3}
		Uranium-238	groundwaterIngestion	0.16	
		Uranium-233/234	groundwaterIngestion	0.14	
	699-13-3A	Tritium	waterInhalationSweatLodge	0.51	$>1 \times 10^{-2}$
		Tritium	groundwaterIngestion	0.36	
		Tritium	waterIngestionSweatLodge	0.12	

6.8.2.2 Radiation Dose. The groundwater pathways radiation dose results for the CTUIR Resident and Yakama Resident scenarios in the 300 Area are 450 mrem/yr and 520 mrem/yr, respectively. Groundwater radiation dose results for these scenarios are presented in Table 6-19.

A summary of the key exposure pathways and analytes contributing 10% or more to groundwater radiation dose above the 4.0 mrem/yr threshold in the 300 Area for the CTUIR Resident and Yakama Resident scenarios is provided below.

Scenario	Analyte	Pathway	Fraction of OU Dose	OU Radiation Dose (mrem/yr)
CTUIR Resident	Uranium-233/234	waterInhalationSweatLodge	0.33	450
	Uranium-238	waterInhalationSweatLodge	0.26	
	Tritium	waterInhalationSweatLodge	0.15	
	Tritium	groundwaterIngestion	0.12	
Yakama Resident	Tritium	waterInhalationSweatLodge	0.80	520

Uranium isotopes are the main contributor to radiation dose in the CTUIR Resident scenario. Inhalation exposure in the sweat lodge dominates for both scenarios, but the CTUIR Resident results include the contribution of inhalation of gas-phase nonvolatile COPCs. As described above, natural radioactive decay of tritium (12.3-year half-life) will result in decreased concentrations of this COPC over time.

Radiation dose for the CTUIR Resident and Yakama Resident scenarios calculated using the data from monitoring wells 699-13-3A, 399-3-1, and 399-2-2 is shown in Table 6-23. No radionuclides were measured in well 399-3-1; consequently, there are no radiation dose results for this well. Radiation dose values for the CTUIR Resident were 350 mrem/yr and 2,300 mrem/yr in wells 399-2-2 and 699-13-3A, respectively. For the Yakama Resident, the analogous doses were 45 mrem/yr and 7,700 mrem/yr. Contaminants of potential concern and exposure pathways contributing to radiation dose at wells 399-2-2 and 699-13-3A are analogous to those described above for cancer risk, with the exception of tetrachloroethene contributions at well 399-2-2.

A summary of the key exposure pathways and analytes contributing 10% or more to groundwater radiation dose above the 4.0 mrem/yr threshold in wells 399-2-2 and 699-13-3A for the CTUIR Resident and Yakama Resident scenarios is provided below.

Scenario	Well	Analyte	Pathway	Fraction of Well Dose	Well Radiation Dose (mrem/yr)
CTUIR Resident	399-2-2	Uranium-233/234	waterInhalationSweatLodge	0.48	350
		Uranium-238	waterInhalationSweatLodge	0.39	
	699-13-3A	Tritium	waterInhalationSweatLodge	0.46	2,300
		Tritium	groundwaterIngestion	0.35	
		Tritium	waterIngestionSweatLodge	0.16	

Scenario	Well	Analyte	Pathway	Fraction of Well Dose	Well Radiation Dose (mrem/yr)
Yakama Resident	399-2-2	Tritium	waterInhalationSweatLodge	0.28	45
		Uranium-233/234	groundwaterIngestion	0.23	
		Uranium-238	groundwaterIngestion	0.22	
		Uranium-233/234	waterIngestionSweatLodge	0.10	
	699-13-3A	Tritium	waterInhalationSweatLodge	0.84	7,700
		Tritium	groundwaterIngestion	0.10	

6.8.2.3 Chemical Hazard. The chemical HI results for a child in the CTUIR Resident and Yakama Resident scenarios in the 300 Area are 15 and 16, respectively. The adult HI results, which include contribution from sweat lodge exposure beginning at the age of 6 years, are higher for the CTUIR Resident due to the contribution of exposure to gas-phase nonvolatile COPCs in the sweat lodge.

A summary of the key exposure pathways and analytes contributing 10% or more to an adult groundwater HI above 1.0 in the 300 Area for the CTUIR Resident (adult) and Yakama Resident (child) scenarios is provided below.

Scenario	Analyte	Pathway	Fraction of OU HI	OU Hazard Index
CTUIR Resident	Manganese	waterInhalationSweatLodge	0.49	21
	Uranium	groundwaterIngestion	0.30	
	Uranium	waterIngestionSweatLodge	0.10	
Yakama Resident	Uranium	groundwaterIngestionChild	0.87	16

Approximately 50% of the CTUIR Resident adult HI values is due to inhalation of manganese in the sweat lodge. For the Yakama Resident scenario, HI is related to ingestion of uranium in domestic drinking water.

Chemical hazard calculated using the data from monitoring wells 699-13-3A, 399-3-1, and 399-2-2 is shown in Table 6-23. For both the CTUIR Resident and Yakama Resident scenarios, child HI generally exceeds adult HI at wells 399-2-2, 399-3-1, and 699-13-3A. CTUIR Resident child HI results are 27 (399-2-2), 34 (399-3-1), and 3.1 (699-13-3A). At well 699-13-3A, the CTUIR Resident adult HI of 4.6 exceeds the result for the child HI. Yakama Resident child HI results are 28 (399-2-2), 35 (399-3-1), and 3.2 (699-13-3A).

A summary of the key exposure pathways and analytes contributing 10% or more to a groundwater HI above 1.0 in wells 699-13-3A, 399-3-1, and 399-2-2 for the CTUIR Resident and Yakama Resident scenarios is provided below.

Scenario	Well	Analyte	Pathway	Fraction of Well HI	Well Hazard Index
CTUIR Resident	399-2-2	Uranium	groundwaterIngestionChild	0.95	27
	399-3-1	Uranium	groundwaterIngestionChild	0.96	34
	699-13-3A	Manganese	waterInhalationSweatLodge	0.32	4.6
		Barium	waterInhalationSweatLodge	0.25	
		Uranium	groundwaterIngestion	0.15	
		Nitrogen in nitrate	groundwaterIngestion	0.11	
Yakama Resident	399-2-2	Uranium	groundwaterIngestionChild	0.95	28
	399-3-1	Uranium	groundwaterIngestionChild	0.96	35
	699-13-3A	Uranium	groundwaterIngestionChild	0.48	3.2
		Nitrogen in nitrate	groundwaterIngestionChild	0.35	
		Fluoride	groundwaterIngestionChild	0.11	

Uranium is the main contributor to child HI values above 1.0 at wells 399-2-2 and 399-3-1. Nitrate and fluoride contribute 35% and 11% to Yakama Resident child HI at well 699-13-3A. The toxicity of uranium is manifested in the kidney, while that of fluoride targets the skeletal system. Nitrate toxicity relates to inhibition of oxygen binding to hemoglobin in the blood. Additivity of effects from uranium, nitrate, and fluoride is unlikely. The Yakama Resident child HI value for well 699-13-3A is likely to be overestimated for this reason.

6.9 UNCERTAINTY ANALYSIS FOR THE GROUNDWATER RISK ASSESSMENT

The uncertainty analysis provides quantitative and qualitative information for evaluating the level of confidence and the possible bias in the risk assessment results described in Sections 6.3 through 6.8. The main purpose of the uncertainty analysis is to put the numerical results in perspective with regard to the assumptions and uncertainties in the risk assessment process. Another purpose of the uncertainty analysis is to provide a basis for recommendations relating to additional information that could be of value for refining the risk estimates.

The principal tool applied in the HHRA for quantifying uncertainties in these risk estimates is the use of RME and CTE parameter values in the risk calculations. The range of these values for those parameters related to behavioral and/or physiological characteristics (e.g., ingestion and inhalation rates, exposure frequency) provide a measure of uncertainty related to the attributes of individual receptors within a receptor population. The groundwater assessment provides estimates of risks that apply to most areas within the OU, as well as estimates for specific locations where risks are highest. In both an entire OU and an individual well, CTE and RME estimates of EPCs were used to provide a measure of importance of the magnitude of the range of COPC concentrations to the risk estimates. A semiquantitative or qualitative assessment of uncertainty is provided for other aspects of the risk assessment that affect the final estimates including the following:

- Uncertainty in data collection and evaluation including analytical data quality and data representativeness

- Uncertainty in the exposure assessment including COPC identification, statistical models for the EPCs, and human exposure models underlying the exposure scenarios

- Uncertainty in the toxicity assessment including models of chemical toxicity and radiation dosimetry, upon which assessment of potential health effects are based.

Both the quantitative and qualitative assessments of uncertainty are directed toward identifying key assumptions and parameters that contribute the most potentially significant human health exposures and effects.

6.9.1 Uncertainties Related to Data Collection and Evaluation for Groundwater

Uncertainty pertaining to data collection and evaluation encompasses sample collection activities, laboratory sample preparation and analysis, and data preparation and analysis. A summary of these issues is presented below.

6.9.1.1 Analytical Data Quality. As discussed in Section 6.2, analytical laboratory, data review, and data validation qualifiers are reported in RCBRA's GiSdT database (http://rcbra.gisdt.org/). Detection status is included as a derived field in the database, where a detect status of TRUE is assigned when a U qualifier (indicating the result was reported below the analytical detection limit) does not occur in any of the qualifier fields pertaining to the analytical laboratory, reviewer, or validator. The number of detections and total number of samples are included as fields in the electronic attachment to Appendix C-3 that provides the groundwater representative concentrations used in the risk calculations.

Uncertainty related to laboratory sample preparation and analysis is not considered to be a significant contributor to overall uncertainty in the groundwater risk assessment results. All groundwater data used in this assessment met the performance requirements defined in the groundwater sampling and analysis plans related to the groundwater monitoring program. *Risk Assessment Guidance for Superfund, Human Health Evaluation Manual, Part A (Interim Final)* (EPA/540/1-89/002) recommends using unfiltered groundwater data for estimating exposure because these data are more representative of unfiltered tap water than filtered samples. However, the use of only unfiltered samples may result in an overestimation of risk because monitoring wells are not pumped on a regular basis. Water samples from infrequently pumped wells may be susceptible to having particulates that would not be present in samples from a domestic supply well that is more frequently pumped.

Method detection limits, quality assurance/quality control measures, and other sampling and analysis requirements can be found in documents prepared for each groundwater OU, as described in Section 6.2.1.

6.9.1.2 Data Representativeness: Spatial and Temporal Distributions of Contaminants. The groundwater data set for each OU was developed by identification of spatially representative monitoring wells and the selection of temporally representative analytical data. Spatially representative wells within each groundwater OU were identified by review of Hanford Site groundwater monitoring well maps and the use of the Hanford Site Geospatial Map Portal. Only wells with unfiltered sample results for potential contaminants (e.g., cations, anions, radionuclides, or organic constituents) were selected for inclusion in the groundwater data set.

The groundwater risk calculations were conducted using groundwater sampling data collected over a period of 5 to 10 years (1998 through 2008). Because wells associated with different monitoring programs have different sampling objectives and protocols, sampling frequencies can vary from a quarterly basis to an annual basis. Therefore, it is difficult to identify one time frame that can adequately represent current concentrations for all monitoring wells and also provide a sufficient number of sampling rounds to support calculation of exposure concentrations. A maximum time period of 10 years for any individual monitoring well was selected to balance concerns of obtaining an adequate number of samples and also representing present-day conditions. Both increasing and decreasing temporal trends in COPC concentrations in a well have been observed in the data set. The significance of such trends for qualifying potential future exposures cannot be ascertained without specific investigation of the conditions in that location.

Analysis of both temporal and spatial trends in contaminant concentrations will be performed in support of proposals for additional groundwater sampling in the RI reports. As is discussed in the context of the results in Sections 6.3 through 6.8, it is reasonable to conclude that health effects related to radionuclides with short half-lives will decrease over time in proportion with radiological decay.

The difference between the 50th and 90th percentiles for certain COPCs in an OU can be very large. This tends to be the case where a COPC is known to be of particular concern, such that areas with the highest groundwater concentrations (central portions of a plume) have very different concentrations than are measured in other locations in the OU. Differences of between tenfold and hundredfold in the CTE and RME OU percentiles are not uncommon. To ensure that RME risks for an OU were not underestimated, when there were many detections of risk-driving COPCs at a well above the 90th percentile for the OU, the well was removed from the OU data set used to calculate percentiles for the OU, and the well was evaluated separately from the rest of the OU data set. The protocol for identifying individual wells for well-specific risk calculations is described in Section 6.2. The differences between the mean and UCL values of COPCs in the wells for which well-specific risks were calculated were small, usually less than a factor of two, indicating low variability in concentrations and relatively high confidence in the estimation of average COPC concentrations in those individual wells.

As described in Section 6.2.3, the use of the 50th and 90th percentiles for estimating groundwater concentrations in an OU was performed for the baseline risk assessment for the 200-ZP-1 OU (DOE/RL-2007-28) and was done for this assessment based on that precedent. These percentiles are used, respectively, to represent CTE and RME COPC concentrations across an entire groundwater OU. However, these percentiles do not necessarily correspond to potential exposure concentrations at any particular location within an OU where a groundwater supply well for a future residence might be located. The groundwater risk assessment results are considered to be screening-level primarily because the 50th and 90th percentiles for an entire groundwater OU do not correspond to estimates of COPC concentrations in a volume of groundwater that might be pumped and used in a manner consistent with the exposure scenario assumptions. Risk assessment results calculated for individual wells where risks are highest represent an upper bound on potential groundwater risks within an OU. As discussed in Section 6.1, the RIs will present an analysis of spatial and temporal trends in groundwater COPC concentrations and propose exposure units for calculating EPCs within each ROD decision area.

Plots of groundwater COPCs that have been the focus of groundwater monitoring, showing concentration isopleths within a ROD decision area, are provided in Sections 6.3 through 6.8. These figures indicate the location of individual monitoring wells and the approximate contours of groundwater COPC concentrations. These figures were originally developed to support the groundwater monitoring program and have been included in this report to facilitate visualization of the groundwater plumes. However, the concentration isopleths of some COPCs were generated to show where groundwater concentrations exceed drinking water standards, rather than to show the complete range of potential exposure concentrations. Some of the contours are, therefore, not ideal for visualizing the boundaries of groundwater concentrations used in the groundwater risk assessment. The figures are nevertheless useful for visualizing the approximate boundaries of risk-relevant COPC contamination in groundwater. More detailed evaluation of spatial patterns of groundwater contamination will be presented in the RI reports.

6.9.2 Uncertainties Related to the Groundwater Exposure Assessment

Uncertainty in exposure parameter values, including behavioral variables and COPC concentrations in groundwater, is assessed by the calculation of CTE and RME risks for the Resident National Monument/Refuge and Subsistence Farmer scenarios. The CTE risks represent an individual with average levels of contact with the exposure media across the various exposure pathways. The RME calculation represents a reasonable maximum exposure condition of contact with the exposure media. As discussed in the text of Sections 6.3 through 6.8, differences between CTE and RME risks varied significantly in some OUs due to differences in the magnitude of the 50th and 90th percentiles used for the CTE and RME calculations. The difference in CTE and RME risk results, due to parameters relating to receptor characteristics (e.g., drinking water consumption, exposure duration), varies between a factor of approximately 3 and 10 depending on the health effect endpoint and scenario.

The exposure assessment for groundwater is limited to direct ingestion, dermal contact, and inhalation exposure routes. Groundwater exposure was not integrated with the soil-mediated agricultural exposure pathways. A discussion of potential risks related to agricultural uses of groundwater is provided in the uncertainty analysis for the soil source term in Section 5.0.

6.9.2.1 Identification of COPCs. As shown in Tables 6-1 through 6-7, the number of samples in a groundwater OU from which EPCs were calculated varies between very few (less than five) to many hundreds and even 1,000 or more. Risk results that are based on very few detections are intrinsically uncertain because a groundwater EPC must pertain to a large volume of water that can serve as a source of exposure over a residential time period. As discussed in the context of COPC identification in Section 6.2, it is considered more feasible to calculate exposure concentrations from relatively few detections for naturally occurring analytes because some distribution of positive values between zero and the detection limit may be presumed for such analytes.

After a review of the boxplots for groundwater concentrations in each OU (Appendix C-12) and consideration of other available information, analytes that were assigned uncertain COPC status in Section 6.2 (Tables 6-8 through 6-14) are those for which a conclusion regarding COPC status was considered to be unsupportable based on the available groundwater data. The specific attributes related to each of these COPC-OU combinations are captured in the column labeled "Explanation" in Tables 6-8 through 6-14. Analytes with uncertain COPC status in each groundwater OU are summarized in Table 6-24. The majority of these analytes are organic chemicals, many with very few associated detections.

Table 6-24 provides information on sample size, number of detections, and both the maximum detected value and a risk-based screening value for tap water published by EPA. In cases where concentrations are near or above risk-based thresholds, additional groundwater data may be necessary to resolve uncertainty regarding the presence and concentrations of the analyte in groundwater. As a general summary, maximum concentrations of organic chemicals listed in Table 6-24 exceed the regional screening level (RSL) values, whereas maximum concentration of metals (with the exception of arsenic) are usually below the RSL values. The information provided in Table 6-24 may be used to inform the selection of analytical methods and numbers of samples for additional groundwater sampling proposed under the RI.

Many of the risk assessment results are related to COPCs for which there are naturally-occurring concentrations in groundwater, or concentrations that may be related to global fallout of man-made radionuclides (e.g., strontium-90). The relative contribution of background levels of these analytes to the tabulated risk results is unknown, but significant background contributions would be expected only for COPCs such as fluoride and nitrate for which risk results are only slightly above thresholds of concern. Therefore, this is not likely to be a significant source of uncertainty in the groundwater risk assessment.

6.9.2.2 Identification of Individual Wells for Supplemental Analysis. The process of identifying individual wells where risk results might exceed those calculated with the 90th percentile for an OU begins with the set of all COPCs that contribute 10% or more to an RME cancer risk greater than or equal to 1×10^{-4}, or HI equal to or greater than 10, in each groundwater OU. However, there were also occasions where there were just a few very high sample results of a COPC and an RME (90th percentile) value that did not result in risk above these thresholds. Those occasions include the following:

- Two results for hexavalent chromium at well 199-K-107A
- One result each for iron and manganese at well 699-43-2 in 100-F/100-IU-2/100-IU-6
- One result for iron at well 199-N-26
- One result for hexavalent chromium at well 199-N-80.

For each of these metals, an RME risk calculation using the one or two available samples would generate a risk result higher than the RME result for the OU. In the case of the 100-N Area in particular, high metal concentrations may be associated with reducing conditions brought on by degradation of petroleum hydrocarbons. Such reducing conditions may result in a change in the equilibrium between metal adsorption on soil and dissolution, favoring increased metal concentrations in solution. Well-specific evaluation of these elevated metal concentrations may be advisable to determine if additional sampling is necessary.

6.9.2.3 Transport Models. As described in Section 3.4.3, a VOC shower volatilization model is used to estimate VOC indoor air concentrations for inhalation exposure. This model does not account for additional contributions of VOCs to indoor air such as dishwashers and washing machines. Therefore, there is a potentially low bias in indoor air VOC exposures and resulting risks for VOC inhalation. The EPA (EPA 600/R-00/096) states that in their experiments the stripping efficiencies for removing VOCs from water ranged from about 6 to 80% for showers, from 20% to 100% for dishwashers, and from 4% to 100% for washing machines. Dishwashers were associated with low but continuous VOC emissions during operation, and the chemical stripping efficiencies for washing machines were observed to be highly sensitive to system operating conditions. Details of the volatilization model and input values used for calculating VOC concentrations in indoor air are provided in Appendix D-1.3.0.

Exposure via direct migration of VOCs from groundwater to indoor air through vadose zone soils was not quantified in the groundwater risk assessment. However, the potential for significant exposure from this pathway was evaluated using EPA's Johnson and Ettinger model (http://www.epa.gov/oswer/riskassessment/airmodel/johnson_ettinger.htm). The methods and results of the modeling conducted to evaluate the potential importance of vapor intrusion into a building are described fully in Appendix D-1.10.0. Vapor intrusion modeling with the Johnson and Ettinger model was conducted for three VOCs (cis-1,2-dichloroethene, tetrachloroethene, and trichloroethene). Trichloroethene was identified as a COPC in four of the groundwater OUs (100-BC-5, 100-KR-4, 100-FR-3, and 300-FF-5); cis-1,2-dichloroethene, and tetrachloroethene were identified as COPCs only in the 300-FF-5 OU. There are uncertainties associated with the modeled indoor air results using the EPA Johnson and Ettinger model because of many assumptions used to represent the subsurface and building conditions. In addition, empirical data are not available to check the modeled output results. However, default assumptions were used for most parameters and the default parameters are designed to provide a conservative estimate of risks associated with potential vapor intrusion. The results of this modeling are summarized below.

The largest potential cancer risk per unit concentration in groundwater was for trichloroethene, which has the highest RME EPC of any of these three VOCs (21 µg/L in the 100-KR-4 OU [100-K ROD decision area]). This concentration corresponds to a potential residential RME lifetime incremental cancer risk of approximately 1×10^{-5} using the Johnson and Ettinger intrusion model. Approximately the same RME cancer risk was calculated for the 100-K ROD decision area using the VOC shower volatilization model. Therefore, cumulative risks from inhalation of VOCs in indoor air (sum of VOCs in indoor air from shower volatilization and vapor intrusion) may be underestimated by approximately a factor of 2. However, these cancer risks are far below the RME Subsistence Farmer groundwater cancer risk of 9×10^{-4} for the 100-K Area, indicating that other groundwater COPCs are a greater contributor to potential risks than VOCs.

Exposure point concentrations for air in a sweat lodge were calculated for the CTUIR Resident and Yakama Resident scenarios. Appendix 4 of Harris and Harper (2004) provides equations for estimating air-phase contaminant concentrations for volatile and semivolatile COPCs in the water used to create steam in the lodge, as well as separate equations for nonvolatile COPCs. As discussed in Section 3.4.3, inhalation exposure to nonvolatile COPCs in the sweat lodge was evaluated in the CTUIR Resident scenario in spite of concerns with the model for calculating these air-phase EPCs. The Harris and Harper (2004) equation for calculating air-phase EPCs for nonvolatile analytes (Equation 3-2) calculates the concentration of a nonvolatile COPC in air as a function of the concentration of water vapor produced by the volatilization of water poured over hot rocks in a sweat lodge. Because nonvolatile contaminants have no vapor pressure, Equation 3-2 does not have a common physical basis with volatile chemicals. It is possible that inhalation of nonvolatile COPCs might occur by an alternative physical model, such as respiration of respirable-size aerosols, if such aerosols were formed when water is poured over the hot rocks in a lodge. However, a model of resuspension of nonvolatile impurities in aerosol form is inconsistent with other mechanical processes involving steam. For example, EPA does not address this pathway in shower volatilization models (EPA 600/R-00/096). It is also inconsistent with the widespread use of steam distillation for commercial water purification.

The results of the groundwater risk assessment calculations for the CTUIR Resident scenario in the six ROD decision areas are shown without the contribution of nonvolatile COPC inhalation exposure in Table 6-25. By comparison to Table 6-19, it is clear that some groundwater risk results for these scenarios are heavily influenced by the nonvolatile COPC inhalation exposure pathway.

6.9.3 Uncertainties Related to the Groundwater Toxicity Assessment

There are a number of general sources of uncertainty pertaining to the assessment of carcinogenicity that were discussed in the uncertainty analysis for the soil source term risk assessments in Sections 4.5 and 5.9.

6.9.3.1 Updating of Toxicity Criteria. Toxicity criteria for chemical COPCs are continually revised by EPA and state agencies through regular review of more recent toxicological data and updates to methods for analyzing the data. Therefore, risk assessment calculations based on these criteria may become outdated over time as our understanding of chemical toxicity improves. The chemical toxicity criteria for this risk assessment were current in spring and summer 2009. As this report was finalized (winter 2009/2010), toxicity criteria published by EPA in December 2009 in support of their Regional Screening Levels were reviewed (http://epa-prgs.ornl.gov/radionuclides/). Revised toxicity criteria in the RSL tables affect two chemicals that are groundwater COPCs in this risk assessment. These chemicals are trichloroethene and hexavalent chromium. The oral CSF for trichloroethene published by the state of California has decreased from 0.013 to 0.0059 per mg/kg-day. This change would result in a twofold decrease to the calculated trichloroethene cancer risk by ingestion of water in this assessment.

The recent publication of oral CSFs for hexavalent chromium could have a significant impact on the results of the groundwater risk assessment if such a value was used in the RCBRA. An oral CSF for hexavalent chromium was independently derived by the states of New Jersey and California in 2009 based on the results of a 2-year rodent study performed by the National Toxicology Program, which is an interagency program within the Public Health Service of the Department of Health and Human Services. Hexavalent chromium did not previously have a published oral CSF. Only inhalation exposure was historically considered to be important for hexavalent chromium carcinogenicity, due to reduction of chromium from the hexavalent to trivalent state in the lungs.

The assessment of hexavalent chromium cancer risk via the ingestion exposure route using the oral CSF published by New Jersey (0.5 per mg/kg-day) or California (0.6 per mg/kg-day) could significantly increase estimated cancer risks for hexavalent chromium. As an approximate measure of the potential significance of this issue, the risk-based tap water RSL published in December 2009 using an oral CSF of 0.5 per mg/kg-day is 0.043 µg/L using a threshold cancer risk of 1×10^{-6}. In the April 2009 RSLs, using an oral RfD of 0.003 per mg/kg-day and an HQ of 1.0, the risk-based tap water screening level was 110 µg/L. The protocol for identifying toxicity criteria for use in the RCBRA, described in Section 3.5.7, states that Tier 3 criteria such as values published by state agencies are generally not used when Tier 1 (IRIS) values exist for the same exposure route. This would be the case for hexavalent chromium because an IRIS oral RfD is published for this COPC.

6.9.3.2 Cancer Risk and Radiation Dose. The main sources of uncertainty pertaining to the assessment of carcinogenicity include (1) high-to-low dose extrapolation, (2) uncertainty in the applicability of the no-threshold model of carcinogenicity, (3) the common use of a UCL (typically 95%) on the slope of the dose-response curve for chemical CSFs, and (4) uncertainty in whether a particular chemical is in fact a human carcinogen. As discussed in Section 3.5.3, most chemical CSFs are based on carcinogenic effects observed at relatively high dose rates in test animals that have been extrapolated to lower dose rates in humans. The human data on which radionuclide CSFs are based are also based on high-to-low-dose extrapolation. The underlying assumption for both chemicals and radionuclides is that even a very low level of exposure carries some risk of carcinogenesis.

In the groundwater risk assessment, significant carcinogenic risk in the Resident National Monument/Refuge and Subsistence Farmer scenarios was mostly related to radionuclides rather than a combination of radionuclides and chemicals. Sources of uncertainty in the CSFs for radionuclides include high-to-low dose extrapolation and uncertainty in the applicability of the no-threshold model of carcinogenesis. Uncertainty in the cancer risk results for the CTUIR Resident scenario may be primarily related to the sweat lodge air EPCs for nonvolatile COPCs, as described in Section 6.9.2.

Uncertainties in extrapolating carcinogenic response from high-to-low dose rates and in the no-threshold model of carcinogenicity are related. The human body has the capability to repair DNA damage that may lead to cancer, as well as a variety of cellular mechanisms for halting the proliferation of cells that, due to DNA damage, have developed characteristics of uncontrolled growth. A certain amount of DNA damage is naturally incurred at all times independent of any exposure to environmental contaminants. At low radiation dose rates common to environmental exposures, it becomes impossible in either human or test animal populations to observe the slight potential increase in cancer incidence that may be related to exposure.

Groundwater cancer risks for the CTUIR Resident and Yakama Resident scenarios were above 1 in 100 in several instances, primarily in relation to inhalation exposure to nonvolatiles in the sweat lodge. Exposure to radionuclides in some OUs and individual wells is associated with effective dose rates of over 1,000 mrem/yr. The radionuclide cancer risk slope factors used in this risk assessment are applicable at relatively low absorbed dose rates of 0.1 mGy/min or less, as discussed in EPA's *Cancer Risk Coefficients for Environmental Exposure to Radionuclides,* Federal Guidance Report No. 13 (EPA 402/R-99/001). The gray (Gy) is a unit of absorbed radiation dose (1 Gy = 100 rad), whereas the rem is a measure of dose that incorporates biological effectiveness. In the case of photon and electron emissions, such as those from tritium, carbon-14, and strontium-90, absorbed and effective doses are equivalent (ICRP 1977, *Recommendations of the International Commission on Radiological Protection*).

Converting to units commensurate with the rem, the radiation slope factors for tritium, carbon-14, and strontium-90 are applicable up to dose rates of approximately 600 mrem/hr:

$$0.1 \text{ mGy/min} \times 60 \text{ min/hr} \times 100 \text{ mrad/mGy} \times 1 \text{ mrem/1 mrad} = 600 \text{ mrem/hr} \qquad \text{Equation 6-1}$$

The groundwater cancer risk results lie well within the range of applicability of these slope factors as defined by the limit of 0.1 mGy/min. Nevertheless, because of the linear dose-response relationship assumed for radionuclide carcinogenesis (EPA 402/R-99/001), this limit cannot necessarily be applied when calculating cancer risks from chronic radiation exposure over the course of many years. For chemical carcinogens, EPA guidance (EPA/540/1-89/002, Section 8.2) suggests 1 in 100 (1×10^{-2}) as the upper limit for applying the linear model to estimate cancer risk. This limit has also been applied to radionuclide cancer risk results. This is consistent with guidance in EPA 402/R-99/001, that states CSFs should be "applied with care when the cumulative radiation dose over time is large."

There are also uncertainties in the estimation of radiation dose, as well as the assessment of carcinogenic risk associated with any particular dose. As described in Section 3.5.5, age-dependent internal dose coefficients published in *Age-Dependent Doses to Members of the Public from Intake of Radionuclides: Part 5, Compilation of Ingestion and Inhalation Dose Coefficients* (ICRP 1996) were used for the radiation dose calculations. For the residential exposure scenarios that include both child and adult receptors, age-dependent dose coefficients that apply to a 30-year exposure period were calculated because this is the exposure period applied in the RME calculations for these scenarios. However, when calculating radiation dose for a child receptor age 1 to 6 years, these DCFs will tend to underestimate the actual dose, because effective dose is usually lower for an adult than a child. A discussion of the calculation of age-dependent DCFs, including a comparison of DCFs for specific age classes to the 30-year exposure DCFs used in this assessment, is provided in Appendix D-3.

6.9.3.3 Summing of Cancer Risks for Chemicals and Radionuclides. Cancer risks for each exposure route and COPC (radionuclides and chemicals) are summed to calculate cancer risk to an individual. This methodology is consistent with CERCLA guidance in OSWER Directive 9200.4-18, "Establishment of Cleanup Levels for CERCLA Sites with Radioactive Contamination," The EPA reiterated their recommendation for summing chemical and radionuclide cancer risks in a 1999 memorandum titled *Distribution of OSWER Radiation Risk Assessment Q&A's Final Guidance,* available online at http://www.epa.gov/superfund/health/contaminants/radiation/radrisk.htm. Separate sums of cancer risk for radionuclides and chemicals are provided in the electronic results files in Appendix D-5, but summary tables in the main report provide cancer risk results for the sum of all COPCs.

One argument for assessing chemical and radionuclide cancer risks separately is that the basis of the CSFs for these classes of analytes is sufficiently different enough that separate presentations are warranted. As discussed in Section 3.5, most CSFs for chemical COPCs are upper 95th percentile estimates, whereas the radionuclide CSF reflects an average estimate of the lifetime risk of cancer associated with exposure to a specific concentration of a carcinogen in an environmental medium. Also, the majority of chemical CSFs are based on animal data, but radionuclide CSFs are derived from the observed effects of radiation doses in humans. Ultimately, chemical and radiological CSFs both involve a similarly high degree of uncertainty related to extrapolation of effects at high doses to hypothetical effects at low (environmental) dose rates. Also, EPA is moving towards a mode-of-action approach for evaluating chemical carcinogenicity that stresses whether a chemical is carcinogenic by mutagenic or nonmutagenic processes (EPA/630/P-03/001F). From that perspective, additivity of cancer risk for mutagenic chemicals and radionuclides may be more defensible in principle than additivity of risk among chemicals that operate by mutagenic and nonmutagenic modes of action.

A separate consideration is that because most CSFs for chemical COPCs are upper 95th percentile estimates, the risks calculated using these criteria are not strictly additive. For this reason, the total cancer risk estimate for chemicals becomes increasingly biased in a conservative manner as the number of summed carcinogens increases. Within this context, the summing of chemical and radionuclide risk does not substantially increase the protective bias of sums across both chemicals and radionuclides because radionuclide CSFs are central tendency rather than upper-bound estimates of dose response.

6.9.3.4 Systemic Toxicity. General sources of uncertainty pertaining to the assessment of systemic toxicity include (1) the application of uncertainty factors on the dose-response data, (2) relying on a single critical effect to measure toxicity, and (3) toxicological interactions among the various COPCs. Uncertainty factors are used to account for several possible sources of uncertainty in developing an RfD including extrapolating from the no observable adverse effect level or lowest observable adverse effect level to a chronic RfD, variability in sensitivity in the human population, interspecies variability between humans and test animals, and inadequate dosing periods in a critical study. These uncertainty and related modifying factors are designed to introduce a protective bias in the toxicity criteria such that the potential for adverse effects in sensitive human subpopulations will not be underestimated.

As described in Section 3.6.1, the calculation of an HI sum by adding the COPC-specific hazard quotients may result in overestimation of systemic hazards when COPC toxicities pertain to different target organs. There are some instances in the groundwater risk assessment where two or more chemicals contributed in a roughly equal manner to an HI above 1.0. Specific examples are discussed in relation to the groundwater risk assessment results presented in Sections 6.3 through 6.8.

Section 3.5.2 provides an overview of the toxicity criteria used to evaluate health effects other than cancer. These include the oral RfD and the inhalation reference concentration (RfC). Each chemical RfD and RfC value is based on dose-response data from animal or human studies. From these observed effect levels, RfD and RfC values are developed by applying uncertainty and modifying factors, which are used as protective multipliers to account for various uncertainties when applying these data to a general human population. The extent to which uncertainty and modifying factors in the chemical RfD and RfC values impact the results of the risk assessment is chemical specific. Uncertainty and modifying factors in oral RfDs for some of the different chemicals for which significant chemical hazard was observed in the Resident National Monument/Refuge and Subsistence Farmer scenarios are as follows.

Contaminant of Potential Concern	Uncertainty Factor	Modifying Factor
Chromium	100	10
Iron	– [a]	– [a]
Fluoride	1	1
Nitrate	1	1
Uranium	100	1
Zinc	3	1

[a] This is a provisional value with no associated factors in the source referenced in Section 3.5.

Calculated hazard quotient values for chromium and uranium are, therefore, associated with a much higher degree of uncertainty, which has been addressed by introducing protective bias to the RfD, than those for fluoride, nitrate, and zinc.

Some of the metal COPCs for which chemical hazard due to water ingestion is quantified in the groundwater risk assessment (i.e., chromium, iron, and zinc) are also essential micronutrients required by the body for normal functioning. The body normally exerts a degree of homeostatic control over the body burdens of these metals following ingestion exposures, meaning that uptake is regulated depending on need. In order for uptake followed by toxicity to occur, the body's control mechanisms on uptake must be overwhelmed or incapacitated. Thus, chronic toxicity calculated as hazard quotients below or only a few times higher than 1.0 for exposure to these metals is probably unrealistic.

6.10 REFERENCES

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7.0 CONCLUSIONS AND RECOMMENDATIONS

This section provides a summary of the human health baseline risk assessment component of the River Corridor Baseline Risk Assessment (RCBRA), including key conclusions pertinent to the remedial investigation/feasibility study (RI/FS) process in the River Corridor. The purpose of the RCBRA is to characterize the current and potential future threats to human health and the environment that may be posed by contaminants in soil, sediment, surface water, and groundwater of the River Corridor, and to provide the baseline risk assessment needed to support records of decision (RODs) for final remedies (DOE/RL-2004-37, *Risk Assessment Work Plan for the 100 Area and 300 Area Component of the RCBRA*). The human health risk assessment in this volume follows the evaluation process described in *Risk Assessment Guidance for Superfund, Volume 1, Human Health Evaluation Manual (Part A) (Interim Final)* (EPA/540/1-89/002).

The human health risk assessment is a key component of the RI/FS process under the *Comprehensive Environmental Response, Compensation, and Liability Act of 1980* (CERCLA) and provides information related to key contaminants and environmental media that may require further evaluation in the FS. The main objectives of a human health risk assessment are to:

- Provide an analysis of baseline risks and help determine the need for action at sites
- Provide a basis for determining levels of chemicals and radionuclides that can remain onsite and still be adequately protective of public health
- Provide a basis for comparing potential health impacts of various remedial alternatives
- Provide a consistent process for evaluating and documenting public health threats at sites.

In addition to meeting the objectives listed above, the RCBRA addresses the following questions to provide information needed by risk managers to support final CERCLA decisions in the River Corridor that ensure protection of human health and the environment.

- Are residual conditions in the River Corridor following cleanup actions completed under the interim action records of decision (IARODs) protective of human health and the environment under the various RCBRA exposure scenarios?
- What are the uncertainties associated with the RCBRA risk results and conclusions?
- Are soil cleanup levels established under the IARODs protective of human health and the environment using current regulatory guidelines?
- What are the preliminary remediation goals (PRGs) for the various RCBRA exposure scenarios?

- Are there any recommendations for additional studies or monitoring that should be considered at this time to reduce uncertainties with specific risk results and conclusions or establishing health-protective PRGs?

The human health risk assessment has evaluated the protectiveness of the existing cleanup levels in the IARODs in accordance with current regulatory guidance. It has also evaluated the residual risks at remediated waste sites and potentially affected areas using a variety of exposure scenarios, including risks from exposure to contaminants present in riparian soils, river sediments and surface water, and groundwater. The results of these evaluations will be used in the 100 and 300 Area RI/FS reports to determine the need for further cleanup at waste sites that underwent interim actions and to identify PRGs that will be used in the RI/FS to evaluate yet-to-be remediated waste sites in the River Corridor. Sections 7.1 through 7.5 provide a summary of the key information presented in the human health risk assessment including a description of the sources of data used in the risk calculations and uncertainties associated with the data, the process for selection of contaminants of potential concern (COPCs), the methodology for calculating representative concentrations (RCs) for individual remediated waste sites and other potentially affected areas of the River Corridor, risk results for a range of exposure scenarios that cover a broad range of possible future uses of the River Corridor, and uncertainties associated with the risk results. In addition, PRGs for soil are presented and discussed for the various exposure scenarios.

The contents of this section are organized as follows:

- Section 7.1, Background Information – Contains a summary of the cleanup strategy used in the River Corridor and how the RCBRA will be used to support the FS and final RODs for the River Corridor.

- Section 7.2, Overview of Human Health Risk Assessment in the River Corridor – Contains a summary of previous risk assessments for the River Corridor, methods used for establishing the IAROD cleanup levels, and the methods used to conduct the RCBRA including a description of the data sets, COPC selection process, and methods for calculating RCs.

- Section 7.3, Quantitative Evaluation of Human Health Risks – Provides an evaluation of risk using the IAROD cleanup level scenarios with updated toxicity criteria and target risk thresholds and a summary of the risk assessment results, including uncertainty analysis and recommendations, for the broad area risk assessment, local-area risk assessment, and screening-level groundwater risk assessment.

- Section 7.4, Preliminary Remediation Goals – Provides tables of PRGs for various exposure scenarios that may be used in the FS for analysis of remedial alternatives.

- Section 7.5, Key Conclusions of the Human Health Risk Assessment – Provides a summary of the key conclusions, including risk-driving COPCs and exposure pathways, key uncertainties, and recommendations for additional evaluation.

- Section 7.6, References.

7.1 BACKGROUND INFORMATION

The following section provides a brief summary of cleanup activities in the River Corridor. Additional detail is provided in Sections 2.5 and 2.6.

In 1989, the 100 Area and the 300 Area were placed on the National Priorities List under the authority of CERCLA. Together, the 100 Area and 300 Area comprise the River Corridor. Placement on the National Priorities List initiated the CERCLA process that would result in the cleanup of contaminated areas that pose a threat to human health and the environment. The U.S. Department of Energy (DOE), the U.S. Environmental Protection Agency (EPA), and the Washington State Department of Ecology (Ecology) (collectively called the Tri-Parties) developed an approach for expedited remediation of the River Corridor in 1991. The Tri-Parties decided that enough information was known about contaminated soil at the Hanford Site to begin cleanup with a focus on protecting groundwater and the Columbia River. Cleanup decisions were established through IARODs based on existing knowledge of the waste sites (e.g., site types, processes, contaminants) and supplemented by limited amounts of characterization. In 1995, cleanup actions were initiated focusing on removal of contaminated soil and debris from waste sites with the highest potential to impact groundwater and the Columbia River. Actions to address existing plumes of groundwater contamination were also initiated.

Waste site and groundwater cleanup actions in the River Corridor have continued from 1995 to date. Cleanup actions at hundreds of waste sites have been completed to date, with a majority of the cleanup work in the River Corridor anticipated to be complete by 2015. At each waste site where remediation has occurred, the goals and objectives of the IARODs have been met as demonstrated by verification documentation that has been completed and submitted to the DOE and approved by the regulatory agencies.

Unacceptable risks are present in the River Corridor at waste sites that are identified in the IARODs but have yet to be remediated. Qualitative risk assessments (QRAs) provided the original determination of unacceptable risk and basis for action for waste sites in the River Corridor. The original determination of the presence of unacceptable risk and basis for action at yet-to-be remediated waste sites is supported by the field experiences and information gathered through implementation of the observation approach based soil cleanup actions in the River Corridor over the past 13 years. The risk associated with the quantity and type of waste that has been excavated confirms unacceptable risks that appropriately drive CERCLA cleanup actions. Additional details of the QRA results are presented in Section 3.1.

This information supports the conclusion that there is currently unacceptable risk at yet-to-be remediated waste sites in the River Corridor. In parallel with establishing final cleanup actions for the River Corridor through the RI/FS process, the DOE is committed to continuing cleanup actions at these sites according to goals and objectives of the IARODs.

Remedial investigation/feasibility study processes under CERCLA have been initiated for the River Corridor to gather and evaluate information needed to make final cleanup decisions. The process to pursue cleanup decisions has been organized into pieces of work that are aligned with Hanford Site operational functions. Six final remedy RODs will be developed associated with the operations areas.

The final remedy decision areas and the size of each are as follows:

- 100-B/C — 11.53 km^2 (4.45 mi^2)
- 100-D/100-H — 20.31 km^2 (7.84 mi^2)
- 100-K — 8.99 km^2 (3.47 mi^2)
- 100-N — 8.88 km^2 (3.43 mi^2)
- 100-F/100-IU-2/100-IU-6 — 376.15 km^2 (145.23 mi^2)
- 300 Area — 145.95 km^2 (56.35 mi^2).

A key element in the RI/FS decision-making process is the performance of a baseline risk assessment. Under CERCLA, a baseline risk assessment is needed to provide risk managers with an understanding of the current and potential future risks posed by a site to support RODs documenting final remedies. The RCBRA is being conducted while cleanup actions at waste sites are under way. As such, baseline conditions that are assessed include a mix of areas where cleanup has been completed in accordance with the IARODs, areas that are currently scheduled for cleanup, and areas that are currently not identified for cleanup actions.

This human health risk assessment (Volume II) provides the human health portion of the RCBRA and presents a comprehensive assessment of the River Corridor, considering all relevant sources of contamination, exposure pathways, and contaminants. The methodologies and exposure scenarios used in the human health risk assessment were developed with stakeholder input through a series of workshops that included the Tri-Parties as well as other stakeholders. Volume II will be used, with a complementary ecological risk assessment (Volume I), to support final cleanup 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 develop final cleanup decisions that will be protective of human health and the environment. Final cleanup decisions, applying to all portions of the River Corridor, will be identified in proposed plans that will undergo public review and be documented in RODs.

7.2 OVERVIEW OF HUMAN HEALTH RISK ASSESSMENT IN THE RIVER CORRIDOR

This section provides a summary of risk assessment activities that have been done for the River Corridor and outlines the human health risk assessment methodology used for the RCBRA. Risk assessment methods have evolved through time and, therefore, there may be differences in approaches used for risk assessment activities that were conducted for the River Corridor in the past as compared to the RCBRA. It is important to understand these differences in approaches when comparing risk results from earlier assessments to the RCBRA results. This section provides information about previous risk assessments that have been conducted for the River Corridor, a description of the basis for the IAROD cleanup levels, and a summary of the methods used for the human health risk assessment component of the RCBRA. Additional detail on past risk assessments is provided in Section 2.5. Details of the RCBRA human health risk assessment methodology is provided in Section 3.0.

7.2.1 Previous Risk Assessments

7.2.1.1 Qualitative Risk Assessments. In 1991, the Tri-Parties adopted a "bias-for-action" approach to expedite the decision-making process and allow cleanup actions in the River Corridor to begin as soon as possible. Known as the *Hanford Past-Practice Strategy* (HPPS) (DOE/RL-91-40), this "bias-for–action" approach streamlined the RI/FS process to begin remediation of contaminated waste sites earlier than typically performed under the traditional CERCLA process. The HPPS incorporated limited field investigations (LFIs), focused feasibility studies, and QRAs, and allowed further site characterization to proceed in tandem with waste site remediation. The waste sites with the highest potential to contribute to contamination of groundwater and the Columbia River were prioritized for the first remediation efforts. The QRAs were performed using limited data and included abbreviated quantitative risk analyses. The approach outlined in the HPPS is consistent with later EPA initiatives developed to support expedited cleanups, such as the *Superfund Accelerated Cleanup Model* (EPA/540/R-98/025) and the *RCRA Facility Stabilization Initiative* (DOE/EH-231-076/0295r).

The QRAs were performed for the high-priority sites in each operable unit (OU). Conservative assumptions, such as highest reported contaminant levels from either the LFI or historical data from *Radiological Characterization of the Retired 100 Areas* (UNI-946), were used in the QRAs. The QRAs provided estimates of human health risks, assuming frequent use and occasional use, and included considerations such as the attenuation of external dose provided by layers of clean gravel fill that overlie many sites. The QRAs identified the human health risk to be primarily from external exposure to the radionuclides cobalt-60, cesium-137, europium-152, and europium-154. Ecological risks were also estimated for a single receptor, the great basin pocket mouse.

The high-priority sites were evaluated using the following criteria to help identify those recommended for remedial actions:

- Magnitude of risk identified in the QRA
- Exceedance of a chemical-specific applicable or relevant and appropriate requirement
- Potential to contaminate groundwater
- Insufficient information for conceptual model
- Multiple exposure pathways
- Expected natural attenuation and radioactive decay.

The QRAs were used to determine whether contaminant concentrations pose an unacceptable risk that warrants remedial action and have been used to establish the basis for action for all waste sites identified in the River Corridor IARODs and RODs.

7.2.1.2 Columbia River Comprehensive Impact Assessment. Information gathered and lessons learned from a prior study called the Columbia River Comprehensive Impact Assessment (CRCIA) were incorporated into the RCBRA. The purpose of CRCIA was to assess the effects of Hanford Site-derived materials and contaminants on the Columbia River environment, river-dependent life, and users of river resources for as long as those contaminants remain intrinsically hazardous. The CRCIA screening assessment scope included current conditions, the Columbia River and adjacent riparian zone between Priest Rapids Dam and McNary Dam, a limited number of contaminants, a limited amount of monitoring data, a limited number of species, and a limited number of scenarios. Several documents were published during the course of the CRCIA project, the most comprehensive of which is *Screening Assessment and Requirements for a Comprehensive Assessment*: *Columbia River Comprehensive Impact Assessment* (DOE/RL-96-16).

The nature of the potential exposure scenarios to be used in the RCBRA has been the subject of numerous discussions among the Tri-Parties and various stakeholders. One outcome of the early discussions was a decision to implement a pilot human health and ecological risk assessment for the 100-B/C Area. The draft pilot assessment is documented in *100-B/C Pilot Project Risk Assessment Report* (DOE/RL-2005-40). The 100-B/C Pilot Project identified five exposure scenarios: a Rural-Residential scenario (called the Subsistence Farmer scenario in this report), Resident Monument Worker scenario, Industrial/Commercial Worker scenario, Recreational Use scenarios (Avid Hunter, Avid Angler, and Casual User applications), and a Native American User scenario. Among these exposure scenarios, contaminant exposure and potential health effects were quantified in the 100-B/C Pilot Project for all except a Native American User scenario.

Information from the previous risk assessment projects in the River Corridor is being integrated into the RCBRA. Results from the 100-B/C Pilot Study and the *100-NR-2 Study Area Ecological Risk Assessment Sampling and Analysis Plan* (100-NR-2 ecological risk assessment) (DOE/RL-2005-22) are part of the data set being used for the RCBRA and are integrated into the report to present a comprehensive picture of current and potential threats to human health and the environment from contaminants in the River Corridor.

7.2.2 IAROD Cleanup Levels

The cleanup levels specified in the IARODs were referred to as remedial action goals (RAGs). The RAGs are contaminant-specific numerical cleanup criteria developed to guide the remedial actions to meet the remedial action objectives. The RAGs used in the IAROD process employed residential and industrial exposure scenarios to evaluate risks from contaminants in soil. Cleanup levels in the 100 Area of the River Corridor are based on a residential exposure scenario For radionuclides, a residential scenario that included food ingestion and drinking water exposure pathways was implemented to develop the RAGs for protection of human health. For chemicals, RAGs for protection of human health were based on the *Model Toxics Control Act* (MTCA) Method B "Soil Cleanup Levels for Unrestricted Land Use" (WAC 173-340-740). The MTCA Method B cleanup levels are based on incidental soil ingestion and use residential exposure frequency assumptions.

Cleanup levels for the 300 Area are based on a mix of residential and industrial exposure scenarios. The RAGs that were applied to interim actions for the waste sites reflected regulatory requirements and guidelines in place in the mid-1990s. Though regulatory guidance has evolved in subsequent years, these cleanup levels have been retained and applied consistently to all waste sites being cleaned up under interim action in the River Corridor (DOE/RL-96-17, *Remedial Design Report/Remedial Action Work Plan for the 100 Area*).

The IAROD RAGs for radionuclides correspond to a 15 mrem/yr radiation dose. An EPA memorandum published in 1997 related to cleanup goals for radionuclides at CERCLA sites (OSWER Directive 9200.4-18, "Establishment of Cleanup Levels for CERCLA Sites with Radioactive Contamination") stated that for sites where a dose assessment is conducted, 15 mrem/yr should generally be the maximum dose limit for humans. The memorandum also stated that 15 mrem/yr equates to approximately 3×10^{-4} risk and is consistent with the upper end of the target risk range of 10^{-6} to 10^{-4} described in the "National Oil and Hazardous Substances Contingency Plan" (NCP) (40 CFR 300.430).

Remedial action goals related to radiation dose were calculated in the IAROD process using the RESidual RADioactivity (RESRAD) computer code, the use of which is consistent with DOE's requirements for assessing doses to the public for the cleanup of residual radioactivity material and the release of property (DOE Order 5400.5, *Radiation Protection of the Public and the Environment*). Residential scenario RAGs were calculated for shallow zone soil (0 to 4.6 m [0 to 15 ft] below ground surface [bgs]) based on a number of exposure pathways including soil ingestion, dust inhalation, external radiation, produce ingestion, beef ingestion, and milk ingestion. The RESRAD calculations assume that the land is irrigated to support the agricultural exposure pathways, and also include modeling of radionuclide leaching to groundwater followed by drinking water and aquatic foods exposures. The RESRAD code was also used to calculate a second set of RAGs specifically related to groundwater protection. These RAGs focused on leaching of residual radionuclide contamination in deep zone soil (>4.6 m [15 ft] bgs).

Remedial action goals related to chemical cancer risk and hazard were based on screening models of the State of Washington's MTCA published in the 1996 WAC Part 173-340. The RAGs for chemicals correspond to a target cancer risk of 1×10^{-6} or a noncancer hazard quotient of 1. Consistent with the MTCA screening models, the chemical RAGs were calculated based on a soil ingestion exposure pathway, using MTCA exposure factor values that were drawn from EPA guidance. A MTCA soil screening model for groundwater protection was also used to establish additional chemical RAGs for soil.

As described in the previous paragraphs, the IAROD RAGs for chemicals and radionuclides were based on two different exposure models. The RAGs for chemical constituents were based on MTCA soil cleanup levels that include only the soil ingestion exposure pathway. The RAGs for radionuclide constituents were calculated using RESRAD and included the soil ingestion pathway and many additional exposure pathways. The differences between the risk models for the chemical and radionuclide RAGs were greatest for the residential scenario, where the RESRAD calculations addressed exposure via dust inhalation, homegrown foodstuffs (e.g., produce, beef, and milk), drinking water ingestion, and fish ingestion.

7.2.3 Summary of Human Health Risk Assessment Methods Used in the RCBRA

As noted above, the human health risk assessment in this volume follows the evaluation process described in EPA/540/1-89/002. The following four steps comprise the EPA baseline human health risk assessment process:

1. Data collection and evaluation – site data are compiled and analyzed and COPCs are identified.

2. Exposure assessment – potentially complete exposure pathways are identified including potential receptors and environmental media to which they may be exposed. Exposure parameters for the various receptors/exposure scenarios are identified.

3. Toxicity assessment – toxicity criteria used in the quantitative risk calculations are compiled.

4. Risk characterization – quantitative risk estimates are calculated and uncertainties associated with the risk estimates are identified and summarized.

The methods used in the RCBRA for each step in the EPA human health risk assessment process are summarized below.

7.2.3.1 Data Collection and Evaluation. Data collection and evaluation includes a summary of the relevant historical operations and releases, a discussion of the sampling and analysis conducted, presentation of data processing protocols, and methods for background comparisons and identification of COPCs. Details of the data collection and evaluation are provided in Section 3.2 (Methodology for Data Evaluation).

Two types of soil data were used to calculate cancer risks and noncancer hazards associated with exposure to areas of the River Corridor: (1) data from cleanup verification soil samples, and (2) characterization data collected specifically for the RCBRA in upland, riparian, and near-shore environments. Cleanup verification soil samples were collected at individual remediated waste sites to document completion of remedial actions according to (primarily) IARODs. Most of the cleanup verification samples were collected by combining numerous individual subsamples into a single composite sample that was submitted for laboratory analysis. These were referred to as "statistical" samples because they were intended to be used to produce statistical estimates of average soil concentrations at the waste site. A second kind of cleanup verification sample, the "focused" samples, are individual samples collected from a single location based on where contamination was either evident or expected. Focused samples were also commonly collected subsequent to a targeted soil removal to confirm that residual concentrations were below IAROD cleanup levels.

The cleanup verification soil data are also distinguished by whether they were collected in shallow zone soil (0 to 4.6 m [0 to 15 ft] bgs) or deep zone soil (>4.6 m [15 ft] bgs). Consistent with the conceptual site model described in Section 3.3, only shallow zone sample results were used to assess human health risks from residual soil contamination at individual remediated waste sites. These cleanup verification soil data are used to conduct the local-area risk assessment described in Section 5.0.

The data used in the human health risk assessment to characterize exposure at locations other than an individual remediated waste site were collected under the *100 Area and 300 Area Component of RCBRA Sampling and Analysis Plan* (RCBRA SAP) (DOE/RL-2005-42), supplemented by data obtained during the 100-B/C Pilot and 100-NR-2 projects. A *MULTI INCREMENT*® sampling (MIS) approach was used for the soil samples collected under the RCBRA SAP (DOE/RL-2005-42). RCBRA multi-increment soil samples represent the top 15.2 cm (6 in.) of soil within an area of approximately 1 ha (2.47 ac) and consist of approximately 50 soil increments collected on a spatial grid. The MIS upland surface soil data are used to calculate broad-area representative concentrations, which describe contaminant concentrations that are applicable over the scale of an entire ROD area or larger. The MIS upland surface soil samples are associated with 20 remediated waste sites that represent a cross section waste site types, contaminants, and remediation methodology. The COPC concentrations for these data are assumed to be a conservative representation of average COPC concentrations over the entire River Corridor upland environment. The number of COPC detected and the concentrations near remediated waste sites are expected to be greater than in other less disturbed locations in the River Corridor. However, because of the small size of this data set, there are

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uncertainties associated with using these data to represent average surface soil concentrations for the entire River Corridor for the broad-area assessment.

The RCBRA SAP originally identified sampling locations within the 100 Area and 300 Area. The SAP was amended during the first year of field sampling to include sampling of the shoreline regions between the operating areas, referred to as the “Inter Areas.” This assessment uses data from both the operating and the “Inter Areas” to characterize contaminant concentrations in the River Corridor and does not differentiate between the original sampling distinctions. Together, these data allow evaluation of exposures within the upland, riparian, and near-shore environments of the River Corridor. The RCBRA data, supplemented by the 100-B/C Pilot and 100-NR-2 project data, are used to conduct the broad-area risk assessment described in Section 4.0.

The COPCs that were carried through the quantitative risk assessment process for the local-area and broad-area risk assessments were identified through a process that is based on approaches and methods presented in the Tri-Parties-approved RCBRA SAP (DOE/RL-2005-42). This process is described in Section 3.2. The RCBRA SAP outlined a process for focusing contaminants based on comparing mean concentrations at study sites to background or reference sites using conclusions and data summaries from LFIs, cleanup verification packages (CVPs), Hanford Site monitoring, and related projects. 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”).

The COPC refinement process includes a number of complementary steps and criteria, including a pre-selected list of contaminants that will be excluded and a list that will be included, as determined and agreed upon among the Tri-Parties. 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. Additional selection steps included evaluation of all of the data according to detection status, statistical comparisons of Hanford Site data to background and reference site data, and an analyte-specific evaluation. The analyte-specific evaluation integrated a variety of information (such as the magnitude and significance of statistical comparisons results, sample results in other media, and sample results for similar analytes) to support a conclusion on COPC identification when the results of statistical comparisons were inconclusive.

The primary data used in the human health risk assessment to characterize groundwater exposure over the separate ROD areas (Section 6.0) were obtained from the Hanford Environmental Information System database. For the purpose of characterizing groundwater exposure to human receptors, decision areas are assigned on the basis of the boundaries of each groundwater OU. The following groundwater OUs are evaluated in this risk assessment: 100-BC-5, 100-KR-4, 100-NR-2, 100-HR-3, 100-FR-3, 300-FF-5, and the groundwater underlying the 100-IU-2 and 100-IU-6 decision area. The groundwater underlying the portion of the 100-IU-2 and 100-IU-6 source area OU is assigned to the 200-PO-1 OU. Groundwater data from unfiltered samples collected during the past 5 to 10 years from a selection of wells representative of groundwater conditions between 1998 and 2008 were used in the human health risk assessment.

7.2.3.2 Exposure Assessment. The exposure assessment focuses on identifying how much of a chemical is present in an environmental medium, determining who might be exposed and how, and quantifying the rate of exposure. Both current and potential future exposure scenarios are considered in the human health risk assessment. Details of the exposure assessment are provided in Section 3.3 (Conceptual Site Model) and Section 3.4 (Methodology for Exposure Assessment).

Present-day exposures in the River Corridor are controlled by access restrictions. Present-day activities in these areas are limited to security surveillance and remedial action and sampling activities conducted by remedial action workers. Activities conducted by Hanford Site workers are managed under a site health and safety plan. This plan addresses worker training and protective measures to minimize potential exposures and requires monitoring of potential radiological exposure, where necessary. Because potential exposures and associated risks are monitored for these workers, they are not considered potential receptors for the RCBRA.

For future land use, a range of exposure scenarios, ranging from the hypothetical situation where recreational users occasionally visit the site to residents who live onsite and consume food items that are predominantly grown or raised onsite, are evaluated in the RCBRA. In order to account for the changes in contaminant concentrations over time due to radionuclide decay, risk assessment results for each exposure scenario are presented for a number of points in time. As discussed in Section 3.4.3, cancer risk and radiation dose are calculated using present-day radionuclide activities in soil, and with radionuclide activities in soil decayed to the years 2075 and 2150.

The exposure scenarios evaluated in the RCBRA have been grouped according to general types of potential exposure and land use. The types of exposure scenarios evaluated in the RCBRA include the following:

- **Recreational and Nonresident Tribal Scenarios**
  - Recreational Use scenarios: Avid Hunter, Avid Angler, and Casual User
  - Nonresidential Tribal scenario.
- **Occupational Scenarios**
  - Industrial Worker scenario
  - Resident National Monument Worker scenario.
- **Residential Scenarios**
  - Subsistence Farmer scenario
  - Confederated Tribes of the Umatilla Indian Reservation (CTUIR) Native American Resident scenario
  - Yakama Nation Native American Resident scenario.

The human health risk assessment includes an evaluation of risks under both reasonable maximum exposure (RME) and central tendency exposure (CTE) conditions. Under RME conditions, risk is evaluated for individuals whose behavioral characteristics may result in higher potential exposure than seen in the average individual. A CTE assessment characterizes potential risk to an average member of the target population. The inclusion of both RME and CTE calculations provides a semiquantitative measure of the range of expected risks that may occur under a particular exposure scenario. The CTE and RME provide risk managers with an estimate of the mean and upper percentile of estimates of exposure.

The RME and CTE exposure parameter values used for the scenarios listed above are based on default EPA values and site-specific assumptions that were developed with input from multiple stakeholders. Identification of potentially complete exposure pathways for each scenario was likewise developed with stakeholder input. In the case of the residential exposure scenarios, these include pathways related to home-produced foods for which there is limited CERCLA risk assessment guidance.

In general, the process used to calculate the RCs for soil follows EPA guidance as provided in the *ProUCL Version 4.0 User Guide* (EPA/600/R-07/038). For groundwater, contaminants were evaluated for a range of concentrations for each COPC, with the high end of the range sufficient to cover the RME to groundwater, rather than on a well-by-well basis. The 50th and 90th percentile values for each COPC from the groundwater data set in an OU were selected to represent a reasonable range of potential exposure concentrations. These representative concentrations were used to evaluate central tendency and reasonable maximum groundwater concentrations for potential groundwater exposures.

7.2.3.3 Toxicity Assessment. The toxicity assessment includes a discussion of the different types of potential adverse health effects associated with chemical and radionuclide exposure. In addition, the toxicity values used in the risk calculations are presented along with the sources for those values. The details of the toxicity assessment are presented in Section 3.5.

The hierarchy of references for chemical toxicity criteria is described in a 2003 memorandum from EPA's Office of Solid Waste and Emergency Response (OSWER Directive 9285.7-53). This hierarchy is followed for the RCBRA, with the primary source of toxicity values being EPA's Integrated Risk Information System database (IRIS) (IRIS 2009). Only toxicity criteria published in IRIS have gone through peer-review and EPA-consensus-review processes. The second tier of toxicity criteria are the provisional peer-reviewed toxicity values published by the National Center for Environmental Assessment in EPA's Office of Research and Development. These values are developed on a chemical-specific basis when requested by EPA's Superfund program, but the documentation for them is generally not citable. The third tier of references include values published in EPA's *Health Effects Assessment Summary Tables* (HEAST) (EPA/540/R-97/036), and other sources such as the California EPA and the Agency for Toxic Substances and Disease Registry.

If no toxicity criteria are published for a chemical, the value(s) for a surrogate chemical may be used and are documented in the tabulation of toxicity criteria. Selection of a surrogate chemical is based on commonality in mechanism of action or molecular structure. Preferentially, the surrogate value selected is a Tier 1 value. Consistent with current EPA practice, route-to-route extrapolation of toxicity criteria is not used in the human health risk assessment.

The EPA has not established any toxicity criteria for lead (IRIS 2009). Instead, potential health risks related to lead exposure are evaluated by modeling blood lead concentrations and comparing these concentrations to published blood lead concentration criteria. The EPA has recommended a residential screening level for lead in soil of 400 mg/kg, derived using the integrated exposure uptake biokinetic model (OSWER Directive 9355.4-12, "Revised Interim Soil Lead Guidance for CERCLA Sites and RCRA Corrective Action Facilities"). The 400 mg/kg residential screening level was used as a protective screening value for lead in soil for the human health risk assessment.

7.2.3.4 Risk Characterization. The risk characterization is the part of a human health risk assessment in which estimates of potential health effects and radiation dose for each exposure scenario are presented. The risk characterization also clarifies which COPCs and exposure pathways are associated with these risks. Uncertainty related to the various assumptions and inputs used in the human health risk assessment is also assessed in the risk characterization step to qualify the risk assessment results. Details of the methodology for risk characterization are presented in Section 3.6 (Methodology for Risk Characterization).

Potential human health risks for nonradiological COPCs are characterized for two health endpoints: (1) the risk of potential carcinogenic (cancer-related) health effects and (2) the potential for noncancer health effects (systemic hazards). For radiological COPCs, radiation dose and carcinogenic risk are calculated. Based on EPA guidance (OSWER Directive 9200.4-18), cancer risk is summed across chemicals and radionuclides.

Cancer risks for each exposure route and COPC (radionuclides and chemicals) are summed to calculate cancer risk to an individual. Calculated incremental cancer risk in a CERCLA risk assessment is generally evaluated relative to the point of departure of 1×10^{-6} and the target risk range of 10^{-6} to 10^{-4} described in the NCP (40 CFR 300.430). Under the MTCA, Washington State evaluates cancer risks due to exposure to multiple chemicals and/or multiple exposure pathways using an incremental cancer risk threshold of 1×10^{-5}.

Noncarcinogenic effects for individual chemicals are expressed as hazard quotients (HQs). Hazard quotients for each chemical may be summed to calculate a hazard index (HI) across chemicals for each exposure pathway if target organs and mechanisms of toxicity are similar. Hazard indices across exposure pathways are summed to calculate an overall HI. As an initial evaluation, HQs may be protectively summed even in situations where target organs and mechanisms of toxicity are dissimilar. An HQ or HI value of greater than 1.0 is indicative of the potential for adverse effects, with higher values being related to greater concern for adverse effects. Unlike cancer risk, the magnitude of an HQ is not a measure of the probability of an effect occurring.

Radiation dose, which is a measure of the amount of energy deposited in body tissues, is calculated as the product of the intake or exposure rate for a single radionuclide and the dose conversion factor for that radionuclide and exposure route. Radiation doses for each exposure route and radionuclide are summed to calculate the total annual dose to an individual. The acceptability of a calculated annual dose for exposures related to soil and foodstuffs is evaluated for human receptors in the RCBRA relative to a threshold dose limit of 15 mrem/yr (OSWER Directive 9200.4-18) for all exposure pathways except groundwater-related exposures, for which a threshold of 4 mrem/yr (related to a groundwater maximum contaminant level) is used. Current EPA policy is to employ cancer risk as a basis for CERCLA cleanup levels rather than radiation dose (OSWER Directive 9200.4-18). However, past assessments at the Hanford Site (including cleanups at 100 and 300 Area waste sites under the IARODs) have employed radiation dose as a basis for action. For comparison purposes, radiation dose is also evaluated in this risk assessment.

7.3 QUANTITATIVE EVALUATION OF HUMAN HEALTH RISKS

The following section presents a summary of the quantitative risk assessment results for a wide range of soil exposure scenarios. In addition, a summary of the evaluation of potential exposure to shoreline springs is presented. Finally, summaries of the screening-level risk assessments for groundwater and the fish ingestion pathway are presented.

The soil exposure scenarios include the recreational and Nonresident Tribal scenarios related to broad-area exposures, and occupational and residential scenarios related to local-area exposures. The risk results for soils are presented in the context of EPA's risk range that is generally used to determine if remedial action is warranted for a site. In addition, cancer risks for chemicals are presented in the context of target risks as described in MTCA.

The EPA guidance memorandum, "Role of the Baseline Risk Assessment in Superfund Remedy Selection Decisions" (OSWER Directive 9355.0-30), states that action is generally not warranted at sites where the cumulative carcinogenic site risk to an individual based on reasonable maximum exposure for both current and future land use is less than 1 x 10^{-4}, the noncarcinogenic hazard quotient is less than 1, and there are no adverse environmental impacts. In addition, the EPA guidance memorandum (OSWER Directive 9355.0-30) states that the upper boundary of the risk range is not a discrete line at 1 x 10^{-4}, although EPA generally uses 1 x 10^{-4} in making risk management decisions. A specific risk estimate around 10^{-4} may be considered acceptable if justified based on site-specific conditions and, in certain cases, EPA may consider risk estimates slightly greater than 1 x 10^{-4} to be protective.
Additional guidance in the EPA memorandum notes that both current and reasonably likely future risks need to be considered in order to demonstrate that a site does not present an unacceptable risk to human health and the environment. The potential land use associated with the highest level of exposure and risk that can reasonably be expected to occur should be addressed in the baseline risk assessment and these same exposure assumptions should be used in developing remediation goals.

For MTCA, the target risk for individual chemicals is 1 x 10^{-6} and the target HQ is 1. For cumulative risk from multiple chemicals, the target risk is 1 x 10^{-5}. These risk results for a wide range of exposure scenarios based on possible future land uses and the target risk levels described above can be used in the RI/FS reports to evaluate remedial alternatives.

7.3.1 IAROD Cleanup Level Scenario Evaluation

As discussed in Section 7.2.2, waste sites in the River Corridor were interim closed using RAGs related to soil exposure and protection of groundwater from contaminants leaching from soil. For radionuclides in the 100 Area, a residential scenario that included food ingestion and drinking water exposure pathways was implemented to develop the RAGs for protection of human health. For chemicals in the 100 Area, RAGs for protection of human health were based on the MTCA Method B "Soil Cleanup Levels for Unrestricted Land Use" (WAC 173-340-740). The MTCA Method B cleanup levels are based on incidental soil ingestion and use residential exposure frequency assumptions. For the 300 Area, RAGs were based on industrial land use for some sites and residential land use for other sites.

These same IAROD scenarios that were used to calculate soil RAGs for radionuclides and chemicals were also used in Section 2.8 of the RCBRA to calculate screening-level cancer risks and noncancer hazards for individual remediated waste sites using the cleanup verification soil data. These waste sites achieved the interim action remedial action objectives as documented in the CVPs and were interim closed. The assessment described in Section 2.8 updates the CVP analyses as described below.

The exposure point concentrations used in the Section 2.8 calculations were the RME values described in Section 3.4, developed using methods consistent with *ProUCL Version 4.0 User Guide* (EPA/600/R-07/038), which differ from the protocols employed to calculate exposure concentrations for the interim closures as presented in the CVPs. As discussed in Section 3.4, when there were five or more detected values in a data set, 95% upper confidence limit on the mean (95% UCL) values were computed using two parameteric and one nonparametric method and the median value was used in the risk assessment. A separate decision logic was employed when there were fewer than five detected values. Noncensored radionuclide data were all treated as detected values, and nondetect values for chemicals were estimated using Kaplan-Meier and regression-on-order statistical methods.

By contrast, different statistical methods were used to calculate 95% UCL values for chemicals and radionuclides for the CVP reports. The methods used for handling nondetect results for the CVP reports for the 100 and 300 Area remediated waste sites also differed from those employed in the RCBRA and included the following steps (DOE/RL-96-22, *100 Area Remedial Action Sampling and Analysis Plan*; DOE/RL-2001-48, *300 Area Remedial Action Sampling and Analysis Plan*):

1. For radionuclides, if the result was qualified as nondetect, the minimum detectable activity was used as a proxy value in the 95% UCL calculations.

2. For chemicals, if the result was qualified as nondetect, one-half the practical quantitation limit was used as a proxy value in the 95% UCL calculations.

3. For uranium isotopes, the 95% UCL values were adjusted by subtracting background levels from the calculated 95% UCL value.

4. For chemicals, if greater than 50% of the results were nondetect, the maximum detected value was used as the "statistical value" in lieu of the 95% UCL.

5. For chemicals, if 100% of the results were nondetect, the practical quantitation limit was used as the "statistical value" in lieu of the 95% UCL.

The MTCA Method B values used in the Section 2.8 IAROD scenario calculations differ from the IAROD RAGs for chemicals only in that current chemical toxicity criteria were used for the MTCA Method B values derived in Section 2.8. For radionuclides, the major difference between the Section 2.8 risk-based concentrations for the IAROD scenario calculations and the IAROD RAGs is that the risk-based concentrations used in the Section 2.8 calculations are based on a cancer risk threshold in accordance with OSWER Directive 9200.4-18. As discussed in Section 7.2.2, the basis of the RAGs for radionuclides in the IARODs was a radiation dose threshold of 15 mrem/yr.

The risk calculations documented in Section 2.8 that use the IAROD residential scenarios were performed to help answer the following question that is included in the list of questions at the beginning of this section: Are soil cleanup levels established under the IARODs protective of human health and the environment using current regulatory guidelines?

Results of this screening-level assessment can be compared to risk assessment results presented in later sections of the RCBRA where a range of exposure scenarios were evaluated.

7.3.1.1 IAROD Cleanup Level Scenario Results for Chemicals. Residual cumulative cancer risks for chemicals for most of the 156 remediated waste sites evaluated in RCBRA are less than 1×10^{-5} using the IAROD residential scenario (i.e., MTCA Method B unrestricted use scenario). For some remediated waste sites, chemical risks up to 3×10^{-5} were calculated, with the main risk driver for these sites being arsenic. Reasonable maximum exposure arsenic soil concentrations exceed the upland reference area RME value (3.2 mg/kg) and sometimes the Hanford Site 90th percentile value (6.47 mg/kg [DOE/RL-92-24, *Hanford Site Background: Part 1, Soil Background for Nonradioactive Analytes*]) at a number of remediated waste sites; consequently, arsenic was identified as a COPC for soil in the 100-K, 100-D/100-H, 100-F/100-IU-2/100-IU-6, and 300 Area ROD decision areas. However, arsenic RME soil concentrations at all remediated waste sites are less than the residential IAROD cleanup level of 20 mg/kg for arsenic established by Ecology as an unrestricted land-use cleanup value. This value is adjusted for natural background in soil (http://apps.leg.wa.gov/WAC/default.aspx?cite=173-340-900). When residual cancer risks for chemicals were calculated without the contribution from arsenic, all 156 remediated waste sites have cancer risk results less than 1×10^{-5}.

The noncancer HIs for chemicals only exceed a threshold of 1.0 at 2 of the 156 remediated waste sites evaluated in RCBRA (the 316-1 site had an HI of 1.4, and the 316-5 waste site had an HI of 1.2) using the MTCA unrestricted use scenario with current toxicity criteria. It is important to note that both of these 300 Area waste sites were remediated to Industrial scenario IAROD cleanup levels. At the 316-5 waste site, total uranium is the main contributor to the HI with an HQ of 1.0. The RME soil concentration for uranium at the 316-5 waste site is 242 mg/kg, which is essentially equal to the Industrial IAROD cleanup level of 240 mg/kg. For the 316-1 waste site, arsenic and mercury are the main contributors to the HI with HQs of 0.6 and 0.5, respectively. Because the toxicity criteria for these chemicals are related to different target organs (vascular and skin toxicity for arsenic, immune system effects for mercury), the target organ-specific HIs at 316-1 are likely below 1.0.

7.3.1.2 IAROD Cleanup Level Scenario Results for Radionuclides. Residual cumulative cancer risks from radionuclides for most remediated waste sites are less than 1×10^{-4} based on the IAROD residential scenario. There are 14 of the 156 remediated waste sites that were evaluated in the RCBRA that have present-day radionuclide risks greater than 1×10^{-4} (between 2×10^{-4} and 5×10^{-4}) using this scenario. Table 7-1 presents a summary of these 14 waste sites. Eleven of the 14 sites with risks greater than 1×10^{-4} are in the 100 Areas: 6 sites are in the 100-B/C Area, 2 sites are in the 100-D/100-H Area, 1 site is in the 100-F/100-IU-2/100-IU-6 Area, and 2 sites are in the 100-K Area. The remaining 3 of the 14 sites with radionuclide cancer risks greater than 1×10^{-4} based on the IAROD residential scenario are in the 300 Area (316-1, 316-2, and 316-5).

For the eleven 100 Area waste sites with present-day radionuclide risks greater than 1×10^{-4}, the risk drivers are europium-152, cesium-137, and strontium-90. The radioactive half-lives of cesium-137 and strontium-90 are approximately 30 years, and the half-life of europium-152 is approximately 13 years. Soil concentrations of cesium-137 and strontium-90 decrease by one-half in about 30 years due to radiological decay, and europium-152 decreases about twice as quickly. All 11 of these waste sites with present-day risk greater than 1×10^{-4} would be expected to have risks less than 1×10^{-4} at year 2075. Each of these remediated waste sites has also been excavated and backfilled with clean soil, such that these residual soil concentrations are not expected on the ground surface and, therefore, exposure would not occur unless there are disturbances to the soil.

Residual conditions in the 300 Area meet the Industrial IAROD cleanup levels. However, when the 300 Area waste sites are evaluated using the IAROD residential scenario, there are three sites with cancer risks greater than 1×10^{-4}. At the 316-1 waste site, the risk driver is cobalt-60, and at the 316-2 and 316-5 waste sites, the risk drivers are long-lived uranium isotopes (uranium-235 and uranium-238). It is important to note that the IAROD cleanup levels for the 316-1, 316-2, and 316-5 waste sites in the 300 Area were based on future industrial land use and that residual soil concentrations of these radionuclides are below the industrial IAROD cleanup levels. In addition, each of these 300 Area sites has also been excavated to a depth of approximately 6 m (20 ft) or more and backfilled with clean soil. The radioactive half-life of cobalt-60 is approximately 5 years, so residual soil concentrations at the 316-1 waste site will decrease by 50% about every 5 years.

7.3.1.3 Conclusions of the IAROD Scenario Risk Calculations. Based on these screening-level cumulative cancer risk and noncancer hazard calculations using the IAROD residential exposure scenarios for radionuclides and chemicals, the conditions at 142 of the 156 remediated waste sites evaluated in the RCBRA are protective of human health based on comparison of results to the following thresholds:

- Cumulative cancer risks for chemicals (not including arsenic) are less than 1 x 10^{-5}
- HIs for chemicals are very near or below 1.0
- Cumulative cancer risks for radionuclides are less than 1 x 10^{-4}.

There are 14 sites that exceeded at least one of these thresholds, including 11 remediated waste sites in the 100 Area and 3 remediated waste sites in the 300 Area. There are 11 remediated waste sites in the 100 Area that have cumulative cancer risks for radionuclides that exceed 1 x 10^{-4}. The cumulative cancer risks for radionuclides at these sites range from 2 x 10^{-4} to 5 x 10^{-4}, with the risk drivers being the short-lived radionuclides europium-152, cesium-137, and strontium-90. Residual conditions in the 300 Area meet the Industrial IAROD cleanup levels, but there are three remediated waste sites that have cumulative radionuclide cancer risks that exceed 1 x 10^{-4} based on the IAROD residential scenario. The risk drivers for these three sites are cobalt-60, uranium-235, and uranium-238. Each of these 14 remediated waste sites has been excavated and backfilled with clean soil such that the residual concentrations evaluated in the screening calculations are not expected on the ground surface.

A key uncertainty related to these risk calculations for the IAROD exposure scenarios is how applicable the cleanup verification soil data at excavated and backfilled sites are for future human exposure. Risks may be overestimated using cleanup verification data from excavated sites (i.e., from sidewalls and bottom of excavations) because these soils are now covered by clean backfill. This is of particular significance because the great majority of remediated waste sites with the highest levels of residual risk, including all 14 of the sites where cancer risks were estimated to exceed 1 x 10^{-4}, are sites that have been excavated and backfilled so verification soil samples are not necessarily representative of soil on or near the surface of the waste site.

Another uncertainty in the IAROD risk calculations is how well the residential exposure assumptions represent the reasonably anticipated future land use. Both IAROD scenarios for radionuclides and chemicals include residential assumptions for direct contact with soil. However, for radionuclides, the IAROD scenario also includes exposure by food ingestion and drinking water pathways. For chemicals, these pathways are not part of the MTCA Method B exposure scenario. Specific uncertainties related to these pathways include the likelihood that intensive home agriculture will exist in the future, whether any particular site is large enough to reasonably support these activities, and whether the models used to estimate plant and animal tissue concentrations from the soil exposure concentrations overstate human exposure. Additional details related to these uncertainties are presented in Section 7.3.3.4.

7.3.2 Broad-Area Risk Assessment Results

The broad-area risk assessment was designed to address the following questions:

- Are residual conditions in the River Corridor after cleanup actions completed under the IARODs protective of human health and the environment based on the various RCBRA exposure scenarios?

- What are the uncertainties associated with the RCBRA risk results and conclusions?

- Are there any recommendations for additional studies or monitoring that should be considered at this time to reduce uncertainties with specific risk results and conclusions or establishing health-protective PRGs?

The broad-area risk assessment used surface soil, sediment, and surface water data from throughout the River Corridor to evaluate potential risks for receptors that may engage in activities where they are exposed to these media over an area larger than an individual waste site. As discussed in Section 4.2, representative concentrations for the upland environment were calculated by combining surface soil data among all six ROD areas. Representative concentrations for the riparian and near-shore environments were calculated separately for each individual ROD area. As noted in Section 7.2.3.1, the representative concentrations for soil and sediment for the upland, riparian, and near-shore environments were calculated using MIS samples that were collected specifically for the RCBRA, supplemented with 100-B/C Pilot and 100-NR-2 project data. These data are assumed to be a conservative representation of the entire footprint of the three environments because they were collected from remediated waste sites (upland) or areas potentially affected by releases from waste sites (riparian and near-shore). However, because of the limited number of MIS samples, there are uncertainties with using these data to calculate representative concentrations for the broad-area assessment.

As described in Section 3.3, human exposure scenarios evaluated in the broad-area risk assessment include the Avid Hunter, Casual User, and Avid Angler applications of a Recreational scenario and a Nonresident Tribal exposure scenario. The Recreational scenarios address child and adult exposures for different types of activities in the upland (Avid Hunter), riparian (Casual User), and near shore (Avid Angler) environments. The Nonresident Tribal scenario is focused on adults and children engaged in a subsistence lifestyle who reside offsite but who use the River Corridor for traditional tribal activities including fishing, hunting, gathering plants, and participating in sweat lodges using river water. Potential risks from exposure to groundwater surfacing at shoreline springs along the Columbia River are also evaluated as part of the broad area risk assessment. Finally, a screening-level assessment of fish ingestion risks related to COPCs in fish tissue is included in the broad-area risk assessment. A more detailed assessment of the fish ingestion pathway will be presented in the RI of Hanford Site releases to the Columbia River.

7.3.2.1 Results for the Recreational and Nonresident Tribal Exposure Scenarios. The results of the broad-area risk assessment for the three Recreational scenarios (Casual User in the riparian environment, Avid Hunter in the upland environment, and Avid Angler in the near-shore environment) indicate minimal potential risks for these types of activities throughout the River Corridor. Total cancer risks (i.e., sum of radionuclide and chemical risks) are generally near or below the 1×10^{-6} de minimis threshold for exposures to soil, sediment, and surface water in the Casual User and Avid Angler scenarios. Radiation dose and chemical hazard are also below thresholds for these scenarios.

The RME total cancer risk for the Avid Hunter scenario (5×10^{-5}) is within the 1×10^{-6} to 1×10^{-4} risk management range. The primary exposure pathway contributing to this risk is ingestion of game meat. Exposure to game meat in the Avid Hunter scenario was evaluated using modeled tissue concentrations. The modeled tissue concentrations were calculated using literature-based uptake factors and CTE and RME soil concentrations from MIS samples associated with remediated waste sites in the upland environment. Key COPCs for the game meat ingestion pathway are arsenic, benzo(a)pyrene, and the polychlorinated biphenyl (PCB) Aroclor-1254; arsenic and benzo(a)pyrene have RME cancer risks at or above 1×10^{-5} and Aroclor-1254 has an RME cancer risk of 9×10^{-6}. The Avid Hunter RME hazard index is approximately equal to the threshold level of 1.0. The Avid Hunter CTE cancer risk is between 1×10^{-6} and 1×10^{-5}, and the HI is less than 1.0.

The results of the broad-area risk assessment for the Nonresident Tribal scenario in the six ROD areas are summarized in Table 7-2. These results indicate potential present-day total cancer risks (i.e., sum of radionuclide and chemical risks) of approximately 1×10^{-2} and chemical HI between about 75 and 95, related primarily to arsenic exposure by the plant ingestion pathway. The arsenic plant pathway contributes over 90% to total cancer risk in all ROD areas except 100-K, where exposure to carbon-14 by plant ingestion in the riparian environment contributes about 25%.

The Nonresident Tribal risk results described above are related to arsenic soil concentrations that are approximately equivalent to levels in areas unaffected by Hanford Site activities. Nonresident Tribal risk results for the reference areas are similar to the ROD area results described in the previous paragraph. For the reference areas, the Nonresident Tribal cancer risk is also 1×10^{-2} and the HI is 44, with arsenic being the main risk driver and contributor to noncancer HI. The RME representative concentration of arsenic in upland soil in the River Corridor based on MIS samples is 4.7 mg/kg, and in the upland reference area is 3.2 mg/kg. The riparian soil RME arsenic concentrations in the six ROD areas based on MIS samples range from 7.1 to 9.6 mg/kg, and the equivalent riparian reference area value is 7.4 mg/kg. These upland and riparian arsenic values are not very different than the Hanford Area 90th percentile arsenic soil background concentration of 6.47 mg/kg.

Because arsenic concentrations in upland and riparian site soils are not very different than reference area values and commonly applied background levels, Nonresident Tribal risks for COPCs other than arsenic may also be of interest for risk management decisions. In the absence of arsenic, present-day cancer risks between 3×10^{-4} and 4×10^{-3} are calculated for the six ROD areas (Table 7-3). These risks are related to several different COPCs (technetium-99, carbon-14, strontium-90, benzo(a)pyrene, and Aroclor-1254) and exposure pathways (mostly native plant ingestion and game ingestion). Nonresident Tribal chemical HI results without arsenic range between 24 and 38 for the six ROD areas with cadmium, zinc, and mercury being the key COPCs and the plant ingestion pathway being the key contributor to the HI (Table 7-4).

Because exposures from plant ingestion dominate the Nonresident Tribal cancer risk and child HI results, a sensitivity analysis was conducted for this exposure pathway. The sensitivity analysis focused on published COPC-specific plant-soil concentration ratios and the percentage of diet that might come from wild plant foods gathered on the Hanford Site. The range of cancer risk and child HI plant ingestion sensitivity analysis results spans between three and four orders of magnitude (see Section 4.1 in Appendix D-1). The RCBRA plant ingestion pathway risk results were within a factor of 10 of the upper end of this range. Differences between low-end and high-end plant-soil concentration ratios were the primary cause of the large range of sensitivity analysis results. Differences between low-end and high-end values for fraction of diet associated with wild plants from the Hanford Site vary by only a factor of approximately 20. A summary of the sensitivity analysis results is provided in Section 4.7, and the complete analysis is provided in Appendix D-1.

The CTE and RME representative concentrations for lead in upland soil are 27 and 45 mg/kg, respectively. Among the six ROD decision areas where individual riparian representative concentrations are calculated, the highest CTE and RME values are 51 and 92 mg/kg, respectively, in the 100-F/100-IU-2/100-IU-6 Area. These representative concentrations are well below the residential screening criterion of 400 mg/kg described in Section 3.5.8.

7.3.2.2 Results for the Evaluation of Shoreline Springs. As described in Section 4.2.1, water data from seeps along the Columbia River have been obtained as part of the 100-B/C Pilot and 100-NR-2 projects, as well as under the Surface Environmental Surveillance Program. These seeps represent surface discharge of the unconfined aquifer that underlies the Hanford Site. As discussed in Section 10.5 of the *Hanford Site Environmental Report for Calendar Year 2008* (PNNL-18427), the presence and flow rate of water at the seeps, as well as concentrations of groundwater COPCs in the seep water, are governed by fluctuations in the level of river water and therefore may vary widely over time.

Key groundwater plume contaminants in the seven groundwater OUs underlying the River Corridor are described in *Hanford Site Groundwater Monitoring for Fiscal Year 2006* (PNNL-16346) and are employed in Section 6.0 of the human health risk assessment to support identification of groundwater COPCs. A subset of these contaminants including chromium, nitrate, tritium, strontium-90, technetium-99, and uranium are discussed in *Hanford Site Environmental Report for Calendar Year 2008* (PNNL-18427) as key contaminants observed in seep water. A crosswalk of these six key seep contaminants with the ROD decision areas where they have been measured at elevated levels includes the following:

- Dissolved chromium in the 100-D/100-H, 100-B/C, 100-K, 100-N, and 100-F/100-IU-2/100-IU-6 Areas

- Nitrate in the 100-D/100-H, 100-B/C, 100-K, 100-F/100-IU-2/100-IU-6, and 300 Areas
- Tritium in the 100-D/100-H, 100-B/C, 100-K, 100-N, 100-F/100-IU-2/100-IU-6, and 300 Areas
- Strontium-90 in the 100-D/100-H, 100-B/C, and 100-K Areas
- Technetium-99 in the 100-D/100-H, 100-B/C, 100-F/100-IU-2/100-IU-6, and 300 Areas
- Uranium in the 100-D/100-H, 100-F/100-IU-2/100-IU-6, and 300 Areas.

Strontium-90 was historically measured in water flowing from certain springs in the 100-N Area, but flow from these springs has not been observed since 1997 due to lowering of the water table that followed the cessation of water discharges from the 105-N Reactor when the reactor was shut down (PNNL-18427).

Potential risks related to use of water from shoreline springs was evaluated comparing water concentrations from these seeps to EPA short-term drinking water health advisories and Washington State and EPA maximum contaminant levels for chronic use consumption of drinking water. The evaluation was focused on chromium, nitrate, tritium, strontium-90, technetium-99, and uranium as described above.

The results of the risk assessment for groundwater contaminants at shoreline springs along the Columbia River indicate that for the majority of the shoreline springs there is negligible risk related to exposure to key groundwater contaminants being released to the Columbia River at these locations. For a subset of springs, concentrations of radiological contaminants may exceed long-term drinking water criteria by factors of between 2 and 10, but this would present negligible exposure for occasional users of a spring. At only one spring (100-F Spring) are concentrations of a contaminant (nitrate) elevated relative to short-term standards that apply to 1-day or 10-day drinking water exposure. There are no short-term health advisories for uranium; however, at Spring 42-2 in the 300 Area, concentrations of uranium are routinely above drinking water standards.

7.3.2.3 Results for the Screening-Level Fish Ingestion Risk Assessment. The screening-level fish ingestion risk assessment for the Avid Angler, Nonresident Tribal, and Residential Exposure scenarios is based on concentrations of near-shore COPCs in tissues of sculpin, shellfish, and crayfish. However, these species were sampled for the purpose of determining the extent to which Hanford Site-related contaminants are accumulated by biota in areas where groundwater plumes emerge at the Columbia River. As discussed in Section 4.6, these species are not plausible food sources for chronic human exposure and, therefore, the results of the screening assessment are applicable only for identifying risk-relevant COPCs and for evaluating relative risk among the six ROD areas. The risk assessment for fish ingestion being prepared for the RI of Hanford Site releases to the Columbia River is intended to address potential human health risks from ingestion of food fish in the Hanford Reach of the Columbia River.

Among near-shore COPCs, carbon-14 is the only potential carcinogen, and it was detected only in one of two sculpin samples at the 100-K Area; there are no carbon-14 reference-area data for sculpin. Screening-level fish ingestion cancer risks were therefore only calculated for the 100-K Area. Because carbon-14 was detected in just one of two sculpin samples from the 100-K Area, these risk estimates are highly uncertain due to the limited amount of data.

The 100-K Area RME cancer risks calculated using the sculpin data are 4×10^{-5} and 9×10^{-6} for the Avid Angler and Subsistence Farmer scenarios, respectively. Cancer risks for the Nonresident Tribal, CTUIR Resident, and Yakama Resident scenarios using the sculpin data are higher (2×10^{-4} or 3×10^{-4}), reflecting the higher fish ingestion rates for these scenarios and a full 70- to 75-year lifetime (rather than 30-year) exposure duration. The following fish ingestion rates were used for the various scenarios:

- Nonresident Tribal and CTUIR Resident scenarios: 620 g/day for adults and 310 g/day for children
- Yakama Resident scenario: 520 g/day for adults and 360 g/day for children
- Subsistence Farmer scenario: 60.2 g/day for adults
- Avid Angler scenario: 231 g/day for adults.

The RME HI values for Avid Angler sculpin ingestion ranged between 1.7 and 4.3 in the six ROD areas, with an RME HI value for the reference area of 1.8. The Nonresident Tribal fish ingestion child HI for sculpin ranged between 10 and 25, with the highest values in the 100-D/100-H ROD area (25) and 100-F/100-IU-2/100-IU-6 ROD area (17). As a point of comparison, the child HI values for the COPC concentrations in reference area sculpin were 11. In the 100-D/100-H Area, the main contributors to the screening-level sculpin HI result of 25 were nickel (60%) and selenium (15%). The main contributors to the HI result of 17 in the 100-F/100-IU-2/100-IU-6 Area were copper (30%) and selenium (20%).

The screening-level assessment of the fish ingestion pathway also includes information from EPA studies conducted in the Columbia River. Two relevant studies have been released by EPA Region 10 during the past 10 years describing contaminant levels in fish tissues in the Columbia River Basin (*Columbia River Basin Fish Contaminant Survey 1996-1998* [EPA/910/R-02/006] and the Upper Columbia River (EPA 2007, *Phase I Fish Tissue Sampling Data Evaluation Upper Columbia River Site CERCLA RI/FS, Final*). Fish ingestion risks were estimated for Native American tribes and the general public in the Columbia River Basin study. The most important contributors to health risks were PCBs, arsenic, and mercury (EPA/910/R-02/006). 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).

7.3.2.4 Key Uncertainties for the Broad-Area Risk Assessment. The key uncertainties for the broad-area risk assessment are summarized below. Additional details of the uncertainty analysis for the broad-area risk assessment are presented in Section 4.7.

One of the key uncertainties in the Nonresident Tribal results relate to exposure assumptions for the receptors and modeled concentrations of metals and radionuclides in native plants. This scenario includes exposure by direct soil contact when Tribal members may come onto the site to hunt, gather plants, or engage in other Tribal activities. The scenario also includes exposure through indirect contact with soil contaminants through the food pathways. The amount of food that Nonresident Tribal members may consume that comes from the Hanford Site is uncertain. Therefore, as a conservative estimate of exposure for this assessment, it was assumed that 80% of plant foods and 100% of fish and game consumed by the receptors over an entire year comes from the Hanford Site. This assumption results in potential overestimation of the exposure that might actually occur if only a portion of food items is obtained from the Hanford Site. In addition, it is uncertain whether sufficient food items would be available from the Hanford Site to sustain the lifestyle assumed in the Nonresident Tribal scenario. Therefore, risk estimates assuming the majority of food consumed by these receptors comes from the Hanford Site may be overestimated based on this additional uncertainty.

There are also considerable uncertainties related to contaminant intake from these food pathways because concentrations in edible plant tissues are modeled using literature values, not site-specific data. As described in Section 7.2.3.1, uncertainty in the value of plant-soil concentration ratios was the primary contributor to a 10,000-fold range of sensitivity analysis results for the plant ingestion exposure pathway of the Nonresident Tribal scenario.

There are uncertainties related to the fish ingestion results for the recreational and Tribal scenarios based on the use of biota tissue samples from localized species in the calculations. The RCBRA, 100-B/C Pilot, and 100-NR-2 investigation biota tissue samples for localized species such as sculpin, clams, and crayfish were largely collected in specific areas where groundwater plumes emerge at the Columbia River. The purpose of this sampling was to determine the extent to which Hanford Site-related contaminants are accumulated by biota at these locations. Although these localized species are not plausible sources for chronic human exposure, sculpin in particular were identified in the RCBRA SAP as an appropriate species for screening human health risks for Hanford Site-related releases (DOE/RL-2005-42, Section 1.4). Sampling of game fish that are more representative of food sources for chronic exposure assessment is being conducted within the Columbia River under the *Remedial Investigation Work Plan for Hanford Site Releases to the Columbia River* (DOE/RL-2008-11). An assessment of risks associated with ingestion of these game fish will be presented in the RI report.

7.3.2.5 Recommendations Related to the Broad-Area Risk Assessment. As noted in the previous subsection, the key uncertainties with the Nonresident Tribal scenario are associated with the risk estimates based on the plant ingestion pathway. Measuring COPC concentrations in edible portions of native plants growing in the River Corridor, particularly plants identified by Tribal representatives as important food resources, could help to reduce uncertainty in the Nonresident Tribal results and provide information for risk communication purposes.

In addition, consultation with Tribal members could help to refine the exposure assumptions used in the Nonresident Tribal scenario to assist in communicating risk results for various levels of intake rates and percent of diet that may be obtained from the Hanford Site.

7.3.3 Summary of Local-Area Risk Assessment Results

The local-area risk assessment was also designed to address the first, second, and third questions listed at the beginning of this section:

- Are residual conditions in the River Corridor after cleanup actions completed under the IARODs protective of human health and the environment under the various RCBRA exposure scenarios?

- What are the uncertainties associated with the RCBRA risk results and conclusions?

- Are there any recommendations for additional studies or monitoring that should be considered at this time to reduce uncertainties with specific risk results and conclusions or establishing health-protective PRGs?

The local-area risk assessment used post-interim action cleanup verification soil data from waste sites to evaluate potential risks for receptors that may be exposed to soil on the scale of an individual waste site. Representative concentrations in soil were calculated separately for each individual remediated waste site. The COPCs were identified independently for each of the six ROD areas.

The Industrial/Commercial scenario and the Resident National Monument/Refuge scenario are the two occupational scenarios evaluated in the local-area risk assessment. Receptors for these scenarios are limited to adult workers. The Industrial/Commercial receptor is assumed to work at a building located on a remediated waste site, but resides offsite. The receptor in the Resident National Monument/Refuge scenario is assumed to live in a residence constructed on a remediated waste site and work outdoors elsewhere on the site. The residential part of the exposure for the Resident National Monument/Refuge scenario is based on remediated waste site data (i.e., local-area exposure) and the occupational part of the exposure is based on the MIS sampling data (i.e., broad-area exposure).

The Subsistence Farmer, CTUIR Resident, and Yakama Resident scenarios are the residential scenarios evaluated in the local-area risk assessment. These scenarios include exposures related to a rural land-use pattern that involves home-produced foods. All residential receptors are assumed to spend effectively all of their time in the area around a residence located on a remediated waste site in order to protectively assign all soil-related exposures to that site.

Summaries of the RME results for the local-area risk assessment for all ROD areas are shown in Tables 7-5 and 7-6 for the occupational and residential scenarios, respectively. Cancer risks are shown in Tables 7-5 and 7-6 as “total risk,” which is the sum of the radionuclide and chemical cancer risks. The risk assessment results for individual remediated waste sites are sometimes shown to exceed thresholds for the exposure scenarios and health effects evaluated. The risk assessment results may vary widely for an individual remediated waste site depending on the land-use scenario and the time (present day, 2075, or 2150) when the scenario is evaluated. For example, present-day Subsistence Farmer total cancer risks exceed 1×10^{-4} for 10 remediated waste sites in the 100-B/C Area due to the short-lived radionuclides europium-152, cesium-137, cobalt-60, and strontium-90; but by year 2075 due to radiological decay, RME risks are above 1×10^{-4} at only one site.

Consideration of the key uncertainties and biases is important when applying the risk assessment results for individual remediated waste site to risk management decisions. For excavated sites, a key uncertainty is the use of cleanup verification soil data that reflects pre-backfill conditions. As the depth of backfill increases, the degree of potential conservative bias in the results increases. The use of linear plant-soil concentration ratios in the residential scenarios when COPC concentrations are well above background levels is also a major source of conservative bias in the risk results for some remediated waste sites.

7.3.3.1 Results for the Industrial/Commercial Exposure Scenario. A summary of the RME cancer risk and HI results that are higher than threshold criteria at the remediated waste sites is provided for the Industrial/Commercial exposure scenario in Table 7-7. The results of the local-area risk assessment for the Industrial/Commercial scenario indicate that RME total cancer risk (i.e., sum of radionuclide and chemical risks) is very rarely above 1×10^{-4} (4 of 156 sites have total cancer risks of 2×10^{-4}) and that RME radiation dose is less than 15 mrem/yr for all the sites evaluated. As noted previously, the IAROD cleanup levels were based on 15 mrem/yr dose. In addition, there are no remediated waste sites where Industrial/Commercial chemical HI results exceed the threshold of 1.0. For the four sites with cancer risks that exceed 1×10^{-4} (166-B-11, 116-B-14, 166-DR-9, and 116-F-14), the RME total cancer risks are all 2×10^{-4} with radionuclides as the key risk drivers.

The highest Industrial/Commercial total cancer risk and dose results are related to external irradiation from short-lived radionuclides, including cesium-137, cobalt-60, and europium-152. Soil concentrations of cobalt decrease by one-half in about 5 years, cesium-137 and strontium-90 decrease by one-half in about 30 years due to radiological decay, and europium-152 decreases about twice as quickly as cesium-137 and strontium-90. As shown in Table 7-7, cancer risks at 166-B-11, 116-B-14, 166-DR-9, and 116-F-14 are between 7×10^{-6} and 3×10^{-5} at year 2075.

7.3.3.2 Results for the Resident National Monument/Refuge Exposure Scenario. A summary of the RME total cancer risk (i.e., sum of radionuclide and chemical risks) and HI results that are higher than threshold criteria at the remediated waste sites is provided for the Resident National Monument/Refuge exposure scenario in Table 7-8. The results of the local-area risk assessment for the Resident National Monument/Refuge scenario indicate that RME total cancer risk and radiation dose are rarely above 1×10^{-4} (11 of 156 sites) and 15 mrem/yr (3 of 156 sites), respectively. There are no remediated waste sites where Resident National Monument/Refuge chemical HI results exceed the threshold of 1.0. Like the Industrial/Commercial scenario, high total cancer risk and dose results for the sites in the 100 Areas are related to short-lived radionuclides and therefore decrease with time.

By the year 2150, Resident National Monument/Refuge RME total cancer risks at or above 1×10^{-5} are limited to three remediated waste sites in the 300 Area (316-1, 316-5, and 316-2). Risks for the Resident National Monument/Refuge scenario at these three sites are primarily due to external irradiation from long-lived uranium isotopes.

7.3.3.3 Results for the Residential Exposure Scenarios. Risk assessment results for three residential exposure scenarios (Subsistence Farmer, CTUIR Resident, and Yakama Resident) are presented in detail in Section 5.0 of this report. The following paragraph summarizes the risk assessment results for the Subsistence Farmer scenario, which are shown in Tables 7-9 and 7-10. The subsequent paragraphs of Section 7.3.3.3 provide an interpretation of the results of the local-area residential scenario risk assessments for all three of the residential scenarios described in Section 5.0.

Reasonable maximum exposure total cancer risk (i.e., sum of radionuclide and chemical risks) and HI results exceeding thresholds for the Subsistence Farmer scenario are summarized in Table 7-9 for remediated waste sites in each ROD decision area. As stated in the Table 7-9 notes, the RME cancer risk (5×10^{-4}) and RME child HI (2.7) results for the Subsistence Farmer scenario for the upland reference area for arsenic also exceeded thresholds due to modeled exposure from produce ingestion. Arsenic was identified as a COPC in the 100-K, 100-D/100-H, 100-F/100-IU-2/100-IU-6, and 300 Decision Areas. Sites where arsenic is the primary contributor to RME results below these reference area values are not included in Table 7-9 because these results are related to naturally occurring levels of arsenic. To support risk management decisions for COPCs that more likely reflect historical Hanford Site operations, a supplemental Subsistence Farmer scenario results table (Table 7-10) is also provided. For those sites where arsenic was a significant contributor to risks above 1×10^{-4} and child HI above 1.0, Table 7-10 shows the site risks without the contribution of arsenic. As shown in Table 7-10, present-day RME cancer risks for many remediated waste sites are still greater than 1×10^{-4} without the contribution of arsenic, but by the year 2075 RME risks are less than 1×10^{-4} with the exception of some 300 Area sites where long-lived uranium isotopes are of concern.

The results of the local-area risk assessment for the three residential scenarios indicate that total cancer risk is frequently greater than 1×10^{-4} and that chemical HI frequently exceeds the threshold of 1.0. Present-day total cancer risks greater than 1×10^{-4} for the residential exposure scenarios are almost entirely related to one of three factors:

1. External irradiation from short-lived radionuclides including europium-152, cesium-137, and cobalt-60

2. Exposure to arsenic from ingestion of garden produce

3. Exposure to the short-lived radionuclide strontium-90 from ingestion of produce and livestock products.

By the year 2075, Subsistence Farmer RME total cancer risks above 1×10^{-4} are related predominantly to arsenic exposure from produce ingestion. The only exceptions are the

116-B-6A and 116-DR-9 waste sites, where concentrations of cesium-137 and strontium-90 are still high enough at year 2075 to dominate cancer risks. Because the CTUIR Resident and Yakama Resident scenarios use very high site-raised food ingestion rates, strontium-90 still plays a significant role in food-related exposures at year 2075 for these scenarios. By year 2150, however, CTUIR Resident and Yakama Resident cancer risks above 1×10^{-4} are dominated by arsenic exposure from ingestion of garden produce. Reasonable maximum exposure arsenic concentrations at remediated waste sites based on cleanup verification soil data range between 1.1 and 17.3 mg/kg.[1]

Higher values of arsenic in this range exceed the upland reference area RME value (3.2 mg/kg) and sometimes the Hanford Site 90th percentile value (6.47 mg/kg [DOE/RL-92-24]). However, the RME values for arsenic are less than the IAROD cleanup value of 20 mg/kg for arsenic established by Ecology as an unrestricted land-use cleanup value; this value is adjusted for natural background in soil (http://apps.leg.wa.gov/WAC/default.aspx?cite=173-340-900). The IAROD arsenic cleanup level does not address exposures related to uptake of arsenic into foodstuffs.

Residential scenario present-day radiation dose results are below the threshold of 15 mrem/yr at most remediated waste sites. For sites where radiation dose exceeds 15 mrem/yr, the pattern is similar to that described for the highest cancer risks under the occupational scenarios. Short-lived radionuclides contribute most to present-day radiation dose above 15 mrem/yr, except in the 300 Area where uranium isotopes are most important. With very few exceptions, radiation dose is below 15 mrem/yr for all residential scenarios by year 2075. For example, there are 19 remediated waste sites where present-day RME Subsistence Farmer radiation dose exceeds 15 mrem/yr, but only 2 sites (316-5 and 316-2) where the Subsistence Farmer radiation dose exceeds 15 mrem/yr at year 2075. An important difference between the radiation dose results that exceed 15 mrem/yr for the occupational and residential scenarios is that exposure to strontium-90 by food ingestion pathways frequently dominates the present-day dose results for the residential scenarios.

Chemical HI results for children in the Subsistence Farmer (4.8), CTUIR Resident (42), and Yakama Resident (27) scenarios were above 1.0 for the reference area. Exposure to arsenic from produce ingestion was the main contributor to these background HI levels. There were 15 remediated waste sites where Subsistence Farmer RME child HI results were more than twice the background level. Although arsenic was also the main risk driver at some of these 15 sites (100-F-37, 100-H-21, 300-10, 618-12, and 300-50), the results at sites where Subsistence Farmer RME child HI was highest were related to food ingestion exposure pathways for other metals including mercury (100-K-30, 1607-H2, 100-K-31, 100-K-33, 100-K-32, 100-B-14:6,

[1] The RME concentration for arsenic for upland soil based on MIS samples is 4.7 mg/kg (Section 7.3.2.1) and the RME concentrations for arsenic at remediated waste sites based on the cleanup verification soil data range from 1.1 to 17.3 mg/kg. These differences in concentrations are most likely related to variability in arsenic concentrations among individual waste sites and also differences in sampling techniques as described in Section 7.2.31. The MIS samples represent average surface soil concentrations over a 1-ha area and are combined to produce an estimate of soil concentrations for all upland sites, whereas the cleanup verification soil samples pertain to an individual remediated waste site.

116-B-10), uranium (316-5 and 316-2), and copper (316-1). Unlike arsenic, concentrations of mercury, uranium, and copper at these remediated waste sites exceed 90th percentile Hanford soil background levels by up to approximately 50 times for mercury and uranium and approximately 65 times for copper.

The highest calculated RME representative concentration for lead in shallow zone soil at any remediated waste site is approximately 210 mg/kg. This value applies to the 100-F-37 remediated waste site in the 100-F/100-IU-2/100-IU-6 Area. No representative concentrations for lead at any waste site exceed EPA's recommended residential screening level of 400 mg/kg for lead in bare soil in a play area or the IAROD residential cleanup level of 350 mg/kg. The basis of EPA's lead soil screening criterion is discussed in Section 3.5.8.

7.3.3.4 Key Uncertainties for the Local-Area Risk Assessment. The key uncertainties associated with the local-area risk assessment are summarized below. Additional details of the uncertainty analysis for the local-area risk assessment are presented in Section 5.9.

A key uncertainty associated with the results for the local-area risk assessment relates to how well the exposure scenarios and associated assumptions represent reasonably anticipated future land use. The local-area scenarios range from occupational exposure (where receptors are assumed to live offsite and work on site) to residential scenarios where farmers or Tribal members live onsite and a large portion of their food items are grown or raised on site. For the residential scenarios, there are additional uncertainties related to the assumptions that underlie the calculation of exposure by agricultural pathways. Specific uncertainties related to these pathways include the likelihood that intensive home agriculture will exist on the Hanford Site in the future, whether any remediated waste site is large enough to reasonably support these activities, and (as described above) the models used to estimate plant and animal tissue concentrations from the soil exposure concentrations. Small sites (less than approximately 50 m^2 [538 ft^2]) are identified in Tables 7-9 and 7-10. In these cases, the implicit assumption that the site is of adequate size to support all exposure pathways is most likely not valid because an area larger than approximately 50 m^2 (538 ft^2) is needed to support the raising of adequate food and livestock to satisfy the assumptions for the Subsistence Farmer, CTUIR Resident, and Yakama Resident scenarios.

A second key uncertainty related to the site-specific results of the local-area risk assessment is how applicable the cleanup verification soil data at excavated and backfilled sites are for future human exposure. Residual contamination at backfilled sites is characterized by confirmation soil samples collected on the sidewalls and (if the excavation depth is <4.6 m [15 ft]) bottom of an excavation. Risks may be overestimated using cleanup verification data from excavated sites to estimate exposure concentrations in surface soil because present-day surface soil at these sites consists of backfill. This is of particular significance because the great majority of remediated waste sites with the highest levels of residual risk are sites that have been excavated and backfilled. The excavation depth of remediated waste sites with risks above threshold values is shown in Tables 7-7 through 7-10.

A third key uncertainty that pertains to the residential scenarios relates to modeled exposure concentrations in foods, particularly garden produce. In the case of the produce ingestion chemical HI results for mercury, uranium, and copper, a large conservative bias is anticipated because a linear plant uptake model was applied to soil concentrations that are far above naturally occurring levels. For arsenic, where the range of site soil concentrations is relatively small, uncertainty in produce concentrations is attributable to intrinsic variability related to soil conditions, plant species and tissue type, harvest time, and other variables. A sensitivity analysis of the plant ingestion exposure pathway for the Subsistence Farmer was conducted for 10 remediated waste sites in a manner analogous to that described for the Nonresident Tribal scenario in Section 7.3.2. The values of COPC-specific plant-soil concentration ratios and the quantity of home-grown fruits and vegetables were varied for this analysis. The range of cancer risk and child HI plant ingestion sensitivity analysis results spans approximately four orders of magnitude (see Section 4.1 in Appendix D-1). Reasonable maximum exposure Subsistence Farmer plant ingestion results are frequently close to the upper end of the sensitivity analysis range, but CTE results are between 10 and 30 times lower than the RME results and are nearer to the midpoint of this range. A summary of the sensitivity analysis results is provided in Section 5.9, and the complete analysis is provided in Appendix D-1.

7.3.3.5 Recommendations Related to the Local-Area Risk Assessment. The following recommendations for further evaluation and sampling are related to uncertainties associated with the local-area risk assessment results.

- The presence and depth of backfill and the size of the remediated waste site should both be considered when interpreting risk assessment results for an individual waste site. Activities are planned for the RI to assist in evaluating the lateral extent of residual contamination at several representative remediated waste sites.

- Additional sampling is planned to help establish the manner in which COPC concentrations in soil decrease beyond the sidewalls of historical excavations. This information may be used to estimate potential exposure concentrations resulting from the mixing of soils that might be assumed to accompany construction of a hypothetical residential or occupational building.

- Measuring COPC concentrations in edible portions of native plants growing in the River Corridor, particularly plants identified by Tribal representatives as important food resources, could help to reduce uncertainty in the Subsistence Farmer, CTUIR, and Yakama results and provide information for risk communication purposes.

7.3.4 Summary of Groundwater Risk Assessment Results

The groundwater risk assessment presents an initial evaluation of potential risks associated with groundwater exposures for each of the seven groundwater OUs within the River Corridor. Exposure to groundwater is evaluated for the three residential scenarios (Subsistence Farmer, CTUIR Resident, and Yakama Resident scenarios) and the Resident National Monument/Refuge exposure scenario. Ingestion and dermal exposure to contaminants in groundwater, and inhalation of volatile contaminants, is evaluated for household uses of groundwater in each of these scenarios. Exposure to groundwater contaminants in a sweat lodge was also evaluated for the CTUIR Resident and Yakama Resident scenarios.

The groundwater risk assessment is considered to be screening level because the representative concentrations used in the calculations may not adequately represent present-day exposure concentrations in particular subareas of a groundwater OU. The 50th percentile and 90th percentile concentrations of a COPC from all wells in an OU were used to represent central and upper-bound concentrations, respectively. Supplemental risk calculations were also performed for a subset of wells where risks might exceed those calculated using a 90th percentile value. Additional groundwater evaluation to supplement this assessment will be presented in the RI/FS reports for the River Corridor ROD areas.

The results of the groundwater screening-level risk assessment indicate potential risks above EPA thresholds within each groundwater OU. Reasonable maximum exposure cancer risks were above the upper end of the EPA target risk range of 1×10^{-6} to 1×10^{-4} for each exposure scenario in all of the groundwater OUs, with the exception of Resident National Monument/Refuge and Subsistence Farmer risks in the 100-HR-3 and 100-FR-3 OUs. With the exception of the Resident National Monument/Refuge scenario in the 100-B/C-5 OU, hazard indices were above EPA's threshold value of 1.0 for each exposure scenario and each groundwater OU.

7.3.4.1 Key Uncertainties for the Groundwater Risk Assessment. There are several key uncertainties for the groundwater results that contribute to qualification of these calculations as "screening level." These are mostly related to the ability of the existing groundwater data set to represent current baseline conditions for possible exposures within each groundwater OU. Analytical data used for the screening-level groundwater risk assessment are obtained from several groundwater-monitoring programs, including the *Atomic Energy Act of 1954* surveillance program, the *Resource Conservation and Recovery Act of 1976* (RCRA) compliance program, and the CERCLA program. Sampling and analysis data from these programs comprehensively define the suite of contaminants associated with existing and potential groundwater contamination sources. However, differences in sampling frequencies (monthly, annually, or tri-annually); differences in analytes analyzed at each monitoring well (radiological and chemical); and differences in method detection limits create uncertainties associated with the spatial, chemical, and temporal representative qualities of the data set used for this risk assessment.

7.3.4.2 Recommendations Related to the Groundwater Risk Assessment. Activities that may help reduce uncertainties, update the conclusions of the screening-level groundwater risk assessment, and ensure that no contaminants were inadvertently overlooked based on use of the existing data set are identified in *Integrated 100 Area Remedial Investigation/Feasibility Study Work Plan* (DOE/RL-2008-46). These activities include the following:

- Identify existing and/or install new monitoring wells that are spatially representative of the ROD area. This set of monitoring wells will represent locations where a receptor potentially could contact groundwater.

- Conduct multiple rounds of sampling to obtain temporal representation of COPC concentrations in the unconfined aquifer. Additional rounds of sampling at spatially representative monitoring wells will represent current groundwater conditions and capture the influence of river fluctuations on COPC concentrations.

- Analyze all spatially representative monitoring wells for a focused list of groundwater COPCs identified for the ROD area for each round of sampling. Analyzing each of the monitoring wells for COPCs will provide a data set that is representative of potential releases to the groundwater.

- Evaluate sample results from characterization activities to support final remedial action decisions for groundwater.

Additional groundwater evaluations will be presented in the RI/FS reports for the River Corridor ROD areas based on the additional data collection activities described above.

7.4 PRELIMINARY REMEDIATION GOALS

Preliminary remediation goals were developed for soil for various scenarios including Recreational, Industrial/Commercial Worker, and Resident National Monument/Refuge Worker scenarios to address the following questions listed at the beginning of this section:

- What are the PRGs for the various RCBRA exposure scenarios?

- Are soil cleanup levels established under the IARODs protective of human health and the environment using current regulatory guidelines?

- Are there any recommendations for additional studies or monitoring that should be considered at this time to reduce uncertainties with specific risk results and conclusions or establishing health-protective PRGs?

Preliminary remediation goals were not developed for the Nonresident Tribal, CTUIR Resident, Yakama Resident, or Subsistence Farmer scenarios. As summarized in Sections 7.2.3.1 and 7.3.3.4, there are significant uncertainties associated with the quantification of exposure for the food pathways. The sensitivity analyses that were conducted for the plant ingestion pathway are provided in Appendix D-1. Because site-specific tissue concentrations for edible plants are not available, and the range of plant-soil concentration ratios that are cited in the literature span up to four orders of magnitude for COPCs that contribute most to risk by this pathway, PRGs that may be calculated for these scenarios are considered too uncertain to provide useful information for decision-making in the River Corridor ROD areas.

EPA risk assessment guidance (EPA/540/1-89/002) states that remedial actions should be based on an estimate of the RME expected to occur under both current and future land-use conditions. The RME is defined as the highest exposure that is reasonably expected to occur at a site. Reasonable maximum exposures are estimated for individual pathways and if individuals may be exposed via more than one pathway, the combination of exposures across pathways should represent the RME. A variety of scenarios were used to develop PRGs presented in this section so that remedial alternatives under various land-use options and RME scenarios can be evaluated in the RI/FS reports for the River Corridor ROD areas. In addition, the PRGs may be used to help determine the final soil cleanup levels for the River Corridor.

The PRGs were developed in accordance with the methodology presented in *Risk Assessment Guidance for Superfund: Volume 1 - Human Health Evaluation Manual (Part B, Development of Risk-Based Preliminary Remediation Goals)* (EPA/540/R-92/003). The same exposure assumptions, toxicity criteria, and equations for each scenario that were used for the "forward risk calculations" (as described in Sections 3.3 through 3.6) were used with the EPA/540/R-92/003 Part B methodology to "back calculate" a risk-based PRG (as shown in Section 3.7). Uncertainties related to the exposure and toxicity assessments for these scenarios are also applicable to the associated PRGs.

The only exception to the equivalence of the "forward" and "backward" risk calculations relates to the radionuclide soil PRGs for the RCBRA scenarios. In Sections 4.0 and 5.0, radionuclide cancer risk is calculated for the present day, and also with radionuclide soil activities decayed to the years 2075 and 2150, in order to show the impact of radionuclide decay on estimated risk. To address radiological decay in the RCBRA soil PRGs, an EPA methodology described in Section 3.7 has been used in their calculation. The use of this methodology is equivalent in this regard between the RCBRA radiological PRGs and the risk-based PRGs calculated using RESRAD or referenced to EPA. The RESRAD calculations that are based on a cancer risk threshold and the EPA radionuclide PRGs also integrate the effect of radiological decay during the presumed exposure period.

The PRGs for individual radionuclides are based on a 1×10^{-4} target cancer risk. The PRGs for individual chemicals are based on a target cancer risk of 1×10^{-6} and/or an HQ of 1 (using an adult or child receptor for HQ depending on the scenario), depending on whether the chemical has published toxicity criteria for carcinogenic and/or noncarcinogenic health effects. Preliminary remediation goals were computed for the superset of COPCs from all broad-area environments (i.e., surface soil and sediment for upland, riparian, and near shore) and shallow zone soil for all six ROD areas.

Tables 7-11 through 7-13 present the PRGs for the Recreational, Occupational (Resident National Monument/Refuge Worker and Industrial/Commercial Worker) and Residential scenarios.

In addition, the PRG tables show the IAROD cleanup levels for radionuclides and chemicals. The IAROD cleanup levels for radionuclides were calculated using the RESRAD computer code (http://web.ead.anl.gov/resrad/home2/) with a radiation dose threshold of 15 mrem/yr. The residential IAROD cleanup levels for chemicals were based on the MTCA Method B (unrestricted use) methodology with toxicity values current at the time. As a point of comparison, the PRG tables also show draft PRGs for radionuclides for the IAROD residential scenario using assumptions and methods developed for the 100 Area and 300 Area RI/FS and a target cancer risk of 1×10^{-4} (ECF-HANFORD-10-0429) and draft PRGs for chemicals for the IAROD residential scenario developed for the 100 and 300 Area RI/FS (ECF-HANFORD-10-0444) based on MTCA Method B (unrestricted use) methodology.[2]

Table 7-13 includes EPA PRGs for radionuclides and chemicals so that comparisons can be made between the standard default EPA residential scenario, the residential IAROD cleanup levels, and the draft PRGs for the 100 and 300 Area RI/FS. For radionuclides, the EPA PRGs include soil ingestion, inhalation, external exposure, and ingestion of homegrown fruits and vegetables. For chemicals, the EPA PRGs include soil ingestion, inhalation, and dermal contact. Finally, the 90th percentile background values are also provided in the tables for comparison purposes.

The Recreational and Resident National Monument/Refuge exposure scenarios evaluated in the RCBRA are based on activities that result in individual exposures across broad areas of the River Corridor. The purpose of evaluating broad-area risks is to determine if contaminant concentrations in surface soil and other accessible environmental media present concerns under these exposure assumptions. Although these scenarios have been used to develop PRGs, assigning sediment exposure to soil if necessary, the PRGs for these scenarios may not be directly applicable for evaluating residual contaminant concentrations at individual waste sites. This is because any particular waste site, even large sites, would reflect only a fraction of the exposure area that an individual would contact under the exposure assumptions governing these scenarios.

7.4.1 Recreational Scenarios

As shown in Table 7-11, radionuclide PRGs (based on target cancer risk of 1×10^{-4}) for the Recreational scenarios are all greater (i.e., less restrictive) than the residential IAROD cleanup levels and the related risk-based values (i.e., draft 100 and 300 Area PRGs), indicating that these residential radionuclide values are generally protective for the recreational scenarios.

For chemicals, all of the Casual User PRGs are greater than the residential IAROD cleanup levels except for two polycyclic aromatic hydrocarbons (PAHs) (i.e., dibenz(a.h)anthracene and phenanthrene). All of the Avid Angler PRGs for chemicals are greater than the residential

[2] The draft 100 and 300 Area PRGs for the IAROD Residential scenario are presented as a preliminary point of comparison for the PRGs based on the RCBRA scenarios. The draft 100 and 300 Area PRGs may be revised based on updates to toxicity values or adjustments to exposure factors during the preparation of the 100 and 300 Area RI/FS report.

IAROD cleanup levels except for dibenz(a.h)anthracene. For carcinogenic PAHs like dibenz(a.h)anthracene, the Casual User PRGs use toxicity equivalency factors from EPA and the IAROD cleanup levels and the draft 100 and 300 Area PRGs use factors consistent with MTCA.

In addition, some of the PRGs for the Avid Hunter scenario are lower (i.e., more restrictive) than the residential IAROD cleanup levels and the draft 100 and 300 Area PRGs values due to the contribution from the game meat ingestion pathway. When only direct contact pathways are considered (incidental soil ingestion, dermal contact, and dust inhalation), the chemical PRGs for the Avid Hunter scenario are all greater (i.e., less restrictive) than the residential IAROD cleanup levels. There is considerable uncertainty associated with exposure to soil contaminants for the game meat ingestion pathway (see Section 4.3.3), and exposure from this pathway is likely to be overestimated.

7.4.2 Occupational Scenarios

The occupational scenario PRGs (Resident National Monument/Refuge and Industrial/Commercial) are shown in Table 7-12. The Industrial/Commercial PRGs are all greater (i.e., less stringent) than the residential IAROD cleanup levels and draft PRGs for the 100 and 300 Area RI/FS for radionuclides and chemicals. Comparisons of Resident National Monument/Refuge PRGs for radionuclides to residential IAROD cleanup levels and draft 100 and 300 Area PRGs indicate that, for most radionuclides, the Resident National Monument/Refuge PRGs are less stringent except for cesium-137, cobalt-60, europium-152, and europium-154. For chemicals, the residential IAROD cleanup levels or the draft 100 and 300 Area PRGs are the most stringent when compared to the occupational PRGs except for dibenz(a,h)anthracene. As noted in Section 7.4.1, there are differences in the toxicity equivalency factors for PAHs used for the IAROD cleanup levels and draft 100 and 300 Area PRGs as compared to the PRGs calculated for the RCBRA scenarios.

The PRGs for the radionuclides cesium-137, cobalt-60, europium-152, and europium-154 are mostly influenced by the contribution of external irradiation, which is a function of the length of time spent onsite (inside and outside of a building) and the gamma shielding factor attributed to the building. There are slight differences in the assumptions used for the Resident National Monument/Refuge PRGs versus the IAROD and draft 100 and 300 Area PRGs that result in small differences in the values (i.e., Resident National Monument Worker/Refuge PRGs are within a factor of 2 of the IAROD cleanup levels and 100 and 300 Area PRGs).

7.4.3 Residential Scenarios

As noted above, PRGs were not calculated for the Subsistence Farmer, CTUIR Resident, or Yakama Resident scenarios because of the high level of uncertainty associated with the food ingestion pathways. Table 7-13 presents the residential IAROD cleanup levels, the draft 100 and 300 Area PRGs that are based on the residential/unrestricted use scenario, and default EPA PRGs for the residential scenario.

Residential IAROD cleanup levels for radionuclides were compared to EPA PRGs (http://www.epa.gov/superfund/health/contaminants/radiation/radrisk.htm) to provide information about the protectiveness of the IAROD cleanup levels as compared to the EPA default residential scenario for radionuclides. As noted above, the residential IAROD cleanup levels for radionuclides include soil ingestion, inhalation of particulates, external exposure to ionizing radiation, and food pathways. The EPA PRGs include soil ingestion, inhalation of particulates, external exposure to ionizing radiation, and ingestion of homegrown fruits and vegetables. As shown in Table 7-13, the residential IAROD cleanup levels are more stringent than the EPA PRGs for radionuclides based on a 10^{-4} target risk.

Residential IAROD cleanup levels for chemicals were compared to EPA PRGs (i.e., residential regional screening levels [RSLs] [http://www.epa.gov/reg3hwmd/risk/human/rb-concentration_table/index.htm]) for chemicals to provide information about the protectiveness of the IAROD cleanup levels as compared to EPA default residential scenarios. As noted above, the residential IAROD cleanup levels for chemicals are based only on the incidental soil ingestion pathway. The EPA residential RSLs are based on EPA default residential exposure assumptions and include the incidental soil ingestion, dust inhalation, and dermal contact pathways. Table 7-14 shows a comparison of the contribution of the various pathways to exposure for the residential IAROD cleanup levels and the EPA residential RSLs for key risk drivers for the remediated waste sites. For the key risk drivers shown, the incidental soil ingestion pathway contributes 69% to 100% of the exposure based on the EPA residential default assumptions. For many chemicals, the incidental soil ingestion pathway is the largest contributor to exposure of the three direct contact pathways (incidental ingestion, inhalation, and dermal contact). Consequently, the residential IAROD cleanup levels will be essentially equal to or somewhat less stringent than the EPA RSLs depending on the chemical.

7.4.4 Summary

A variety of PRGs were developed based on the RCBRA exposure scenarios to provide information for the analysis of remedial alternatives in the RI/FS reports for the ROD areas of the River Corridor. The PRGs cover a wide range of exposure scenarios ranging from the hypothetical situation where recreational users may occasionally visit the site to residents living onsite and consuming food items that are predominantly grown or raised onsite.

The PRGs were also calculated to provide a comparison to the IAROD cleanup levels that were used for interim actions performed in the River Corridor. In general, the PRGs for the recreational and occupational scenarios are less stringent than the residential IAROD cleanup levels indicating the IAROD cleanup levels are protective for those scenarios.

Preliminary remediation goals were not developed for the Nonresident Tribal, CTUIR Resident, Yakama Resident, or Subsistence Farmer scenarios because of the high level of uncertainty associated with the food pathways. Measuring COPC concentration in edible portions of native plants, particularly plants identified by Tribal representatives as important food resources, may help to reduce food pathway uncertainties in these scenarios and provide useful information for risk communication purposes.

7.5 KEY CONCLUSIONS OF THE HUMAN HEALTH RISK ASSESSMENT

The RCBRA provides a summary of the QRAs that were performed to determine whether contaminant concentrations posed an unacceptable risk that warranted remedial action and have been used to establish the basis for action for all waste sites identified in the River Corridor IARODs and RODs. The original determination of the presence of unacceptable risk and basis for action at yet-to-be remediated waste sites is supported by the field experiences and information gathered through implementation of the observation approach-based soil cleanup actions in the River Corridor over the past 13 years.

The RCBRA evaluated potential cumulative cancer risks and noncancer hazards associated with post-interim action conditions at wastes sites and potentially affected areas within the River Corridor. A wide range of exposure scenarios was included in the human health risk assessment, ranging from recreational scenarios where receptors occasionally visit the site to residential scenarios where receptors spend effectively 100% of their time on a remediated waste site and consume food items that are predominantly produced onsite. Soil PRGs were also developed for most of these scenarios, and compared to IAROD RAGs, published EPA screening criteria, and soil background levels, to provide information for analysis of alternatives in the feasibility study.

The key conclusions for the different risk evaluations included in Volume II of the RCBRA are as follows.

- IAROD Scenario Calculations: Of the 156 remediated waste sites evaluated in the RCBRA using exposure assumptions consistent with those used to develop the residential IAROD RAGs, all except 14 sites have cumulative risks for radionuclides within EPA's risk management range of 10^{-6} to 10^{-4}, chemical risks (without arsenic) less than 10^{-5}, and HIs approximately equal to or below 1.0. All of the 14 sites with cancer risks that exceed 10^{-4} have been excavated and residual contamination covered by clean backfill. For all but 2 of 14 sites, the key risk drivers are short-lived radionuclides including europium-152, cesium-137, and strontium-90. Soil concentrations of cesium-137 and strontium-90 decrease by one-half in about 30 years due to radiological decay, and europium-152 decreases about twice as quickly.

- Broad-Area Risk Assessment: The results of the risk assessment for groundwater contaminants at shoreline springs along the Columbia River indicate that, with the exception of Spring 42-2 in the 300 Area and 100-F Spring, there is negligible risk from potential consumption of spring water. For the three recreational scenarios (Casual User, Avid Hunter, and Avid Fisherman), total RME cancer risk estimates (i.e., sum of radionuclide and chemical risk) are within EPA's risk management range of 10^{-6} to 10^{-4}, and RME HIs are less than 1.0 for all six ROD areas. For the Nonresident Tribal scenarios, the total cancer risk estimates exceed 10^{-4} and HIs exceed 1.0 for all ROD areas, mostly due to exposures that are associated with ingestion of plants assumed to be gathered from the Hanford Site. A large proportion of Nonresident Tribal cancer risk and HI is related to arsenic soil concentrations that are approximately equivalent to levels in areas unaffected by Hanford Site activities. When cancer risk estimates are calculated without the contribution of arsenic, the total cancer risk estimates still exceed 10^{-4} for all six ROD areas. The key risk drivers other than arsenic are technetium-99, carbon-14, strontium-90, benzo(a)pyrene, and Aroclor-1254, predominantly by the plant and game ingestion pathways. There is a high degree of uncertainty related to contaminant intake from these food pathways because concentrations in edible plant tissues and game tissue are modeled using literature uptake factors, not site-specific data.

- Local-Area Risk Assessment: For the Industrial/Commercial scenario, all but 4 of the 156 remediated waste sites evaluated in the RCBRA have total RME cancer risks within EPA's risk management range and RME HIs less than 1.0. For the four sites with total cancer risks that exceed 10^{-4}, the key risk drivers are short-lived radionuclides (cesium-137, cobalt-60, and europium-152). For the Resident National Monument/Refuge scenario, no remediated waste sites exceed the HI threshold of 1.0, but total RME cancer risk exceeds 10^{-4} at 11 of the 156 sites. However, these cancer risks are related to short-lived radionuclides and will naturally decrease with time. For the Subsistence Farmer scenario, many sites have total RME cancer risks greater than 10^{-4} and HIs greater than 1.0. At many of these sites, risks greater than 10^{-4} are related to external irradiation from short-lived radionuclides that will decay to low levels with time. At some sites, exposure to arsenic or other metals through homegrown produce and livestock products are responsible for RME risk and HI above thresholds. As with the Nonresident Tribal results, there is a high degree of uncertainty and a likely conservative bias related to contaminant intake from these food pathways. Also, the great majority of remediated waste sites where residual risks are highest have been excavated and residual soil contamination is now covered with clean backfill.

- Groundwater Risk Assessment: The screening-level groundwater assessment indicates potential cancer risks and chemical hazard above EPA thresholds within each of the seven groundwater OUs. However, the existing groundwater data set used in this screening assessment may not represent current baseline conditions for possible exposures within each groundwater OU. Additional groundwater evaluations will be presented in the RI/FS reports for the ROD area based on additional data collection activities that are being conducted as part of the remedial investigation.

There are several key uncertainties associated with the use of the human health risk assessment results for supporting risk management decisions and selection of soil PRGs. The most important of these are noted in the summary of results for the IAROD scenario calculations and for the broad-area and local-area risk assessments.

For the occupational and residential exposure scenarios evaluated in the local-area risk assessment, a key uncertainty is the use of soil exposure concentrations at excavated waste sites where this soil is now covered by clean backfill. The nature of uncertainty in the use of historical soil data at excavated waste sites is to produce a conservative bias in the risk assessment results.

A second key uncertainty for the local-area risk assessment, where the highest cancer risks are related to short-lived radionuclides, is the appropriate time frame and land-use assumptions for risk management. These two key uncertainties are not independent, because the short-lived radionuclides are predominantly present at backfilled sites where land use consistent with intrusive activities may be necessary in order for external radiation exposure to occur.

A third key type of uncertainty relates to the food pathways, which applies to the Nonresident Tribal and residential exposure scenarios. Specific uncertainties related to these pathways include the likelihood that intensive home agriculture will exist on the Hanford Site in the future, whether any remediated waste site is large enough to reasonably support these activities, and (as described above) the models used to estimate plant and animal tissue concentrations from the soil exposure concentrations. The magnitude of the uncertainty may be measured by a factor of 10, 100, or more in the risk assessment results pertaining to these pathways. As has been discussed here and in Sections 4.0 and 5.0, there is evidence that the human health risk assessment results for these pathways are not only uncertain but are likely to have a considerable conservative bias.

For the local-area residential risk assessment results, uncertainty and bias related to contaminant uptake in foods is compounded by uncertainty and bias related to the use of historical (pre-backfill) soil data at excavated waste sites. As noted in Sections 7.3.1 and 7.3.3, additional specific uncertainties related to residential agricultural pathways include the likelihood that intensive home agriculture will exist in the future, and whether any particular site is large enough to reasonably support these activities. The compounding of these uncertainties, and the conservative bias introduced by how some of these uncertainties were addressed in the risk calculations, necessitates that caution be used when applying these scenarios for risk management at individual waste sites.

There are several types of data that could be collected to reduce uncertainties related to the risk assessment results. Measuring COPC concentrations in edible portions of native plants could help reduce uncertainties related to risks associated with the use of plant uptake pathways in the Nonresident Tribal scenario. Additional sampling at excavated remediated waste sites could be helpful in establishing COPC concentration gradients beyond the sidewalls of excavations. Information on how residual soil concentrations attenuate with distance from the sidewalls could be helpful in estimating exposure concentrations associated with the mixing of soils that may result from future construction of a structure. Additional groundwater data collection efforts have been identified for the RI to help reduce uncertainties related to spatial, chemical, and temporal representativeness of the groundwater data.

The PRGs that are presented in Section 7.4 represent a broad range of scenarios. Based on the comparison of PRGs to residential IAROD cleanup levels, the residential IAROD cleanup levels are generally protective for the recreational and occupational scenarios. The results of the human health risk assessment, the uncertainty analyses related to these results, and the PRGs will be used in the FS to evaluate remedial alternatives and to support development of final cleanup goals for the River Corridor ROD areas.

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