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A Risk-based Screening Analysis for Radionuclides Released to the Columbia River from Past Activities at the U.S. Department of Energy Nuclear Weapons Site in Hanford, Washington

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Authors

**Helen A. Grogan, Ph.D., Cascade Scientific, Inc.
Arthur S. Rood, K-Spar, Inc.
Jill Weber Aanenson, Scientific Consulting, Inc.
Edward B. Liebow, Ph.D., Environmental Health and Social Policy Center**

Principal Investigator

John E. Till, Ph.D., Risk Assessment Corporation

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A RISK-BASED SCREENING ANALYSIS FOR RADIONUCLIDES RELEASED TO THE COLUMBIA RIVER FROM PAST ACTIVITIES AT THE U.S. DEPARTMENT OF ENERGY NUCLEAR WEAPONS SITE IN HANFORD, WASHINGTON

EXECUTIVE SUMMARY

The Hanford Environmental Dose Reconstruction (HEDR) Project reconstructed doses to offsite members of the public resulting from radionuclide releases since 1944 from operation of U.S. Government facilities at the Hanford Nuclear Site, in Washington State. This report focuses on radionuclide releases to the Columbia River only. Initially, the HEDR Project considered all radionuclides released to the Columbia River between 1944 and 1972. Following a series of scoping calculations, doses were calculated for five radionuclides: sodium-24, phosphorus-32, zinc-65, arsenic-76, and neptunium-239. The radionuclide exposure pathways were also selected on the basis of scoping calculations.

In a review of the HEDR dose estimates for the Agency for Toxic Substances and Disease Registry (ATSDR), it was suggested that ^{131}I , ^{60}Co , and ^{90}Sr should have been included in the HEDR dose calculations for the Columbia River. The objective of this report is to develop and apply a risk-based screening methodology that can be used to evaluate this recommendation. However, the screening methodology is applied to 23 radionuclides to ensure a comprehensive evaluation, rather than limiting the screening to only the 5 HEDR radionuclides with dose estimates and the 3 radionuclides identified in the ATSDR review.

A two-stage screening methodology is developed. In the first-stage, radionuclides and pathways are compared with a predefined risk-based screening value. Only those radionuclides and pathways with screening values above the screening criterion are considered further in the analysis. Conservative assumptions¹ are made to characterize the exposure pathways and the radionuclide parameter values to ensure that no potentially important ones are removed from the analysis. Eight exposure pathways are considered. These are designed to account for the different types of individuals, activities, and practices that may have resulted in exposure to radionuclides released to the Columbia River. Explicit consideration is given to Native American tribes potentially impacted by releases from the Hanford Site because they lived in close proximity to the river and their lifestyle activities were intimately linked with the river. Due to the conservative assumptions, the screening values are generally over estimates of risk to the most at-risk individuals and are expected to overestimate the risks to all real individuals.

In the second-step, exposure scenarios are defined to represent the most exposed river users. More than one exposure scenario is required because of the differing habits and activities of the various river users. Three exposure scenarios are developed: Native American, local resident, and migrant worker. The risk-based screening values do not represent an actual risk because there is

¹ A conservative assumption in this type of analysis is one that is unlikely to underestimate the exposure to a certain nuclide, but may, in fact, overestimate the exposure to that nuclide. For example, if it is known that the average person on the river spent between 2 and 8 hours a day by the riverside, a conservative assumption might be that we assume everyone spent 8 hours a day there.

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still significant conservatism built into the calculations. The conservatism includes factors such as location and time. However, because this is consistent across pathways, we can develop a relative ranking and, therefore, prioritization of the radionuclides and exposure pathways.

A 2-dimensional advection-dispersion river transport model is developed to calculate radionuclide concentrations in river water and sediment from the downstream Hanford Site boundary to the McNary Dam below the confluence of the Snake River. Transport of radionuclides in both the dissolved and sorbed phase is considered, and is coupled to conservative assumptions for receptor exposure pathways to estimate screening risk values. The river transport model is calibrated to measured river water and sediment concentrations.

An absolute risk-based criterion of 10^{-4} is recommended² for the initial screening of 23 radionuclides released to the Columbia River from the Hanford Nuclear Site. Application of this criterion identifies 4 radionuclides (⁴⁵Ca, ⁵¹Cr, ⁹³Y, and ¹²²Sb) that can be removed from further consideration. It also identifies a number of exposure pathways that can be eliminated from the analysis including external exposure to contaminated sediments, ingestion of contaminated sediments and inhalation of contaminated aerosols. Combined, these pathways contribute less than 0.5% to the total screening risk value calculated for all pathways. Of the remaining pathways, exposure to contaminated sediments through dermal contact, and ingestion of contaminated river water during swimming are low priority.

The screening results support the HEDR Project conclusion that fish ingestion is the dominant exposure pathway for releases to the Columbia River. For most radionuclides, fish ingestion is the dominant exposure pathway, in a few cases (⁵⁶Mn, ⁹³Y, ¹³³I) water ingestion is dominant, and for ²⁴Na, boating is the dominant pathway. For total risk (all nuclides), fish ingestion accounts for over 90% of the total risk. Most of the exposure is incurred over the years 1952 to 1964. These years correspond to the years of highest release from the Hanford reactors.

The relative ranking of radionuclides in the second-stage screening based on the 3 scenarios (local resident, migrant worker, and Native American) shows that some radionuclides are more significant than others. In all 3 scenarios, ⁷⁶As is the highest contributor to risk. In addition to the five radionuclides (⁷⁶As, ³²P, ²³⁹Np, ⁶⁵Zn, and ²⁴Na) for which detailed dose calculations were made in the HEDR Project, ⁶⁹Zn and ⁹⁵Zr emerge as important risk contributors for all 3 scenarios. Strontium-89 and ⁹⁰Sr are important for the Native American scenario where they contribute ~16% and ~5%, of the total risk, respectively, because consumption of whole fish rather than fish filets is assumed. If further evaluation of risks from radionuclides released to the Columbia River is undertaken, these nine radionuclides should be considered as most important for the analysis.

The significance of fish ingestion for Native American users of the river may have been underestimated in the HEDR Project because fish consumption rates reported for Native Americans tend to be higher than the value assumed for the maximum representative individual in the HEDR Project. Furthermore, contrary to the HEDR assumptions, it is reasonable to assume the entire fish was consumed which increases the dose and risk for a number of radionuclides, in particular ^{89,90}Sr. The second-stage screening also indicates that ⁶⁰Co can be eliminated from further analysis because it contributes <1% to the total risk for all pathways in all 3 scenarios.

² We apply this screening value to demonstrate its application, and its use does not represent endorsement by the Centers for Disease Control and Prevention.

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ACRONYMS

API	American Petroleum Institute
ATSDR	Agency for Toxic Substances and Disease Registry
BCF	bioconcentration factor
CDC	Center for Disease Control and Prevention
EDE	effective dose equivalent
EPA	U.S. Environmental Protection Agency
FB	fractional bias
HEDR	Hanford Environmental Dose Reconstruction (Project)
ICRP	International Commission on Radiological Protection
IDA	individual dose assessment
K_d	sorption coefficient ($m^3 g^{-1}$)
NCRP	National Council on Radiation Protection and Measurements
NESHAPs	National Emissions Standards for Hazardous Air Pollutants
NID	negligible individual dose
NMSE	normalized mean square error
NRC	U.S. Nuclear Regulatory Commission
<i>P/O</i>	predicted-to-observed
RM	River Mile
STRRM	Source Term River Release Model
TSP	Technical Steering Panel
<i>UF</i>	uncertainty factor
USGS	U.S. Geological Survey

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OBJECTIVE

The primary purpose of the Hanford Environmental Dose Reconstruction (HEDR) Project was to reconstruct doses to offsite members of the public resulting from radionuclide releases since 1944 from operation of U.S. Government facilities at the Hanford Nuclear Site, in Washington State. The Columbia River Dosimetry Code (Farris et al. 1994) was developed as part of the HEDR Project to calculate radiation doses for hypothetical individual users of the Columbia River at various locations on the river. Initially, the HEDR Project considered all radionuclides released from the Hanford Nuclear Site between 1944 and 1972. Ultimately, doses were calculated for five radionuclides: ^{24}Na , ^{32}P , ^{65}Zn , ^{76}As , and ^{239}Np . The water concentrations for these radionuclides were estimated by the CHARIMA computer code (Walters et al. 1994).

The Technical Steering Panel of the HEDR Project selected the five radionuclides on the basis of a series of scoping or screening calculations (Napier 1993). The radionuclide exposure pathways considered in the dose calculations were also selected on the basis of these scoping calculations. In a review of the HEDR dose estimates for the Agency for Toxic Substances and Disease Registry (ATSDR), Hoffman et al. (1998) suggested that ^{131}I , ^{60}Co , and ^{90}Sr should also have been included in the HEDR dose calculations and in the Hanford individual dose assessment (IDA)^a process for the Columbia River.

This report develops and applies a screening methodology to the radionuclide releases to the Columbia River (see Appendix A for the original statement of work). To ensure a comprehensive evaluation, the screening methodology is applied to a total of 23 radionuclides rather than just the five radionuclides with dose estimates in the HEDR Project, and the 3 radionuclides suggested by Hoffman et al. (1998).

A risk-based decision criterion is recommended and applied to determine if any radionuclides and/or exposure pathways can be eliminated from further consideration using conservative assumptions about exposure to Columbia River water. Following the initial screening, three scenarios are used to represent the river users most at risk and to prioritize the radionuclides and exposure pathways. This report compares these results with the original HEDR analysis and the recommendations of Hoffman et al. (1998).

Screening Methodologies

Screening identifies the most important radionuclide releases to the Columbia River, in terms of direct or indirect exposure risks to individuals. Typically, screening is conducted early in a study before detailed estimates of radionuclide releases exist, to identify where effort and resources should be allocated. Because this screening is taking place after the HEDR Project was completed, detailed release estimates are available for 11 radionuclides. Two general approaches

^a The Hanford IDA Project is designed to allow individuals exposed to Hanford radiation releases historically to estimate their individual radiation doses.

can be used for screening: one is based on an absolute screening and the other on a prioritization (relative ranking).

Using the absolute screening approach, radionuclides and pathways are screened against an absolute screening value. Only those radionuclides and pathways with screening values above the predefined screening criterion are considered further in the analysis. For this approach to be effective, it is essential that no radionuclide or exposure pathway that is potentially important is removed from the analysis. To ensure this is the case, conservative assumptions are made to characterize the exposure pathways and the radionuclide parameter values.

Using the prioritization approach, radionuclides and pathways are evaluated and ranked in order of significance. To allow a relative ranking it is important that the parameter values used to characterize the exposure pathways and the radionuclides are selected in a consistent manner to avoid biasing the results. Assigning realistic parameter values is preferred because it is difficult to define parameter values with the same degree of conservatism consistently. Furthermore, the relative importance of exposure pathways may depend on the concentration of the radionuclide in the environment.

Both approaches have advantages and disadvantages. Using the absolute screening approach radionuclides and pathways of no potential significance are removed from the analysis. However, the conservatism associated with the absolute screening may result in few radionuclides and/or exposure pathways being eliminated. Also, this initial screening provides little information about the relative importance of specific exposure pathways or radionuclides. On the other hand, prioritization may require excessive effort to achieve a defensible relative ranking. Also, relative ranking provides no information about the absolute significance of any radionuclide or exposure pathways.

Screening Approach for Radionuclide Releases to the Columbia River

To screen the radionuclides released to the Columbia River, we employ both approaches. Initially, we defined a risk-based screening criterion. We make conservative assumptions about the exposure location and pathway characteristics. The objective of this initial screening is not to underestimate the potential risk to any individual for a given radionuclide or a given exposure pathway. The resulting risk-based screening values are compared to the risk-based screening criterion, and all radionuclides and exposure pathways that fall above the criterion remain in the analysis (Figure 1).

In the second-step, a number of exposure scenarios are defined to represent the most exposed river users. More than one exposure scenario is required to cover the range of river users because of the differing habits and activities of the various groups. In this case, three exposure scenarios are defined. The types of exposure pathways, the characteristics of the exposed individual and the risk-based screening value do not represent an actual risk because there is still significant conservatism built into the calculation. The conservatism includes factors such as location and time. However, because this is consistent across pathways, we can develop a relative ranking and, therefore, prioritization of the radionuclides and exposure pathways (Figure 1).

A number of inputs are required to apply the screening methodology. These include the radionuclide release estimates to the Columbia River from 1944–1971, an environmental transport model, the exposure pathways of potential significance with regard to health-risk, and a

risk-based decision criterion. Each is discussed briefly below and in detail in later sections of this report.

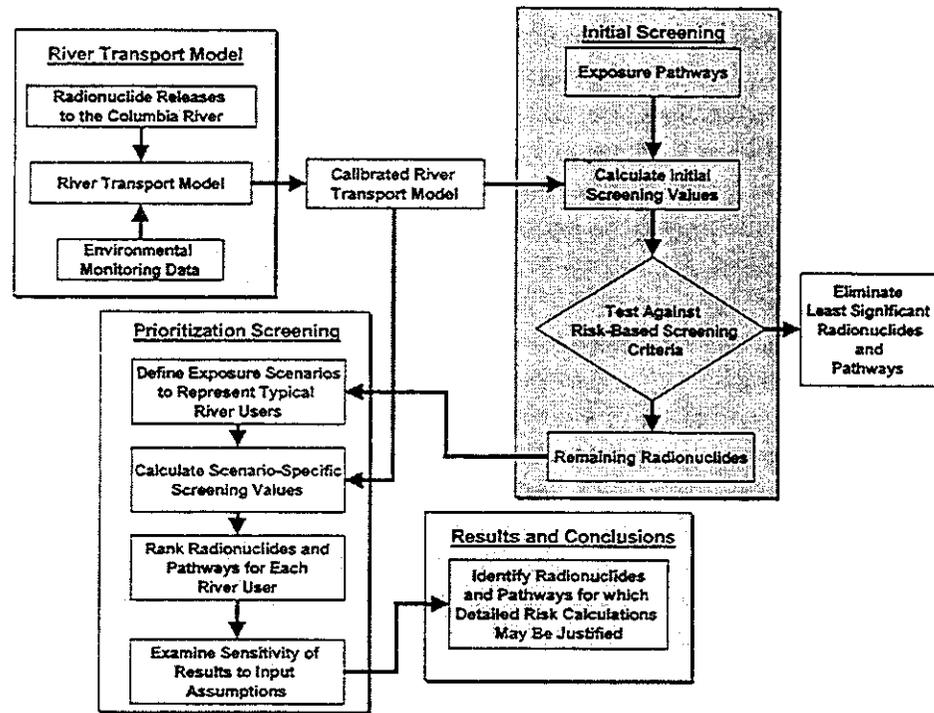


Figure 1. Overview of the screening methodology.

Estimates of the quantities of radionuclides released to the Columbia River throughout the operation of the Hanford Site provide the source term for any exposure and risk calculations. A summary of reactor operations and estimates of radionuclide releases to the Columbia River is provided in a separate section.

An environmental transport model is required in the screening methodology to account for the transport of radionuclides downstream from the Hanford Site in both the water and sediment, sediment accumulation, and transfer into other environmental media. Historical measurements of radionuclide concentrations in the various media are important for calibrating and testing the environmental transport model. Because this is a screening methodology, a detailed river model that estimates radionuclide concentrations at numerous locations downstream and in the Pacific Ocean is not required. The highest radionuclide concentrations in media, primarily river water and sediment, are required to ensure that exposure consequences are not underestimated. For the river water, this is a short distance downstream of the reactor outfall locations. For sediment, this may be somewhat further downstream, in a location of sediment accumulation. For the current analysis, three locations are considered: Ringold, 300 Area Boundary, and Richland. Ringold is the closest offsite location on the far bank that is downstream of the Hanford Site. Access near the 300 Area is downstream of Ringold but also on the near shore where reactor effluent was released. Richland is a short distance downstream on the near bank of the Columbia River.

The different ways in which people were exposed to radionuclides released into the Columbia River, referred to as exposure pathways, must be identified. People who used the Columbia River as a source of drinking water, ate fish or waterfowl from the river, and swam, fished, and boated in the river could have been exposed to radionuclides. The different groups of people who made use of the river and their different activities are considered to ensure that no important exposure pathway is omitted from the analysis or that the parameters used to quantify the exposure pathway are not underestimated. For example, fish consumption rates for Native American tribes that fished the river tend to be significantly higher than other residents along the Columbia River. For the initial screening, each exposure pathway is considered in isolation with regard to the potential for exposure, and parameter values that represent the most exposed individuals are selected. Following the initial screening, scenarios are defined to represent specific river users, with consistent exposure pathways and parameter values.

Another input to the screening methodology is a risk-based decision criterion. In this case, a risk-based decision criterion is defined to identify those radionuclides that are below some minimum level of concern. If the initial screening values for a radionuclide for all exposure pathways that conservatively characterize the most exposed groups of individuals are below the predefined risk-based criterion, that radionuclide can be eliminated from further analysis. Likewise, if the initial screening values are below the predefined risk-based criterion for all radionuclides for a given exposure pathway, that pathway may also be removed from subsequent stages of the screening analysis.

Risk-based Decision Criteria

Many radionuclides were discharged into the Columbia River as a result of operations at the Hanford nuclear facility. However, not all the radionuclides pose a significant exposure risk. For example, radionuclides with short half-lives decay rapidly resulting in no or minimal potential exposure. A screening methodology is used to identify and focus resources on the most important radionuclides and pathways. A risk-based decision criterion is applied in the methodology as an initial screening tool to identify those radionuclides and exposure pathways that are below a minimum level of concern.

This section reviews risk-based decision criteria that have been used at other locations for similar projects and by other agencies, and it concludes by recommending a risk-based screening value for this study.

The National Research Council (1995) has suggested a decision criterion of 0.07 Sv for a whole-body lifetime dose for identifying sites where a dose reconstruction may be warranted. This value is based on the Federal Registry 10 CFR 20 maximum annual dose limit of 0.001 Sv to any individual at a nuclear site boundary, multiplied by 70 years to give a whole-body lifetime dose of 0.07 Sv. In terms of risk, this is roughly equivalent to a lifetime excess cancer incidence risk of 5×10^{-3} .

The Oak Ridge Health Agreement Steering Panel, of the Oak Ridge Dose Reconstruction study, established a decision criterion of 10^{-4} lifetime excess cancer incidence risk for the study as a whole (Theissen et al. 1996). For screening releases of radionuclides to the aquatic pathways (Clinch River), a lifetime excess cancer incidence risk criterion of 10^{-5} , which is a factor of 10 lower, was applied (Apostoaiei et al. 1999). The lower value was used because each radionuclide was compared to the decision guide independently for each exposure pathway rather than

combining the exposure risk from all pathways. The calculated screening index was a conservatively biased estimate of excess lifetime risk to the most at-risk individual and was, therefore, expected to overestimate the risk to most or all real individuals (Apostoaeci et al. 1999, page 3-1).

In the HEDR Project, one of the criteria used to define the physical area to be included in the study calculations (study domain) was a thyroid dose of 1 rad (0.01 Gy) to a child or infant (Shleien 1992). This dose represents an increased lifetime risk for radiation-induced thyroid cancer in the order of 2×10^{-4} .

For continuous exposures to ionizing radiation, the National Council on Radiation Protection and Measurements (NCRP) recommends an annual limit for members of the public of 1 mSv effective dose (NCRP 1993). This is the same as the value recommended by the International Commission on Radiological Protection (ICRP) (ICRP 1991). This dose limit corresponds to a lifetime risk of about 4×10^{-3} , assuming the risk per sievert from fatal and nonfatal cancers is 6×10^{-2} (ICRP 1991, Table 3) and a 70-year lifetime exposure. The NCRP also defines an annual negligible individual dose^b (NID), which establishes a boundary below which the dose can be dismissed from consideration and sets the NID at 0.01 mSv effective dose. This corresponds to a lifetime risk of about 4×10^{-5} using the same assumptions as above.

The U.S. Environmental Protection Agency (EPA) has specified an upper bound individual lifetime cancer risk "target range" for carcinogens of 10^{-4} to 10^{-6} , within which EPA strives to manage risks as a part of a Superfund cleanup. The risk estimates are determined using reasonable maximum exposure assumptions for either current or future land use (EPA 1991).

Once a decision has been made to cleanup, EPA has expressed a preference for cleanups achieving the more protective end of the range (i.e., 10^{-6}). However, the upper boundary of the risk range (10^{-4}) is somewhat flexible, although EPA generally uses 10^{-4} in making risk management decisions. The EPA has stated that a specific risk estimate around 10^{-4} may be considered acceptable if justified based on site-specific conditions (EPA 1991). For example, in a Clean Air Act rulemaking establishing National Emissions Standards for Hazardous Air Pollutants (NESHAPs) for U.S. Nuclear Regulatory Commission (NRC) licensees, U.S. Department of Energy facilities, and many other kinds of sites, EPA concluded that a risk level of 3×10^{-4} is essentially equivalent to 1×10^{-4} . EPA explicitly rejected a risk level of 5.7×10^{-4} in the case of elemental phosphorus plants in this rulemaking. EPA has consistently concluded that levels of 15 mrem y^{-1} effective dose equivalent (EDE) (which EPA equates to approximately a 3×10^{-4} increased lifetime cancer risk) or less is protective and achievable (EPA 1997). EPA has explicitly rejected levels above 15 mrem y^{-1} EDE as being not sufficiently protective. For example, the EPA has found the NRC dose limit of 25 mrem y^{-1} (equivalent to approximately 5.7×10^{-4} increased lifetime risk) specified in NRC's Radiological Criteria for License Termination (decommissioning rule) to be beyond the upper bound of the risk range generally considered protective under the Comprehensive Environmental Response, Compensation and Liability Act (EPA 1997).

The EPA approach has been adapted to identify and prioritize potential remediation sites at the Idaho National Engineering and Environmental Laboratory using a target risk level of 10^{-6} . The scenarios evaluated are based on current residential or occupational exposure conditions with exposure durations of 30 and 25 years, respectively. The pathways evaluated are ingestion of

^b Per source or practice.

drinking water, inhalation of contaminated particulates, ingestion of contaminated soil, and external exposure to soils. Each pathway is evaluated independently (Fromm 1996).

Recommendation

For the initial screening a risk-based screening value of 10^{-4} is recommended for use as a decision criterion to identify those radionuclides and exposure pathways for further analysis. The screening values are conservatively biased estimates of risk to the most at-risk individuals and are expected to overestimate the risks to all real individuals. We apply this screening value to demonstrate its application, and its use does not represent endorsement by the CDC.

RADIONUCLIDE RELEASES TO THE COLUMBIA RIVER

Radionuclides were released into the Columbia River primarily in the cooling-water effluent from eight^c once-through-cooled reactors at the Hanford Site (Figure 3). Radionuclides also entered the river along the shoreline as a result of retention basin leakage and by leaks transmitted through the groundwater to the river (Walters et al. 1992). The following section provides a brief overview of reactor operations, which is summarized from two reports produced as part of the HEDR Project (Walters et al. 1992; Heeb and Bates 1994), and discusses the existing radionuclide release estimates and source terms for the current screening methodology.

History of Reactor Operations

Releases of radioactivity to the Columbia River began in September 1944 when the 100-B reactor, located farthest upstream at River Mile (RM) 384 above the mouth of the Columbia River, came online. The 100-D reactor (RM 377.6) began operating in December of the same year, and the 100-F reactor (RM 369) came online in February 1945. The 100-H reactor (RM 372.5) was the fourth reactor to come online in November 1949. In October 1950, 100-DR (RM 377.6) came online, followed by 100-C (RM 383.6) in November 1952. The last of the once-through-cooled reactors, 100-KW (RM 381.8) and 100-KE (RM 381.4), came online in January and April 1955, respectively.

^c A ninth reactor (100-N or N-reactor) did not discharge directly into the Columbia River because it had a different design.

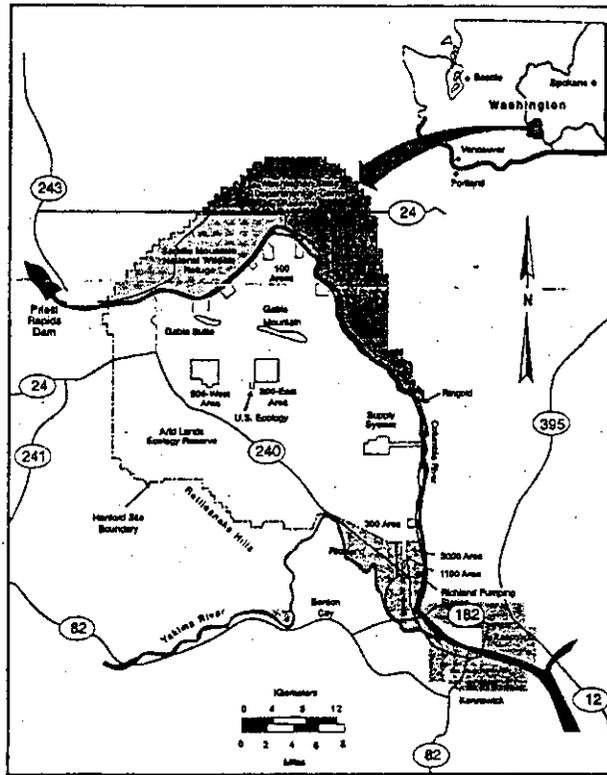


Figure 2. The Hanford reach of the Columbia River (taken from Walters et al. 1992, Plate 2).

Between 1964 and January 1971, all eight reactors were taken offline permanently, starting with 100-DR in December 1964. The 100-H and 100-F reactors were closed in mid-1965. A labor strike caused all the remaining reactors to be closed down temporarily during July and August 1966. In June 1967, 100-D was taken offline, followed by 100-B in 1968, 100-C in April 1969, and 100-KW in February 1970. In January 1971, the last of the once-through-cooled reactors, 100-KE, was shut down permanently.

All eight reactors used raw river water drawn from the Columbia River to cool the reactor fuel elements during operation. Water from the Columbia River was pumped into the water treatment plant. Chemicals were added to adjust the pH, decrease turbidity, and inhibit corrosion of the supply piping and reactor process tubes. The processed river water was filtered, held in clear wells, and pumped into large holding tanks. From the tanks it was pumped through the reactor. The water took 1 to 2 seconds to pass through the reactor core region, during which time it was heated to over 100°C (212°F) in the highest power tubes. The hot effluent water was discharged from the reactor into external retention basins located near the Columbia River, where it was stored temporarily to allow thermal cooling and the shortest-lived radionuclides to decay. The water was discharged to the river via a spillway system to outfall lines. The radionuclide composition and activity level of cooling water discharged to the Columbia River varied considerably as a result of several factors, including

- The number of reactors operating and their power levels
- Seasonal changes in the chemical composition of the raw river water

- Chemicals used in water treatment
- Corrosion rates of reactor piping and fuel element cladding
- Purging of radioactive film from reactor components
- Fuel element failures (ruptures)
- The length of time effluent was retained in basins before discharge to the river.

Radionuclide concentrations and distribution in the Columbia River were also determined by seasonal fluctuations in the hydrologic characteristics, and were greatly impacted by the construction of dams across the Columbia River. For example, McNary Dam (RM 292) did not exist when the first reactors came online during the 1940s. Figure 4 (taken from Walters et al. 1992) shows the river profile as it was in 1944 and the sequence of dam construction from 1953 to 1967.

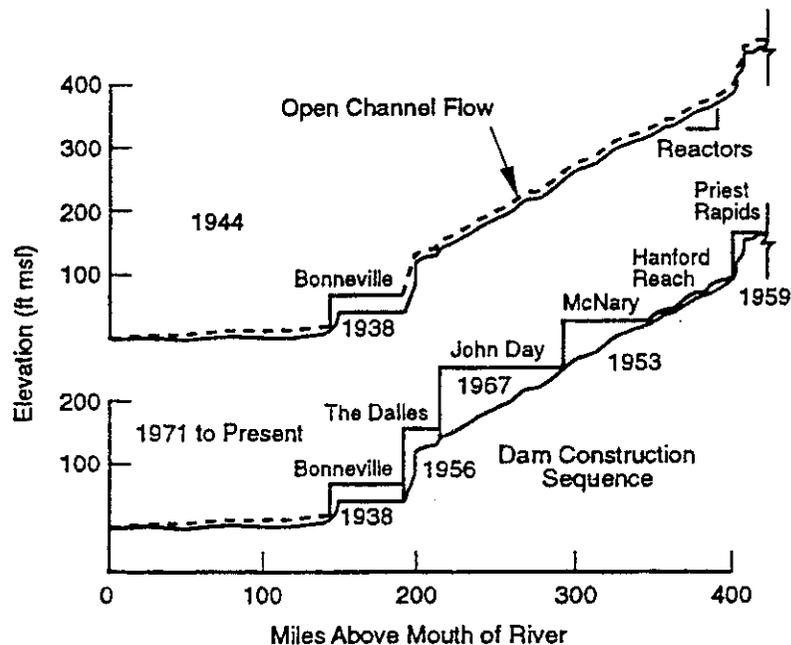


Figure 3. Profile of the Columbia River showing conditions in 1944 as compared to 1971 to present. Sequence of dam and reservoir construction is indicated (taken from Walters et al. 1992, Figure 3.1).

Existing Radionuclide Release Estimates for the Columbia River

Although a large number of different radionuclides were discharged into the Columbia River, most of them had very small inventories and/or very short half-lives (Heeb and Bates 1994). Napier (1993) screened 19 of these radionuclides, and based on the results, the Technical Steering Panel (TSP) of the HEDR Project identified 11 radionuclides (^{24}Na , ^{32}P , ^{46}Sc , ^{51}Cr , ^{56}Mn , ^{65}Zn , ^{72}Ga , ^{76}As , ^{90}Y , ^{131}I , and ^{239}Np) and gross nonvolatile beta activity for further study. Heeb and Bates (1994) went on to estimate distributions of total annual and monthly releases to the Columbia River from the eight single-pass Hanford production reactors for these 11 radionuclides and gross nonvolatile beta activity for the years 1944 through 1971. The release estimate

distributions are based on 100 Monte Carlo simulations for each radionuclide, and the minimum, median, and maximum values are reported. These were the final source term estimates for the HEDR Project and were reconstructed using the Source Term River Release Model (STRRM), where reactor operating history and measurements of radionuclide concentrations (if available) provided the necessary input. Missing data were reconstructed using a statistical analysis of existing data coupled with Monte Carlo modeling techniques. Of the 11 radionuclides for which detailed source term estimates were made, the TSP designated that downriver water and biota concentrations and associated doses be estimated for the five most significant radionuclides (^{24}Na , ^{32}P , ^{65}Zn , ^{76}As , and ^{239}Np) (Napier 1993). Farris et al. (1994) presents the HEDR methodology and dose calculations.

The initial screening of the 19 radionuclides was based on interim monthly release estimates generated by Dr. Maurice Robkin for 1944–1971 (Napier 1993). No attempt was made to adjust for missing data; therefore, gaps existed in the data, especially for the early years. Few data are available before mid-1958. Appendix B provides a summary of HEDR scoping study reports and other reports directly related to this issue.

Starting Point for Screening Methodology

To ensure that the current screening methodology is comprehensive, 23 radionuclides are included in the analysis (Table 1). This list includes

- The 19 radionuclides screened by Napier (1993) in the HEDR Project (^{24}Na , ^{32}P , ^{45}Ca , ^{46}Sc , ^{51}Cr , ^{56}Mn , ^{60}Co , ^{64}Cu , ^{65}Zn , ^{69}Zn , $^{69\text{m}}\text{Zn}$, ^{72}Ga , ^{76}As , ^{89}Sr , ^{90}Sr , ^{90}Y , ^{93}Y , ^{95}Zr and ^{239}Np) where 1961 was identified as the year of maximum releases and, therefore, doses. This provided the basis for screening decisions. However, no release estimates were available for ^{60}Co and ^{95}Zr for that year and as a result these two radionuclides appear to have been forgotten in the HEDR Project calculations and dropped from the analysis.
- The 3 radionuclides identified by Hoffman et al. (1998) (^{60}Co , ^{90}Sr , ^{131}I),
- Three additional radionuclides identified by Hoffman (1999) in a subsequent report in support of a legal case related to the Hanford Site (^{122}Sb , ^{133}I , and ^{137}Cs),
- One radionuclide identified in early scoping studies (Napier 1991; PNL 1991) for the HEDR Project but not in Napier (1993) (^{64}Cu).

Because the same radionuclide may be identified by more than one reference, these numbers do not add up to 23.

The final HEDR Project release estimates that exist for 11 of the 23 radionuclides (Table 1) are used as the source term input for the current screening methodology. In the absence of detailed source term information for the remaining radionuclides, we adopted a simple scaling approach that is based on the monthly median source term estimates provided by Heeb and Bates (1994) and on relative concentration in reactor effluent water reported by Soldat in 1969 (Napier 1991, Appendix E). This approach was used by Hoffman (1999) to estimate releases of ^{60}Co , ^{64}Cu , ^{90}Sr , ^{122}Sb , ^{133}I , and ^{137}Cs to the Columbia River. The source terms for radionuclides that are activation products are estimated as a function of the Heeb and Bates (1994) monthly median source term estimate for ^{32}P , which is an activation product also. This is selected because the ^{32}P source term appears least affected by changes in process (e.g., treatment of effluent water). An uncertainty factor of 5 is applied to these source terms. Source terms for the fission products ^{89}Sr ,

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^{90}Sr , ^{93}Y , ^{95}Zr , and ^{137}Cs are estimated as a function of the Heeb and Bates (1994) monthly median source term estimate for the fission product ^{90}Y . This allows for both fission products from fuel element failures and from fission of natural uranium in river water. Again, an uncertainty factor of 5 (see the "Uncertainty" section later in the report) is applied to these source terms. Source terms for isotopes of the same element estimated by Heeb and Bates (1994) are estimated as a function of that element (e.g., ^{69}Zn , $^{69\text{m}}\text{Zn}$, ^{93}Y , and ^{133}I). A smaller uncertainty factor of 2 is assigned because the same element is being considered in each case.

Table 1. Radionuclide Source Terms for Columbia River Releases

Radionuclide	Source term and data source	Uncertainty factor
^{24}Na	Heeb and Bates (1994)	Distribution ^a
^{32}P	Heeb and Bates (1994)	Distribution
^{45}Ca	$0.05 \times ^{32}\text{P}$ based on Soldat (1969)	5
^{46}Sc	Heeb and Bates (1994)	Distribution
^{51}Cr	Heeb and Bates (1994)	Distribution
^{56}Mn	Heeb and Bates (1994)	Distribution
^{60}Co	$0.02 \times ^{32}\text{P}$ based on Soldat (1969)	5
^{64}Cu	$60 \times ^{32}\text{P}$ based on monitoring data ^b	5
^{65}Zn	Heeb and Bates (1994)	Distribution
$^{69}\text{Zn}^{\text{c}}$	Equal to $^{69\text{m}}\text{Zn}$	2
$^{69\text{m}}\text{Zn}$	$4 \times ^{65}\text{Zn}$	2
^{72}Ga	Heeb and Bates (1994)	Distribution
^{76}As	Heeb and Bates (1994)	Distribution
^{89}Sr	$0.2 \times ^{90}\text{Y}$ based on Soldat (1969)	5
^{90}Sr	$0.01 \times ^{90}\text{Y}$ based on Soldat (1969)	5
^{90}Y	Heeb and Bates (1994)	Distribution
^{93}Y	$2 \times ^{90}\text{Y}$ based on Soldat (1969)	5
^{95}Zr	Equal to ^{90}Y based on Soldat (1969)	5
^{122}Sb	$0.5 \times$ to ^{32}P based on Soldat (1969)	5
^{131}I	Heeb and Bates (1994)	Distribution
^{133}I	$10 \times ^{131}\text{I}$ based on Soldat (1969)	2
^{137}Cs	$0.01 \times ^{90}\text{Y}$ based on Soldat (1969)	5
^{239}Np	Heeb and Bates (1994)	Distribution

^a Based on 100 Monte Carlo simulations (Heeb and Bates 1994).

^b Based on model calibration to water monitoring data at Ringold and Richland. Hoffman (1999) assumed a ratio of 20.

^c Short-lived daughter would have grown to equilibrium activity by the time the water left the holding pond.

ENVIRONMENTAL MONITORING DATA

During the HEDR Project, environmental data were reviewed and compiled for use in the HEDR study. Although environmental data existed for a range of time periods, a few critical years were used in the HEDR Project to calculate preliminary estimates of dose and provide screening estimates for maximally and typically exposed individuals. It is primarily these data that made up the information contained in the HEDR files that were obtained at the outset of this project. Because these data were compiled only in report form, we compiled a descriptive spreadsheet containing the data. This spreadsheet is used to calibrate the river transport model developed for this screening analysis (see the "Environmental Transport Model" section). The spreadsheet and data are described in this section. We compiled environmental monitoring data relevant for assessing radionuclide releases to the Columbia River electronically in a Microsoft Excel[®] Workbook (filename: hanford data.xls). This workbook contains four important types of historical radionuclide measurements: annual average radionuclide concentrations in river water, bi-weekly radionuclide concentrations in river water grab samples, bi-weekly cumulative concentrations in river water, and radionuclide concentrations in sediment. Each dataset is important for different reasons. The measured annual average concentrations in river water allow the river transport model to be calibrated on a macro-scale temporally and show the concentrations of long-lived radionuclides averaged over the course of a year. For the early years, radionuclide-specific measurement techniques are not available to discern the different radionuclides within a river sample. The data for these early years show total beta activity concentrations. The bi-weekly cumulative data reveal another level of detail, and they provide insight into the fluctuations in river water radionuclide concentrations throughout the year. The impact of seasonal variations in the volume and velocity of the river water on radionuclide concentrations can be examined. Because some of the radionuclides released from the reactors have very short half-lives, the bi-weekly grab samples are useful for estimating the transport of these nuclides. Finally, sediment data are important for revealing some information about how the radionuclides in the river sorb onto the sediments. Unfortunately, few sediment data were collected during the period of interest for the study because the Site geared much of the environmental monitoring toward estimating annual doses to potential receptors. Sediment measurements didn't contribute directly to dose estimates; instead, pocket ionization chambers were placed outside to measure the external dose (Walters et al. 1992). This information has limited usefulness in terms of assessing the radionuclide sediment load.

The river upstream of McNary Dam, probably at the location of Ringold, has the highest radionuclide concentrations in river water. McNary Dam was completed in 1953 and is logically the location of highest sediment concentrations because it was the first dam downstream of the Hanford reactors. Environmental data gathered in the river stretch from the last Hanford reactor to McNary Dam are the focus of the environmental data compiled in the workbook.

The first worksheet (name: annual averages) in the Microsoft Excel workbook includes the annual average radionuclide concentration data for river water at different locations downstream of the Hanford reactors. Annual average beta activity concentration measurements at Pasco are included for the years 1945–1971. Radionuclide concentrations were documented for different years at different locations. Annual average concentrations in river water are compiled in this worksheet for locations at the Richland Pumping Station (1963–1989), the Pasco Pumping Station (1959–1965), and the McNary Dam (1964–1969).

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Grab sample measurements of radionuclide concentrations in river water are compiled in the second worksheet (name: grab samples) for the years 1964–1966 at various locations. Grab samples were collected and analyzed for ^{24}Na , ^{32}P , ^{51}Cr , ^{64}Cu , ^{65}Zn , ^{76}As , ^{90}Sr , ^{131}I , ^{72}Ga , ^{56}Mn , $^{69\text{m}}\text{Zn}$, and ^{239}Np . Locations of interest where the data were available include Richland and Ringold.

Cumulative data were collected over 2-week periods during which the sample chamber collected continuously from the river water, and they are compiled in the third worksheet (name: cumulative data). These data represent concentrations of longer lived radionuclides at various locations. Radionuclides collected and analyzed in this manner include ^{32}P , ^{51}Cr , ^{54}Mn , ^{60}Co , ^{65}Zn , ^{90}Sr , and ^{131}I . For some of the radionuclides, only limited data exist. The time period spanned by the compiled data is 1964–1966. Locations between Hanford and McNary Dam where continuous data were collected were the 300 Area, Richland, Pasco, and the upstream side of McNary Dam.

The sediment data are primarily compiled from special studies conducted by other agencies. A number of studies and their results are outlined in the fourth worksheet (name: sediment data). One study was conducted after the reactors were shut down and radionuclide concentrations in surface sediments in April 1971 and August 1976 were measured. Surface sediment concentrations decreased dramatically over this time period, and it was estimated that sediments uncontaminated with radionuclides released from the Hanford sites were being deposited on top of the contaminated sediments at a rate of 38 to 76 cm (15 to 30 in. y^{-1}).

During the early 1960s, the U.S. Geological Survey (USGS) conducted a number of sediment studies to examine the role of sediments in the uptake and transport of radionuclides in the Columbia River. The first study, documented in Nielsen and Perkins (1957), attempted to assess the magnitude of radionuclide uptake by sediments between the reactors and McNary Dam. This study showed the percent loss of different radionuclides between the reactors and Pasco and between Pasco and Vancouver. It also reported a few radionuclide concentrations behind McNary Dam.

In the second USGS sediment study documented here (Nelson et al. 1964), water, suspended sediment, and surficial streambed samples were collected several times per week at different locations. Transport rates calculated for the radionuclides showed that only 30% of ^{51}Cr was lost to sediment, but that ^{65}Zn was almost entirely sorbed by sediment and was resuspended during periods of high river flow. This study documented the radionuclides associated with aqueous phase and sediments, as well as concluding that 75% of the depletion of radionuclides by sediments occurred behind McNary Dam. Total inventories of radioactive material in sediments were estimated.

Concentrations of Radionuclides in Fish

Consumption of fish was identified as the dominant exposure pathway in the HEDR assessment of Columbia River doses (Farris et al. 1994). Measured concentrations of radionuclides in fish are not compiled in electronic format because they are not used for the model calibration or testing. Radionuclides accumulate in fish to varying degrees depending on the species of fish, and not all river users consume the same types of fish. Therefore, it is important to identify the uptake characteristics of various fish categories, the different groups that consumed fish from the Columbia River, and the types and quantities of fish that were consumed

(see the "Exposure Pathways" section). The following paragraphs present a general overview of data regarding radionuclide concentration measurements in fish.

Some of the resident fish in the Columbia River are primarily bottom feeders (e.g., white sturgeon) that reside year round in the Hanford reach. These fish take up radionuclides by directly ingesting contaminated sediments and also via the aquatic foodchain. Watson et al. (1970) reports concentrations measured in 1966 and 1967.

Other resident fish include species such as whitefish, small mouth bass, crappie, channel catfish, walleye, and yellow perch. Large populations of rough fish also live in the Hanford reach, including carp, shiner, sucker, and squawfish. Concentrations of ^{32}P in whitefish collected at Ringold appear to peak in May or June and again in August, September, or October. The peak concentration of phytoplankton and periphyton (benthic microflora) is observed in April and May, with a secondary peak in late summer/early autumn. The spring pulse is probably related to increasing light and water temperature rather than to nutrient availability. Zooplankton population densities are lowest in the winter and highest in summer. Whitefish exhibited the highest radionuclide concentrations of the sport fish. However, of all the resident species, suckers had the highest concentrations but were rarely eaten. Native Americans have reported eating suckers and using all parts of the fish.

Anadromous fish use the Columbia River as a migration route. Species of this type of fish include Chinook salmon, sockeye salmon, coho salmon, and steelhead trout. They are carnivorous fish that actively feed on juveniles in the river and as they mature in the ocean, but they do not feed during the spawning migration. Because of this lack of feeding during the time that the fish spend in the Columbia River, it has traditionally been more difficult to predict concentrations of radionuclides in anadromous fish.

ENVIRONMENTAL TRANSPORT MODEL

This section presents the equations used to calculate concentrations of radionuclides in river water and sediment. First, a conceptual model is developed. The conceptual model is then translated into the mathematical model. Key assumptions, processes, and parameter values are discussed.

Conceptual Model

The screening approach used in this analysis is designed to calculate river water and sediment concentrations from the downstream Hanford Site boundary to the McNary Dam below the confluence of the Snake River and couple this to conservative assumptions for receptor exposure pathways to estimate screening risk values. To achieve this objective, the river transport model is calibrated to measured river water and sediment concentrations. The Columbia River domain of interest extends from the Hanford Site boundary near the Vernita Bridge (RM 385) to its confluence with the Snake River (around RM 325). Along a 20-mile stretch of the river from RM 385 to RM 365, eight reactors released radionuclides into the Columbia River along its southern bank. Observation of the contaminant plume in the Columbia River indicated that complete horizontal mixing occurred near Pasco (RM 327). Downstream from the Snake River confluence, water is backed up as a result of the McNary Dam, which was completed in 1953.

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Sediment deposition behind the Dam is an important sink for radionuclides sorbed onto sediments.

For model calculations we consider radionuclides in both the dissolved and sorbed phase. For dissolved phase radionuclides, dispersion in the longitudinal (parallel to river flow) and transverse (perpendicular to river flow) is considered because complete mixing cannot be assumed in the river sections. River flow is assumed to be a steady state (within the time step of the calculation), with no sources or sinks above the Yakima River confluence. Partitioning between the sorbed and dissolved phases is described by the linear sorption isotherm or sorption coefficient (K_d) assumed to be a function of the river water concentration. In these areas, sediments are continuously scoured, deposited, and remobilized, resulting in sediment concentrations that reflect current concentrations in the river water. In areas where sediments are accumulating, such as behind the McNary Dam, radionuclides sorb to sediment deposit from the water column as the velocity of the water slows. The radionuclides sorbed onto sediment are considered fixed and do not repartition into the water, forming a sink of radionuclides in sediment. In reality, some redistribution occurs, but repartitioning is mainly a function of the aqueous phase concentration in the sediment pore water and not the river water concentration.

The model for transport in the river from Hanford Site boundary to the confluence with the Yakima River is described by a 2-dimensional advection-dispersion model for transport of contaminants in the dissolved phase and sorbed suspended sediment transport. The same model is used for receptors downstream of the Yakima River confluence; however, concentrations are modified by a dilution factor that accounts for the additional dilution from the inflow of the Snake and Yakima Rivers. Behind McNary Dam and at other points in the model domain, a first order kinetic model is used to predict concentrations in deposited sediments.

The river transport model (Figure 4) assumes a constant river channel width (W) and depth (D). A Cartesian coordinate system is defined having its origin at the near-shore point of discharge into the river. Releases are described by a vertically averaged point source at a point defined by the coordinates $0, y_0$. The distance x is the distance from the source to the receptor.

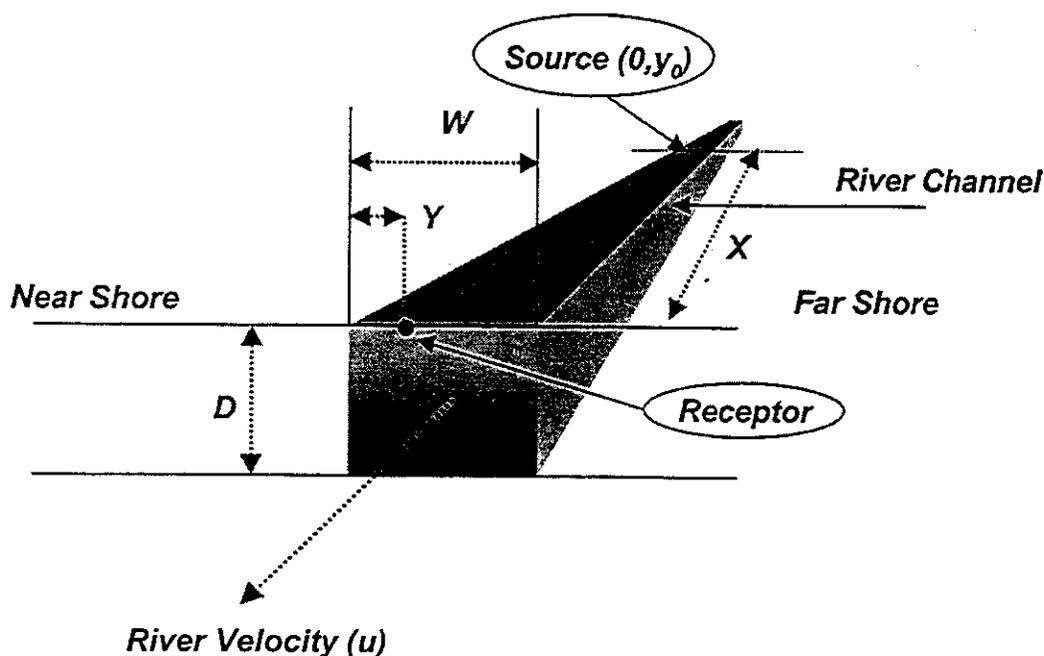


Figure 4. Conceptual representation of river transport model for screening radionuclides released to the Columbia River from the Hanford Site.

Mathematical Model

The mathematical model is based on the work of Codell et al. (1982) and Jirka et al. (1983), which developed semi-analytical solutions for the advection-dispersion equation in river channels. The mass-balance equation for a vertically-averaged radionuclide concentration in a uniform flow field may be written as (Codell et al. 1982)

$$\frac{\partial C}{\partial t} = E_x \frac{\partial^2 C}{\partial x^2} + E_y \frac{\partial^2 C}{\partial y^2} - u \frac{\partial C}{\partial x} - \lambda C \quad (1)$$

where

- C = radionuclide concentration (aqueous and sorbed) in the river water (Ci m^{-3})
- E_x = longitudinal turbulent dispersion coefficient ($\text{m}^2 \text{s}^{-1}$)
- E_y = transverse turbulent dispersion coefficient ($\text{m}^2 \text{s}^{-1}$)
- u = river flow velocity (m s^{-1})
- λ = decay rate constant (s^{-1}).

The initial and boundary conditions are given by

- $C = 0$, at $t = 0$
- $C = 0$ at $x = \pm \infty$
- $\partial C / \partial y = 0$, at $y = 0, y = W$ (width of river channel).

Assuming a straight rectangular channel of width W , cross sectional area A , and steady-state velocity u , the solution to Equation (1) for the concentration at a point x, y downstream resulting from an instantaneous unit release at the point, $(0, y_0)$ is given by (Codell et al. 1982)

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$$C = \frac{1}{\sqrt{4\pi E_x t DW}} \exp\left[-\frac{(x-ut)^2}{4E_x t} - \lambda t\right] \left[1 + 2 \sum_{n=1}^{\infty} \exp\left(-\frac{n^2 \pi^2 E_y t}{W^2}\right) \cos\left(n\pi \frac{y_s}{W}\right) \cos\left(n\pi \frac{y}{W}\right)\right] \quad (2)$$

where

- D = effective river depth (m)
- W = effective river width (m)
- x = downstream distance from source (m)
- y = transverse distance from near-field shoreline (m)
- y_s = transverse distance of source from near-field shoreline (m).

Equation (2) gives the concentration for an instantaneous release at time = 0. The more generalized solution for an arbitrary release occurring over time is given by the convolution integral.

$$C_a(x, y, t) = \int_0^t C(x, y, t-\tau) f(\tau) d\tau \quad (3)$$

where

- $C(t-\tau)$ = instantaneous concentration at time $(t-\tau)$ for a release at time $(t-\tau) = 0$
- $f(\tau)$ = source release rate (Ci s^{-1}).

Releases to the Columbia River occurred not from a single point, but eight reactors separated by a distance of ~26 km (~16 mi). Concentrations at points downstream are then a function of the sum of the contributions from each individual reactor and can be solved using methods of superposition as given in Equation (4).

$$C_a(x, y, t) = \sum_{i=1}^n \int_0^t C_i(x-x_o, y, t-\tau) f(\tau)_i d\tau \quad (4)$$

where

- C_i = concentration from the i^{th} reactor source
- n = number of reactors
- x_o = distance downstream from a central frame of reference of reactor source i (m)
- $f(\tau)_i$ = source release rate (Ci s^{-1}).

The central frame of reference was defined as RM 385, which is upstream of all the reactors.

Treatment of Nonsteady Flow and Changing River Dimensions

The mathematical model presented in Equations (1-4) assumes the river flow rate is at steady state. However, water flow in the Columbia River varies seasonally (Figure 5), changing the extent of radionuclide dilution and downstream travel times. Travel times from the 100-D Reactor (RM 377.6) to the Pasco pumping station (RM 330) as a function of river discharge were estimated by Soldat (1962) and reported in Walters et al. (1992). Travel times ranged from 0.43 to 1 day (Table 2). Because the travel times are short compared to the time resolution of the release history (1 month), steady-state conditions within the model domain (Hanford Site

boundary to the McNary Dam) would be achieved within about 1 day. Therefore, seasonal fluctuations in the flow rate can be incorporated into the model much the same way as variable wind vectors are incorporated into the Gaussian Plume air dispersion model. For any given day of simulation, the monthly-average flow rate is used to calculate u , W , and D . Mean river velocity as a function of river flow rate is calculated by dividing the distance from 100-D reactor to the Pasco pumping station (76 km [47 mi]) by the travel times reported in Soldat (1962). These data are then regressed to give estimates of u for other flow rates. A power function is fit to these data ($r^2 = 0.99$) given by

$$u = 0.028158F^{0.456338} \quad (5)$$

where F = the flow rate ($\text{m}^3 \text{s}^{-1}$).

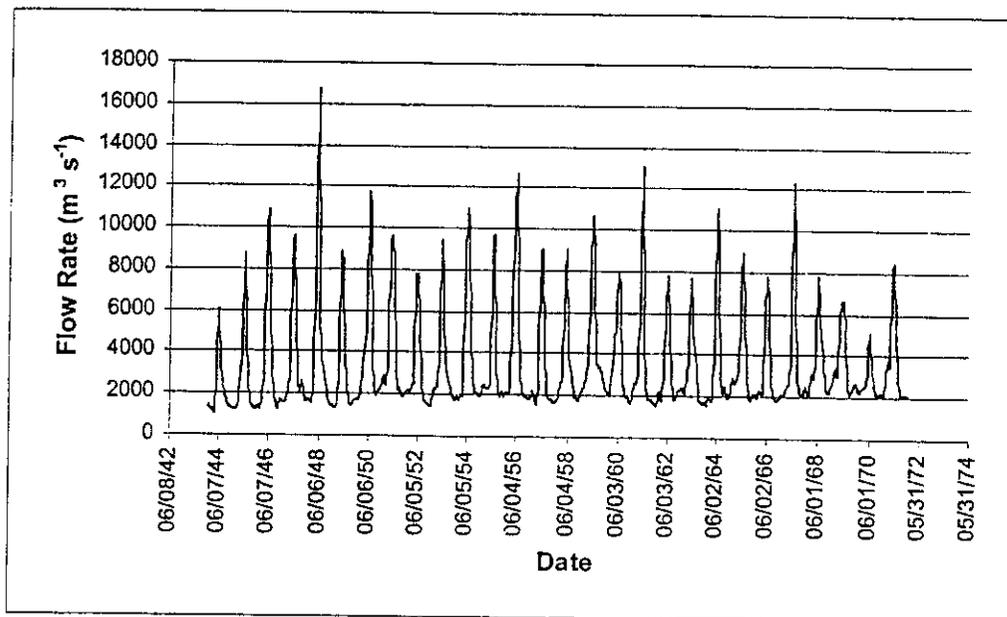


Figure 5. Monthly average flow rate in the Columbia River near the Hanford Site.

The width and depth of the river are calculated using an equation for depth as a function of flow rate developed by Leopold et al. (1964). Recall that the river width and depth are held spatially constant in the model domain but may change with season. The river depth as a function of the flow rate is estimated by

$$D = 10 F^{0.46} \quad (6)$$

Using the relationship $F = D \times W \times u$, the effective width can be calculated for any flow rate using Equation (7).

$$W = \frac{F}{u D} \quad (7)$$

Table 2. Flow Rate and Travel Times between 100-D Reactor and Pasco Pumping Station^a

Flow rate (m ³ s ⁻¹) ^b	Travel time (days)	<i>u</i> (m s ⁻¹) ^c
1840	1	0.879
4899	0.67	1.31
8750	0.48	1.83
12232	0.43	2.05

^aData from Soldat (1962) as reported in Walters et al. (1992)

^bUnits have been changed from cubic feet per second to cubic meters per second

^c*u* = 76 km (47 mi)/travel time

Dispersion Coefficients

Contaminant dispersion is a function of differential shear flow and cross-sectional turbulent mixing. Longitudinal dispersion coefficients may be calculated using equations developed by Fischer et al. (1979) as reported in Till and Meyer (1983).

$$E_x = \frac{0.011 u^2 W^2}{D u_*}, E_y = \beta_y u_* D \quad (8)$$

where

*u*_{*} = shear velocity (m s⁻¹)

β_y = unitless coefficient having a value of 0.6 ± 0.3.

The shear velocity is estimated by

$$u_* = \sqrt{g D s} \quad (9)$$

where

g = gravitational acceleration (9.8 m s⁻²)

s = channel slope (m m⁻¹).

The model assumes a constant channel width, depth, velocity, and slope over the release period. Channel width, depth, and velocity are calculated based on the monthly average flow rates in Figure 5. The effective channel slope of 2.55 × 10⁻⁴ between RM 390 and RM 320 is estimated from Figure 3.1 in Walters et al. (1992). Equations (8) and (9) produce *E*_x and *E*_y values on the order of 3000–6000 m² s⁻¹ (32,300–64,600 ft² s⁻¹) and 0.1–1.0 m² s⁻¹ (1.1–10.8 ft² s⁻¹),

respectively, for Columbia River flows within the model domain and using a β value of 0.3. The HEDR model (Walters et al 1994) used substantially smaller longitudinal dispersivity values, on the order of $46 \text{ m}^2 \text{ s}^{-1}$ ($500 \text{ ft}^2 \text{ s}^{-1}$) to $232 \text{ m}^2 \text{ s}^{-1}$ ($2500 \text{ ft}^2 \text{ s}^{-1}$), and concluded that the predicted river water concentrations were not very sensitive to this parameter. Except for short-lived isotopes such as ^{56}Mn (half-life = 0.107 d) average concentrations are insensitive to longitudinal dispersivity. Concentrations are shown, however, to be sensitive to the transverse dispersivity depending on the downstream distance from the reactors. A transverse dispersivity value of $\sim 0.6 \text{ m}^2 \text{ s}^{-1}$ ($\sim 6.5 \text{ ft}^2 \text{ s}^{-1}$) essentially provides complete mixing across the width of the river at all locations (Figure 6). We use both the transverse and longitudinal dispersivity as model calibration parameters. A value of $50 \text{ m}^2 \text{ s}^{-1}$ ($538 \text{ ft}^2 \text{ s}^{-1}$) is selected for the longitudinal dispersivity and the transverse dispersivity uses a value between 0.1 and $0.6 \text{ m}^2 \text{ s}^{-1}$ (1.1 and $6.5 \text{ ft}^2 \text{ s}^{-1}$) depending on the nuclide. The calibration is discussed further in the "Model Parameters and Calibration" section.

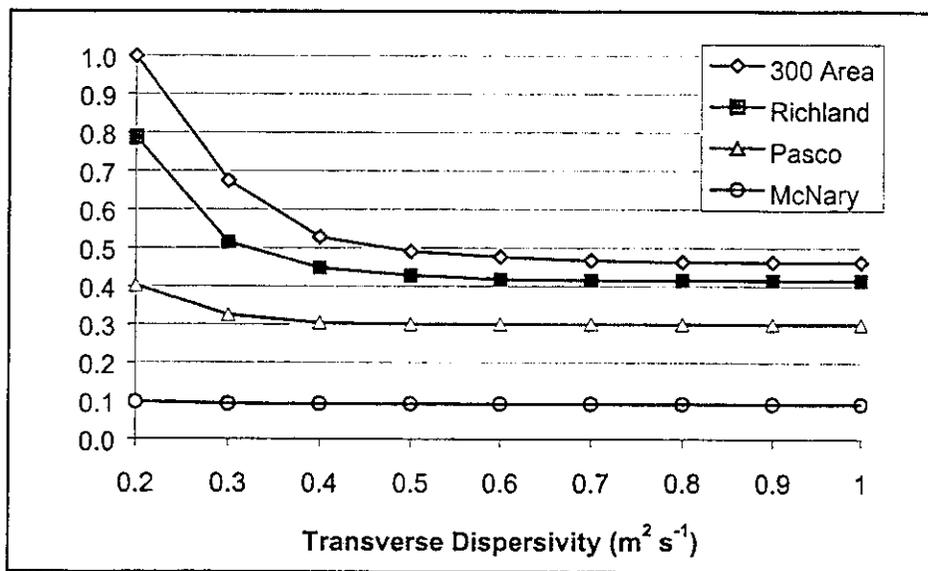


Figure 6. Sensitivity of river water concentration to the transverse dispersivity at selected downstream distances. Concentrations were evaluated at a point 50 m (55 yd) from the near shore. Concentrations were normalized to the concentration at the 300 Area for a transverse dispersivity of $0.2 \text{ m}^2 \text{ s}^{-1}$ ($2.2 \text{ ft}^2 \text{ s}^{-1}$).

Source Term

Details related to developing the radionuclide release estimates (also called the source term) are discussed in the "Radionuclide Releases to the Columbia River" section. This section describes how the source term is implemented into the transport model. Monthly release quantities to the Columbia River for a subset of the radionuclides examined from *all* reactors are provided in Heeb and Bates (1994) and compiled in a Microsoft Excel spreadsheet. Unfortunately, release quantities were not segregated by reactor. For the HEDR study, this was not a major issue because the model domain extended $\sim 480 \text{ km}$ ($\sim 300 \text{ mi}$). However, the screening model domain is considerably smaller ($\sim 105 \text{ km}$ [$\sim 65 \text{ miles}$]) and the distance

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separating each reactor can be significant. Therefore, it is necessary to segregate release quantities by reactor. Releases are apportioned to each reactor based on the monthly power production rate reported in Appendix A of Heeb and Bates (1994). We assume that the quantity of radioactivity released to the Columbia River is proportional to the power production from the reactors. The validity of this assumption is illustrated in Figure 7, which shows the monthly gross beta activity plotted against the monthly power production for all reactors. The method only approximates the monthly release from each reactor and does not account for events such as fuel-element failure. The fraction of total activity released from each reactor by month is equal to the monthly power production for a given reactor divided by the total power production from all reactors. This fraction is then multiplied by the total activity for a given nuclide released for the month to calculate the nuclide-specific activity released for the given reactor.

For some nuclides, source terms are not available in Heeb and Bates (1994) but are instead approximated by the ratios between fission-activation product production rates in reactors. Nuclides for which these approximations are made include ^{45}Ca , ^{60}Co , ^{64}Cu , $^{69\text{m}}\text{Zn}$, ^{89}Sr , ^{93}Y , ^{95}Zr , ^{122}Sb , ^{133}I , and ^{137}Cs .

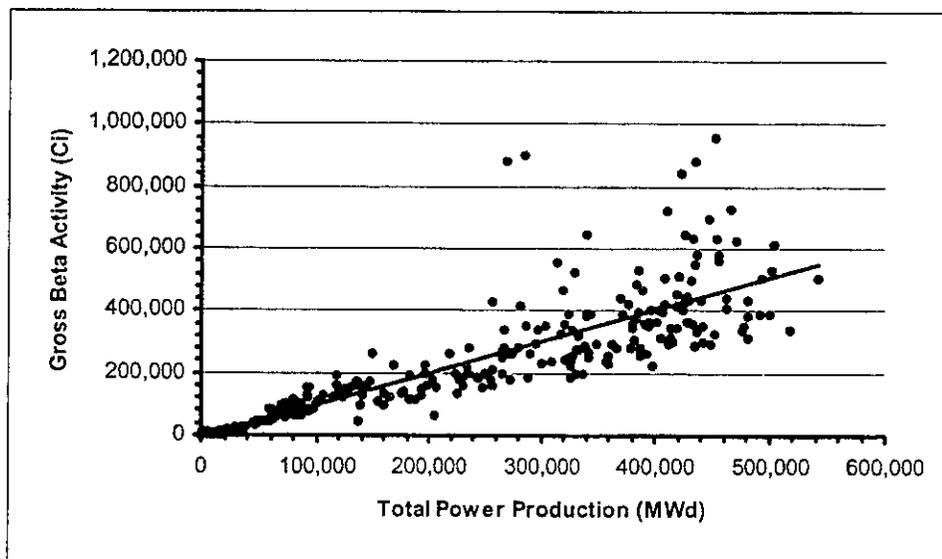


Figure 7. Gross monthly beta activity versus total monthly power production from all eight reactors. The regression coefficient (r^2) is 0.76.

Sediment Effects

A criticism of the HEDR Project was the exclusion of contaminated sediments as a pathways of exposure (Hoffman et al. 1998). To address this concern, sediment effects are incorporated into our screening model. Partitioning of radionuclides in the aqueous and sorbed phases is described by the linear sorption coefficient and the sediment load in the river water. Derivation of the concentration in each phase begins with the mass balance equation for radionuclides in aqueous and sorbed phase equilibrium.

$$Q_w = C_w V_w + C_s S_e V_w \quad (10)$$

where

- Q_w = total radionuclide inventory in a unit volume of water (Ci)
 C_w = radionuclide concentration in aqueous phase (Ci m⁻³)
 C_s = radionuclide concentration in sediment (Ci g⁻¹)
 V_w = unit volume of water (m³)
 Sc = sediment load in river water (g m⁻³).

The radionuclide concentration on sediments can be described by

$$C_s = C_w K_d \quad (11)$$

where

K_d = linear sorption coefficient (m³ g⁻¹).

Substitution of Equation (11) into Equation (10) and rearrangement gives

$$C_w = \frac{Q_w}{V_w(1 + K_d Sc)} \quad (12)$$

The sediment load (Sc) has two components: the sediments suspended in river water and the fixed sediment bed that is in contact with river water. Therefore, Sc can be written as

$$Sc = Ss + \rho \frac{Tb}{D} \quad (13)$$

where

- Ss = suspended sediment load (g m⁻³)
 ρ = bulk density of sediment bed (g m⁻³)
 Tb = thickness of fixed sediment bed (m)
 D = depth of river (m).

The quantity Q_w/V_w is the total concentration in sediment and water calculated with Equation (4). We ignore the water held in the pore spaces of the bed sediments. The aqueous phase concentration is calculated using Equation (12) and the sediment concentration is calculated using Equation (11).

The USGS monitors suspended sediment loads in rivers at numerous locations around the country and posts these data on their Web page (<http://webserver.cr.usgs.gov/sediment/>). Suspended sediment was only monitored at Vancouver, Washington which is many miles downstream from Richland, below the Bonneville Dam. Sediment loads in the Yakima River were also available at a monitoring station about 8 km (5 mi) west of Richland near Kiona, Washington. Average sediment loads for the Yakima River from 1977 to 1980 were about 60 mg L⁻¹. Most of the load (~90%) was comprised of fine sand and silt (0.062–0.0039 mm). Sediment loads in the Columbia River near Vancouver from 1963 to 1969 averaged 34 mg L⁻¹. No data

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were found for the Snake River. Suspended sediment loads in the Columbia River near Hanford were discussed with Pacific Northwest Laboratories hydrologist, Marshal Richmond.^d He suggested a value of around 10 mg L^{-1} for the Columbia River above its confluence with the Yakima and Snake Rivers. The Snake River is believed to substantially increase the sediment load in the Columbia River. In our model, we use a value of 10 mg L^{-1} for locations above and 35 mg L^{-1} for locations below the confluence with the Snake River.

The thickness of fixed bed sediments that interact with the river water was initially based on the work of Onishi et al. (1982), which modeled sediment and radionuclide transport for two rivers in New York State. They used a value of 5.99 cm (2.36 in.) in the SERATRA code to represent the thickness of the top layer of cohesive sediments.^e Marshall Richmond^d suggested a value of several grain diameters of the fixed bed sediments. The bed sediment thickness would be in the range of 1 to 4 mm assuming the bed sediment is comprised of coarse to very coarse sand ($\frac{1}{2}$ to 2 mm). A value of 1 mm is used in the model based on model calibration with measured ⁶⁵Zn aqueous phase and sediment concentrations. Model calibration is discussed in a later section ("Model Parameters and Calibration").

Many other complex physical processes not included in this model occur during sorbed phase transport, including deposition and suspension of sediment, temporal and spatial fluctuation of both the quantity and characteristics of the sediment load, and changes in water chemistry that affect the sorption process. Models that incorporate such processes (Onishi et al. 1982) often require calibration to detailed site-specific sediment data that are not historically available for the Columbia River. The net effect of sorption on the fixed sediment bed is to reduce aqueous-phase river water concentrations and provide a source term for shoreline exposure. The sorption coefficient is used as a calibration parameter to match predicted radionuclide concentrations in river water to corresponding measured values. A separate sediment submodel discussed in the next section is used to estimate radionuclide inventory and concentrations in deposited sediments.

Sediment Submodel

The treatment of sediment effects discussed in previous sections does not account for radionuclides that are sorbed onto suspended sediments that are later deposited and either covered by clean sediments or remain exposed. In either case, desorption from the sediments back into river water is restricted because sediments are no longer in contact with river water. To address this potential pathway, a separate sediment submodel is developed. This submodel is limited in that the activity that is removed from the system through sediment deposition is not subtracted from the activity in the river system. Therefore, mass balance is not achieved. The submodel is an adaptation of the shoreline exposure models described in Soldat et al. (1974) and Strenge et al. (1986) and implemented in NCRP (1996). The model is described by a first-order compartment model where activity sorbed to suspended sediments accumulates and radioactive decay is the only loss mechanism considered.

^d Personal communication with Marshal C. Richmond, Pacific Northwest Laboratories, August 24, 2000.

^e The value of 5.99 cm was taken from the SERATRA output file on page C.3

$$\frac{dQ_s}{dt} = v_d C_w K_d S_s - \lambda Q_s \quad (14)$$

where

Q_s = radionuclide inventory in deposited suspended sediment (Ci m^{-2})

v_d = deposition velocity in river water (m d^{-1}).

The radionuclide concentration in water (C_w) varies as a function of time and, therefore, Equation (14) must be solved numerically. The deposition velocity is assigned a value of 0.07 m d^{-1} based on the work of Soldat et al. (1974), which used sediment and water concentrations in the Columbia River between Richland, Washington and Tillamook Bay, Oregon, to estimate deposition of contaminated sediments. The quantity Q_s is used to calculate the risk from external exposure to shoreline sediments, inadvertent ingestion of sediment, and dermal contact.

Computation Details

Equations (1–14) are coded into a FORTRAN program (RVRDSP) that

- Reads model inputs and performs initial unit conversions
- Computes the convolution integral (Equation 4) and source superposition
- Calculates sediment effects and solves Equation (14)
- Writes output to ASCII files.

Input file formats and user instructions are presented in Appendix D. The convolution integral is solved using Simpson Rule integration as described in Press et al. (1992). To reduce computational time, terms that add little to integrand are removed from the computation. This is accomplished by calculating integration limits, t_1 and t_2 , provided in Codell et al. (1982)

$$t_{1,2} = t - 0.5 \left[\frac{2xu + 4E_x\gamma}{u^2 + 4E_x\gamma} \pm \sqrt{\left(\frac{2xu + 4E_x\gamma}{u^2 + 4E_x\gamma} \right)^2 - \frac{4x^2}{u^2 + 4E_x\lambda}} \right] \quad (15)$$

where γ is an arbitrary number chosen to be 50. The infinite series calculation is performed for 20 terms or until no significant change in the returned value is achieved. Equation (14) is solved using a 4th order Runge-Kutta solver described in Press et al. (1992).

Model Verification

Model verification is defined here as confirmation that the model has been coded and implemented in the computer code correctly. To do this, we compare model results of a sample problem with the results from another code that employs the same model. The computer code RIVLAK, as described in Codell et al. (1982), provides such a model and sample problem. The RIVLAK code contains the primary elements of RVRDSP; however, it lacks treatment of sediment effects, superposition of sources, and is cumbersome to use. Table 3 shows the results of the sample problem described on page 2.56 and Figure 2.17 of Codell et al. (1982). The river in the sample problem has the following properties

- $D = 25$ ft (7.62 m)
- $W = 500$ ft (152 m)
- $u = 1$ ft s⁻¹ (0.3048 m s⁻¹)
- $t_{1/2} = 5000$ sec
- $E_x = 11.5$ ft² s⁻¹ (1.07 m² s⁻¹)
- $E_y = 0.45$ ft² s⁻¹ (0.0418 m² s⁻¹)
- $y_s = 0$
- $x = 7000$ ft (2133 m)
- $y = 0$.

The source is represented by a linearized source table (Table 2.3 in Codell et al. 1982), representing a total of 39,217 Ci released over 13 s. Table 3 compares the two models. While there are differences between the two models, they are considered minor.

Table 3. Comparison of RIVLAK and RVRDSP for the Sample Problem in Codell et al. (1982)

Time (days)	RIVLAK (Ci m ⁻³)	RVRDSP (Ci m ⁻³)	Percent difference
0.05787	9.76×10^{-12}	1.01×10^{-11}	3.3
0.08102	2.09×10^{-4}	2.12×10^{-4}	1.19
0.11574	3.26×10^{-13}	3.23×10^{-13}	-0.79
0.13889	1.41×10^{-24}	1.38×10^{-24}	-1.86

Model Parameters and Calibration

Model calibration is defined in this report as the process of parameter adjustment to match model predictions with observations. The sorption coefficient (K_d), fixed bed sediment thickness (Tb), and dispersivity (E_x and E_y) are treated as calibration parameters. The parameters Tb , E_x , and E_y are nuclide independent parameters and are expected to be the same for all nuclides. However, it was difficult to calibrate the model to measured concentrations for some of the short-lived nuclides without making some adjustments to the transverse dispersivity.

The calibration procedure involves first calibrating the fixed bed sediment thickness for nuclides where both sediment and water concentrations are available. Then, using this thickness,

the K_d and E_y values are calibrated to the measured concentration for the remainder of the radionuclides.

The term y_s represents the release point of reactor effluent in the Columbia River as measured from the near shore. All reactors were situated on the near shore. For most reactors, effluent was first discharged to retention ponds to allow thermal cooling and decay some of the short-lived fission products. The effluent was then discharged via gravity flow through a 42 to 60-in. pipe that extended into the center of the river channel, about 100 to 200 m (109 to 219 yd) from the reactor shoreline (Walters et al. 1992). However, during seasons of high river flows, problems developed in this discharge system because of the hydraulic head differential between the discharge basin and the water-surface level. In cases of high river flow, reactor effluent was discharged to the river at the shoreline. Two exceptions were noted for this discharge system: the 100-F reactor discharge pipe only extended ~50 m (55 yd) from the shoreline, and, at the 100-D and 100-DR reactors, the river channel is divided by an island and effluent was discharged over the island and into the far channel. Table 4 presents the modeled distances (y_s) from the near shore for reactor effluent. Shoreline distances are kept at their minimum estimated value to account for the times of high flow where effluent was discharged to the shoreline.

Radionuclide concentrations are evaluated at downstream sampling locations that included the 300 Area, Ringold, Richland and Pasco Pumping Stations, and McNary Dam (Table 5). The downstream distance could be estimated from Plate 2 in Walters et al. (1992) and from Walters et al. (1994). However, the distance from the shoreline where samples were taken was not reported. Assuming samples were taken from pump water and the intakes for these pumps were located within the channel, we assume a near shore distance of 50 m (55 yd) for all sample locations except those taken at Ringold, where samples were taken from the far shore. For the Ringold location, concentrations are evaluated at 400 m (437 yd) from the near shore.

Table 4. Distance from Near Shore that Reactor Effluent Was Discharged

Reactor	Discharge distance from near shore (y_s , meters)
100-B	100
100-C	100
100-KW	100
100-KE	100
100-N	100
100-D	300
100-DR	300
100-H	100
100-F	50

Table 5. Distances to Downstream Sampling Locations

Downstream location	Downstream distance ^a (m)	Distance from near shore (m)
Ringold	48899	400
300 Area	64376	50
Richland	72420	50
Pasco	93341	50
McNary Dam	148864	50

^a Distance downstream measured from RM 385

Calibration of Bed Sediment Thickness

The bed sediment thickness (T_k) and the K_d are both parameters that affect the water and sediment phase concentration. Generally higher T_k and K_d values yield lower water phase concentrations, and higher K_d values yield higher sediment phase concentrations. However, as shown in Figure 8, the overall sensitivity of one parameter depends of the value of the other. In general, low T_k values result in a lower sensitivity of the K_d . Determination of the bed sediment thickness required estimates of the K_d value along with measured concentrations in sediment and water.

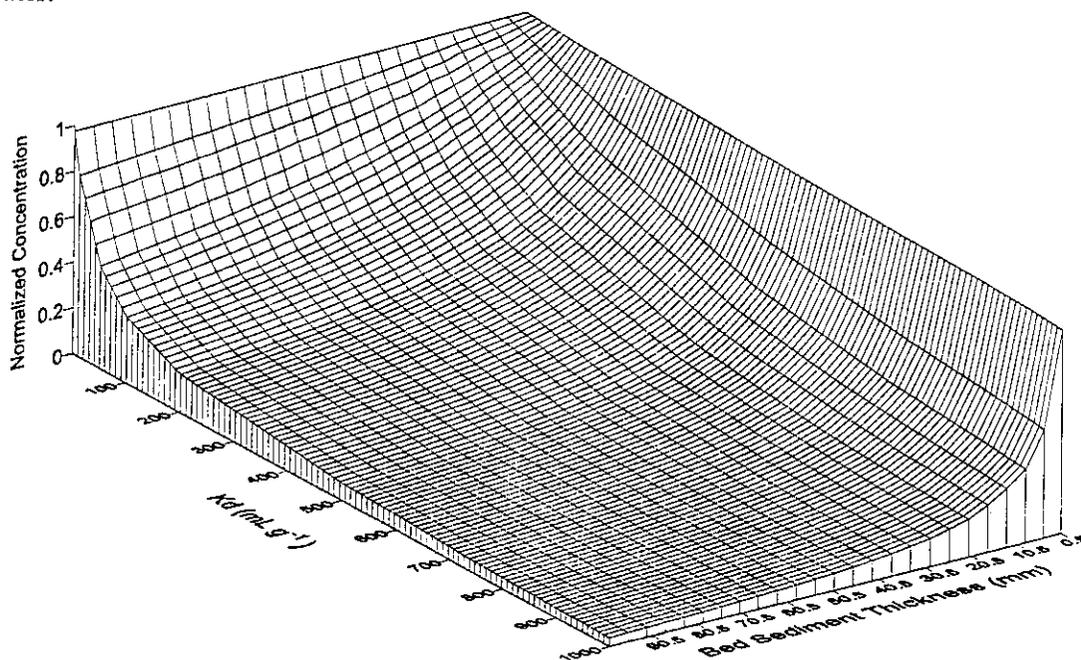


Figure 8. Sensitivity of the water phase concentration to the linear sorption coefficient (K_d) and the bed sediment thickness (T_k). Concentrations have been normalized to the concentration using a K_d of zero.

Although it is reported that numerous radionuclide measurements in sediment were made (Walters et al. 1992), only a few measurements were found in the literature search. Measurements that were obtained were taken behind the McNary Dam in 1957 and 1971. Corresponding water concentrations at McNary Dam were limited to ^{65}Zn and ^{51}Cr for 1964 through 1969. Consequently, there are no concurrent measurements of sediment and water phase concentrations with which to estimate a K_d value using its fundamental relationship

$$K_d = \frac{C_s}{C_w} \quad (16)$$

where C_s is the concentration in sediments (pCi g^{-1}) and C_w is the concentration in water (pCi mL^{-1}). Because the annual average water concentration at McNary dam was not measured, the ^{65}Zn concentration is estimated by multiplying the annual average water concentration in 1964 (0.077 pCi mL^{-1}) by the ratio of the median estimated release of ^{65}Zn in 1957 (27,560 Ci) to the median estimated release in 1964 (15,710 Ci). This calculation yields an estimated annual average water concentration of 0.135 pCi mL^{-1} for 1957. The estimated K_d for zinc calculated using Equation (16) is $\sim 2600 \text{ mL g}^{-1}$. This same procedure is applied to ^{51}Cr , yielding an estimated K_d value of 35 mL g^{-1} . With an estimate of the K_d value, the predicted concentrations in sediment and water can be calibrated to the corresponding measured data using the effective bed sediment thickness (T_k) as a calibration parameter. The USGS inverse modeling code, UCODE (Poeter and Hill 1998) assists in this task. UCODE determines optimal parameter values using nonlinear regression techniques. Both the K_d and T_k are used in the regression.

Calibration results from UCODE are not used verbatim (Table 6). Other considerations, including measured concentrations at other locations and the literature range of K_d values, are also considered in defining the final parameter values. In addition, we want to set the bed sediment thickness constant for every nuclide. After taking these factors into consideration, the calibrated bed sediment thickness is 0.001 m, and the ^{65}Zn and ^{51}Cr K_d values are 2400 and 30 mL g^{-1} , respectively.

Table 6. Calibration Results from UCODE and Final Parameter Values Used in the Model

Parameter	UCODE-calibrated value	Final value
T_k for ^{65}Zn	$9.3 \times 10^{-4} \text{ m}$	$1.0 \times 10^{-3} \text{ m}$
K_d for ^{65}Zn	2810 mL g^{-1}	2400 mL g^{-1}
T_k for ^{51}Cr	$1.0 \times 10^{-3} \text{ m}$	$1.0 \times 10^{-3} \text{ m}$
K_d for ^{51}Cr	30 mL g^{-1}	30 mL g^{-1}

Longitudinal Dispersivity Values

Concentrations in river water are shown to be insensitive to the longitudinal dispersivity except for radionuclides that have short half-lives relative to their transit time. Because transit times in the model domain are anywhere from 0.5 to 2 days, the longitudinal dispersivity is only sensitive for ^{56}Mn (half-life 0.107 d). In general, the model tends to overpredict most short-lived isotopes. Decreasing the longitudinal dispersivity has the net effect of pulling these concentrations closer to their measured values. For this reason, we use the lower-bound estimated

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longitudinal dispersivity of $50 \text{ m}^2 \text{ s}^{-1}$ from the HEDR Project. Concentrations of radionuclides are found to be sensitive to the transverse dispersivity and this parameter is reserved for calibration.

Calibration of Nuclide-specific K_d and Transverse Dispersivity

For each nuclide where measured data in the Columbia River existed, predicted concentrations are calibrated to their corresponding measured values. The K_d and transverse dispersivity (E_y) are used as calibration parameters. The metrics used to evaluate model calibration incorporate several performance measures commonly used in evaluation of atmospheric transport models (Fox 1981; EPA 1988; Cox and Tikvart 1990). These measures are the fractional bias (FB) and normalized mean square error ($NMSE$). The FB is given by

$$FB = \frac{2(\overline{C_o} - \overline{C_p})}{\overline{C_o} \overline{C_p}} \quad (17)$$

where C_p and C_o are the predicted and observed concentrations, respectively. Overbars indicate averages over the sample. The $NMSE$ is given by

$$NMSE = \frac{\overline{(C_o - C_p)^2}}{\overline{C_o} \overline{C_p}} \quad (18)$$

where C_p and C_o are the predicted and observed concentrations, respectively. Overbars indicate averages over the sample. The FB is a measure of the mean bias. A FB of 0.6 is equivalent to model underprediction by about a factor of 2. A negative value indicates model overprediction. The $NMSE$ is a measure of model variance. A $NMSE$ value of 1.0 indicates that the typical difference between predictions and observations is approximately equal to the mean. A perfect model would have a FB and $NMSE$ of zero. The goal of the calibration is to adjust the K_d and E_y values so that the FB and $NMSE$ are as close to zero or slightly negative in the case of FB , thereby assuring model overprediction.

Limitations are put on the possible values for K_d and E_y . The K_d value is to stay within the range of measured K_d values reported in the literature (Table 7). As shown earlier, water concentrations become insensitive to the E_y value at Pasco and McNary Dam. Radionuclide concentrations in water are also insensitive in E_y values greater than $1 \text{ m}^2 \text{ s}^{-1}$ at all locations, which means almost complete horizontal mixing across the width of the river. Therefore, E_y values are limited to values in the 0.1 to $1.0 \text{ m}^2 \text{ s}^{-1}$ range. For calibration purposes, we use the median estimate of the radionuclide release rate to the Columbia River as the source term.

Table 7. Summary of Linear Sorption Coefficients (K_d) Reported in the Literature (mL g^{-1})

Element	Sheppard and Thibault ^a				Baes ^b	NCRP ^{c,d}
	Sand	Silt	Clay	Organic		
As					200	110
Ca	5	20	1	70	4	6.7
Co ^e	50	1300	550	1000	45	60
Cr	70	30	1500	270	850	30
Cs ^f	280	4600	1900	270	1000	270
Cu					35	30
Ga					1500	
I	1	5	1	25	60	1
Mn	50	750	180	150	65	50
Na					100	76
Np					30	5
P	5	25	35	90	3.5	8.9
Sb					45	45
Sc					1000	310
Sr	15	20	110	150	35	15
Y	170	720	1000	2600	500	190
Zn	200	1300	2400	1600	40	200
Zr	600	2200	3300	7300	3000	580

^a Sheppard and Thibault (1990).

^b Baes et al. (1984).

^c NCRP (1996).

^d The units stated NCRP 1996 of $\text{m}^3 \text{kg}^{-1}$ are incorrect. The correct units are mL g^{-1} .

^e A 1971 sediment measurement behind McNary Dam was reported to be 27 pCi g^{-1} . Calibration of the model to this measurement yielded a K_d value of 1300 mL g^{-1} .

^f A 1971 sediment measurement behind McNary Dam was reported to be 4 pCi g^{-1} . Calibration of the model to this measurement yielded a K_d value of 4000 mL g^{-1} .

Results of the calibration (Table 8) show that the model tends to overpredict concentrations for short-lived radionuclides. Several possibilities exist for this overprediction. First, retention times in the holding ponds could be underestimated, resulting in an overestimation of the activity released to the river. Second, sorption on sediments could have the net effect of retarding the transport sufficiently so that more radioactive decay occurred during transport. Based on the information provided in Heeb and Bates (1994) for retention time, it does not appear likely that any additional investigation will improve upon their estimates. The second possibility would require a detailed sediment transport model such as the CHARIMA (Holly et al. 1993) model used in the HEDR Project. Models such as CHARIMA require substantially longer preparation and computer run times. These calculations are intended to be screening in nature and, therefore do not warrant the use of a complex model such as CHARIMA. In any case, concentrations are overestimated and will provide conservative estimates of exposure and risk.

Strontium-90 concentrations are also overpredicted substantially, but this nuclide has a half-life of 29-years so its unlikely that holdup times in the ponds would have had any serious impact on the release rates.

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Figures 9 through 11 provide plots of concentration verses time for several nuclides. Figures 12 and 13 present predicted verses measured concentrations in river water for monthly averaged continuous data and annual average data. Most of the model predictions were within a factor of 2 of the observations (the shaded area of the graph).

Table 8. Calibrated K_d and E_y Values and FB and $NMSE$

Radionuclide/measured data	K_d (mL g ⁻¹)	E_y (m ² s ⁻¹)	FB	$NMSE$
⁷⁶ As	200	0.35		
Annual average data			0.013	0.067
⁵¹ Cr	30	0.25		
Grab samples at Ringold			0.53	0.49
Continuous data, 300 Area			-0.02	0.14
Continuous data, Richland			-0.29	0.07
Continuous data, Pasco			-0.093	0.067
Annual average data			-0.391	0.078
⁶⁴ Cu	30	0.25		
Grab samples at Ringold			0.0051	0.456
Grab samples at Richland			-0.699	0.32
⁷² Ga	1500	0.6		
Grab samples at Ringold			-2.1	1.27
¹³¹ I	60	0.6		
Continuous data, 300 Area			-2.8	1.5
Continuous data, Richland			-2.56	1.4
Continuous data, Pasco			-4.4	2.2
Annual average data			-3.38	0.388
⁵⁶ Mn	750	0.6		
Annual average data			-3.4	1.3
²⁴ Na	100	0.6		
Annual average data Richland			-0.71	1.5
Annual average data Pasco			-3.4	1.1
²³⁹ Np	30	0.4		
Grab samples at Ringold			-0.12	0.53
Continuous data, 300 Area			-0.48	0.12
Continuous data, Richland			0.032	0.14
³² P	10	0.3		
Continuous data, 300 Area			-0.001	0.11
Continuous data, Richland			0.0049	0.19
Continuous data, Pasco			-0.96	0.28
Annual average data			-0.18	0.075
¹²² Sb	45	0.6		
Grab samples Richland			a	a
⁹⁰ Sr	110	0.6		
Grab samples, 300 Area			-1.7	0.54
Grab samples, Richland			-1.4	0.71

Table 8. Calibrated K_d and E_y Values and FB and $NMSE$

Radionuclide/measured data	K_d (mL g ⁻¹)	E_y (m ² s ⁻¹)	FB	$NMSE$
Grab samples, Pasco			-1.7	0.64
Annual average data			-5.4	1.62
⁴⁶ Sc	1000	0.2		
Annual average data			-0.74	0.61
⁶⁵ Zn	2400	0.1		
Continuous data, 300 Area			0.082	0.056
Continuous data, Richland			0.19	0.356
Continuous data, Pasco			0.039	0.13
Annual average data			-0.21	0.493
^{69m} Zn	2400	0.1		
Ringold grab samples			-0.16	12

^a Only one grab sample taken June 13, 1968 at Richland was found in Napier (1991). The measured concentration was 79 pCi L⁻¹ and the corresponding predicted concentration was 114 pCi L⁻¹.

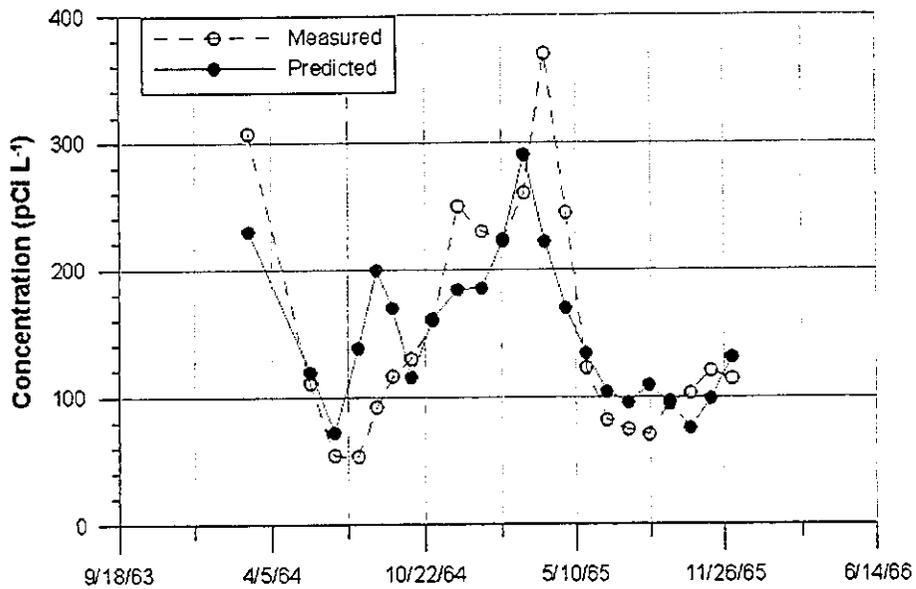


Figure 9. Monthly averaged ⁶⁵Zn concentration in water at Pasco as a function of time

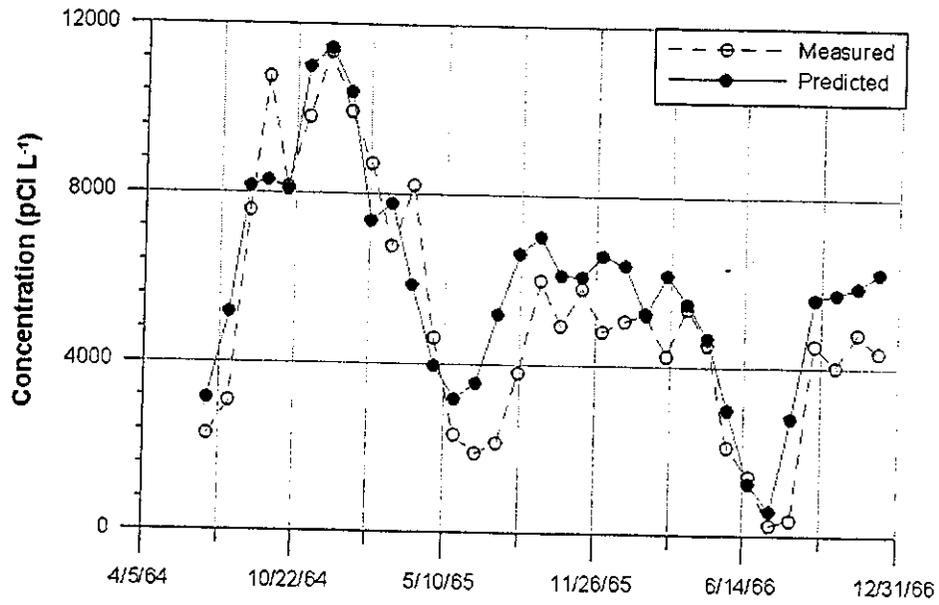


Figure 10. Monthly averaged ^{51}Cr concentration in water at Richland as a function of time.

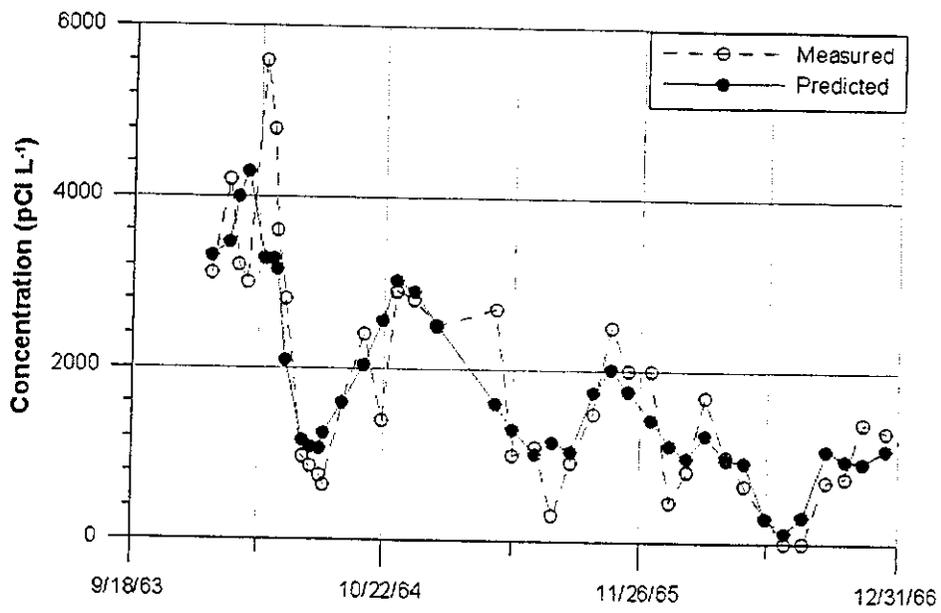


Figure 11. Monthly averaged ^{239}Np concentration in water at Richland as a function of time.

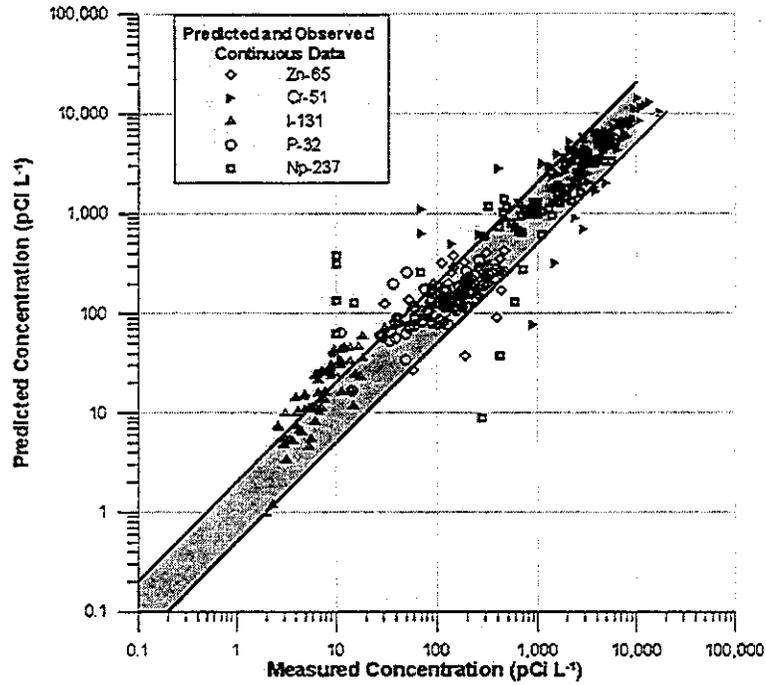


Figure 12. Predicted concentration in water as a function of measured concentration for monthly-averaged continuous data. The shaded area represents model predictions that are within a factor of 2 of the observations.

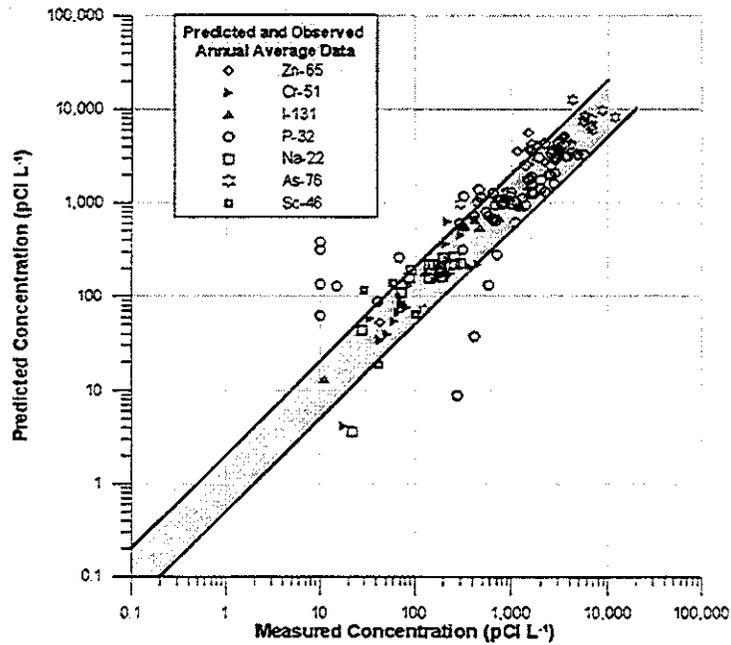


Figure 13. Predicted concentration in water as a function of measured concentration for monthly-averaged annual average data. The shaded area represents model predictions that are within a factor of 2 of the observations.

Model Uncertainty

Calculation of lifetime cancer incidence screening risk values for a continuously exposed person requires an estimate of the time-integrated concentration over the period a person is exposed. Exposure periods included the years from 1952 to 1965, during which time the highest releases from the reactors occurred. For some pathways, such as fish ingestion and swimming, the monthly average concentration is important because exposure is limited to several months of the year. Therefore, uncertainty must consider both monthly and annual-average concentrations.

The distribution of predicted-to-observed ratios (P/O) for annual and monthly average concentrations (Figure 14) provides a measure of the overall uncertainty in the model. As expected, monthly average P/O ratios exhibit more variability than annual average P/O ratios. The model exhibits positive bias, overpredicting concentrations in about 70% of the cases. Ninety-five percent of the model predictions are between a factor of 0.45 and 3.4 of the observations.

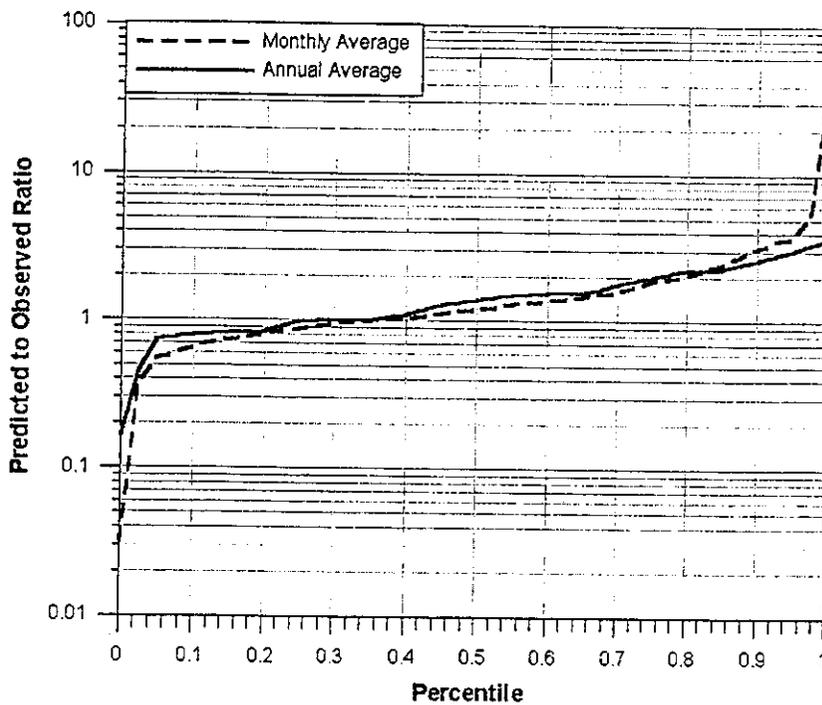


Figure 14. Distribution of monthly and annual average predicted-to-observed ratios.

The distribution of a multiplicative uncertainty factor (UF) was defined from the distribution of P/O ratios by

$$UF = \frac{1}{P/O} \quad (19)$$

The uncertainty factor applies not only to river transport but the source term as well. These data are developed from nuclide-specific river water measurements. For some nuclides, no river water measurements exist (^{95}Zr , ^{93}Y , ^{133}I , ^{45}Ca , ^{60}Co , and ^{137}Cs). In these cases, the uncertainty factor developed in Equation (19) is assumed to apply without modification, although it is recognized that the uncertainty for these nuclides is expected to be larger.

The uncertainty factor is applied to a limited number of radionuclides that have low screening risks and could potentially be removed from consideration. The uncertainty factor is applied to risk estimates outside the FORTRAN program and within a Microsoft Excel[®] spreadsheet using the Crystal Ball software (Decisioneering 2000).

Screening Risk Value Calculation

Calculation of lifetime cancer incidence screening risk values is performed by multiplying the water or sediment concentration by an exposure factor and summing over the exposure period.

$$R_l = \sum_{j=1}^k \sum_{i=m}^n C_{i,j} EF_{i,l} \Delta t \quad (20)$$

where

- R_l = Incremental lifetime cancer incidence screening risk value for the l^{th} pathway
- $C_{i,j}$ = Concentration in environmental media for the i^{th} day of year and j^{th} year of the simulation (Ci m^{-3} or Ci m^{-2} for sediment pathways)
- $EF_{i,l}$ = exposure factor for i^{th} day of year and l^{th} pathway ($\text{m}^3 \text{Ci}^{-1} \text{d}^{-1}$ or $\text{m}^2 \text{Ci}^{-1} \text{d}^{-1}$ for sediment pathways)
- Δt = time step (1 day).

Derivation of exposure factors is discussed in the next section. Exposure factors include media intake, exposure, and risk and are expressed in terms of the incremental cancer screening risk per day per unit concentration in environmental media. These factors are calculated on a monthly basis and are selected based on the current day of year of the simulation year. Incremental cancer screening risk is calculated on a daily basis and summed to yield to total cancer screening risk for the exposure period. Exposure factors for nuclides that would be in secular equilibrium with their parent (specifically, ^{90}Y derived from the decay and ^{90}Sr , $^{69\text{m}}\text{Zn}$ and ^{69}Zn , and ^{137}Cs and $^{137\text{m}}\text{Ba}$) are added together into a single exposure factor.

EXPOSURE PATHWAYS

The ways in which people were exposed to radionuclides released into the Columbia River are called exposure pathways. The different groups of people who used the river and their various activities are considered to identify exposure pathways to ensure that all important pathways are addressed in the screening analysis and that the parameters used to quantify the exposure pathway are not underestimated. The intent of the screening methodology is to produce screening estimates of risk for each pathway that are very unlikely to underestimate the actual risk to exposed individuals, and, for most situations, overestimate the risks. Through this process, those

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radionuclides and/or exposure pathways that are above the predefined risk decision criterion of 10^{-4} (discussed in the Risk-based Decision Criteria section) can be identified for further study, and those that fall below that level may be excluded from further analysis. Resources and effort can focus on those radionuclides and pathways where the screening risk values are in excess of the decision criterion.

The exposure pathways included in this screening analysis and the parameter values used to quantify them are discussed below. Each pathway is identified based on knowledge about the transfer characteristics of the radionuclides, the exposure pathways to humans from the Columbia River environment, and historical evidence that suggests the pathway represents an opportunity for actual human exposures. We reviewed the literature carefully to select parameter values for the initial screening that represent realistic maximum exposures so the potential exposure from any pathway is not underestimated. Where there is variability or uncertainty associated with a parameter value, a value from the upper end of the distribution was selected.

In this methodology, screening risk values are calculated at locations with the highest predicted radionuclide concentrations in river water and sediment. This conservatism is applied to reduce the likelihood that the risks associated with any exposure pathway are underestimated.

Ringold, located at RM 356, appears to be the first potential exposure location downstream of the eight once-through-cooled reactors located between RM 384 and 369. The 300 Area boundary and the pumping stations^f located in Richland at RM 339 and Pasco at RM 328 are other potential exposure sites. Effluent releases to the river from retention basins came from outfall lines (pipes) near the river bottom and took the form of a narrow plume that gradually spread and dispersed downstream. Because the reactor outfalls were located along the same shoreline and were in relatively close proximity, these plumes tended to coalesce and hug the Richland side of the river. Under some flow conditions, the contaminant plume was not entirely mixed over the full river width until it approached Pasco. Figure 15 shows the centerline of the effluent plume from the 100-B reactor to Pasco (Walters et al. 1992, Plate 2). The maximum beta activity generally occurred near the Hanford Ferry Landing, where the plume was about 8 km (5 mi) long and 152 m (500 ft) wide. Downstream of the Hanford Ferry Landing, the mixing across the river was more complete, although the plume could still be discerned along the shore at Richland.

Eight exposure pathways are considered in the screening calculations. These are designed to account for the different types of individuals, activities, and practices that may have resulted in exposure to radionuclides released to the Columbia River. Explicit consideration is given to Native American tribes potentially impacted by releases from the Hanford Site because they lived in closest proximity to the river and their lifestyle activities were intimately linked with the river.

The exposure pathways considered are

- (1) Ingestion of drinking water (untreated)
- (2) Inhalation of river water aerosols (sweat lodge activities and river water spray)
- (3) Sediment exposure (ingestion, external exposure, and dermal contact)
- (4) Swimming (immersion and inadvertent ingestion)
- (5) Boating
- (6) Consumption of fish (entire fish)
- (7) Consumption of waterfowl

^f Not operational until late 1963 (Napier 1993, p. 24).

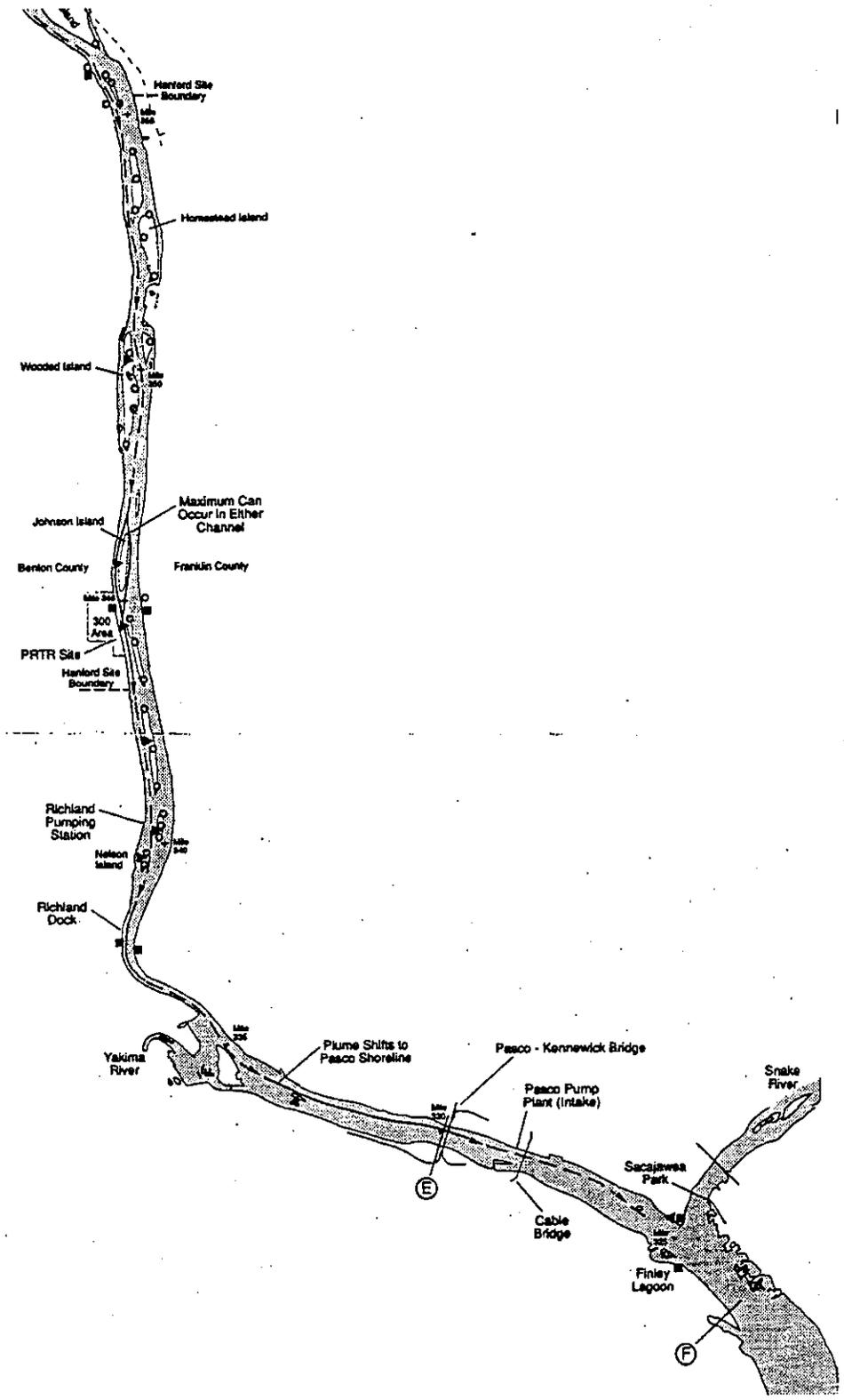


Figure 15. Radionuclide plume path from the Hanford boundary to Pasco.

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- (8) Irrigation of pasture/crops with river water and
- Milk consumption (also assume cattle consume river water and buildup of contaminants from irrigation in soil in which pasture grass grows)
 - Meat consumption (also assume cattle consume river water and buildup of contaminants from irrigation in soil in which pasture grass grows)
 - Food crop consumption (also assume buildup of contaminants from irrigation in soil).

In the "Exposure Scenario" section, we describe three exposure scenarios developed to further explore the impact and sensitivity of each of these pathways. These exposure scenarios were developed to represent an average individual in each population. The three scenarios are a Native American, a local resident of Richland, and a migrant worker.

Drinking Water Ingestion

The most direct exposure pathway for the Columbia River is to use it as a source of drinking water. The EPA recommends drinking water intake rates of 2 L d⁻¹ for adults for exposure assessment (EPA 1999a). These values represent upper percentile tapwater intake rates and include drinking water consumed in the form of juices and other beverages containing tapwater, such as coffee. Because the tribes reported to CDC a maximum estimate of 2.1 L d⁻¹ for drinking water intake (CDC 2000), we assume this drinking water intake rate (U_w) (2.1 L d⁻¹) for the screening analysis. This value accounts for water used by Native Americans to prepare teas from *Ledum groenlandicum*, mint (*Mentha arvensis*), wild bergamot (*Monarda fistulosa*), wild rose stems, and various flowers. We further assume that 100% of the drinking water is obtained directly from the Columbia River without treatment or holdup time and that drinking water is consumed at the same rate all year long (i.e., $F_{cw} = 1$; $EF = 365$ d y⁻¹).

The drinking water ingestion screening factor ($SF_{ing,water}$) is given by

$$SF_{ing,water} = C_w \cdot U_w \cdot F_{cw} \cdot EF \cdot ED \cdot RF_{ing,w} \quad (21)$$

where

- C_w = radionuclide concentration in river water (Bq L⁻¹)
 U_w = daily consumption rate of drinking water (L d⁻¹)
 F_{cw} = fraction of water consumed that is contaminated (unitless)
 EF = exposure frequency (d y⁻¹)
 ED = exposure duration (y)
 $RF_{ing,w}$ = lifetime morbidity risk coefficient for ingestion of water (Risk Bq⁻¹).

Lifetime morbidity risk coefficient values for this and all pathways are taken from EPA Federal Guidance Report 13 (EPA 1999b) unless otherwise stated.

Inhalation of River Water Aerosols

A number of activities may have resulted in the inhalation of aerosols of river water. Some of these are specific to Native Americans and others are applicable to a wider group of river users. Inhalation of aerosols of river water may occur during fishing activities, such as from dip-net platforms near waterfalls (Hewes 1998) or from inside sweat lodges when river water is applied to hot rocks (Harris and Harper 1997). For both of these activities, we estimate exposure as exposure to volatilized radionuclides in a shower. For the river water spray, we allow this air to be mixed with ambient air. For the sweat lodge, we assume that the air inside the sweat lodge is well-mixed but not mixed with air outside the sweat lodge. These activities and their exposure factors are considered below.

River Water Spray

In estimating the exposure to river water spray, we use the American Petroleum Institute (API) model for exposure to volatilized chemicals in a shower, replacing chemical parameters with similarly selected radionuclide parameters (API 1999).

For inhalation rate, we use the EPA exposure factor standard breathing rate of $20 \text{ m}^3 \text{ d}^{-1}$. The hourly breathing rate is then $0.83 \text{ m}^3 \text{ h}^{-1}$ (EPA 1999a). We assume that 2 h d^{-1} throughout the year are spent doing activities in locations where river spray could be a factor. These locations would most likely be near waterfalls, rocky areas where river flow is increased, or possibly dam outlets when water is being released.

The river water spray inhalation screening factor (SF_{spray}) is given as

$$SF_{\text{spray}} = IR \cdot C_{\text{spray}} \cdot ET \cdot EF \cdot ED \cdot R_{\text{inh}} \quad (22)$$

where

IR	=	inhalation rate ($\text{m}^3 \text{ h}^{-1}$)
C_{spray}	=	concentration of radionuclide in air due to river water spray (Bq m^{-3})
ET	=	exposure time (h d^{-1})
EF	=	exposure frequency (d y^{-1})
ED	=	exposure duration (y)
R_{inh}	=	lifetime morbidity risk coefficient for inhalation (Risk Bq^{-1}).

The contaminant concentration in river water spray is estimated using the API model mentioned above. We adapted this model for the river spray conditions because the shower model assumes no mixing with outside air, and in a location where river spray was a factor, mixing with ambient air would be a factor. Using simple first-order mixing, the contaminant concentration in river spray is estimated by the following equation:

$$C_{\text{spray}} = \frac{R}{F} \quad (23)$$

where

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- R = rate of contaminant release/aerosol production (Bq min^{-1})
 F = flow rate of air through the system ($\text{m}^3 \text{min}^{-1}$).

We assume the system to be a $1 \times 1 \times 2\text{-m}$ ($3.3 \times 3.3 \times 6.6\text{-ft}$) cell in which the receptor is located and river water spray is produced. The flow rate of air through the system is given by taking the cross-sectional area of the "cell" of air and multiplying that by the wind speed. The cross sectional area is 2 m^2 and the wind speed is assumed to be 2 m s^{-1} for a flow rate of air through the system of $4 \text{ m}^3 \text{ s}^{-1}$ or $240 \text{ m}^3 \text{ min}^{-1}$.

The rate of contaminant release or aerosol production is given by the following equation:

$$R = f_v \cdot Q \cdot C_w \quad (24)$$

where

- f_v = efficiency of contaminant release (unitless)
 Q = volumetric flow rate of water (L min^{-1})
 C_w = contaminant concentration in water (Bq L^{-1}).

We assume the volumetric flow rate of water is 10 L min^{-1} , similar to flow rate in showers. The efficiency of contaminant release estimates the volatilization of the contaminant by the following equation:

$$f_v = 1 - e^{\left(\frac{-K'_L t}{(d/6)3600}\right)} \quad (25)$$

where

- K'_L = the overall mass transfer coefficient at the temperature of the water (cm h^{-1})
 t = the water droplet drop time (s)
 d = the representative diameter of the water droplet (cm)
 3600 = conversion factor from hours to seconds.

For the water droplet time and diameter of a water droplet, we use values that are representative of showering situations, assuming that the river spray situation would be similar. The water droplet time used is 2 s and the diameter used is 0.2 cm.

The overall mass transfer coefficient is estimated using the two-film boundary theory, as shown in the following equation:

$$K_L = \left(\frac{1}{k_l} + \frac{1}{H \cdot k_g} \right)^{-1} \quad (26)$$

where

- K_L = overall mass transfer coefficient at a known calibration temperature, T_c (cm h^{-1})
 k_l = liquid-phase mass transfer coefficient (cm h^{-1})
 H' = Henry's Law constant (dimensionless)
 k_g = gas-phase mass transfer coefficient (cm h^{-1}).

Because we are estimating the volatilization of water containing dissolved radionuclides, we calculate the Henry's Law constant for water and assume the vapor concentration of the nuclide is the same as its liquid phase. The Henry's Law constant for water is shown below (Lyman et al. 1990).

$$H = \frac{P_{vp}}{S} \quad (27)$$

where

- P_{vp} = vapor pressure of water at temperature T (atm)
 S = solubility (mol m^{-3}).

The dimensionless Henry's Law constant (H') is then given by

$$H' = \frac{H}{R \cdot T} \quad (28)$$

where

- R = universal gas constant ($8.2 \times 10^{-5} \text{ atm m}^3 \text{ mol}^{-1} \text{ K}^{-1}$)
 T = temperature (K).

For the river water aerosols, the temperature of the river water is assumed to be 15°C (288 K). The values for the liquid- and gas-phase mass transfer coefficients are calculated, at a calibration temperature of 20°C (293 K) by the following equations:

$$k_g = 3000 \cdot \left(\frac{18}{MW_{cont}} \right)^{0.5} \quad (29)$$

$$k_l = 20 \cdot \left(\frac{44}{MW_{cont}} \right)^{0.5} \quad (30)$$

where

- MW_{cont} = the molecular weight of the contaminant,

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The values of 3000 and 20 represent the mass transfer coefficients of water and carbon dioxide, respectively, and 18 and 44 are the molecular weights of water and carbon dioxide, respectively.

Finally, the overall mass transfer coefficient must be adjusted to the temperature of the water using the following equation:

$$K'_L = K_L \left(\frac{T_{water} \cdot \mu_c}{T_c \cdot \mu_{water}} \right)^{0.5} \quad (31)$$

where

- T_{water} = temperature of the water (K)
 μ_c = viscosity of water at T_c (g s m^{-1})
 T_c = calibration temperature (K)
 μ_{water} = viscosity of water at T_{water} (g s m^{-1}).

The calibration temperature used for these calculations is 20°C (293 K), at which the viscosity of water is 1.002 g s m^{-1} . The viscosity of water at T_{water} is calculated, when $T < 20^\circ\text{C}$, as

$$\mu_w = 100 \cdot 10^y \quad (32)$$

$$y = \left[\frac{1301}{998.33 + 8.1855(T - 20) + 0.00585(T - 20)^2} \right] - 3.30233 \quad (33)$$

The viscosity of water when $T_{water} > 20^\circ\text{C}$ is

$$\mu_w = 1.002 \cdot 10^y \quad (34)$$

$$y = \frac{-1.3272(T - 20) - 0.001053(T - 20)^2}{T + 105} \quad (35)$$

For the case of the river water spray, the temperature of the river water is assumed to be 15°C (289 K).

Sweat Lodges

For the example of the sweat lodge, we also estimate volatilization of the contaminants using the API shower model. For this case, however, we assume that there is no mixing with outside

air. For inhalation rate, we use the EPA exposure factor standard breathing rate of $20 \text{ m}^3 \text{ d}^{-1}$ (EPA 1999a). The hourly breathing rate is then $0.83 \text{ m}^3 \text{ h}^{-1}$. We assume that 1 h d^{-1} throughout the year is spent in sweat lodge activities.

The sweat lodge inhalation screening factor (SF_{lodge}) is given below.

$$SF_{lodge} = IR \cdot C_{lodge} \cdot ET \cdot EF \cdot ED \cdot R_{inh} \quad (36)$$

where

- IR = inhalation rate ($\text{m}^3 \text{ h}^{-1}$)
- C_{lodge} = concentration of radionuclide in sweat lodge air (Bq m^{-3})
- ET = exposure time (h d^{-1})
- EF = exposure frequency (d y^{-1})
- ED = exposure duration (y)
- R_{inh} = lifetime morbidity risk coefficient for inhalation (Risk Bq^{-1}).

For the sweat lodge, because air does not mix with outside air, the concentration in air is given by

$$C_{lodge} = \frac{A_{lodge}}{V_{lodge}} \quad (37)$$

where

- A_{lodge} = activity of contaminants released into sweat lodge air (Bq)
- V_{lodge} = volume of sweat lodge (m^3).

The volume of the sweat lodge is assumed to be 20 m^3 . The activity of contaminants in sweat lodge air is estimated using the following equation:

$$A_{lodge} = f_v \cdot Q \cdot C_w \cdot t_{sl} \quad (38)$$

where

- f_v = efficiency of contaminant release (unitless)
- Q = volumetric flow rate of water (L min^{-1})
- C_w = contaminant concentration in water (Bq L^{-1})
- t_{sl} = time water is flowing within the sweat lodge (min).

The efficiency of contaminant release is calculated in the same manner as for the river water spray, except the temperature of the sweat lodge water is assumed to be 100°F (37°C), or 310 K . The flow rate of water is 10 L min^{-1} , and the time water is flowing within the sweat lodge is assumed to be 60 min .

Sediment Exposure Pathways

Several potential exposure pathways are associated with the accumulation of contaminated sediments along the shores or in shallower sections of the river with slow moving waters. A variety of river users may be exposed to contaminated sediment along the shores of the Columbia River. These include Native Americans, recreational fishermen, hikers, campers, and swimmers. The sediment exposure pathways are discussed below.

External Exposure from Sediments

The natural discharge into the Columbia River exhibits a marked seasonal fluctuation, with the largest discharges occurring during the summer months (June, July, and August) and the smallest discharges during the winter season (November, December, and January). This effect is reflected in a shoreline radiation survey (McConnon 1962) conducted in 1961 and 1962 between Ringold and Richland where beaches surveyed in October were submerged during the July survey. The Columbia River discharge also varies from year to year; therefore, the extent to which beaches and other areas of sediment are exposed varies not only throughout the year, but also between years. Despite these fluctuations, some beaches and areas of sediment accumulation will have remained accessible throughout the year. For this reason, we assume that external exposure to contaminated sediments could occur throughout the year. However, it may not be reasonable to assume that a person would be exposed to contaminated sediment for 24 hours a day every day of the year. Harris and Harper (1997) assumed an exposure duration and frequency of 12 h d^{-1} for 180 d y^{-1} (2160 h y^{-1}) to shoreline sediment in defining a Native American exposure scenario. The NCRP (1996) recommends an exposure time of 2000 h y^{-1} for screening calculations, which is roughly equivalent to 5.5 h d^{-1} for 365 d y^{-1} . The EPA does not address this issue specifically but recommends a value of 1.5 h d^{-1} for the time an adult spends outdoors as compared to 5 to 7 h d^{-1} for children (3 to 11 years of age).

Screening calculations for historical radionuclide releases to the Clinch River from X-10 on the Oak Ridge Reservation in Tennessee (Apostoaie et al. 1999) assumed a person (fisherman or camper) was exposed to sediment 25% of the year (~91 days). A distinction was made between low and high water levels, which uncover more or less of the sediment, respectively, and a unitless shielding factor (0.6) was applied to the high water level conditions.

For these screening calculations, we assume that exposure occurs each day throughout the year for 6 h d^{-1} for a total of 2190 h y^{-1} . We assume no shielding. The screening factor for external exposure to sediments ($SF_{ext, sed}$) is given by the following equation:

$$SF_{ext, sed} = C_{sed} \cdot ET \cdot F_{si} \cdot RF \cdot CF \cdot EF \cdot ED \quad (39)$$

where

- C_{sed} = time integrated sediment concentration (Bq kg^{-1})
- ET = exposure time (h d^{-1})
- F_{si} = sorption adjustment factor (dimensionless) for radionuclide i
- RF = risk per unit dose ($\text{Risk m}^2 \text{ Bq}^{-1} \text{ s}^{-1}$)

CF = conversion factor ($s\ h^{-1}$)
 EF = exposure frequency ($d\ y^{-1}$)
 ED = exposure duration (y).

Sediment Ingestion

Activities occurring where river sediments have accumulated may have resulted in the inadvertent ingestion of some sediment. Such activities could include sitting, playing, grubbing for worms, and collecting driftwood. Furthermore, materials such as reeds collected from along the shores and banks of the Columbia River may have sediments closely associated with them. Activities such as basket and mat weaving could result in sediment ingestion because of oral contact to wet the reed tips. Similarly, the use of roots, tubers, or vegetation gathered from areas of river sediment and soil for food preparation or medicinal purposes could result in sediment ingestion.

While data on sediment ingestion rates are lacking, data regarding soil ingestion rates may be relevant. EPA recommends a central estimate value of $0.05\ g\ d^{-1}$ for daily soil ingestion by adults and suggests a value of $0.1\ g\ d^{-1}$ as a conservative central estimate (EPA 1999a). However, data on soil ingestion rates are limited, particularly in adults and, therefore, they are uncertain. NCRP recommends a soil ingestion rate of $0.25\ g\ d^{-1}$ for screening calculations.

For the screening methodology, we recognize the uncertainty associated with the documented ingestion rates and adopt a conservative approach. A sediment ingestion rate of $0.25\ g\ d^{-1}$ is used for the screening calculation. The exposure frequency for this ingestion rate is assumed to be each day from April through September, for a total of approximately $180\ d\ y^{-1}$, based on the amount of time Harris and Harper (1997) estimated Native Americans in the Columbia River Basin region spend in various subsistence activities.

The equation that describes the screening factor for ingestion of sediment ($SF_{ing, sed}$) is shown below.

$$SF_{ing, sed} = C_{sed} \cdot \frac{1}{d \cdot \rho} \cdot U_{sed} \cdot F_{csed} \cdot EF \cdot ED \cdot RF_{ing, d} \quad (40)$$

where

C_{sed} = concentration of sediments ($Bq\ m^{-2}$)
 d = depth of sediment (m)
 ρ = density of sediment ($g\ m^{-3}$)
 U_{sed} = ingestion rate of sediment ($g\ d^{-1}$)
 F_{csed} = fraction of sediment ingested that is contaminated
 EF = exposure frequency ($d\ y^{-1}$)
 ED = exposure duration (y)
 $RF_{ing, d}$ = lifetime morbidity risk coefficient for dietary ingestion (Risk Bq^{-1}).

Dermal Absorption

Although the skin is permeable to a large number of primarily lipophilic toxicants, it is relatively impermeable to most ions and aqueous solutions. Therefore, dermal absorption is unlikely to be a significant exposure pathway for radioactive contaminants released into the Columbia River from the Hanford Site.

Dermal Contact

Activities such as reed gathering and driftwood collection along the shoreline could result in contaminated sediment adhering to the skin and allowing exposure of the skin to penetrating radiations (e.g., electrons). Electrons would probably not be energetic enough to be the cause of much external exposure from standing on the shoreline, but when sediment is applied directly to the skin, exposure becomes more likely. This exposure pathway is referred to as dermal contact. Harris and Harper (1997) suggests a daily adherence rate of 1 mg cm^{-2} over 5000 cm^{-2} , which is approximately 25% of the total skin surface area (EPA 1992) as a reasonable value. An exposure frequency of 180 d y^{-1} is suggested.

For the purposes of this screening analysis, we consider dermal contact as a special case because no risk factors exist for these types of exposures. Our ability to assess this pathway according to recommended exposure parameters is limited, but we use the information available on the dose delivered by dermal contact to assess the potential risk due to this pathway.

Dose rate conversion factors have been estimated by Kocher and Eckerman (1987) for some of the nuclides considered for this work. Dose coefficients are not identified for radionuclides with shorter half-lives (on the order of about 1 day or less), so risks are not calculated for those nuclides. Kocher and Eckerman assume that radioactivity is uniformly distributed over the entire body surface instead of just over some fraction of the body's surface area.

For this exposure, we calculate a risk screening factor (SF_{dermal}) using the following equation:

$$SF_{dermal} = C_{sed} \cdot ET \cdot EF \cdot ED \cdot DCF_{dermal} \cdot CF_t \cdot CF_a \quad (41)$$

where

C_{sed}	=	average sediment concentration over exposure period (Bq m^{-2})
ET	=	exposure time (hr d^{-1})
EF	=	exposure frequency (d y^{-1})
ED	=	exposure duration (y)
DCF_{dermal}	=	dose rate conversion factor (Sv y^{-1} per Bq cm^{-2})
CF_t	=	conversion factor for time (y hr^{-1})
CF_a	=	conversion factor for area ($\text{m}^2 \text{cm}^{-2}$).

The average sediment concentrations over the exposure period are calculated using our river model. We assume exposure to occur 1 h d^{-1} , 180 d y^{-1} , for 30 years.

Swimming

A swimmer in the Columbia River is directly exposed to radionuclides from immersion in the contaminated water and as a result of inadvertent ingestion of river water while swimming. This exposure pathway accounts for any activity where an individual is partly or totally immersed in the river water, for example, bathing and washing of plant materials. Exposure from activities where someone is only partly immersed would be overestimated.

In the HEDR Project, early screening calculations for this pathway assumed a "maximum individual," swam 100 h y^{-1} as compared to 10 h y^{-1} for a "typical individual" (Napier 1993). In the final HEDR dose calculations for the Columbia River pathway, a "maximum representative individual" defined as a significant user of the Columbia River, was assumed to swim for 5 hours every month from April through November giving a total of 40 h y^{-1} (Farris et al. 1994). Walker and Pritchard (1999) defines a "maximum river user" scenario for Native American fishermen who swim 42 h mo^{-1} from May through September (210 h y^{-1}). The Native American exposure scenario developed by Harris and Harper (1997) assumes 2.6 h d^{-1} is spent swimming for 70 d y^{-1} (180 h y^{-1}).

Migrant farm workers have been identified as another group of river users where this exposure pathway could be significant as a result of bathing and swimming in the vicinity of the Columbia River, predominantly in irrigation ditches. The months when this occurred coincide with those defined by Walker and Pritchard for Native American fishermen. We assume that the irrigation water comes from the Columbia River, and that the concentration of the irrigation water would, at most, be equal to the concentration in the Columbia River. Given the same exposure parameters, the risk to the migrant workers would be equivalent to the risk to Native Americans. We incorporate this risk into the migrant worker scenario, described in the "Exposure Scenarios" section. For this screening analysis, we assume the river user swims 1.4 h d^{-1} from May through September ($\sim 210 \text{ h y}^{-1}$).

The equation that describes the screening factor for immersion (SF_{imm}) in river water is shown below.

$$SF_{imm} = C_w \cdot ET_s \cdot DCF_{imm} \cdot EF \cdot ED \cdot RC_{imm} \cdot CF \quad (42)$$

where

- C_w = concentration of radionuclide in water (Bq L^{-1})
- ET_s = exposure time for swimming (h d^{-1})
- DCF_{imm} = dose conversion factor for immersion (Sv s^{-1} per Bq L^{-1})
- EF = exposure frequency (d y^{-1})
- ED = exposure duration (y)
- RC_{imm} = lifetime risk coefficient (Risk Sv^{-1})
- CF = units conversion (s h^{-1}).

We took dose conversion factors for swimming exposure from EPA Federal Guidance Report No. 12 (EPA 1993). EPA Federal Guidance Report 13 (EPA 1999b) does not provide morbidity risk coefficients for immersion therefore a lifetime risk coefficient of $7.3 \times 10^{-2} \text{ Sv}^{-1}$ was

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assumed based on ICRP Publication 60 (1991). This risk coefficient is an aggregated detriment that includes the probability of severe hereditary effects in addition to fatal and non-fatal cancers.

Swimming exposure can also result in some inadvertent ingestion of river water. The quantity ingested would not be very large, certainly not as large as the amount of water ingested for dietary reasons each day. The EPA recommends an incidental ingestion rate of 0.05 L h⁻¹ (EPA 1999a). The screening factor for ingestion of river water ($SF_{ing, inad}$) is shown below.

$$SF_{ing, inad} = C_w \cdot U_{wi} \cdot F_{cw} \cdot ET_{sw} \cdot EF_{sw} \cdot ED \cdot RF_{ing, w} \quad (43)$$

where

- C_w = radionuclide concentration in river water (Bq L⁻¹)
- U_{wi} = inadvertent ingestion rate of river water while swimming (L h⁻¹)
- F_{cw} = fraction of water ingested that is contaminated
- ET_{sw} = exposure time for swimming (h d⁻¹)
- EF_{sw} = exposure frequency for swimming (d y⁻¹)
- ED = exposure duration (y)
- $RF_{ing, w}$ = lifetime morbidity risk coefficient for ingestion of water (Risk Bq⁻¹).

These two screening factors, for immersion in and ingestion of river water can be summed to obtain the total screening factor for the swimming pathway.

Boating

The dose rate in a boat located on contaminated water is about one-half that of swimming in the same water. However, the number of hours that an individual may spend boating in a year is considerably larger than for swimming. We evaluate the external exposure from the boating pathway using the same approach used for the swimming immersion pathway. However, we use a dose rate that is one-half the dose rate for swimming.

Although swimming probably occurs only during a limited portion of the year, it is possible that boating activities can take place on the river throughout the entire year, especially in the Northwest regions of the country. For Native American populations, Wolfe and Walker (1987) recommends using a boating exposure of 240 h mo⁻¹ during April through October, totaling 1680 h y⁻¹. The HEDR Project dose calculations (Farris et al. 1994) assumed exposure every month of the year for 42 h mo⁻¹ for a total of 504 h y⁻¹.

For these screening calculations, we assume boating exposure occurs 2 h d⁻¹ for the entire year, for a total exposure of 730 h y⁻¹. The Wolfe and Walker exposure seemed excessive, but it was appropriate to use an exposure time larger than the HEDR calculations.

The screening factor for boating exposure (SF_{boat}) is shown below.

$$SF_{boat} = C_w \cdot ET_b \cdot \frac{1}{2} DCF_{imm} \cdot EF \cdot ED \cdot RC_{imm} \cdot CF \quad (44)$$

where

- C_w = concentration of radionuclide in water (Bq L⁻¹)
 ET_b = exposure time for boating (h d⁻¹)
 DCF_{imm} = dose conversion factor for immersion (Sv s⁻¹ per Bq L⁻¹)
 EF = exposure frequency (d y⁻¹)
 ED = exposure duration (y)
 RC_{imm} = lifetime risk coefficient (Risk Sv⁻¹)
 CF = units conversion (s h⁻¹).

Fish Consumption

Fish consumption is one of the primary exposure pathways identified for radionuclide releases to the Columbia River, and there has been concern that the parameters used in the HEDR Project dose calculations (Farris et al. 1994) underestimate the significance of this pathway for Native American users of the river. Based on studies by the Columbia River Inter-Tribal Fish Commission (CRITFC 1994) and by Wolfe and Walker (1987) on harvest rates of subsistence communities in Alaska, the EPA (1999a) recommends a mean fish consumption value of 70 g d⁻¹ and a 95th percentile value of 170 g d⁻¹ for Native American subsistence populations.

Table 9 presents the fish consumption rates and holdup times used by Walker and Pritchard (1999) for a "maximum river user," and they correspond to an annual consumption of 237 kg. For comparison, Table 10 presents the fish consumption rates and holdup times assumed in the HEDR dose calculations for the maximum representative individual, and they correspond to an annual consumption of 42.1 kg (Farris et al. 1994).

Table 9. Fish Consumption Rates (kg) and Holdup Times (d) for a Maximum River User^a

Fish category ^b	Month												Total	Holdup ^c (days)
	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sept	Oct	Nov	Dec		
Omnivore	4	4	4	2	2	2	2	2	2	2	4	4	34	3
1 st order predator	--	--	--	--	--	--	--	--	--	--	--	--	0	0
2 nd order predator	4	4	4	2	2	2	2	2	2	2	4	4	34	3
Salmon	3	3	3	22	22	22	22	22	22	22	3	3	169	14

^a From Walker and Pritchard (1999).

^b Omnivorous fish include bullhead, catfish, suckers, whitefish, chiselmouth, chub, sturgeon, minnows, and shiners. First-order predators include perch, crappie, punkinseed, and bluegill. Second-order predators include bass, trout, and squawfish.

^c The time between obtaining fish from the river and consuming it.

Table 10. Fish Consumption Rates (kg) and Holdup Times (d) for a Maximum Representative Individual^a

Fish category ^b	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sept	Oct	Nov	Dec	Total	Holdup ^c (days)
Omnivore	3.0	2.0	0.2	0.2	0	0	0	0.2	1.1	2.0	2.0	3.0	13.7	7
1 st order predator	0.3	1.2	2.8	2.8	3	3	3	2.8	2.0	1.2	1.2	0.3	23.6	2
2 nd order predator	0	0.1	0.3	0.3	0.3	0.3	0.3	0.3	0.2	0.1	0.1	0	2.3	2
Salmon	--	--	--	--	--	--	--	--	2.5	--	--	--	2.5	15

^a From Farris et al. (1994).

^b Omnivorous fish include bullhead, catfish, suckers, whitefish, chiselmouth, chub, sturgeon, minnows, and shiners. First-order predators include perch, crappie, punkinseed, and bluegill. Second-order predators include bass, trout, and squawfish.

^c The time between obtaining fish from the river and consuming it.

To define a Native American exposure scenario for risk assessment purposes, Harris and Harper (1994) uses a fish consumption rate of 540 g d⁻¹ comprised of 135 g d⁻¹ of fresh fish and 135 g d⁻¹ of dried fish. It was assumed that 405 g of fresh fish yield 135 g of dried fish, which is equivalent to an annual fish consumption of 197.1 kg. They considered this a reasonable intake for subsistence fishing based on a review of the literature and interviews with tribal members.

A number of Native American tribes in the Columbia River region have reported fish consumption values to CDC. Mean annual consumption rates ranged from 17 to 110 kg. Upper bound estimates were generally in the range of 220 kg, with one value as large as 411 kg. These estimates include fresh fish, stored fish, and shellfish. The highest consumption rates were reported for the spring season.

A distinction is usually made between the different types of fish that are consumed because the radionuclide concentrations vary. Resident fish in the Columbia River downstream of Hanford tend to have higher concentrations of a given radionuclide than nonresident fish because resident fish spend their entire lives in the Columbia River and have more time to accumulate radionuclides (Hanf et al. 1992). Furthermore, omnivorous fish tend to have higher radionuclide concentrations than predator fish.

The nonresident or anadromous species hatch in freshwater, grow and migrate to the ocean and eventually return to freshwater to spawn. Anadromous species that use the Columbia River as a migration route include the chinook salmon, sockeye salmon, coho salmon, and steelhead trout. These are important fish for Native Americans as well as sport fishermen.

In the HEDR Project, resident fish of importance to Native Americans and sport fishermen were identified as including mountain whitefish, white sturgeon, smallmouth bass, crappie, channel catfish, walleye, and yellow perch (Walters et al. 1992). Resident fish that are not usually eaten (carp, shiners, suckers, and squawfish) were not considered further in the HEDR Project. However, the highest reported concentrations of radionuclides in large fish are for suckers (Davis et al. 1958 cited in Walters et al. 1992), which Native Americans do consume. To account for the possibility of ingestion of fish with higher concentrations, we use the largest bioconcentration factors, or fraction of a given radionuclide that is concentrated from the water into fish muscle, available for all fish assumed to be consumed.

Based on the information provided in Walker and Pritchard (1999) and the information provided by the Native American tribes, a reasonable upper bound screening estimate for ingestion of fish is 238 kg annually. We assume an annual consumption of 68 kg of resident fish and 170 kg anadromous fish. Monthly consumption rates are assumed to vary in the same manner as estimated by Walker and Pritchard (1999). Daily consumption rates during each month used in the screening calculations are shown in Table 11.

Table 11. Daily Consumption Rates for Fish Used in Screening Calculations (kg d⁻¹)

Fish type	January	February	March	April	May	June	July	August	September	October	November	December	Annual total
Freshwater	0.27	0.27	0.27	0.13	0.13	0.13	0.13	0.13	0.13	0.13	0.27	0.27	~68
Anadromous	0.10	0.10	0.10	0.72	0.72	0.72	0.72	0.72	0.72	0.72	0.10	0.10	~170

Table 12. Bioconcentration Factors in Fish Used for Screening Calculations

Radionuclide	Freshwater (L kg ⁻¹)		Anadromous (L kg ⁻¹)	
	Cool	Warm	Cool	Warm
²⁴ Na	1.0	1.0	1.0	1.0
³² P	320	1100	60	680
⁴⁵ Ca	170	170	170	170
⁴⁶ Sc	75	75	75	75
⁵¹ Cr	5	5	5	5
⁵⁶ Mn	0.25	0.25	0.25	0.25
⁶⁰ Co	300	300	300	300
⁶⁴ Cu	3.5	3.5	3.5	3.5
⁶⁵ Zn	160	330	90	150
⁶⁹ Zn	1.3	1.3	1.3	1.3
^{69m} Zn	19	19	19	19
⁷² Ga	7.7	7.7	7.7	7.7
⁷⁶ As	300	300	300	300
⁸⁹ Sr – filet	40	40	40	40
⁹⁰ Sr – filet	60	60	60	60
⁸⁹ Sr – whole fish	1500	1500	1500	1500
⁹⁰ Sr – whole fish	2400	2400	2400	2400
⁹⁰ Y	2.5	2.5	2.5	2.5
⁹³ Y	0.4	0.4	0.4	0.4
⁹⁵ Zr	220	220	220	220
¹²² Sb	8.2	8.2	8.2	8.2
¹³¹ I	8.4	8.4	8.4	8.4
¹³³ I	1.1	1.1	1.1	1.1
¹³⁷ Cs	2000	2000	100	100
²³⁹ Np	25	25	25	25

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Table 12 shows the radionuclide-specific bioconcentration factors selected for these screening calculations for resident/freshwater fish and anadromous fish. These values are selected to err on the conservative side, and the most conservative value for freshwater fish (e.g., omnivorous fish) is used for the freshwater bioconcentration factor. A seasonal difference is observed for zinc and phosphorus, with greater uptake in the warm season. For some elements, NCRP suggests bioconcentration factors that are too conservative for the short-lived isotopes of that nuclide. In those cases, the element-specific bioconcentration factors can be adapted by the biological half-life and radiological half-life of the nuclide to produce nuclide-specific factors. The element-specific bioconcentration factor can be multiplied by a factor (K), calculated using the following equation:

$$K = \frac{\lambda_b}{\lambda_i + \lambda_b} \quad (45)$$

where

- λ_b = biological decay constant = $0.693t_b^{-1}$ (d^{-1})
- λ_i = radiological decay constant = $0.693t_i^{-1}$ (d^{-1})
- t_b = biological half-life (d)
- t_i = radiological half-life (d).

A biological half-life of 30 days is assumed (NCRP 1996).

Several sources of information were used in selecting bioconcentration factors including Napier (1993), NCRP (1996), ATSDR (1998), Till and Meyer (1983), IAEA (1994), Theide et al. (1994), Walker and Pritchard (1999), and Farris et al. (1994).

Although resident fish are consumed sooner after being caught than salmon, no holdup is assumed between catch and consumption of the fish for the screening analysis. The fish ingestion screening factor ($SF_{ing, fish}$) is given by the following equation:

$$SF_{ing, fish} = [(C_{w,i} \cdot BCF_{fw,i} \cdot U_{fw}) + (C_{w,i} \cdot BCF_{an,i} \cdot U_{an})] \cdot EF \cdot ED \cdot RF_{ing,d} \quad (46)$$

where

- $C_{w,i}$ = concentration of radionuclide i in river water ($Bq L^{-1}$)
- $BCF_{fw,i}$ = bioaccumulation factor for radionuclide i in freshwater fish ($L kg^{-1}$)
- U_{fw} = daily consumption rate of freshwater fish ($kg d^{-1}$)
- $BCF_{an,i}$ = bioaccumulation factor for radionuclide i in anadromous fish ($L kg^{-1}$)
- U_{an} = daily consumption rate of anadromous fish ($kg d^{-1}$)
- EF = exposure frequency ($d y^{-1}$)
- ED = exposure duration (y)
- $RF_{ing,d}$ = lifetime morbidity risk coefficient for dietary ingestion (Risk Bq^{-1}).

Waterfowl Consumption

In the HEDR dose calculations, the annual consumption of waterfowl for the maximum representative individual was estimated as 20 kg (fresh weight) (Napier 1993). Consumption was

not constant throughout the year with October and November assumed to have the highest consumption rates (4 kg mo^{-1}). For December through May, the consumption rate was assumed to be 2 kg mo^{-1} , and no consumption was assumed for June through September. In all cases it was assumed no days elapsed before the duck was consumed.

A number of Native American tribes in the Columbia River region have reported to CDC a mean annual consumption rate for wildfowl of 6.2 kg with an upper bound estimate of 222 kg. These values include upland birds and waterfowl. Based on reports of actual hunting success in game management regions around DOE's Hanford Site cited in Harris and Harper (1997), it is estimated that approximately 80% of waterfowl ingestion is from waterfowl and 20% from upland birds. This suggests a mean annual consumption for waterfowl of 5 kg, with an approximate upper bound estimate of 178 kg. Harris and Harper (1997) estimates an intake rate of 35 g d^{-1} for waterfowl (meat and eggs), equivalent to an annual consumption of 12.8 kg for defining a Native American exposure scenario.

For the current screening analysis, it was more appropriate to assume an annual consumption of 20 kg for waterfowl based on the maximum representative individual defined in the HEDR Project. We assume the variation in consumption throughout the year of 4 kg mo^{-1} in October and November, 2 kg mo^{-1} in December through May, and no consumption in June through September. Historical data from Hanford on radionuclide concentrations in waterfowl identified ^{32}P , ^{40}K , ^{65}Zn , ^{90}Sr , and ^{137}Cs in waterfowl (Hanf et al. 1992). HEDR calculated bioconcentration factors for only ^{32}P and ^{65}Zn . We use the observations about the relation of bioconcentration factors in fish to those in waterfowl to estimate factors for ^{90}Sr and ^{137}Cs . Table 13 shows the factors used for our screening calculations.

Table 13. Bioconcentration Factors in Waterfowl for Screening Calculations

Radionuclide	BCF for waterfowl (L kg^{-1})
^{32}P	800
^{65}Zn	75
^{90}Sr	30
^{137}Cs	1000

The screening factor for waterfowl ingestion ($SF_{ing,fowl}$) is given in the following equation:

$$SF_{ing,fowl} = C_{w,i} \cdot BCF_{w,i} \cdot U_{wf} \cdot F_{cwf} \cdot EF \cdot ED \cdot RF_{ing,d} \quad (47)$$

where

- $C_{w,i}$ = concentration of radionuclide I in river water (Bq L^{-1})
- $BCF_{w,i}$ = bioaccumulation factor for radionuclide I in waterfowl (L kg^{-1})
- U_{wf} = daily consumption rate of waterfowl (kg d^{-1})
- F_{cwf} = fraction of waterfowl consumed that is contaminated (unitless)
- EF = exposure frequency (d y^{-1})
- ED = exposure duration (y)
- $RF_{ing,d}$ = lifetime morbidity risk coefficient for dietary ingestion (Risk Bq^{-1}).

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Irrigation

Irrigation with Columbia River water is known to have occurred in fields used for grazing cows in the Riverview area near Pasco, Washington, and in the Ringold area. Measurable concentrations of ^{65}Zn were reported in milk from these cows (Foster and Junkins 1960). We also performed a screening analysis on beef ingestion and food crop ingestion, assuming that beef cattle were grazed on contaminated lands and that crops were irrigated with river water. The irrigation pathway may not relate to Native Americans, but it may be significant for other users of the Columbia River and is, therefore, important to include in the screening methodology. Irrigation with Columbia River water is a pathway for radioactivity to reach milk, meat, and food crops consumed by an individual. Contamination is transferred to food crops consumed by humans and forage consumed by cattle by direct deposition from irrigation and by buildup in the soil from regular irrigation and uptake via the root systems of plants.

Milk Consumption

Radionuclide contamination of milk from the Columbia River could occur because of dairy cattle ingesting contaminated river water and contaminated forage. We use the NCRP screening models methodology (NCRP 1996) to calculate the concentration in forage due to direct irrigation and buildup in soil over a 30-year time period.

Milk ingestion, for the purposes of this screening calculation, is assumed to occur each day throughout the year. The EPA Exposure Factors Handbook (EPA 1999a) indicates that the median intake of milk for the U.S. population is $8 \text{ g kg}^{-1} \text{ d}^{-1}$. For the average 71.8-kg adult, this is approximately 0.6 L d^{-1} of milk consumption. The distribution of values for this parameter has a 95th percentile value of 2.3 L d^{-1} . NCRP suggests a usage value for milk ingestion of 300 L y^{-1} , or approximately 0.8 L d^{-1} (NCRP 1996). For these screening calculations, we assume milk ingestion of 0.8 L d^{-1} , with all the milk that is being consumed contaminated.

NCRP also recommends values for dairy cattle ingestion of water and forage of 60 L d^{-1} and 16 kg d^{-1} , respectively. The irrigation rate recommended by NCRP for these calculations is $2 \text{ L m}^{-2} \text{ d}^{-1}$ (NCRP 1996).

The screening factor for milk consumption ($SF_{\text{ing,milk}}$) is given by

$$SF_{\text{ing,milk}} = [C_{\text{milk(water)}} + C_{\text{milk(for)}}] \cdot U_{\text{milk}} \cdot F_{\text{cd}} \cdot EF \cdot ED \cdot RF_{\text{ing,d}} \quad (48)$$

where

- U_{milk} = daily milk ingestion (L d^{-1})
- F_{cd} = fraction of consumed milk that is contaminated
- EF = exposure frequency (d y^{-1})
- ED = exposure duration (y)
- $RF_{\text{ing,d}}$ = lifetime morbidity risk coefficient for dietary ingestion (Risk Bq^{-1}).

and

$$C_{\text{milk(water)}} = C_w \cdot Q_{\text{wd}} \cdot F_{\text{cw}} \cdot F_m \quad (49)$$

$$C_{milk(for)} = C_{for} \cdot Q_{fd} \cdot F_{cf} \cdot F_m \quad (50)$$

where

- $C_{milk(water)}$ = radionuclide concentration in milk due to cattle ingestion of contaminated water ($Bq L^{-1}$)
- C_w = radionuclide concentration in water ($Bq L^{-1}$)
- Q_{wd} = ingestion rate of water by dairy cattle ($L d^{-1}$)
- F_{cw} = fraction of consumed water that is contaminated
- F_m = transfer coefficient ($d L^{-1}$)
- $C_{milk(for)}$ = radionuclide concentration in milk due to cattle ingestion of contaminated forage ($Bq L^{-1}$)
- C_{for} = radionuclide concentration in forage ($Bq kg^{-1}$)
- Q_{fd} = ingestion rate of forage by dairy cattle ($kg d^{-1}$)
- F_{cf} = fraction of consumed forage that is contaminated
- F_m = transfer coefficient to milk ($d L^{-1}$)

and

$$C_{for} = C_w \cdot F_{ir} \cdot CF_{for,i} \quad (51)$$

where

- C_{for} = concentration of contamination in forage ($Bq kg^{-1}$)
- C_w = concentration of water ($Bq L^{-1}$)
- F_{ir} = irrigation rate ($L m^{-2} d^{-1}$)
- $CF_{for,i}$ = transfer factor for radionuclide i , including buildup in soil ($Bq kg^{-1}$ per $Bq m^{-2} d^{-1}$).

Meat Consumption

Radionuclide contamination of meat could occur when beef cattle ingest contaminated Columbia River water and contaminated forage. We use the NCRP screening models methodology (NCRP 1996) to calculate the concentration in forage due to direct irrigation and buildup in soil over a 30-year time period in the same manner as for milk.

Meat ingestion, for the purposes of this screening calculation, is assumed to occur each day throughout the year with no holdup time between butchering the cattle and ingestion of the beef. The EPA Exposure Factors Handbook (EPA 1999a) indicates that the median intake of beef for the U.S. population is $2.1 g kg^{-1} d^{-1}$. For the average 71.8-kg adult, this is approximately $0.15 kg d^{-1}$ of meat consumption. The distribution of values for this parameter has a 95th percentile value of $0.37 kg d^{-1}$. NCRP suggests a usage value for meat ingestion of $100 kg y^{-1}$, or approximately $0.27 kg d^{-1}$ (NCRP 1996). For these screening calculations, we assume meat ingestion of $0.3 kg d^{-1}$, with all the meat that is being consumed contaminated.

NCRP also recommends values for beef cattle ingestion of water and forage of $50 L d^{-1}$ and $12 kg d^{-1}$, respectively. The irrigation rate recommended by NCRP for these calculations is $2 L m^{-2} d^{-1}$ (NCRP 1996).

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The screening factor for meat consumption ($SF_{ing,meat}$) is given by

$$SF_{ing,meat} = [C_{meat(water)} + C_{meat(for)}] \cdot U_{meat} \cdot F_{cb} \cdot EF \cdot ED \cdot RF_{ing,d} \quad (52)$$

where

- U_{meat} = daily meat ingestion (kg d^{-1})
- F_{cd} = fraction of consumed meat that is contaminated
- EF = exposure frequency (d y^{-1})
- ED = exposure duration (y)
- $RF_{ing,d}$ = lifetime morbidity risk coefficient for dietary ingestion (Risk Bq^{-1})

and

$$C_{meat(water)} = C_{water} \cdot Q_{wb} \cdot F_{cw} \cdot F_b \quad (53)$$

$$C_{meat(for)} = C_{for} \cdot Q_{fb} \cdot F_{cf} \cdot F_b \quad (54)$$

where

- $C_{meat(water)}$ = radionuclide concentration in meat due to cattle ingestion of contaminated water (Bq kg^{-1})
- C_{water} = radionuclide concentration in water (Bq L^{-1})
- Q_{wb} = ingestion rate of water by beef cattle (L d^{-1})
- F_{cw} = fraction of consumed water that is contaminated
- F_b = transfer coefficient (d kg^{-1})
- $C_{meat(for)}$ = radionuclide concentration in meat due to cattle ingestion of contaminated forage (Bq kg^{-1})
- C_{for} = radionuclide concentration in forage (Bq kg^{-1})
- Q_{fb} = ingestion rate of forage by beef cattle (kg d^{-1})
- F_{cf} = fraction of consumed forage that is contaminated
- F_b = transfer coefficient to beef (d kg^{-1})

and

$$C_{for} = C_w \cdot F_{ir} \cdot CF_{for,i} \quad (55)$$

where

- C_{for} = concentration of contamination in forage (Bq kg^{-1})
- C_w = concentration of water (Bq L^{-1})
- F_{ir} = irrigation rate ($\text{L m}^{-2} \text{d}^{-1}$)

$CF_{for,i}$ = transfer factor for radionuclide i , including buildup in soil (Bq kg⁻¹ per Bq m⁻² d⁻¹)

Food Crop Consumption

Food crops consumed by individuals could become contaminated by irrigation by both direct interception of contaminated water and from uptake of radionuclides through roots growing in contaminated soils. We use the NCRP screening models methodology (NCRP 1996) to calculate the concentration in fresh vegetables due to direct irrigation and buildup in soil over a 30-year time period.

Fresh vegetable ingestion, for the purposes of this screening calculation, is assumed to occur each day throughout the year. The EPA Exposure Factors Handbook (EPA 1999a) indicates that the median intake of vegetables for the U.S. population is 4.3 g kg⁻¹ d⁻¹. For the average 71.8 kg adult, this is approximately 0.31 kg d⁻¹ of vegetable consumption. The distribution of values for this parameter has a 95th percentile value of 0.72 kg d⁻¹. NCRP suggests a usage value for meat ingestion of 200 kg y⁻¹, or approximately 0.55 kg d⁻¹ (NCRP 1996). For these screening calculations, we assume vegetable ingestion of 0.55 kg d⁻¹, with all the vegetables that are being consumed contaminated.

The screening factor for ingestion of contaminated food crops ($SF_{ing,crop}$) is shown below.

$$SF_{ing,crop} = C_{veg} \cdot U_{prod} \cdot F_{cp} \cdot ED \cdot EF \cdot RF_{ing,d} \quad (56)$$

where

U_{prod} = ingestion rate of contaminated produce (kg d⁻¹)
 F_{cp} = fraction of consumed produce that is contaminated
 ED = exposure duration (d y⁻¹)
 EF = exposure frequency (y)
 $RF_{ing,d}$ = lifetime morbidity risk coefficient for dietary ingestion (Risk Bq⁻¹)

and

$$C_{veg} = C_w \cdot F_{ir} \cdot CF_{veg,i} \quad (57)$$

where

C_{veg} = concentration of contamination in vegetables (Bq kg⁻¹)
 C_w = concentration of water (Bq L⁻¹)
 F_{ir} = irrigation rate (L m⁻² d⁻¹)
 $CF_{veg,i}$ = transfer factor for radionuclide i , including buildup in soil (Bq kg⁻¹ per Bq m⁻² d⁻¹).

EXPOSURE SCENARIOS

To further explore exposure pathways, we developed scenarios for Native Americans, local residents, and migrant workers. These scenarios were developed to evaluate less conservative situations of exposure for these river users. Although the same fundamental equations are used as for the initial screening, the exposure parameters are adjusted to be less conservative to explore the relative importance of the different exposure pathways more thoroughly. Table 14 shows the parameter values used for the initial screening, which included all exposure pathways, and for the Native American, resident, and migrant worker exposure scenarios. We show the screening values described above for comparison.

Table 14. Exposure Scenarios for the Columbia River^a

Pathway	Screening	Native American	Resident	Migrant worker
Drinking water (L d⁻¹)	2.1	2.1	2.1	2.1 (Apr–Oct)
Fraction contaminated	1	1	0.5	1
Total L y⁻¹ cont. intake	770	770	380	210
Fish ingestion (kg d⁻¹)				
Freshwater (Nov–Mar)	0.27	0.11	0.015	0
Freshwater (Apr–Oct)	0.13	0.06	0.008	0.008
Anadromous (Nov–Mar)	0.1	0.03	0.004	0
Anadromous (Apr–Oct)	0.72	0.35	0.05	0.05
Fraction contaminated	1	1	1	1
Total kg y⁻¹ cont. intake	240	109	15	12
Swimming (h d⁻¹)	1.4	1	0.5	1
(May–Sept)	(May–Sept)	(May–Sept)	(Jun–Aug)	(May–Sept)
Total h y⁻¹ swimming	214	153	46	153
Swimming ingestion (L h⁻¹)	0.05	0.05	0.05	0.05
(May–Sept)	(May–Sept)	(May–Sept)	(Jun–Aug)	(May–Sept)
Total L y⁻¹ ingestion	11	8	2	8
Waterfowl ingestion (kg d⁻¹)				
(Oct–Nov)	0.13	0.045	0.02	0.01 (Oct)
(Dec–May)	0.07	0.02	0.01	0
(Jun–Sept)	0	0	0	0.01
Fraction contaminated	1	1	1	1
Total kg y⁻¹ ingestion	21	6.4	3.0	2.1
Sediment external exposure (h d⁻¹)	6	4	1	4
(Jun–Aug)			(Jun–Aug)	(Apr–Oct)
Total h y⁻¹ exposure	2190	1460	92	856
Sediment ingestion (g d⁻¹)	0.25	0.25	0.25	0.25
(Apr–Sept)	(Apr–Sept)	(Apr–Sept)	(Jun–Aug)	(Apr–Oct)
Total g y⁻¹ ingestion	46	46	23	54
Dermal contact exposure (h d⁻¹)	1	1	0.5	1
(Apr–Sept)	(Apr–Sept)	(Apr–Sept)	(Jun–Aug)	(Apr–Oct)
Total h y⁻¹ exposure	183	183	46	214
River water aerosols (h d⁻¹)	2	2	0	1

Table 14. Exposure Scenarios for the Columbia River^a

Pathway	Screening	Native American	Resident	Migrant worker
				(Apr–Oct)
<i>Total h y⁻¹ inhalation</i>	730	730	0	214
Sweat lodge (h d ⁻¹)	1	1	0	0
<i>Total h y⁻¹ inhalation</i>	365	365	0	0
Boating exposure (h d ⁻¹)	2	1	1	0
			(Jun–Aug)	
<i>Total h y⁻¹ exposure</i>	730	365	92	0
Milk ingestion (L d ⁻¹)	0.8	0	0.6	0.6
				(Apr–Oct)
Fraction contaminated	1	0	1	1
<i>Total L y⁻¹ ingestion</i>	292	0	219	128
Meat ingestion (kg d ⁻¹)	0.3	0	0.15	0.15
				(Apr–Oct)
Fraction contaminated	1	0	1	1
<i>Total kg y⁻¹ ingestion</i>	110	0	55	32
Produce ingestion (kg d ⁻¹)	0.55	0.3	0.25	0.3
		(Jun–Oct)	(Jun–Oct)	(Jun–Oct)
Fraction contaminated	1	1	1	1
<i>Total kg y⁻¹ ingestion</i>	200	46	38	46

Values are shown to 2 significant digits to show consistent mathematical additivity.

SCREENING RISK ESTIMATES

We used an absolute risk-based criterion of 10^{-4} for the initial screening of 23 radionuclides (Table 1) released to the Columbia River from the Hanford Nuclear Site. Results of the calculation (Table 15) indicate that 4 radionuclides (^{45}Ca , ^{51}Cr , ^{93}Y , and ^{122}Sb) can be removed from further consideration because their screening risk value is less than 10^{-4} . Pathways of least importance include inadvertent water ingestion from swimming, external exposure to shoreline sediments, inadvertent sediment ingestion, and aerosol inhalation. Combined, these pathways contribute less than 0.5% to the total screening risk value calculated for all pathways (Table 16). Screening risk values tend to be slightly higher at the 300 Area location because the receptor is closer to the plume centerline compared to the receptor at the Ringold far shore. In most cases, fish ingestion is the dominant pathway for individual radionuclide risks (Table 15), in a few cases, water ingestion is the dominant pathway, and in one case, boating is the dominant pathway (see Appendix E for a detailed accounting of screening risk results). For total risk (all nuclides), fish ingestion accounts for over 90% of the total risk. Most of the exposure is incurred over the years 1952 to 1964 (Figure 17). These years correspond to the years of highest release from the Hanford reactors.

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Table 15. Initial Screening Risks and Primary Exposure Pathways by Radionuclide

Nuclide	300 Area	Ringold	Primary pathway	% contribution
⁷⁶ As	2.2×10^{-2}	1.5×10^{-2}	Whole fish ingestion	99
⁸⁹ Sr	9.5×10^{-3}	4.0×10^{-3}	Whole fish ingestion	>99
⁶⁹ Zn	5.3×10^{-3}	2.6×10^{-3}	Whole fish ingestion	74
²³⁹ Np	3.6×10^{-3}	2.9×10^{-3}	Whole fish ingestion	90
⁹⁰ Sr	2.9×10^{-3}	1.2×10^{-3}	Whole fish ingestion	99
⁶⁵ Zn	2.6×10^{-3}	1.2×10^{-3}	Whole fish ingestion	82
³² P	2.0×10^{-3}	9.3×10^{-4}	Whole fish ingestion	80
⁹⁵ Zr	1.7×10^{-3}	7.8×10^{-4}	Whole fish ingestion	92
²⁴ Na	7.0×10^{-4}	2.9×10^{-4}	Boating	27
⁷² Ga	4.9×10^{-4}	2.6×10^{-4}	Whole fish ingestion	70
¹³⁷ Cs	3.6×10^{-4}	1.9×10^{-4}	Whole fish ingestion	73
¹³¹ I	3.4×10^{-4}	1.5×10^{-4}	Whole fish ingestion	59
⁴⁶ Sc	2.9×10^{-4}	1.4×10^{-4}	Whole fish ingestion	85
⁹⁰ Y	2.2×10^{-4}	9.4×10^{-5}	Whole fish ingestion	44
⁶⁴ Cu	1.7×10^{-4}	8.1×10^{-5}	Whole fish ingestion	50
¹³³ I	1.5×10^{-4}	6.8×10^{-5}	Water ingestion	42
⁵⁶ Mn	1.3×10^{-4}	5.4×10^{-5}	Water ingestion	39
⁶⁰ Co	1.1×10^{-4}	5.9×10^{-5}	Whole fish ingestion	82
⁵¹ Cr	9.3×10^{-5}	4.0×10^{-5}	Whole fish ingestion	41
¹²² Sb	5.1×10^{-5}	3.6×10^{-5}	Whole fish ingestion	71
⁴⁵ Ca	2.5×10^{-5}	1.2×10^{-5}	Whole fish ingestion	98
⁹³ Y	2.2×10^{-5}	1.1×10^{-5}	Water ingestion	78
Total	5.3×10^{-2}	3.0×10^{-2}	Whole fish ingestion	91

^a Includes ^{69m}Zn

Table 16. Percentage Contribution of Exposure Pathways to Total Screening Risk

Exposure pathway	Percentage contribution to total (all nuclides) screening risk
Direct ingestion	3.0
Fish ingestion	91.1
Swimming-immersion	0.3
Swimming-ingestion	<0.1
Waterfowl	0.9
Sediment-external	0.2
Sediment dermal	0.3
Sediment ingestion	<0.1
Aerosol inhalation	<0.1
Boating	0.6
Produce ingestion	0.9
Meat ingestion	1.7
Milk ingestion	0.8

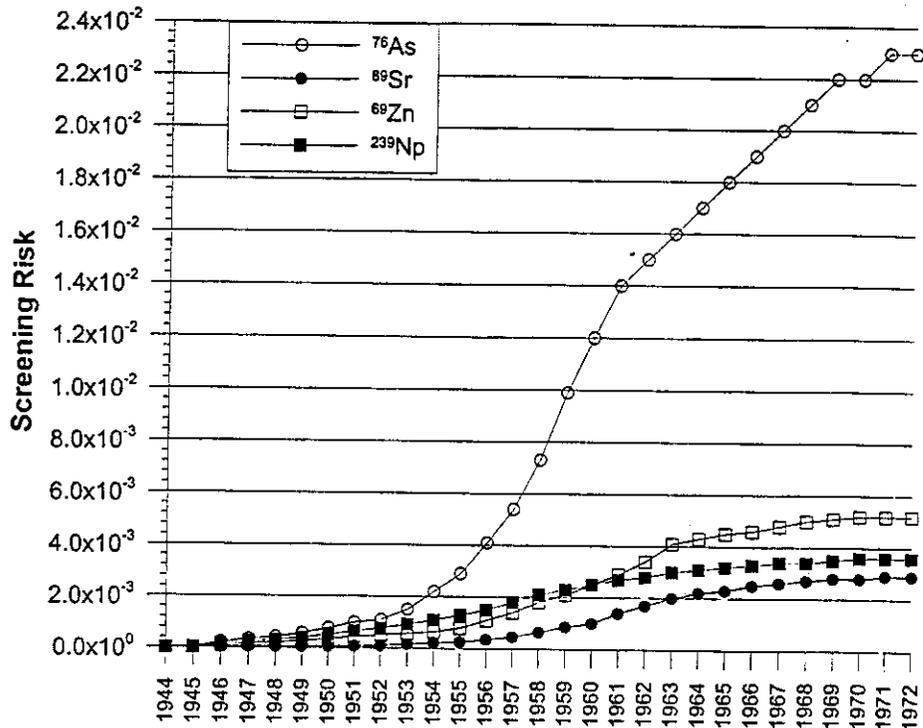


Figure 16. Cumulative screening risk at the 300 Area as a function of year for the initial screening. The four nuclides illustrated are the dominant risk contributors.

We used the three scenarios (local resident, migrant worker, and Native American) to prioritize the remaining nuclides according to each nuclide's contribution to the total risk for the given scenario (Table 17). In all scenarios, ⁷⁶As was the highest risk contributor. The strontium isotopes (⁸⁹Sr and ⁹⁰Sr) are important for the Native American scenario but are relatively unimportant for the local resident and migrant worker scenarios because the whole fish was assumed to be ingested for the Native American scenario compared to only ingestion of the filet for the local resident and migrant worker scenarios. The bioconcentration factor for strontium in fish bone is relatively high. Other important radionuclides include ⁶⁹Zn, ³²P, ²³⁹Np, ⁶⁵Zn and ²⁴Na.

Prioritization of the radionuclides resulted in different sets of significant nuclides for each exposure scenario. If we use a 1% cutoff (nuclides that contribute <1% to the total risk are not considered important), ⁶⁰Co is the only nuclide that is eliminated from further consideration. A detailed accounting of nuclide-specific risk by pathway can be found in Appendix E.

Table 17. Percent of Total Risk for the Local Resident, Native American, and Migrant Worker Scenarios

Local Resident		Native American			Migrant Worker			
Nuclide	300 Area	Richland	Nuclide	300 Area	Ringold	Nuclide	300 Area	Ringold
⁷⁶ As	34.9%	35.0%	⁷⁶ As	43.2%	49.9%	⁷⁶ As	34.1%	36.1%
⁶⁹ Zn	21.4%	20.2%	⁸⁹ Sr	18.6%	13.8%	⁶⁹ Zn	22.6%	18.6%
³² P	8.4%	8.9%	⁶⁹ Zn	9.1%	7.7%	²³⁹ Np	9.0%	12.2%
²³⁹ Np	8.1%	8.9%	²³⁹ Np	6.9%	10.0%	²⁴ Na	6.5%	4.3%
⁶⁵ Zn	7.2%	7.5%	⁹⁰ Sr	5.6%	4.1%	⁹⁵ Zr	5.6%	8.0%
²⁴ Na	6.1%	5.7%	⁶⁵ Zn	4.3%	3.9%	⁶⁵ Zn	5.0%	4.6%
⁹⁵ Zr	3.0%	3.2%	⁹⁵ Zr	3.5%	3.1%	³² P	2.7%	1.9%
⁷² Ga	1.8%	1.7%	³² P	3.4%	2.8%	⁷² Ga	2.6%	2.3%
¹³¹ I	1.7%	1.8%	²⁴ Na	1.1%	0.8%	¹³⁷ Cs	2.0%	3.6%
¹³³ I	1.5%	1.4%	⁷² Ga	1.0%	0.9%	⁵⁶ Mn	1.9%	1.3%
⁹⁰ Y	1.3%	1.3%	¹³⁷ Cs	0.7%	0.8%	⁹⁰ Y	1.6%	1.2%
⁶⁴ Cu	1.0%	0.9%	⁴⁶ Sc	0.6%	0.6%	¹³³ I	1.5%	1.2%
⁵⁶ Mn	0.9%	0.6%	¹³¹ I	0.5%	0.4%	⁶⁴ Cu	1.3%	1.0%
¹³⁷ Cs	0.8%	0.8%	⁹⁰ Y	0.4%	0.3%	¹³¹ I	1.1%	0.8%
⁸⁹ Sr	0.7%	0.7%	⁶⁴ Cu	0.3%	0.3%	⁴⁶ Sc	1.1%	1.4%
⁴⁶ Sc	0.6%	0.6%	⁵⁶ Mn	0.3%	0.2%	⁸⁹ Sr	0.6%	0.4%
⁹⁰ Sr	0.3%	0.3%	⁶⁰ Co	0.2%	0.3%	⁶⁰ Co	0.6%	1.0%
⁶⁰ Co	0.2%	0.2%	¹³³ I	0.2%	0.2%	⁹⁰ Sr	0.2%	0.1%

Uncertainty

We did not perform a comprehensive evaluation of uncertainty. Rather, we performed a limited uncertainty analysis to evaluate the potential for identifying false negatives during the screening process (i.e., removing a nuclide from consideration when it should have been retained). We applied the uncertainty factor derived earlier in this report to three radionuclides (⁷⁶As, ⁵¹Cr, and ¹²²Sb) considered in the initial screening. Arsenic-76 was chosen because it was the highest risk contributor. Chromium-51 and ¹²²Sb were chosen because they had screening risk values just below the 10⁻⁴ screening risk decision criterion.

The uncertainty expressed here only represents the estimated uncertainty in the source term and river transport model. Additional uncertainty also exists in the risk coefficients, transfer factors, and bioconcentration factors. Because of the screening nature of this study, uncertainty in these parameters was accounted for by treating these parameters deterministically and choosing conservative values for the initial screening. The HEDR Project considered uncertainty in the source term, transfer coefficients, bioconcentration factors, and exposure scenario parameters. Uncertainty in the transport model was considered insignificant and was not considered in the HEDR Project evaluation.

Uncertainty was evaluated by sampling from the distribution of annual-average *P/O* ratios for each year of the simulation, calculating the uncertainty factor, multiplying the uncertainty factor by the annual risk, and summing the risks for all exposure years. Distributions of total risk were developed from 2000 model trials. The uncertainty factor was assumed to be independent from year-to-year and nuclide-to-nuclide. Distributions of total risk (Figure 17) show that the

maximum calculated risk for ^{51}Cr is greater than the 10^{-4} risk decision criterion, but the 75th percentile of the distribution is less than 10^{-4} . For this reason, there is an argument for not excluding ^{51}Cr from further analysis. The maximum calculated screening risk for ^{122}Sb is less than the 10^{-4} risk decision criterion, indicating it is very unlikely that this nuclide will add to the overall risk in a detailed risk assessment. The minimum value from the distribution of ^{76}As risks was 1.0×10^{-2} which is substantially above the 1×10^{-4} risk decision criterion.

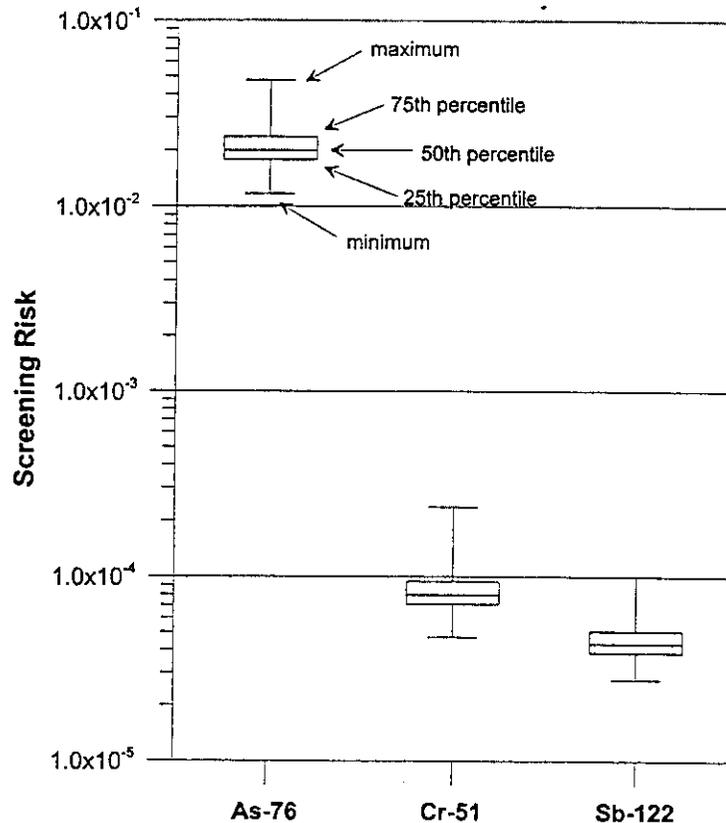


Figure 17. Uncertainty in the screening risk for the initial screening at the 300 Area location. Distributions were developed from 2000 trials.

Discussion

The five nuclides considered in the original HEDR dosimetry report (Farris et al. 1994) (^{24}Na , ^{32}P , ^{65}Zn , ^{76}As , and ^{239}Np) all contributed greater than 1% to the total risk in the initial screening calculations. However, other nuclides not considered in Farris et al. (1994) but included in this analysis were significant risk contributors, specifically $^{89,90}\text{Sr}$, ^{69}Zn , and ^{95}Zr . The screening risks for these radionuclides were primarily driven by the fish ingestion pathway. Furthermore, these radionuclides consistently ranked high in the subsequent prioritization that was based on three different exposure scenarios.

A comparison of exposure estimates from the HEDR Project (Farris et al. 1994) and estimates calculated in this study can be made by converting the HEDR Project estimated cumulative effective dose equivalent to risk for a given exposure scenario. In the HEDR Project a

maximum representative individual was defined to estimate the doses to a significant user of the Columbia River. This hypothetical individual approximated a segment of the general population who had maximum or near maximum ingestion rates for resident fish and waterfowl and spent time in or on the river. Of the three exposure scenarios defined in this study to prioritize the screened radionuclides, the Native American scenario represents the maximum exposed individual and is therefore compared to the maximum representative individual in the HEDR Project. A *median* annual fish consumption rate of 109 kg was assumed for the Native American scenario compared to a *maximum* annual fish consumption rate of 42.1 kg in the HEDR Project. This can be compared to the *maximum* annual fish consumption rate of 240 kg assumed for the initial screening. The median HEDR Project cumulative effective dose equivalent for exposures from 1950 to 1971 for the maximum representative individual at Richland was ~2000 mrem or 0.02 Sv (from Figure 5.2 in Farris et al. 1994). The 5th and 95th percentiles of the distribution were ~1500 mrem and ~4300 mrem, respectively. Assuming a risk of $7.3 \times 10^{-2} \text{ Sv}^{-1}$, the corresponding median risk to this individual is 1.5×10^{-3} , ranging from 1.1×10^{-3} to 3.2×10^{-3} .

In comparison, the total screening risk for the Native American scenario at the 300 Area for exposures from 1944 to 1972 was 2.3×10^{-2} (see Appendix E). This is roughly a factor of 15 higher than the median risk to the maximum representative individual in the HEDR Project. Most of the difference can be attributed to the fish consumption rates and the assumption that whole fish were ingested. Also, in the HEDR Project a holdup time between catch and consumption was assumed for the maximum representative individual. Holdup times were not used in this study. For some radionuclides, particularly ^{76}As , holdup times make a significant difference in the amount of radionuclide ingested via this pathway. Assuming whole fish ingestion in the Native American scenario resulted in the increased importance of the strontium isotopes as compared to the HEDR Project. Combined, these nuclides ($^{89,90}\text{Sr}$) contributed about 25% to the total risk. Another difference included the location of the individual. However, differences in river water concentrations between Richland and the 300 Area are minor compared to differences in fish consumption rates (109 kg y^{-1} compared to 42 kg y^{-1} for HEDR), and the inclusion of the strontium isotopes and ^{69}Zn in the screening risk calculation.

Prioritization of nuclides, as illustrated for the local resident, Native American, and migrant worker scenarios in Table 17 demonstrates the difficulties of using this approach in the absence of an absolute risk decision criterion to identify radionuclides of little significance to risk. Differences among receptor scenarios and locations result in different rankings of the radionuclides. For example, ^{90}Sr was ranked 18th for the local resident scenario but was ranked 4th for the Native American scenario and 11th for the migrant worker scenario. However, using a less restrictive threshold, for example 5% of the total risk, and a qualitative evaluation of the ranking yields several nuclides that consistently show up at the bottom of the ranking. These nuclides include ^{46}Sc , ^{90}Y , ^{137}Cs , ^{60}Co , ^{133}I , ^{131}I , and ^{56}Mn . Another way to analyze the results is to identify those radionuclides that consistently rank high. Applying this procedure to the local resident, Native American, and migrant worker scenarios identifies the same five radionuclides (^{32}P , ^{76}As , ^{65}Zn , ^{24}Na , and ^{239}Np) that were identified in the HEDR Project along with $^{89,90}\text{Sr}$, ^{69}Zn and ^{95}Zr as potentially significant in terms of overall risk.

In a review of the HEDR dose estimates for ATSDR, Hoffman et al. (1998) suggested that ^{60}Co , ^{90}Sr and ^{131}I should also have been included in the HEDR dose calculations for the Columbia River. The concern with ^{60}Co related to the potential buildup of ^{60}Co in sediments, which is accounted for explicitly in the river transport model used in this study. Although ^{60}Co

remained after the initial screening, the screening risk value based on all potential exposure pathways was just above the 10^{-4} risk decision criterion that we applied. Furthermore, the screening risk value for external exposure to shoreline sediments via all radionuclides was below 10^{-4} , indicating this is not a significant exposure pathway for historical radionuclide releases to the Columbia River. For the three exposure scenarios (local resident, Native American and migrant worker) ^{60}Co was consistently one of the least important radionuclides. Therefore the current screening results do not indicate that ^{60}Co should have been included in the HEDR dose calculations.

Hoffman et al. (1998) was concerned that exposure to ^{90}Sr from consuming whole fish (including the bones), and not just fish filets may have resulted in the risks to certain groups of Columbia River users being underestimated in the HEDR Project. In this study this was identified as a realistic potential exposure pathway, for Native Americans in particular, and the screening analysis supports the suggestion of Hoffman et al. (1998) that ^{90}Sr should have been included in the HEDR dose calculations.

Hoffman et al. (1998) considered that for assessing exposures to ^{131}I the dose to the thyroid is the appropriate endpoint as compared to the effective dose equivalent. Based on the screening risk values calculated in this study, ^{131}I was not screened out if a 10^{-4} risk decision criterion was applied. In all three scenarios (local resident, migrant worker, and Native American) used to prioritize the remaining radionuclides, ^{131}I accounted for less than 2% of the total risk, and consistently ranked outside the top 8. Therefore, our results indicate that ^{131}I for the Columbia River pathway does not merit high priority should further analyses of risk be undertaken.

CONCLUSIONS

Based on the screening analysis results presented in this report, there appear to be a number of radionuclides beyond the five for which dose estimates were calculated in the HEDR Project that may have warranted additional analysis. The initial screening was applied to 23 radionuclides released to the Columbia River using a risk criterion of 10^{-4} and indicated that only ^{45}Ca , ^{51}Cr , ^{93}Y , and ^{122}Sb could be eliminated from the analysis.

The screening results support the HEDR Project conclusion that fish ingestion is the dominant exposure pathway for releases to the Columbia River. However, the significance of this pathway for Native American users of the river may have been underestimated in the HEDR Project because fish consumption rates reported for Native Americans tend to be higher than the value assumed for the maximum representative individual in the HEDR Project. Evaluation of the exposure pathways also indicated it is reasonable to assume the entire fish was consumed. This increases the dose and risk for a number of radionuclides, in particular $^{89,90}\text{Sr}$. The screening analysis demonstrated that a number of exposure pathways could be eliminated from the analysis including, external exposure to contaminated sediments, ingestion of contaminated sediments and inhalation of contaminated aerosols. Of the remaining pathways, exposure to contaminated sediments through dermal contact, and ingestion of contaminated river water during swimming were low priority.

To prioritize radionuclides and pathways, three different exposure scenarios were developed for Native Americans, local residents, and migrant workers which evaluated less conservative situations of exposure for these river users. Using a criterion of <1% contribution to the total risk for all pathways identified ^{60}Co for all 3 scenarios. Therefore, ^{60}Co could be eliminated from

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further analysis. The relative ranking of the radionuclides for the 3 scenarios (Table 17) in terms of percentage contribution to the total risk showed that some radionuclides are more significant than others. In addition to the five radionuclides (^{76}As , ^{32}P , ^{239}Np , ^{65}Zn , and ^{24}Na) for which detailed dose calculations were made in the HEDR Project, ^{69}Zn and ^{95}Zr emerged as important risk contributors for all three scenarios, and ^{89}Sr and ^{90}Sr are clearly of high priority for the Native American scenario. If further evaluation of risks from radionuclides released to the Columbia River is undertaken, these nine radionuclides should be considered as most important for the analysis.

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APPENDIX A
STATEMENT OF WORK

APPENDIX A — STATEMENT OF WORK

Introduction

The Centers for Disease Control and Prevention (CDC) is sponsoring two major environmental and health impact studies relating to operation of U.S. Government facilities at the Hanford Nuclear Site, in Washington State. These studies are known as the Hanford Environmental Dose Reconstruction (HEDR) Project, and the Hanford Thyroid Disease Study (HTDS). In addition, CDC sponsors a Cooperative Agreement with the Washington Department of Health to develop and administer the Hanford Individual Dose Assessment (IDA) Project.

The HEDR Project is funded as part of a Memorandum of Understanding between the Department of Energy (DOE) and the Department of Health and Human Services (DHHS). The CDC has been assigned the role of directing the HEDR Project. The primary purpose of the HEDR Project is to reconstruct doses to offsite members of the public resulting from radionuclide releases since 1944. The HTDS is funded directly through Congress; its primary purpose is to determine if individuals exposed to radioactive Iodine (primarily 1-131) released from the Hanford facility have an increased incidence of thyroid disease. The Hanford IDA Project will allow individuals exposed to Hanford radiation releases to estimate their individual radiation doses.

CDC and the Agency for Toxic Substances and Disease Registry (ATSDR) have established the Hanford Health Effects Subcommittee (HHES), a Federal advisory committee, whose members provide advice to CDC on community concerns about CDC's activities in their community. Contractors for the HEDR Project must work with the Subcommittee.

Task Description

The HEDR Project developed the Columbia River Dosimetry Code (Farris et al., 1994) to calculate radiation doses for hypothetical individual users of the Columbia River at various locations on the river. Initially, the HEDR Project considered all radionuclides released from the Hanford Nuclear Site between 1944 and 1972. --Ultimately, doses were calculated for five radionuclides: Sodium-24, Phosphorus-32, Zinc-65, Arsenic-76, and Neptunium-239. The water concentrations for these radionuclides that were used in the dose calculations were provided by the CHARIMA computer code (Walters et al., 1994).

The five radionuclides listed above for which dose calculations were made were selected by the Technical Steering Panel of the HEDR Project from the initial list of released radionuclides on the basis of a series of scoping or screening calculations (Napier, 1993). The radionuclide exposure pathways considered in the dose calculations were also selected on the basis of scoping calculations. Hoffman et al. (1997) suggest that Iodine-131, Cobalt-60, and Strontium-90 should also be considered in a Hanford IDA process for the Columbia River. The objective of this task is to perform screening calculations that can be used to evaluate this recommendation.

Responsibilities of the Contractor

All plans and reports will be submitted to CDC as draft reports for review and approval. The contractor shall be responsible for the deliverables listed below. Specifically, the contractor shall perform the following:

1. Review all of the available HEDR Project documents related to the published Columbia River dose calculations, and select the best available information relating to the quantities of each of the eight radionuclides listed above that were released to the Columbia River between 1944 and 1972. The contractor shall NOT develop any new information on estimates of the radionuclide releases to the Columbia River without the approval of the Project officer.
2. Review risk-based screening limits which might be used by CDC as a decision criteria for choosing radionuclides for further consideration in the development of an individual dose assessment code for the Columbia River pathway.
3. Develop a screening methodology that accounts for all potential pathways of exposure for each of the eight radionuclides listed above. Organ-specific health risk and not just radiation dose should be the end point of the screening calculations. All mathematical models and parameter values' selected for use in the methodology should be carefully justified and thoroughly referenced.
4. After the developed screening methodology has been approved by the Project Officer, perform screening calculations for the eight radionuclides referenced above. Screening calculations for a limited number of additional radionuclides may be proposed, but no screening calculations will be performed for any additional radionuclides without the concurrence of the Project Officer.
5. As a result of these screening calculations, formulate for CDC's consideration recommendations with regard to the inclusion of further radionuclides and pathways in future Columbia River individual dose calculations.
6. During the performance of this task, contractor staff will work with the health agencies of the States of Washington, Oregon, and Idaho, the Hanford Health Effects Subcommittee, the Agency for Toxic Substances and Disease Registry, Native American tribes and the public to insure full public participation in the decision making process associated with the performance of these screening calculations. This includes:
 - a. Attending up to six public meetings in the northwest United States to explain the status of the work, review all documents and major decisions, and respond to questions.
 - b. Publication of one fact sheet at the end of the task to explain the objectives of the task and the significance of the findings.

In addition, the contractor shall provide appropriate representation at periodic contract status meetings and/or meetings with other Government contractors as may be determined appropriate by the Project Officer.

The contractor shall also deliver to the Project Officer periodic letter status reports, as described in Item 11. Letter status reports shall be brief two-to-three page documents summarizing activities and verifying in detail all expenditures.

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Farris, WT, Napier, BA, Simpson, JC, Snyder, SF, Shipler, DB. (1994) Columbia River Pathway Dosimetry Report, 1944-1992. PNWD2227 HEDR, Battelle Pacific Northwest National Laboratories, Richland, Washington.

Napier, BA. (1993) Determination of key radionuclides and parameters related to dose from the Columbia River pathway. BN-SA3768 HEDR, Battelle Pacific Northwest National Laboratories, Richland, Washington.

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APPENDIX B
ANNOTATED BIBLIOGRAPHY OF SELECTED REPORTS
RELATED TO THE HEDR PROJECT

**APPENDIX B — ANNOTATED BIBLIOGRAPHY OF SELECTED REPORTS
RELATED TO THE HEDR PROJECT**

Napier, B.A. 1991. *Selection of Dominant Radionuclides for Phase I of the Hanford Environmental Dose Reconstruction Project*. PNL-7231 HEDR UC-707, Pacific Northwest Laboratory, Richland, Washington. July.

This report documents the selection of the dominant radionuclides (those that may have resulted in the largest portion of the received doses) in the source term for atmospheric releases (1944–1947) and surface water releases (1964–1966). Because the early releases from Hanford operations were largely continuous and each radionuclide retained the same relative fractional contribution to the total released activity, radionuclides were ranked based on a unit source term release.

Actual measurements of radionuclide concentrations in river water were used to determine the dominant radionuclides. Appendix E provides measurements. Contributions from groundwater migration to the river are implicit in the measured values for surface water, therefore no additional calculations were performed for groundwater releases.

A range of potential exposure pathway conditions and individual exposure mechanisms were investigated. Potential variability in the source term (reactor power levels and fuel conditions) was addressed because monitoring data for a number of years was reviewed. The selection of dominant radionuclides was made based on those frequently occurring in the resulting lists. GENII code was used. Appendix C provides parameter inputs and results.

Exposure pathways: drinking contaminated water, recreation in or near contaminated water, consumption of fish, irrigation with contaminated river water and consumption of contaminated produce, exposure to soils contaminated by the water, inhalation of resuspended dusts from such soils. Three variations were considered: drinking water only, exposure from shoreline and river recreational activities, Columbia River fish ingestion as main dietary source. Consumption rates for this were taken from Hunn and Bruneau (1989). "Estimations of Traditional Native American Diets in the Columbia Plateau."

Inclusion of the following five radionuclides in the dose calculations was considered essential: ^{32}P , ^{239}Np , ^{65}Zn , ^{76}As , and ^{64}Cu , with the following four highly desirable: ^{56}Mn , ^{24}Na , ^{46}Sc , and ^{51}Cr . All intermediate calculations are presented in the appendices to the document.

Pacific Northwest Laboratory. 1991. *Columbia River Pathway Report: Phase I of the Hanford Environmental Dose Reconstruction Project*. PNL-7411 HEDR Rev 1, Pacific Northwest Laboratory, Richland, Washington.

This document outlines Phase I of the HEDR Project, including the Phase I screening calculations. For Phase I, the years 1964–1966 were analyzed because of the wealth of environmental data, independent measurements, relatively high river concentrations, and the Richland population having been recently exposed to contamination via groundwater. Phase I still looked at the eight radionuclides identified as important in the initial screening— ^{32}P , ^{65}Zn , ^{76}As , ^{239}Np , ^{56}Mn , ^{51}Cr , and ^{64}Cu —because they were estimated to deliver more than 80% of the total dose to a maximally exposed individual. A simple routing model using only effluent measurement and river discharge as inputs used radioactive decay and mixing to estimate concentrations at downstream locations.

This document contains the routing equation used to calculate downstream concentrations, assuming

- Flow and transport can be represented as steady-state on a monthly basis
- Effluent discharge rates are constant each month
- Radionuclides are completely mixed in a cross section of the river at any location between Priest Rapids Dam and McNary Dam
- Effluent spent a short time in retention basins (~4 hours)
- Radionuclide sources and sinks are neglected (e.g., no sediment buildup).

The highest doses were estimated for individuals who consumed large quantities of fish and who drank untreated river water.

Napier, B.A. and A.J. Brothers. 1992. *Recommendations to the Technical Steering Panel Regarding Approach for Estimating Individual Radiation Doses Resulting from Releases of Radionuclides to the Columbia River. Volume 1: Recommendations*. PNWD-1977 HEDR Vol.1. Battelle Pacific Northwest Laboratories, Richland, Washington, July.

This document includes information regarding the decision process used to evaluate what work and level of effort should be undertaken throughout the remainder of the HEDR Project related to the Columbia River Pathway. Weighed as they impacted the study were minimizing cost, maximizing utility of derived information, being as complete as possible, minimizing the uncertainty of the results, and maintaining consistency with the Technical Steering Panel (TSP) guidance dose level of 100 mrem y^{-1} . When this dose criterion was exceeded, the TSP recommended that some additional effort go into characterizing the dose.

Napier, B.A. 1993. *Determination of Key Radionuclides and Parameters Related to Dose from the Columbia River Pathway*. BN-SA3768 HEDR, Battelle Pacific Northwest National Laboratories, Richland, Washington.

Appendix A contains incomplete individual reactor source terms ($Ci\ d^{-1}$) for 1959–1971 by month for 19 radionuclides. There are many gaps in the data. Appendix C contains the release estimates from all eight reactors ($Ci\ d^{-1}$) by month for same radionuclides (with the same gaps in the data), river flow rate (cfs), and estimated travel time (d). Exposure factors and doses are provided.

Incomplete source term information for 19 radionuclides for 1959–1970 was used for the scoping calculations. Effective doses for “maximum” and “average” individuals were calculated based on 200 realizations for external exposures via swimming and boating; ingestion via drinking water, and fish consumption. The year 1961 was identified as the peak dose year for the period studied. Five radionuclides, ^{24}Na , ^{32}P , ^{65}Zn , ^{76}As , and ^{239}Np , were identified for further study in HEDR. The following radionuclides were eliminated from further analysis because only a few percent of the total dose came from them: ^{45}Ca , ^{46}Sc , ^{56}Mn , ^{51}Cr , ^{69}Zn , ^{69m}Zn , ^{89}Sr , and ^{90}Sr .

Heeb, C.M. and D.J. Bates. 1994. *Radionuclide Releases to the Columbia River from Hanford Operations, 1944-1971*. PNWD-2223 HEDR UC-000. Battelle Pacific Northwest National Laboratories, Richland, Washington. May.

The curie quantities of 11 radionuclides and gross nonvolatile beta activity discharged to the Columbia River are estimated on a monthly basis for the period 1944-1971. This covers the entire operating history of the eight Hanford single-pass reactors. Uncertainties in the estimates were determined. All the release estimates are made on either activity concentration measurements made during the time period of the release or on inferred values resulting from a statistical analysis of data from other time periods. One hundred Monte Carlo STRRM realizations of the Columbia River releases were made to generate release distributions. Scoping calculations were repeated to confirm that the five radionuclides used in the detailed HEDR dose calculations were the most important ones. Appendix B provides a tabulation of minimum, median, maximum monthly release estimates for ^{24}Na , ^{32}P , ^{65}Zn , ^{76}As , ^{239}Np , ^{131}I , ^{90}Y , ^{72}Ga , ^{51}Cr , ^{56}Mn , and ^{46}Sc .

Walters, W, M.C. Richmond, and B.G. Gilmore. 1994. *Reconstruction of radionuclide concentrations in the Columbia River from Hanford, Washington to Portland Oregon, January 1950-January 1971*. PNWD2225 HEDR, Battelle Pacific Northwest National Laboratories, Richland, Washington. May.

Monthly average water concentrations were reconstructed at 12 locations along the Columbia River downstream of the Hanford Site for ^{24}Na , ^{32}P , ^{65}Zn , ^{76}As , and ^{239}Np based on the recommendations in Napier (1993). The calculated concentrations are presented in Appendix A. The ^{51}Cr concentrations were also computed for model validation purposes (not for dose estimates: low contribution to dose).

A 21-year period from January 1950 through January 1971 was evaluated for the Columbia River from Priest Rapids Dam near Hanford to just downstream of the Willamette River confluence at Portland, Oregon. The TSP approved this period of study because

- Period of highest releases was from 1955-1965 when production was at its maximum.
- Five years were added to each end of this time period to ensure adequate coverage.
- The last of the single-pass production reactors was shut down in January 1971.

WSU-CHARIMA, a 1-D finite difference model that simulates unsteady flow hydraulics and nonuniform sediment transport in open channel systems was used to compute water concentrations. The CHARIMA model was modified to include radioactive decay in the transport equation, and the version was named WSU-CHARIMA. Model testing indicated that correction for sediment uptake and release was not feasible. This omission was considered to have a negligible impact except for the Portland location where concentrations for ^{65}Zn were overestimated. The Portland location is also influenced by tidal-effects.

The relatively short-lived radionuclides ^{24}Na , ^{76}As , and ^{239}Np were sensitive to downstream travel time. Transport velocities were greatly reduced after dams were constructed below the Snake River. The water concentrations of these three radionuclides at the downstream locations were much lower than they would have been under open channel conditions. Because of their longer half-lives, ^{32}P and ^{65}Zn were not affected by dam construction to any significant extent.

Farris, W.T., B.A. Napier, J.C. Simpson, S.F. Snyder, and D.B. Shipler. 1994. *Columbia River Pathway Dosimetry Report, 1944–1992*. PNWD2227 HEDR, Battelle Pacific Northwest National Laboratories, Richland, Washington.

This report contains overview information on the technical approach, model development, final bioconcentration factor data, transmission factors (for fraction of radionuclides that pass through treatment process). It documents the simple equations used in the Columbia River Dosimetry code to calculate radionuclide concentrations and doses from each pathway, and lists the scenario parameters used in the HEDR exposure scenarios.

Hunn, E. no date. *Estimations of Traditional Native American Diets in the Columbia River Plateau*. PNL-SA-17296 HEDR.

A draft report prepared for the HEDR Project that provides weekly per-capita consumption levels by season for adult male Native Americans within a 10-county target area. Traditional and nontraditional food categories for three tribal groups (River Yakima, Nez Perce, and Colville) are presented. The food categories are exposed vegetables; other vegetables; grains; fruits/berries; wild bird eggs; game; wild birds; anadromous fish (salmon, steelhead trout, and lamprey eels); other fish (suckers, trout, and whitefish); shellfish; blacktree moss (an exposed vegetable); and water.

Hoffman F.O., A.I. Apostoaei, J.S. Hammonds, K.M. Thiessen, B.G. Blaylock, and B.A. Thomas. 1998. *Estimation of Health Risks Based on Revised Estimates of HEDR Doses for Maximum Representative Individuals Consuming Fish and Waterfowl from the Columbia River: An Evaluation of HEDR Reports on the Columbia River*. SENES, Oak Ridge, Inc., Oak Ridge, Tennessee.

This report reviews the HEDR dose estimates and presents revised estimates of the doses from ingestion of fish and waterfowl based on modified estimates of bioconcentration factors (BCFs) for fish and waterfowl and on expanded organ-specific dosimetry. Estimates of relative risk and lifetime risk were made from the revised dose estimates. The HEDR reports were reviewed and the following possible sources of bias in the HEDR dose calculations were identified:

- Scoping studies – by looking at annual average dose for only 3 years, Hoffman et al. indicate that it is possible to overlook some pathways that might contribute more significantly after radionuclide concentrations have built up (e.g., irrigation pathway and external exposure to shoreline sediments). Scoping studies may also have ruled out radionuclides because of the scoping methodology (e.g., ⁹⁰Sr in fish bones and ⁶⁰Co in sediments).
- Bioconcentration factors – the methodology used to create BCFs distributions was flawed because only natural variability was addressed; the uncertainty about a measured mean was not addressed. Hoffman et al. suggest that it is better to use the available data and take the arithmetic mean to produce a median value about which a distribution determined using

APPENDIX C
COLUMBIA RIVER EXPOSURE SCENARIO ACTIVITY
CATEGORIES

- available scientific knowledge. Also, BCFs based on fish filets rather than on the entire organism would tend to underestimate the doses for certain radionuclides (^{90}Sr)
- In the HEDR Project the BCF for salmon was estimated to be the same as that for a second-order predator fish. This overestimates the dose from salmon ingestion.
 - In the HEDR Project the holdup times between fish harvest and consumption were calculated assuming a combination of fresh fish consumption and frozen or dried fish consumption. This underestimates the doses for individuals who consumed fresh fish only.
 - The ingestion rates (annual and seasonal) for fish and waterfowl for a maximum representative individual in the HEDR Project could underestimate the intake for individuals whose primary source of food was the Columbia River.
 - Target organs were not correctly identified in the HEDR Project. The most recent publication of ICRP dose conversion factors across a wide range of organs makes this easier.
 - Uncertainty may not have been properly evaluated.
 - There was no age specific evaluation of dose in the HEDR Project.

Most of the recommendations revolve around the calculation of dose and not the source term and transport calculations.

APPENDIX C — COLUMBIA RIVER EXPOSURE SCENARIO ACTIVITY CATEGORIES

This list identifies general activity categories that may result in distinctive exposures to Columbia River borne contaminants. For each category, at least one source is mentioned that attributes the activity to the Native peoples of the Columbia River Plateau. A more extensive bibliography follows.

Further discussion is needed about seasonal variation in consumption/exposure and about deriving specific range of consumption/exposure values for different age/gender/tribal cohorts.

It is notable that very few plant and animal categories listed in comprehensive inventories collected in 1883 (Everette) and Curtis (1907–1930) could not be verified by Native consultants in 1976–1992 (Hunn et al. 1998: 525).

Ingestion

Drinking water: untreated river water, treated river water

Fresh salmon and steelhead, lamprey (mostly skin), smelt (mostly skin, but also organs), shad (Columbia River Inter-Tribal Fish Commission 1994; Hewes 1998: 623-624)

Organs, bones, eggs, head, skin, fillet

Fried, baked, broiled, or roasted

Air-dried salmon and steelhead, lamprey, smelt, shad (Columbia River Inter-Tribal Fish Commission 1994)

Organs, bones, eggs, head, skin, fillet

Smoked salmon and steelhead - fillets cooked by the fire or whole fish wrapped in leaves or mud and cooked in the coals (Wallulatum 1977: 187)

Salmon pemmican, or "sugared salmon" (soaked in steelhead oil and set on tule mats to dry and drain excess oil) (Stern 1998: 643)

Fresh resident fish - omnivorous (bullhead, catfish, suckers, whitefish, chiselmouth, carp, sturgeon), first-order predators (trout, whitefish, walleye, squawfish), second-order predators (e.g., sucker) (Columbia River Inter-Tribal Fish Commission 1994)

Organs, bones, eggs, head, skin, fillet

Baked or broiled

Air dried resident fish - bottom feeders, first-order predators, second-order predators (Columbia River Inter-Tribal Fish Commission 1994)

Organs, bones, eggs, head, skin, fillet

Salmon oil also reported as antidote for poisoning from Indian hellebore, water hemlock (*Cicuta douglassii*), death camas (*Zigadenus venenosus*) and baneberry (*Actaea arguta*) (Hunn et al. 1998: 535).

Waterfowl - e.g., puddle ducks (ATSDR 1998)

Watercress (*Rorippa nasturtium-aquaticum*) and other vegetation (Hunn et al. 1998: 527)

Acorns of garry oak (*Quercus garryana*) were also of some significance in the Columbia Gorge area, where they were baked underground after leaching in "blue" mud (Hunn et al. 1998: 530)

Camas bulbs, cooked in a pit (wood burned under rocks, with wet willow branches followed by clumps of wet alfalfa and rye grasses, followed by wet sacks of bulbs, covered by dirt) (Hunn et al. 1998: 529).

Seeds of yellow pond-lily (*Nuphar polysepalum*) known among the Klamath as “wokas” (Hunn et al. 1998: 530-531).

Teas (Labrador tea – *Ledum groenlandicum*; mint – *Mentha arvensis*; wild bergamot – *Monarda fistulosa*; wild rose stems and flowers). Mint, wild bergamot and some wormwoods (*Artemisia* spp.) were also used as preservatives to repel flies and other insects from meat, fish or berries being dried or stored. (Hunn et al. 1998: 535)

Basket and mat-weaving (oral contact to wet reed tips while weaving) – woven for mats (for berry-drying and fish draining) and bags of various types; tule stems (*Scirpus lacustris*), cattail leaves (*Typha latifolia*) and stems of common reed grass (*Phragmites australis*). Stem fiber of Indian hemp (*Apocynum cannabinum*) was used for cordage – fishnets, woven bags, capes – and also for a “time ball” of twine used to record key events in a person’s life (Uebelacker and Wilson 1984).

Cattail bags lined with salmon skin were used to store dried fish flesh that had been pounded into a powdered meal (up to 45 kg per bag) (Hunn et al. 1998: 540).

The oil that our people used to prepare the sugared salmon came from steelhead. Red salmon was air dried and eaten dry for lunches because the other methods of preparing fresh salmon used a lot of wood which was very difficult to get along the river. Large wooden troughs were needed with many heated rocks to bring water to a boil. Salmon fillets were put on sticks and cooked by the fire, and sometimes fish were wrapped in leaves or mud and cooked in the coals. (Wallulatum 1977: 187).

Inhalation

Aerosolized vapors from dip-net platforms near water falls (Hewes 1998: 623-624)

Smoke from camp-fires with fuel wood from the river's edge

Sweat lodge (river water vaporizes when applied to hot rocks in well-insulated enclosure) (Harris and Harper 1997: 794)

Sweat lodge – aromatic plants such as juniper, wormwoods and yarrow inhaled as vapor as treatment for respiratory ailments or fever (Hunn et al. 1998: 535).

Bioavailability/Dermal Absorption

Wading, Swimming – especially for setting fishing nets (Hunn et al. 1998, Hewes 1998)

Boating – especially for fishing (Hunn et al. 1998, Hewes 1998)

Dip-net platforms near water falls (Hunn et al. 1998, Hewes 1998)

Sweat lodge (river water vaporizes when applied to hot rocks in well-insulated enclosure) (Harris and Harper 1997: 794); external washes of plant solutions as treatments for arthritis, rheumatism and muscular pains reported in Hunn et al. (1998: 535).

Fish belly fat is rendered and used as a base for body paint (Harris and Harper 1997: 794; Hunn et al. 1998: 534)

Basket-making (cuts on hands from sharp edges while weaving) (Harris and Harper 1997: 794)

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APPENDIX D
INPUT FILE FORMATS AND USER INSTRUCTIONS
FOR THE RVRDSP

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Risk Assessment Corporation
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APPENDIX D — INPUT FILE FORMATS AND USER INSTRUCTIONS FOR THE RVRDSP CODE

Concentrations in river water and sediments and lifetime cancer incidence risk calculations were performed within the code, RVRDSP that was written specifically for this project. The code is written in FORTRAN 77 and compiled using the Lahey EM32 on a personal computer with the Microsoft Windows 98 operating system. The program operates within the DOS command prompt. The code has also been compiled on a Linux workstation using the Lahey LF-95 Express compiler. Execution of the code is performed on the command line using the command argument

[*path*] RVRDSP [*filename*]

where *path* is the full or relative path to the executable (RVRDSP.EXE) and *filename* is the name of the input file. The filename argument is optional and if no filename is provided, the code will look for the default input file name called RVRDSP.PAR. If the input file name is not provided and the default input file is not found in the working directory, the code will abort.

Construction of the main input file is described in Table D-1. All input files are free-form ASCII, which may be created in any standard text editor. Each card represents one or more lines of input. Comments may be inserted between cards by placing a dollar sign (\$) in the first column. Suggested default values are identified in parentheses if applicable in Table D-1.

At least two other files are required by the code. The first file (Table D-2) contains the river flow rate, width, and, depth as a function of time. The remaining files required depend on the number of individual sources in the simulation. Each source requires a separate file that describes the effluent release rate as a function of time (Table D-3). The names of the flow rate file and source file are specified in the main input file on cards 3 and 9 respectively. A third file containing exposure factors used in the risk calculation (Table D-4) is optional.

The variables *jmax*, *jstart*, and *eps* in the main input file are used with the Simpson's rule integration routine. The routine evaluates the integral for a variable number of points and monitors the accuracy of the solution. The accuracy is checked by computing the integral using 2^n number of middle points plus the two end points and then adding 2^{n+1} number of middle points and comparing the result with the previous evaluation. The variable *jmax* defines the maximum number of iterations allowed before the integration routine is terminated. On the first iteration, the crudest approximation to the integral is performed by evaluating the function at the upper and lower limits. On each successive iteration, the number of middle points (points between the upper and lower limits of integration) evaluated are increased by a factor of 2 starting with one middle point added during the second iteration. The number of additional middle points is given by 2^{n-2} where n = the iteration number. The routine evaluates the integral for at least *jstart* number iterations before the $n-1$ solution is checked for convergence with the n^{th} solution. For example, if the variable *jstart* is set equal to 6, then 6 iterations are performed before convergence checking occurs. The number of middle points added on the 6th iteration is $2^{6-2} = 16$. Convergence is checked by calculating the *eps* value and comparing it to the user input *eps* value. The *eps* variable is given by

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$$EPS = \frac{|\Theta_p - \Theta_c|}{\Theta_c} \quad (D-1)$$

where

Θ_p = previous evaluation of the integral

Θ_c = current evaluation of the integral.

If the calculated eps is less than the user input eps, then the routine is terminated and the current evaluation of the integral is returned. If the calculated eps is greater than the user input eps, then 2^{n-2+1} number of middle points are added and a new value of the integral is calculated and checked. If convergence is not achieved in jmax number of iterations, then the routine is terminated and the current value of the integral is returned. A message warns the user of non-convergence and the current eps value is also printed. An adequate solution does not necessarily require convergence to be met and depends on the values of the integration variables jmax, jstart, and eps. Unacceptable solutions are usually detected by observing the concentration verses time output. If there are perturbations in the concentration versus time curve that are not accounted for by the source release model, then the eps value should be set lower and jmax increased.

Table D-1. Parameter Definition File for the RVRDSP Program

Card	Code variable	Type/format	Units	Description
1	Title	CHAR/A80		Title of run
2	Fileout	CHAR/A60		Output file name
3	Fileflow	CHAR/A60	$M^3 s^{-1}$	File containing river flow rate as a function of time
4	Jstart	INT/*		Number of iterations to perform in the Simpson's rule integration before convergence is checked (6)
4	jmax	INT/*		Maximum number of iterations to perform in the Simpson's rule integration (12)
4	eps	REAL/*		Convergence criteria for Simpson's rule integration
5	nsrc	INT/*		Number of sources (maximum = 10)
5	nrec	INT/*		Number of receptors (maximum = 1000)
5	ntimes	INT/*		Number of output time periods (maximum = 50)
5	idisp	INT/*		Flag variable (0) = use fixed dispersivity values; (1) = calculate dispersivity values based on width, depth, and flow rate
5	ised	INT/*		Flag variable (0) = do not calculate activity in accumulating sediment; (1) = calculate activity in accumulating sediment
5	irisk	INT/*		Flag variable (0) = do not calculate risk; (1) calculate risk. NOTE if irisk=1 then an exposure factor file is needed.
NOTE: Card 6a is read only if IDISP = 1				
6a	slope	REAL/*	$M m^{-1}$	average channel slope
6a	Fyakima	REAL/*	$M^3 s^{-1}$	Annual average flow rate of the Yakima River
6a	Fsnake	REAL/*	$M^3 s^{-1}$	Annual average flow rate in the Snake River
6a	beta	REAL/*		Unitless coefficient used to calculate transverse dispersivity (0.6)
NOTE: Card 6b is read only if IDISP = 0				

Table D-1. Parameter Definition File for the RVRDSP Program

Card	Code variable	Type/format	Units	Description
6b	Ex	REAL/*	$m^2 s^{-1}$	Longitudinal dispersivity
6b	Ey	REAL/*	$m^2 s^{-1}$	Transverse dispersivity
6b	Fyakima	REAL/*	$m^3 s^{-1}$	Annual average flow rate of the Yakima River (105)
6b	Fsnake	REAL/*	$m^3 s^{-1}$	Annual average flow rate in the Snake River (1530)
7	thalf	REAL/*	days	Radionuclide half-life
7	rho	REAL/*	$g cm^{-3}$	Bulk density of bed sediments (1.2)
7	vd	REAL/*	$m d^{-1}$	Deposition velocity of suspended sediments (0.07)
7	yshore	REAL/*	m	Distance from channel centerline where exposure to shoreline sediments are computed (200).
NOTE: Card 8 is read ntimes number of times. The time is the number of days from January 1, 1944				
8	t1(i)	REAL/*	days	Beginning time of simulation for time period <i>i</i>
8	t2(i)	REAL/*	days	Ending time of simulation for time period <i>i</i>
8	tp(i)	REAL/*	days	Print time step of simulation for time period <i>i</i>
NOTE: Card 9 is read nsrc number of times				
9	Xa(i)	REAL/*	m	Downstream distance (as measured from River Mile 385) for the <i>ith</i> source
9	Ya(i)	REAL/*	m	Transverse distance from near shore for the <i>ith</i> source
9	filesrc(i)	CHAR/A60	$Ci d^{-1}$	Source file names for the <i>ith</i> source.
NOTE: Card 10 is read nrec number of times				
10	xdist(i)	REAL/*	m	Downstream distance (as measured from River Mile 385) for the <i>ith</i> receptor
10	ydist(i)	REAL/*	m	Transverse distance from near shore for the <i>ith</i> receptor
10	Tk(i)	REAL/*	m	Bed sediment thickness for the <i>ith</i> receptor
10	Sc(i)	REAL/*	$g m^{-3}$	Suspended sediment load at the for the <i>ith</i> receptor
10	kd(i)	REAL/*	$mL g^{-1}$	Sorption coefficient at the for the <i>ith</i> receptor
NOTE: Card 11 is read only if irisk = 1				
11	filerisk	CHAR/*		Exposure factors/risk coefficients file

Table D-2. Description of the Flow Rate, Width and Depth Input File

Line number	Code variable	Description
1	Junk	Column header (discarded)
2-n+1 ^a	F(i,1)	Days from January 1, 1944 for the <i>i</i> th record
2-n+1 ^a	F(i,2)	Flow rate ($m^3 s^{-1}$) for the <i>i</i> th record
2-n+1 ^a	F(i,3)	River width (m) for the <i>i</i> th record
2-n+1 ^a	F(i,4)	River depth (m) for the <i>i</i> th record

^a *n* is the number of time, flow rate, width, and depth records. A minimum of two records are needed to operate the code.

Table D-3. Description of the Source Term Input File

Line number	Code variable	Description
1	Junk	Column header (discarded)
$2-n+j^a$	$Q(j, i, 1)$	Days from January 1, 1944 for the i^{th} record and j^{th} source.
$2-n+j^a$	$Q(j, i, 2)$	Release rate for the i^{th} record and j^{th} source (Ci d^{-1}).

^a n is the number of time, release rate records. A minimum of two records are needed to operate the code.

Table D-4. Description of the Exposure Factor File

Line number	Code variable	Units	Description
1	junk		Column header (discarded)
2	ef(1,j)	$\text{L d}^{-1} \text{Ci}^{-1}$	Exposure factors/risk coefficients for the drinking water pathway, for j^{th} month (j=1, January; j=12, December)
3	ef(2,j)	$\text{L d}^{-1} \text{Ci}^{-1}$	Exposure factors/risk coefficients for the fish ingestion pathway, for j^{th} month (j=1, January; j=12, December)
4	ef(3,j)	$\text{L d}^{-1} \text{Ci}^{-1}$	Exposure factors/risk coefficients for the swimming immersion pathway, for j^{th} month (j=1, January; j=12, December)
5	ef(4,j)	$\text{L d}^{-1} \text{Ci}^{-1}$	Exposure factors/risk coefficients for the swimming ingestion pathway, for j^{th} month (j=1, January; j=12, December)
6	ef(5,j)	$\text{L d}^{-1} \text{Ci}^{-1}$	Exposure factors/risk coefficients for the waterfowl pathway, for j^{th} month (j=1, January; j=12, December)
7	ef(6,j)	$\text{m}^2 \text{d}^{-1} \text{Ci}^{-1}$	Exposure factors/risk coefficients for the sediment external pathway, for j^{th} month (j=1, January; j=12, December)
8	ef(7,j)	$\text{m}^2 \text{d}^{-1} \text{Ci}^{-1}$	Exposure factors/risk coefficients for the sediment dermal contact pathway, for j^{th} month (j=1, January; j=12, December)
9	ef(8,j)	$\text{m}^2 \text{d}^{-1} \text{Ci}^{-1}$	Exposure factors/risk coefficients for the sediment ingestion contact pathway, for j^{th} month (j=1, January; j=12, December)
10	ef(9,j)	$\text{L d}^{-1} \text{Ci}^{-1}$	Exposure factors/risk coefficients for the aerosol pathway, for j^{th} month (j=1, January; j=12, December) ^a
11	ef(10,j)	$\text{L d}^{-1} \text{Ci}^{-1}$	Exposure factors/risk coefficients for the produce ingestion pathway, for j^{th} month (j=1, January; j=12, December)
12	ef(11,j)	$\text{L d}^{-1} \text{Ci}^{-1}$	Exposure factors/risk coefficients for the meat ingestion pathway, for j^{th} month (j=1, January; j=12, December)
13	ef(12,j)	$\text{L d}^{-1} \text{Ci}^{-1}$	Exposure factors/risk coefficients for the milk ingestion pathway, for j^{th} month (j=1, January; j=12, December)

^a The aerosol pathway includes inhalation of aerosols generated at a waterfall or rapids and inhalation of steam generated during a sweat bath

Table E-4. Screening Risk Values for the Local Resident Scenario at Richland

Nuclide	Direct-Ing	Fish-Ing	Swim-			Sed-				Produce-				Total	% of Total
			Imm	Swim-Ing	Waterfowl	Sed-Ext	Dermal	Sed-Ing	Aerosol	Boating	Ing	Meat-Ing	Milk-Ing		
Zn-65	2.20E-05	1.60E-04	9.60E-08	1.00E-07	1.60E-05	1.50E-06	1.30E-06	6.50E-08	0.00E+00	9.60E-08	1.40E-05	4.80E-05	2.10E-05	2.84E-04	7.61%
Na-24	4.60E-05	2.30E-06	1.60E-05	2.30E-07	0.00E+00	1.80E-08	0.00E+00	1.30E-11	0.00E+00	1.60E-05	1.10E-06	3.80E-05	9.00E-05	2.10E-04	5.61%
I-131	1.30E-05	1.10E-05	7.60E-09	4.90E-08	0.00E+00	7.90E-11	6.60E-09	4.60E-11	0.00E+00	7.60E-09	5.60E-06	1.70E-05	1.90E-05	6.57E-05	1.76%
Y-93	6.60E-06	1.50E-07	9.20E-09	3.40E-08	0.00E+00	1.70E-11	0.00E+00	2.70E-12	0.00E+00	9.20E-09	1.20E-07	1.40E-07	2.10E-08	7.08E-06	0.19%
Cr-51	7.90E-06	2.10E-06	1.10E-07	3.50E-08	0.00E+00	1.90E-09	0.00E+00	2.60E-11	0.00E+00	1.10E-07	3.40E-06	4.90E-06	1.40E-06	2.00E-05	0.53%
As-76	7.70E-05	1.20E-03	2.50E-07	3.00E-07	0.00E+00	1.20E-09	0.00E+00	6.40E-11	0.00E+00	2.50E-07	3.30E-06	1.80E-05	4.30E-07	1.30E-03	34.81%
Ca-45	1.70E-07	1.40E-06	7.10E-13	5.90E-10	0.00E+00	4.00E-14	2.50E-08	6.60E-12	0.00E+00	7.10E-13	9.20E-08	8.10E-09	5.30E-08	1.75E-06	0.05%
Co-60	3.40E-07	5.00E-06	3.70E-09	1.20E-09	0.00E+00	2.40E-07	2.30E-06	3.50E-09	0.00E+00	3.70E-09	2.30E-07	3.30E-07	9.40E-08	8.55E-06	0.23%
Cs-137	4.50E-07	1.30E-05	7.60E-10	2.10E-09	4.60E-06	2.10E-07	1.00E-05	2.10E-08	0.00E+00	7.60E-10	8.20E-07	7.00E-07	5.90E-07	3.04E-05	0.81%
Cu-64	2.30E-05	4.10E-06	5.10E-07	9.30E-08	0.00E+00	1.80E-10	0.00E+00	1.40E-12	0.00E+00	5.10E-07	4.50E-07	2.50E-06	2.40E-06	3.36E-05	0.90%
Ga-72	4.10E-05	1.80E-05	2.40E-06	2.40E-07	0.00E+00	4.00E-08	0.00E+00	2.00E-10	0.00E+00	2.40E-06	1.10E-06	1.30E-07	2.10E-08	6.53E-05	1.75%
I-133	2.50E-05	1.50E-06	8.40E-08	1.10E-07	0.00E+00	4.90E-11	2.80E-09	2.70E-12	0.00E+00	8.40E-08	8.80E-07	1.10E-05	1.30E-05	5.17E-05	1.38%
Mn-56	1.50E-05	2.40E-07	3.80E-06	1.20E-07	0.00E+00	6.10E-09	0.00E+00	8.90E-12	0.00E+00	3.80E-06	7.50E-08	1.50E-07	2.20E-07	2.34E-05	0.63%
P-32	1.20E-05	2.40E-04	4.30E-10	4.20E-08	3.90E-05	3.60E-12	9.60E-09	5.30E-12	0.00E+00	4.30E-10	3.30E-06	1.10E-05	1.90E-05	3.24E-04	8.69%
Sc-46	3.70E-06	1.40E-05	1.00E-07	1.70E-08	0.00E+00	2.10E-07	2.70E-06	1.50E-09	0.00E+00	1.00E-07	2.00E-06	1.60E-07	2.20E-08	2.30E-05	0.62%
Sr-89	6.90E-06	1.30E-05	1.90E-10	3.00E-08	0.00E+00	8.40E-12	2.70E-08	2.30E-11	0.00E+00	1.90E-10	3.30E-06	1.50E-06	1.30E-06	2.61E-05	0.70%
Sr-90	1.50E-06	4.00E-06	1.80E-11	6.50E-09	4.80E-07	4.70E-11	7.70E-08	2.70E-10	0.00E+00	1.80E-11	3.70E-06	9.60E-07	7.90E-07	1.15E-05	0.31%
Zn-69	2.00E-04	2.00E-04	4.30E-06	1.10E-06	3.20E-06	1.20E-07	0.00E+00	1.40E-09	0.00E+00	4.30E-06	4.90E-06	2.20E-04	1.00E-04	7.38E-04	19.76%
Zr-95	7.50E-06	8.90E-05	1.00E-07	3.50E-08	0.00E+00	4.80E-07	1.70E-05	7.00E-09	0.00E+00	1.00E-07	3.70E-06	1.70E-10	4.40E-10	1.18E-04	3.16%
Sb-122	4.90E-06	2.00E-06	1.30E-08	1.60E-08	0.00E+00	3.40E-11	0.00E+00	1.90E-12	0.00E+00	1.30E-08	4.20E-07	6.50E-08	3.00E-08	7.46E-06	0.20%
Np-239	1.30E-04	1.90E-04	2.90E-07	4.80E-07	0.00E+00	4.10E-10	1.40E-07	3.60E-11	0.00E+00	2.90E-07	1.20E-05	1.90E-06	8.80E-08	3.35E-04	8.98%
Y-90	3.90E-05	5.20E-06	1.30E-09	1.80E-07	0.00E+00	3.70E-11	7.20E-08	8.60E-11	0.00E+00	1.30E-09	3.90E-06	1.10E-06	1.50E-07	4.96E-05	1.33%
Total	6.4E-04	2.2E-03	2.8E-05	3.0E-06	6.3E-05	2.8E-06	3.4E-05	1.0E-07	0.0E+00	2.8E-05	6.4E-05	3.8E-04	2.7E-04	3.68E-03	100%
% of Total	17.48%	58.92%	0.76%	0.08%	1.72%	0.08%	0.91%	<0.01%	0.00%	0.76%	1.75%	10.22%	7.31%	100%	

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Table E-5. Screening Risk Values for the Native American Scenario at the 300 Area

Nuclide	Swim-					Sed-				Produce-				Total	% of Total
	Direct-Ing	Fish-Ing	Imm	Swim-Ing	Waterfowl	Sed-Ext	Dermal	Sed-Ing	Aerosol	Boating	Ing	Meat-Ing	Milk-Ing		
Zn-65	2.40E-05	9.20E-04	3.60E-07	3.80E-07	3.80E-05	2.70E-05	5.80E-06	1.50E-07	7.30E-07	5.50E-07	1.90E-05	0.00E+00	0.00E+00	1.04E-03	4.43%
Na-24	5.80E-05	2.00E-05	6.70E-05	9.50E-07	0.00E+00	4.40E-07	0.00E+00	3.30E-11	4.40E-06	9.60E-05	1.70E-06	0.00E+00	0.00E+00	2.48E-04	1.06%
I-131	1.40E-05	9.10E-05	3.20E-08	2.10E-07	0.00E+00	2.40E-09	3.60E-08	1.20E-10	2.10E-07	5.30E-08	7.70E-06	0.00E+00	0.00E+00	1.13E-04	0.48%
Y-93	8.60E-06	1.40E-06	3.90E-08	1.50E-07	0.00E+00	4.20E-10	0.00E+00	7.20E-12	1.30E-07	5.50E-08	1.90E-07	0.00E+00	0.00E+00	1.06E-05	0.05%
Cr-51	9.00E-06	1.70E-05	4.50E-07	1.40E-07	0.00E+00	5.00E-08	0.00E+00	6.90E-11	4.30E-07	6.70E-07	4.70E-06	0.00E+00	0.00E+00	3.24E-05	0.14%
As-76	8.90E-05	1.00E-02	1.10E-06	1.30E-06	0.00E+00	3.60E-08	0.00E+00	1.70E-10	1.70E-06	1.80E-06	4.60E-06	0.00E+00	0.00E+00	1.01E-02	43.22%
Ca-45	1.90E-07	1.10E-05	2.90E-12	2.40E-09	0.00E+00	7.60E-13	1.20E-07	1.50E-11	5.60E-08	5.40E-12	1.20E-07	0.00E+00	0.00E+00	1.15E-05	0.05%
Co-60	3.80E-07	4.00E-05	1.50E-08	5.00E-09	0.00E+00	4.20E-06	1.00E-05	7.90E-09	1.20E-07	2.70E-08	3.10E-07	0.00E+00	0.00E+00	5.51E-05	0.24%
Cs-137	5.00E-07	1.10E-04	3.00E-09	8.10E-09	1.10E-05	3.60E-06	4.60E-05	4.60E-08	6.00E-08	4.40E-09	1.10E-06	0.00E+00	0.00E+00	1.72E-04	0.74%
Cu-64	2.90E-05	3.70E-05	2.20E-06	4.10E-07	0.00E+00	5.40E-09	0.00E+00	4.10E-12	9.30E-07	3.70E-06	6.90E-07	0.00E+00	0.00E+00	7.39E-05	0.32%
Ga-72	5.00E-05	1.60E-04	9.60E-06	9.70E-07	0.00E+00	7.80E-07	0.00E+00	5.00E-10	9.10E-07	1.20E-05	1.70E-06	0.00E+00	0.00E+00	2.36E-04	1.01%
I-133	3.10E-05	1.30E-05	3.60E-07	4.60E-07	0.00E+00	1.40E-09	1.50E-08	7.20E-12	2.00E-07	5.80E-07	1.30E-06	0.00E+00	0.00E+00	4.69E-05	0.20%
Mn-56	2.60E-05	2.90E-06	2.10E-05	6.40E-07	0.00E+00	1.30E-07	0.00E+00	3.20E-11	5.50E-07	2.00E-05	1.60E-07	0.00E+00	0.00E+00	7.14E-05	0.31%
P-32	1.40E-05	6.70E-04	1.80E-09	1.70E-07	9.30E-05	1.10E-10	5.70E-08	1.60E-11	1.40E-06	3.30E-09	4.40E-06	0.00E+00	0.00E+00	7.83E-04	3.35%
Sc-46	4.00E-06	1.10E-04	4.00E-07	6.50E-08	0.00E+00	4.30E-06	1.30E-05	3.60E-09	9.10E-07	5.90E-07	2.60E-06	0.00E+00	0.00E+00	1.36E-04	0.58%
Sr-89	7.70E-06	4.30E-03	7.40E-10	1.20E-07	0.00E+00	1.90E-10	1.40E-07	5.90E-11	7.30E-07	1.10E-09	4.50E-06	0.00E+00	0.00E+00	4.31E-03	18.46%
Sr-90	1.70E-06	1.30E-03	7.00E-11	2.60E-08	1.10E-06	8.30E-10	3.50E-07	6.10E-10	1.30E-07	1.10E-10	5.00E-06	0.00E+00	0.00E+00	1.31E-03	5.60%
Zn-69	2.50E-04	1.80E-03	1.70E-05	4.30E-06	8.40E-06	2.70E-06	0.00E+00	3.50E-09	8.70E-06	2.30E-05	7.20E-06	0.00E+00	0.00E+00	2.12E-03	9.08%
Zr-95	8.40E-06	7.10E-04	4.10E-07	1.30E-07	0.00E+00	1.00E-05	8.10E-05	1.70E-08	1.50E-06	6.10E-07	5.00E-06	0.00E+00	0.00E+00	8.17E-04	3.50%
Sb-122	5.30E-06	1.60E-05	5.50E-08	7.10E-08	0.00E+00	1.10E-09	0.00E+00	5.50E-12	1.00E-07	9.80E-08	5.50E-07	0.00E+00	0.00E+00	2.22E-05	0.09%
Np-239	1.40E-04	1.40E-03	1.20E-06	2.00E-06	0.00E+00	1.20E-08	7.40E-07	9.40E-11	2.90E-06	2.00E-06	1.60E-05	0.00E+00	0.00E+00	1.56E-03	6.70%
Y-90	4.50E-05	4.20E-05	5.20E-09	7.10E-07	0.00E+00	9.50E-10	3.70E-07	2.20E-10	8.50E-07	7.90E-09	5.50E-06	0.00E+00	0.00E+00	9.44E-05	0.40%
Total	8.2E-04	2.2E-02	1.2E-04	1.3E-05	1.5E-04	5.3E-05	1.6E-04	2.3E-07	2.8E-05	1.6E-04	9.4E-05	0.0E+00	0.0E+00	2.34E-02	100%
% of Total	3.49%	93.17%	0.52%	0.06%	0.65%	0.23%	0.67%	<0.01%	0.12%	0.69%	0.40%	0.00%	0.00%	100%	

Table E-2. Screening Risk Values for the Initial Screening at Ringold

Nuclide	Swim-					Sed-				Produce-				Total	% of Total
	Direct-Ing	Fish-Ing	Imm	Swim-Ing	Waterfowl	Sed-Ext	Dermal	Sed-Ing	Aerosol	Boating	Ing	Meat-Ing	Milk-Ing		
Zn65	2.30E-05	9.70E-04	2.00E-07	2.20E-07	6.50E-05	4.60E-05	6.70E-06	1.70E-07	3.50E-07	5.30E-07	4.50E-05	5.10E-05	1.50E-05	1.22E-03	4.04%
Na24	4.80E-05	1.80E-05	3.20E-05	4.60E-07	0.00E+00	9.10E-07	0.00E+00	4.40E-11	1.80E-06	8.10E-05	3.50E-06	4.00E-05	6.30E-05	2.89E-04	0.95%
I131	1.30E-05	8.50E-05	1.60E-08	1.10E-07	0.00E+00	4.30E-09	4.20E-08	1.40E-10	9.00E-08	4.60E-08	1.70E-05	1.70E-05	1.30E-05	1.45E-04	0.48%
Y93	8.40E-06	1.40E-06	2.20E-08	8.30E-08	0.00E+00	9.50E-10	0.00E+00	1.00E-11	6.10E-08	5.40E-08	4.60E-07	1.80E-07	1.80E-08	1.07E-05	0.04%
Cr51	7.90E-06	1.60E-05	2.30E-07	7.40E-08	0.00E+00	8.80E-08	0.00E+00	8.00E-11	1.90E-07	5.90E-07	9.40E-06	4.90E-06	9.50E-07	4.03E-05	0.13%
As76	1.20E-04	1.50E-02	8.50E-07	1.00E-06	0.00E+00	7.80E-08	0.00E+00	2.40E-10	1.20E-06	2.50E-06	1.70E-05	2.90E-05	4.60E-07	1.52E-02	50.13%
Ca45	1.80E-07	1.10E-05	1.50E-12	1.30E-09	0.00E+00	1.30E-12	1.40E-07	1.80E-11	2.60E-08	5.00E-12	3.30E-07	8.40E-09	3.60E-08	1.17E-05	0.04%
Co60	3.50E-07	3.80E-05	7.80E-09	2.60E-09	0.00E+00	7.20E-06	1.20E-05	9.10E-09	5.50E-08	2.50E-08	8.30E-07	3.40E-07	6.40E-08	5.89E-05	0.19%
Cs137	4.30E-07	1.10E-04	1.50E-09	4.00E-09	1.70E-05	6.30E-06	5.30E-05	5.30E-08	2.60E-08	3.80E-09	2.30E-06	6.70E-07	3.80E-07	1.90E-04	0.63%
Cu64	2.90E-05	3.90E-05	1.20E-06	2.30E-07	0.00E+00	1.20E-08	0.00E+00	5.70E-12	4.70E-07	3.70E-06	1.90E-06	3.20E-06	2.00E-06	8.07E-05	0.27%
Ga72	5.40E-05	1.80E-04	6.10E-06	6.20E-07	0.00E+00	1.60E-06	0.00E+00	6.60E-10	5.00E-07	1.30E-05	4.00E-06	1.80E-07	1.90E-08	2.60E-04	0.86%
I133	2.80E-05	1.30E-05	2.00E-07	2.50E-07	0.00E+00	2.70E-09	1.90E-08	9.30E-12	9.00E-08	5.30E-07	3.00E-06	1.30E-05	1.00E-05	6.81E-05	0.22%
Mn56	2.20E-05	2.50E-06	1.10E-05	3.20E-07	0.00E+00	5.50E-07	0.00E+00	8.20E-11	2.40E-07	1.70E-05	3.10E-07	2.30E-07	2.20E-07	5.44E-05	0.18%
P32	1.30E-05	7.30E-04	9.30E-10	9.00E-08	1.50E-04	2.00E-10	6.60E-08	1.80E-11	6.60E-07	3.10E-09	1.20E-05	1.10E-05	1.30E-05	9.30E-04	3.07%
Sc46	3.70E-06	1.10E-04	2.10E-07	3.50E-08	0.00E+00	7.40E-06	1.50E-05	4.20E-09	4.20E-07	5.40E-07	5.60E-06	1.70E-07	1.50E-08	1.43E-04	0.47%
Sr89	6.80E-06	4.00E-03	3.80E-10	6.10E-08	0.00E+00	3.40E-10	1.60E-07	6.90E-11	3.20E-07	1.00E-09	1.00E-05	1.50E-06	8.80E-07	4.02E-03	13.28%
Sr90	1.50E-06	1.20E-03	3.60E-11	1.30E-08	1.70E-06	1.40E-09	4.00E-07	7.10E-10	5.60E-08	9.60E-11	1.10E-05	9.50E-07	5.20E-07	1.22E-03	4.02%
Zn69	2.60E-04	1.90E-03	1.00E-05	2.60E-06	1.60E-05	5.80E-06	0.00E+00	4.60E-09	4.60E-06	2.40E-05	1.80E-05	2.90E-04	9.10E-05	2.62E-03	8.66%
Zr95	7.30E-06	6.50E-04	2.10E-07	6.80E-08	0.00E+00	1.80E-05	9.40E-05	2.00E-08	6.60E-07	5.20E-07	1.10E-05	1.60E-10	2.80E-10	7.82E-04	2.58%
Sb122	7.90E-06	2.50E-05	4.60E-08	5.90E-08	0.00E+00	2.20E-09	0.00E+00	7.20E-12	7.50E-08	1.50E-07	2.40E-06	1.10E-07	3.30E-08	3.58E-05	0.12%
Np239	2.30E-04	2.60E-03	1.20E-06	1.90E-06	0.00E+00	2.30E-08	9.30E-07	1.20E-10	2.40E-06	3.40E-06	6.90E-05	3.30E-06	1.00E-07	2.91E-03	9.62%
Y90	4.00E-05	4.00E-05	2.70E-09	3.70E-07	0.00E+00	1.70E-09	4.40E-07	2.60E-10	3.80E-07	7.00E-09	1.20E-05	1.10E-06	1.00E-07	9.44E-05	0.31%
Total	8.8E-04	2.8E-02	6.3E-05	8.2E-06	2.5E-04	9.4E-05	1.8E-04	2.6E-07	1.4E-05	1.5E-04	2.4E-04	4.7E-04	2.1E-04	3.03E-02	100%
% of Total	2.92%	91.52%	0.21%	0.03%	0.83%	0.31%	0.60%	<0.01%	0.05%	0.49%	0.81%	1.54%	0.70%	100%	

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Table E-3. Screening Risk Values for the Local Resident Scenario at the 300 Area

Nuclide	Direct-Ing		Swim-			Sed-				Produce-			Total	% of Total	
	Direct-Ing	Fish-Ing	Imm	Swim-Ing	Waterfowl	Sed-Ext	Dermal	Sed-Ing	Aerosol	Boating	Ing	Meat-Ing			Milk-Ing
Zn-65	2.40E-05	1.70E-04	1.00E-07	1.10E-07	1.80E-05	1.60E-06	1.40E-06	7.10E-08	0.00E+00	1.00E-07	1.50E-05	5.30E-05	2.30E-05	3.06E-04	7.02%
Na-24	5.80E-05	2.90E-06	1.90E-05	2.70E-07	0.00E+00	2.10E-08	0.00E+00	1.50E-11	0.00E+00	1.90E-05	1.40E-06	4.70E-05	1.10E-04	2.58E-04	5.90%
I-131	1.40E-05	1.30E-05	8.20E-09	5.30E-08	0.00E+00	8.50E-11	7.10E-09	4.90E-11	0.00E+00	8.20E-09	6.40E-06	1.90E-05	2.20E-05	7.45E-05	1.71%
Y-93	8.60E-06	2.00E-07	1.10E-08	4.10E-08	0.00E+00	2.10E-11	0.00E+00	3.30E-12	0.00E+00	1.10E-08	1.60E-07	1.90E-07	2.70E-08	9.24E-06	0.21%
Cr-51	9.00E-06	2.40E-06	1.20E-07	3.80E-08	0.00E+00	2.00E-09	0.00E+00	2.80E-11	0.00E+00	1.20E-07	3.90E-06	5.50E-06	1.60E-06	2.27E-05	0.52%
As-76	8.90E-05	1.40E-03	3.00E-07	3.50E-07	0.00E+00	1.40E-09	0.00E+00	7.50E-11	0.00E+00	3.00E-07	3.80E-06	2.10E-05	4.90E-07	1.52E-03	34.72%
Ca-45	1.90E-07	1.50E-06	7.60E-13	6.30E-10	0.00E+00	4.50E-14	2.80E-08	7.30E-12	0.00E+00	7.60E-13	1.00E-07	9.10E-09	5.90E-08	1.89E-06	0.04%
Co-60	3.80E-07	5.50E-06	4.00E-09	1.30E-09	0.00E+00	2.60E-07	2.60E-06	3.90E-09	0.00E+00	4.00E-09	2.60E-07	3.70E-07	1.00E-07	9.48E-06	0.22%
Cs-137	5.00E-07	1.50E-05	8.20E-10	2.20E-09	5.20E-06	2.30E-07	1.10E-05	2.30E-08	0.00E+00	8.20E-10	9.30E-07	7.80E-07	6.60E-07	3.43E-05	0.79%
Cu-64	2.90E-05	5.10E-06	6.00E-07	1.10E-07	0.00E+00	2.10E-10	0.00E+00	1.70E-12	0.00E+00	6.00E-07	5.70E-07	3.20E-06	3.00E-06	4.22E-05	0.97%
Ga-72	5.00E-05	2.20E-05	2.80E-06	2.80E-07	0.00E+00	4.70E-08	0.00E+00	2.40E-10	0.00E+00	2.80E-06	1.40E-06	1.60E-07	2.60E-08	7.95E-05	1.82%
I-133	3.10E-05	1.80E-06	9.50E-08	1.20E-07	0.00E+00	5.60E-11	3.10E-09	3.10E-12	0.00E+00	9.50E-08	1.10E-06	1.40E-05	1.60E-05	6.42E-05	1.47%
Mn-56	2.60E-05	4.10E-07	6.20E-06	1.90E-07	0.00E+00	9.80E-09	0.00E+00	1.40E-11	0.00E+00	6.20E-06	1.30E-07	2.70E-07	3.80E-07	3.98E-05	0.91%
P-32	1.40E-05	2.70E-04	4.60E-10	4.50E-08	4.40E-05	3.80E-12	1.00E-08	5.70E-12	0.00E+00	4.60E-10	3.70E-06	1.20E-05	2.10E-05	3.65E-04	8.36%
Sc-46	4.00E-06	1.60E-05	1.10E-07	1.80E-08	0.00E+00	2.20E-07	2.90E-06	1.60E-09	0.00E+00	1.10E-07	2.20E-06	1.80E-07	2.40E-08	2.58E-05	0.59%
Sr-89	7.70E-06	1.50E-05	2.00E-10	3.20E-08	0.00E+00	9.10E-12	2.90E-08	2.50E-11	0.00E+00	2.00E-10	3.80E-06	1.70E-06	1.50E-06	2.98E-05	0.68%
Sr-90	1.70E-06	4.50E-06	1.90E-11	7.00E-09	5.40E-07	5.20E-11	8.60E-08	3.10E-10	0.00E+00	1.90E-11	4.20E-06	1.10E-06	8.80E-07	1.30E-05	0.30%
Zn-69	2.50E-04	2.50E-04	5.00E-06	1.30E-06	4.00E-06	1.50E-07	0.00E+00	1.60E-09	0.00E+00	5.00E-06	6.00E-06	2.70E-04	1.30E-04	9.21E-04	21.11%
Zr-95	8.40E-06	1.00E-04	1.10E-07	3.70E-08	0.00E+00	5.20E-07	1.80E-05	7.60E-09	0.00E+00	1.10E-07	4.10E-06	1.90E-10	4.90E-10	1.31E-04	3.01%
Sb-122	5.30E-06	2.20E-06	1.40E-08	1.90E-08	0.00E+00	3.90E-11	0.00E+00	2.20E-12	0.00E+00	1.40E-08	4.60E-07	7.00E-08	3.30E-08	8.11E-06	0.19%
Np-239	1.40E-04	2.00E-04	3.30E-07	5.30E-07	0.00E+00	4.60E-10	1.60E-07	4.10E-11	0.00E+00	3.30E-07	1.30E-05	2.00E-06	9.30E-08	3.56E-04	8.17%
Y-90	4.50E-05	5.90E-06	1.40E-09	1.90E-07	0.00E+00	4.10E-11	7.90E-08	9.40E-11	0.00E+00	1.40E-09	4.60E-06	1.20E-06	1.70E-07	5.71E-05	1.31%
Total	8.2E-04	2.5E-03	3.5E-05	3.7E-06	7.2E-05	3.1E-06	3.6E-05	1.1E-07	0.0E+00	3.5E-05	7.7E-05	4.5E-04	3.3E-04	4.36E-03	100%
% of Total	18.69%	57.36%	0.80%	0.09%	1.64%	0.07%	0.83%	<0.01%	0.00%	0.80%	1.77%	10.37%	7.58%	100%	

APPENDIX E — RADIONUCLIDE AND PATHWAY-SPECIFIC SCREENING RISKS

This appendix contains the radionuclide and pathway-specific screening risks for the initial screening calculations at the 300 Area and Ringold (Tables E-1, E-2), and for the three exposure scenarios (Tables E-3 through E-8). The risks are expressed in terms of the incremental lifetime cancer incidence risk for exposure from 1945 to 1972. There are two receptor locations for each exposure scenario. Nuclides are listed in rows and pathways of exposure in columns. The exposure pathways are (in order of appearance)

- Direct ingestion of contaminated river water (Direct Ing)
- Ingestion of contaminated fish (Fish Ing)
- Immersion in contaminated river water (Swim-Imm)
- Ingestion of contaminated river water during swimming (Swim-Ing)
- Ingestion of contaminated waterfowl (Waterfowl)
- External exposure to contaminated sediments (Sed-Ext)
- Exposure to contaminated sediments through dermal contact (Sed-Dermal)
- Ingestion of contaminated sediments (Sed-Ing)
- Inhalation of contaminated aerosols (Aerosol)
- External exposure while boating (Boating)
- Ingestion of produce irrigated with contaminated river water (Produce-Ing)
- Ingestion of meat from cattle drinking contaminated river water and consuming feed that was irrigated with contaminated river water (Meat-Ing)
- Ingestion of milk from cows drinking contaminated river water and consuming feed that was irrigated with contaminated river water (Milk-Ing)

Table E-1. Screening Risk Values for the Initial Screening at the 300 Area

Nuclide	Swim-					Sed-				Produce-				Total	% of Total
	Direct-Ing	Fish-Ing	Imm	Swim-Ing	Waterfowl	Sed-Ext	Dermal	Sed-Ing	Aerosol	Boating	Ing	Meat-Ing	Milk-Ing		
Zn65	4.80E-05	2.10E-03	5.10E-07	5.30E-07	1.20E-04	4.00E-05	5.80E-06	1.50E-07	7.30E-07	1.10E-06	9.30E-05	1.10E-04	3.00E-05	2.55E-03	4.81%
Na24	1.10E-04	4.50E-05	9.40E-05	1.30E-06	0.00E+00	6.50E-07	0.00E+00	3.30E-11	4.40E-06	1.90E-04	8.30E-06	9.40E-05	1.50E-04	6.98E-04	1.32%
I131	2.90E-05	2.00E-04	4.50E-08	2.90E-07	0.00E+00	3.60E-09	3.60E-08	1.20E-10	2.10E-07	1.10E-07	3.90E-05	3.90E-05	2.90E-05	3.37E-04	0.64%
Y93	1.70E-05	3.10E-06	5.50E-08	2.00E-07	0.00E+00	6.20E-10	0.00E+00	7.20E-12	1.30E-07	1.10E-07	9.30E-07	3.80E-07	3.60E-08	2.19E-05	0.04%
Cr51	1.80E-05	3.80E-05	6.30E-07	2.00E-07	0.00E+00	7.50E-08	0.00E+00	6.90E-11	4.30E-07	1.40E-06	2.10E-05	1.10E-05	2.20E-06	9.29E-05	0.18%
As76	1.80E-04	2.20E-02	1.50E-06	1.80E-06	0.00E+00	5.30E-08	0.00E+00	1.70E-10	1.70E-06	3.60E-06	2.40E-05	4.20E-05	6.60E-07	2.23E-02	42.00%
Ca45	3.80E-07	2.40E-05	4.10E-12	3.40E-09	0.00E+00	1.10E-12	1.20E-07	1.50E-11	5.60E-08	1.10E-11	7.20E-07	1.80E-08	7.80E-08	2.54E-05	0.05%
Co60	7.60E-07	8.90E-05	2.10E-08	7.00E-09	0.00E+00	6.20E-06	1.00E-05	7.90E-09	1.20E-07	5.50E-08	1.80E-06	7.40E-07	1.40E-07	1.09E-04	0.21%
Cs137	1.00E-06	2.60E-04	4.10E-09	1.10E-08	3.50E-05	5.40E-06	4.60E-05	4.60E-08	6.00E-08	8.70E-09	5.40E-06	1.60E-06	8.80E-07	3.55E-04	0.67%
Cu64	5.70E-05	8.20E-05	3.10E-06	5.70E-07	0.00E+00	8.10E-09	0.00E+00	4.10E-12	9.30E-07	7.50E-06	3.90E-06	6.30E-06	4.00E-06	1.65E-04	0.31%
Ga72	9.90E-05	3.40E-04	1.30E-05	1.40E-06	0.00E+00	1.20E-06	0.00E+00	5.00E-10	9.10E-07	2.30E-05	7.30E-06	3.30E-07	3.50E-08	4.86E-04	0.92%
I133	6.20E-05	2.80E-05	5.10E-07	6.50E-07	0.00E+00	2.00E-09	1.50E-08	7.20E-12	2.00E-07	1.10E-06	6.50E-06	2.80E-05	2.20E-05	1.49E-04	0.28%
Mn56	5.10E-05	6.30E-06	3.00E-05	9.00E-07	0.00E+00	1.90E-07	0.00E+00	3.20E-11	5.50E-07	4.00E-05	7.20E-07	5.30E-07	5.10E-07	1.31E-04	0.25%
P32	2.70E-05	1.60E-03	2.50E-09	2.40E-07	3.00E-04	1.70E-10	5.70E-08	1.60E-11	1.40E-06	6.60E-09	2.50E-05	2.40E-05	2.90E-05	2.01E-03	3.79%
Sc46	8.10E-06	2.50E-04	5.60E-07	9.20E-08	0.00E+00	6.40E-06	1.30E-05	3.60E-09	9.10E-07	1.20E-06	1.20E-05	3.60E-07	3.20E-08	2.93E-04	0.55%
Sr89	1.50E-05	9.50E-03	1.00E-09	1.70E-07	0.00E+00	2.90E-10	1.40E-07	5.90E-11	7.30E-07	2.30E-09	2.30E-05	3.40E-06	2.00E-06	9.54E-03	18.01%
Sr90	3.40E-06	2.90E-03	9.70E-11	3.60E-08	3.70E-06	1.20E-09	3.50E-07	6.10E-10	1.30E-07	2.20E-10	2.50E-05	2.20E-06	1.20E-06	2.94E-03	5.54%
Zn69	4.90E-04	3.90E-03	2.40E-05	6.00E-06	2.70E-05	4.10E-06	0.00E+00	3.50E-09	8.70E-06	4.60E-05	3.50E-05	5.50E-04	1.70E-04	5.26E-03	9.93%
Zr95	1.70E-05	1.60E-03	5.70E-07	1.90E-07	0.00E+00	1.50E-05	8.10E-05	1.70E-08	1.50E-06	1.20E-06	2.40E-05	3.70E-10	6.50E-10	1.74E-03	3.28%
Sb122	1.10E-05	3.60E-05	7.60E-08	9.90E-08	0.00E+00	1.60E-09	0.00E+00	5.50E-12	1.00E-07	2.00E-07	3.20E-06	1.40E-07	4.40E-08	5.09E-05	0.10%
Np239	2.70E-04	3.20E-03	1.70E-06	2.80E-06	0.00E+00	1.80E-08	7.40E-07	9.40E-11	2.90E-06	4.00E-06	8.30E-05	4.00E-06	1.20E-07	3.57E-03	6.74%
Y90	9.00E-05	9.40E-05	7.20E-09	1.00E-06	0.00E+00	1.40E-09	3.70E-07	2.20E-10	8.50E-07	1.60E-08	2.70E-05	2.40E-06	2.20E-07	2.16E-04	0.41%
Total	1.6E-03	4.8E-02	1.7E-04	1.8E-05	4.9E-04	7.9E-05	1.6E-04	2.3E-07	2.8E-05	3.2E-04	4.7E-04	9.2E-04	4.4E-04	5.30E-02	100%
% of Total	3.03%	91.14%	0.32%	0.03%	0.92%	0.15%	0.30%	<0.01%	0.05%	0.61%	0.89%	1.74%	0.83%	100%	

APPENDIX E
RADIONUCLIDE AND PATHWAY-SPECIFIC SCREENING RISKS

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Table E-6. Screening Risk Values for the Native American Scenario at Ringold

Nuclide	Swim-					Sed-				Produce-				Total	% of Total
	Direct-Ing	Fish-Ing	Imm	Swim-Ing	Waterfowl	Sed-Ext	Dermal	Sed-Ing	Aerosol	Boating	Ing	Meat-Ing	Milk-Ing		
Zn-65	1.20E-05	4.30E-04	1.50E-07	1.50E-07	2.00E-05	3.10E-05	6.70E-06	1.70E-07	3.50E-07	2.60E-07	8.60E-06	0.00E+00	0.00E+00	5.09E-04	3.89%
Na-24	2.40E-05	8.10E-06	2.30E-05	3.30E-07	0.00E+00	6.10E-07	0.00E+00	4.40E-11	1.80E-06	4.10E-05	6.80E-07	0.00E+00	0.00E+00	9.95E-05	0.76%
I-131	6.30E-06	3.80E-05	1.20E-08	7.60E-08	0.00E+00	2.80E-09	4.20E-08	1.40E-10	9.00E-08	2.30E-08	3.30E-06	0.00E+00	0.00E+00	4.78E-05	0.37%
Y-93	4.20E-06	6.40E-07	1.60E-08	5.90E-08	0.00E+00	6.40E-10	0.00E+00	1.00E-11	6.10E-08	2.70E-08	8.90E-08	0.00E+00	0.00E+00	5.09E-06	0.04%
Cr-51	3.90E-06	7.10E-06	1.70E-07	5.30E-08	0.00E+00	5.80E-08	0.00E+00	8.00E-11	1.90E-07	3.00E-07	2.00E-06	0.00E+00	0.00E+00	1.38E-05	0.11%
As-76	6.10E-05	6.50E-03	6.10E-07	7.10E-07	0.00E+00	5.20E-08	0.00E+00	2.40E-10	1.20E-06	1.30E-06	3.00E-06	0.00E+00	0.00E+00	6.57E-03	50.11%
Ca-45	8.80E-08	4.70E-06	1.10E-12	9.00E-10	0.00E+00	8.80E-13	1.40E-07	1.80E-11	2.60E-08	2.50E-12	5.40E-08	0.00E+00	0.00E+00	5.01E-06	0.04%
Co-60	1.70E-07	1.70E-05	5.50E-09	1.80E-09	0.00E+00	4.80E-06	1.20E-05	9.10E-09	5.50E-08	1.30E-08	1.30E-07	0.00E+00	0.00E+00	3.42E-05	0.26%
Cs-137	2.10E-07	4.80E-05	1.10E-09	2.90E-09	5.20E-06	4.20E-06	5.30E-05	5.30E-08	2.60E-08	1.90E-09	4.50E-07	0.00E+00	0.00E+00	1.11E-04	0.85%
Cu-64	1.40E-05	1.70E-05	8.80E-07	1.60E-07	0.00E+00	7.80E-09	0.00E+00	5.70E-12	4.70E-07	1.90E-06	3.30E-07	0.00E+00	0.00E+00	3.47E-05	0.27%
Ga-72	2.70E-05	8.00E-05	4.30E-06	4.40E-07	0.00E+00	1.10E-06	0.00E+00	6.60E-10	5.00E-07	6.40E-06	8.70E-07	0.00E+00	0.00E+00	1.21E-04	0.92%
I-133	1.40E-05	5.60E-06	1.40E-07	1.80E-07	0.00E+00	1.80E-09	1.90E-08	9.30E-12	9.00E-08	2.60E-07	5.80E-07	0.00E+00	0.00E+00	2.09E-05	0.16%
Mn-56	1.10E-05	1.20E-06	7.70E-06	2.30E-07	0.00E+00	3.70E-07	0.00E+00	8.20E-11	2.40E-07	8.70E-06	6.70E-08	0.00E+00	0.00E+00	2.95E-05	0.23%
P-32	6.30E-06	3.10E-04	6.60E-10	6.40E-08	4.70E-05	1.30E-10	6.60E-08	1.80E-11	6.60E-07	1.50E-09	1.90E-06	0.00E+00	0.00E+00	3.66E-04	2.79%
Sc-46	1.90E-06	4.90E-05	1.50E-07	2.50E-08	0.00E+00	5.00E-06	1.50E-05	4.20E-09	4.20E-07	2.70E-07	1.10E-06	0.00E+00	0.00E+00	7.29E-05	0.56%
Sr-89	3.40E-06	1.80E-03	2.70E-10	4.30E-08	0.00E+00	2.30E-10	1.60E-07	6.90E-11	3.20E-07	5.00E-10	1.90E-06	0.00E+00	0.00E+00	1.81E-03	13.78%
Sr-90	7.50E-07	5.30E-04	2.60E-11	9.50E-09	5.50E-07	9.60E-10	4.00E-07	7.10E-10	5.60E-08	4.80E-11	2.10E-06	0.00E+00	0.00E+00	5.34E-04	4.07%
Zn-69	1.30E-04	8.70E-04	7.40E-06	1.90E-06	4.80E-06	3.90E-06	0.00E+00	4.60E-09	4.60E-06	1.20E-05	3.60E-06	0.00E+00	0.00E+00	1.04E-03	7.92%
Zr-95	3.60E-06	2.90E-04	1.50E-07	4.80E-08	0.00E+00	1.20E-05	9.40E-05	2.00E-08	6.60E-07	2.60E-07	2.00E-06	0.00E+00	0.00E+00	4.03E-04	3.07%
Sb-122	3.90E-06	1.10E-05	3.30E-08	4.20E-08	0.00E+00	1.50E-09	0.00E+00	7.20E-12	7.50E-08	7.30E-08	3.90E-07	0.00E+00	0.00E+00	1.55E-05	0.12%
Np-239	1.10E-04	1.10E-03	8.60E-07	1.40E-06	0.00E+00	1.50E-08	9.30E-07	1.20E-10	2.40E-06	1.70E-06	1.30E-05	0.00E+00	0.00E+00	1.23E-03	9.39%
Y-90	2.00E-05	1.80E-05	1.90E-09	2.60E-07	0.00E+00	1.20E-09	4.40E-07	2.60E-10	3.80E-07	3.50E-09	2.30E-06	0.00E+00	0.00E+00	4.14E-05	0.32%
Total	4.4E-04	1.2E-02	4.6E-05	5.9E-06	7.8E-05	6.3E-05	1.8E-04	2.6E-07	1.4E-05	7.4E-05	4.6E-05	0.0E+00	0.0E+00	1.31E-02	100%
% of Total	3.35%	92.75%	0.35%	0.05%	0.59%	0.48%	1.40%	<0.01%	0.11%	0.57%	0.35%	0.00%	0.00%	100%	

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Table E-7. Screening Risk Values for the Migrant Worker Scenario at the 300 Area

Nuclide	Direct-Ing		Swim-			Sed-			Produce-			Total	% of Total		
	Direct-Ing	Fish-Ing	Imm	Swim-Ing	Waterfowl	Sed-Ext	Dermal	Sed-Ing	Aerosol	Boating	Ing			Meat-Ing	Milk-Ing
Zn-65	2.40E-05	8.70E-05	3.60E-07	3.80E-07	1.00E-05	1.50E-05	6.70E-06	1.70E-07	1.40E-13	0.00E+00	4.50E-09	1.10E-05	5.50E-06	1.60E-04	4.89%
Na-24	5.80E-05	2.20E-06	6.70E-05	9.50E-07	0.00E+00	2.30E-07	0.00E+00	4.00E-11	8.90E-13	0.00E+00	2.40E-06	2.40E-05	5.80E-05	2.13E-04	6.50%
I-131	1.40E-05	9.30E-06	3.20E-08	2.10E-07	0.00E+00	1.10E-09	4.70E-08	1.60E-10	3.80E-14	0.00E+00	1.70E-07	5.80E-06	6.90E-06	3.65E-05	1.11%
Y-93	8.60E-06	1.50E-07	3.90E-08	1.50E-07	0.00E+00	2.20E-10	0.00E+00	8.50E-12	2.60E-14	0.00E+00	9.60E-07	1.10E-07	1.60E-08	1.00E-05	0.31%
Cr-51	9.00E-06	1.80E-06	4.50E-07	1.40E-07	0.00E+00	2.50E-08	0.00E+00	8.60E-11	8.60E-14	0.00E+00	1.60E-08	1.40E-06	4.50E-07	1.33E-05	0.41%
As-76	8.90E-05	1.00E-03	1.10E-06	1.30E-06	0.00E+00	1.70E-08	0.00E+00	2.20E-10	3.20E-13	0.00E+00	3.70E-06	9.50E-06	2.30E-07	1.10E-03	33.76%
Ca-45	1.90E-07	1.10E-06	2.90E-12	2.40E-09	0.00E+00	4.30E-13	1.40E-07	1.80E-11	9.70E-15	0.00E+00	4.90E-11	1.60E-09	1.20E-08	1.45E-06	0.04%
Co-60	3.80E-07	3.90E-06	1.50E-08	5.00E-09	0.00E+00	2.40E-06	1.20E-05	9.20E-09	2.10E-14	0.00E+00	8.70E-12	5.20E-08	1.70E-08	1.88E-05	0.57%
Cs-137	5.00E-07	6.20E-06	3.00E-09	8.10E-09	2.90E-06	2.10E-06	5.30E-05	5.40E-08	1.20E-14	0.00E+00	2.00E-12	1.10E-07	1.10E-07	6.50E-05	1.99%
Cu-64	2.90E-05	3.80E-06	2.20E-06	4.10E-07	0.00E+00	2.50E-09	0.00E+00	5.10E-12	1.70E-13	0.00E+00	2.40E-06	1.60E-06	1.50E-06	4.09E-05	1.25%
Ga-72	5.00E-05	1.80E-05	9.60E-06	9.70E-07	0.00E+00	4.60E-07	0.00E+00	5.80E-10	2.10E-13	0.00E+00	4.50E-06	1.00E-07	1.60E-08	8.36E-05	2.56%
I-133	3.10E-05	1.30E-06	3.60E-07	4.60E-07	0.00E+00	6.60E-10	1.90E-08	9.10E-12	3.70E-14	0.00E+00	1.60E-06	6.80E-06	8.00E-06	4.95E-05	1.51%
Mn-56	2.60E-05	3.70E-07	2.10E-05	6.40E-07	0.00E+00	9.30E-08	0.00E+00	3.40E-11	1.60E-13	0.00E+00	1.20E-05	3.00E-07	4.00E-07	6.08E-05	1.86%
P-32	1.40E-05	4.30E-05	1.80E-09	1.70E-07	2.00E-05	5.20E-11	7.00E-08	1.90E-11	2.40E-13	0.00E+00	4.10E-08	2.90E-06	5.60E-06	8.58E-05	2.62%
Sc-46	4.00E-06	1.20E-05	4.00E-07	6.50E-08	0.00E+00	2.30E-06	1.50E-05	4.20E-09	1.80E-13	0.00E+00	2.40E-09	4.20E-08	6.10E-09	3.38E-05	1.03%
Sr-89	7.70E-06	1.10E-05	7.40E-10	1.20E-07	0.00E+00	1.00E-10	1.60E-07	7.00E-11	1.40E-13	0.00E+00	7.30E-09	3.80E-07	3.70E-07	1.97E-05	0.60%
Sr-90	1.70E-06	3.30E-06	7.00E-11	2.60E-08	2.90E-07	4.90E-10	4.00E-07	7.10E-10	2.40E-14	0.00E+00	6.60E-12	7.30E-08	6.90E-08	5.86E-06	0.18%
Zn-69	2.50E-04	1.90E-04	1.70E-05	4.30E-06	2.50E-06	1.50E-06	0.00E+00	4.10E-09	1.80E-12	0.00E+00	2.70E-05	1.60E-04	7.60E-05	7.28E-04	22.25%
Zr-95	8.40E-06	7.50E-05	4.10E-07	1.30E-07	0.00E+00	5.50E-06	9.50E-05	2.00E-08	3.00E-13	0.00E+00	6.50E-09	4.30E-11	1.20E-10	1.84E-04	5.64%
Sb-122	5.30E-06	1.60E-06	5.50E-08	7.10E-08	0.00E+00	4.90E-10	0.00E+00	6.80E-12	1.80E-14	0.00E+00	8.70E-08	2.60E-08	1.20E-08	7.15E-06	0.22%
Np-239	1.40E-04	1.50E-04	1.20E-06	2.00E-06	0.00E+00	5.50E-09	9.30E-07	1.20E-10	5.20E-13	0.00E+00	3.00E-06	7.80E-07	3.70E-08	2.98E-04	9.10%
Y-90	4.50E-05	4.40E-06	5.20E-09	7.10E-07	0.00E+00	4.70E-10	4.50E-07	2.70E-10	1.60E-13	0.00E+00	8.30E-07	4.90E-07	7.00E-08	5.20E-05	1.59%
Total	8.2E-04	1.6E-03	1.2E-04	1.3E-05	3.6E-05	3.0E-05	1.8E-04	2.6E-07	5.5E-12	0.0E+00	5.9E-05	2.3E-04	1.6E-04	3.27E-03	100%
% of Total	24.93%	49.67%	3.70%	0.40%	1.09%	0.91%	5.62%	0.01%	<0.01%	0.00%	1.79%	6.89%	4.99%	100%	

Table E-8. Screening Risk Values for the Migrant Worker Scenario at Ringold

Nuclide	Swim-					Sed-				Produce-				Total	% of Total
	Direct-Ing	Fish-Ing	Imm	Swim-Ing	Waterfowl	Sed-Ext	Dermal	Sed-Ing	Aerosol	Boating	Ing	Meat-Ing	Milk-Ing		
Zn-65	1.20E-05	3.70E-05	1.50E-07	1.50E-07	4.30E-06	1.80E-05	7.80E-06	2.00E-07	6.10E-14	0.00E+00	1.90E-09	4.80E-06	2.30E-06	8.67E-05	4.57%
Na-24	2.40E-05	8.10E-07	2.30E-05	3.30E-07	0.00E+00	3.00E-07	0.00E+00	5.40E-11	3.30E-13	0.00E+00	8.70E-07	9.00E-06	2.10E-05	7.93E-05	4.18%
I-131	6.30E-06	3.70E-06	1.20E-08	7.60E-08	0.00E+00	1.30E-09	5.50E-08	1.90E-10	1.50E-14	0.00E+00	6.70E-08	2.30E-06	2.70E-06	1.52E-05	0.80%
Y-93	4.20E-06	6.40E-08	1.60E-08	5.90E-08	0.00E+00	3.20E-10	0.00E+00	1.20E-11	1.10E-14	0.00E+00	4.10E-07	4.70E-08	6.70E-09	4.80E-06	0.25%
Cr-51	3.90E-06	7.20E-07	1.70E-07	5.30E-08	0.00E+00	2.90E-08	0.00E+00	1.00E-10	3.40E-14	0.00E+00	6.40E-09	5.50E-07	1.80E-07	5.61E-06	0.30%
As-76	6.10E-05	6.10E-04	6.10E-07	7.10E-07	0.00E+00	2.40E-08	0.00E+00	3.10E-10	1.90E-13	0.00E+00	2.20E-06	5.70E-06	1.30E-07	6.80E-04	35.89%
Ca-45	8.80E-08	4.20E-07	1.10E-12	9.00E-10	0.00E+00	5.00E-13	1.60E-07	2.10E-11	3.80E-15	0.00E+00	1.90E-11	6.50E-10	4.70E-09	6.74E-07	0.04%
Co-60	1.70E-07	1.50E-06	5.50E-09	1.80E-09	0.00E+00	2.80E-06	1.40E-05	1.10E-08	8.20E-15	0.00E+00	3.40E-12	2.00E-08	6.50E-09	1.85E-05	0.98%
Cs-137	2.10E-07	2.30E-06	1.10E-09	2.90E-09	1.10E-06	2.50E-06	6.20E-05	6.20E-08	4.50E-15	0.00E+00	7.40E-13	4.20E-08	4.00E-08	6.83E-05	3.60%
Cu-64	1.40E-05	1.60E-06	8.80E-07	1.60E-07	0.00E+00	3.50E-09	0.00E+00	7.00E-12	7.30E-14	0.00E+00	1.00E-06	6.90E-07	6.50E-07	1.90E-05	1.00%
Ga-72	2.70E-05	8.50E-06	4.30E-06	4.40E-07	0.00E+00	6.10E-07	0.00E+00	7.80E-10	1.00E-13	0.00E+00	2.20E-06	4.90E-08	7.60E-09	4.31E-05	2.27%
I-133	1.40E-05	5.50E-07	1.40E-07	1.80E-07	0.00E+00	8.60E-10	2.40E-08	1.20E-11	1.50E-14	0.00E+00	6.60E-07	2.80E-06	3.30E-06	2.17E-05	1.14%
Mn-56	1.10E-05	1.40E-07	7.70E-06	2.30E-07	0.00E+00	2.40E-07	0.00E+00	8.90E-11	5.90E-14	0.00E+00	4.50E-06	1.10E-07	1.50E-07	2.41E-05	1.27%
P-32	6.30E-06	1.70E-05	6.60E-10	6.40E-08	8.00E-06	6.00E-11	8.20E-08	2.30E-11	9.80E-14	0.00E+00	1.60E-08	1.20E-06	2.20E-06	3.49E-05	1.84%
Sc-46	1.90E-06	4.90E-06	1.50E-07	2.50E-08	0.00E+00	2.70E-06	1.70E-05	4.90E-09	7.30E-14	0.00E+00	9.80E-10	1.70E-08	2.50E-09	2.67E-05	1.41%
Sr-89	3.40E-06	4.30E-06	2.70E-10	4.30E-08	0.00E+00	1.20E-10	1.90E-07	8.20E-11	5.40E-14	0.00E+00	2.80E-09	1.50E-07	1.40E-07	8.23E-06	0.43%
Sr-90	7.50E-07	1.30E-06	2.60E-11	9.50E-09	1.10E-07	5.60E-10	4.70E-07	8.30E-10	9.40E-15	0.00E+00	2.50E-12	2.80E-08	2.70E-08	2.70E-06	0.14%
Zn-69	1.30E-04	8.90E-05	7.40E-06	1.90E-06	1.10E-06	2.00E-06	0.00E+00	5.50E-09	8.40E-13	0.00E+00	1.20E-05	7.40E-05	3.50E-05	3.52E-04	18.59%
Zr-95	3.60E-06	2.90E-05	1.50E-07	4.80E-08	0.00E+00	6.40E-06	1.10E-04	2.30E-08	1.10E-13	0.00E+00	2.50E-09	1.70E-11	4.70E-11	1.49E-04	7.87%
Sb-122	3.90E-06	1.00E-06	3.30E-08	4.20E-08	0.00E+00	6.40E-10	0.00E+00	9.00E-12	1.10E-14	0.00E+00	5.60E-08	1.70E-08	7.90E-09	5.06E-06	0.27%
Np-239	1.10E-04	1.10E-04	8.60E-07	1.40E-06	0.00E+00	6.90E-09	1.20E-06	1.50E-10	3.90E-13	0.00E+00	2.20E-06	5.80E-07	2.80E-08	2.26E-04	11.94%
Y-90	2.00E-05	1.70E-06	1.90E-09	2.60E-07	0.00E+00	5.60E-10	5.40E-07	3.20E-10	6.40E-14	0.00E+00	3.30E-07	1.90E-07	2.70E-08	2.30E-05	1.22%
Total	4.4E-04	9.2E-04	4.6E-05	5.9E-06	1.5E-05	3.6E-05	2.1E-04	3.1E-07	2.5E-12	0.0E+00	2.6E-05	1.0E-04	6.8E-05	1.87E-03	100%
% of Total	23.37%	49.33%	2.43%	0.32%	0.78%	1.90%	11.37%	0.02%	<0.01%	0.00%	1.40%	5.45%	3.62%	100%	

