REPORTS OF THE OAK RIDGE DOSE RECONSTRUCTION, Vol. 6
The Report of Project Task 7 • July 1999

Screening-Level Evaluation of Additional Potential Materials of Concern

Submitted to the Tennessee Department of Health by

ChemRisk
A Service of McLaren/Hart

OAK RIDGE HEALTH STUDIES
OAK RIDGE DOSE RECONSTRUCTION

– TASK 7 REPORT –

SCREENING-LEVEL EVALUATION OF ADDITIONAL POTENTIAL MATERIALS OF CONCERN

July 1999

Submitted to the Tennessee Department of Health by

ChemRisk
A Service of McLaren/Hart

This document was prepared as a team effort by the following individuals:

Project Manager

Thomas E. Widner[1]

Task 7 Managers

Jennifer K. Cockroft[1]
Kathleen M. Thiessen[2]

Principal Authors

Gretchen M. Bruce[3]
John E. Buddenbaum[4]
Jennifer K. Cockroft
Susan M. Flack[1]
Regan E. Burmeister[5]
Talaat Ijaz[4]
Thomas E. Widner

---

[1] ChemRisk®, a Service of McLaren/Hart, Inc., 1320 Harbor Bay Parkway, Suite 100, Alameda, California 94502. (510) 521-5200. E-mail for the project team can be sent to: Tom_Widner@mclaren-hart.com.

[2] SENES Oak Ridge, 102 Donner Drive, Oak Ridge, Tennessee 37830

[3] ChemRisk®, a Service of McLaren/Hart, Inc., Alameda, California. Ms. Bruce is currently with Intertox, Inc., Seattle, Washington.

[4] ChemRisk®, a Service of McLaren/Hart, Inc., 5900 Landerbrook Drive, Cleveland, Ohio, 44124.

[5] Shonka Research Associates, Inc., 4939 Lower Roswell Road, Suite 106, Marietta, Georgia, 30068.

TABLE OF CONTENTS

TABLE OF CONTENTS *(Continued)*

TABLE OF CONTENTS *(Continued)*

LIST OF APPENDICES

LIST OF TABLES

LIST OF TABLES *(Continued)*

LIST OF FIGURES

EXECUTIVE SUMMARY

In the Oak Ridge Dose Reconstruction Feasibility Study, investigators took an intense and comprehensive, but relatively quick, "look through the key hole" at past operations on the Oak Ridge Reservation (the ORR). They performed screening calculations to identify those operations and materials that warranted detailed investigation in terms of potential off-site exposures. At the close of the Feasibility Study, the Tennessee Department of Health (TDH) and the Oak Ridge Health Agreement Steering Panel (ORHASP) recommended that detailed dose reconstructions be conducted for a number of historical activities tied to environmental releases (iodine-131 from X-10 radioactive lanthanum processing, mercury from Y-12 lithium enrichment, PCBs in the environment near Oak Ridge, and radionuclides released from White Oak Creek on the X-10 site to the Clinch River). They called for the study to also include systematic searching of historical records, an evaluation of the quality of historical uranium effluent monitoring data, and additional screening of some materials that could not be evaluated during the Feasibility Study. In addition, several questions raised during the Feasibility Study that could not be answered at the time were slated for evaluation during the dose reconstruction project.

This report presents the methods and results of the Task 7 screening of additional potential materials of concern. The Task 7 investigation included quantitative, screening-level evaluations of 10 materials or classes of materials and less detailed evaluations of 18 others. The purpose of screening in the Oak Ridge Dose Reconstruction was to permit attention and resources to be focused on the most important contaminants and to avoid dilution of resources by identifying situations that are obviously of only minor importance.

Three different methods were used by the Task 7 project team to evaluate the importance of materials in terms of their potential to pose off-site health hazards. The method selected to evaluate a given material was dependent on the quantity of the material present on-site, the form and manner in which the material was used, and the availability of environmental monitoring and release data, as well as whether the material was classified *per se* (i.e., its mere presence on the ORR remained classified). Fortunately, before this project was completed, the presence of any material on the ORR (at the site level) could be publicly revealed.

The methods used to screen materials were as follows:

*Qualitative screening*S all materials identified as having been used on the ORR were subject to qualitative screening; for some materials, the project team determined that based on evaluation of quantities used, forms used, and/or manners of usage, it was unlikely that off-site releases of the material could have been sufficient to pose off-site health hazard; these materials were not subject to quantitative screening.

Small-quantity materials included chemicals and radionuclides used as calibration standards or check sources for laboratory instruments or analytical methods. Materials used in forms not conducive to off-site release include carbon fibers and glass fibers that were received at the K-25 site as premanufactured filaments wound on spools. These fibers were used in construction of rotors used in the centrifuge method of uranium enrichment, in a process by which they were wound on a spool and a plastic binder applied to form the tall, cylindrical rotors. Examples of cases where manners of usage minimized the potential for significant off-site release include liquids, gases, or powders that were kept sealed in cylinders or were processed in containment systems that included multiple barriers against release.

*The threshold quantity approach***S** accurate estimates of inventory quantities of materials used at the Oak Ridge complexes are often not available, or in some cases not publicly releasable. It is typically much easier to determine, based on historical records or interviews of active or retired workers, if inventory quantities of a material were below a calculated threshold quantity. For a number of materials, project investigators used conservative assumptions to calculate a "threshold quantity" below which a material was highly unlikely to have posed a risk to human health through off-site releases. Threshold quantities were calculated using the following approach:

1. The *maximum allowable air concentration or water concentration* of a material was calculated based on the maximum allowable daily dose (assumed to be equivalent to the noncarcinogenic reference dose (RfD) or the dose that would lead to a cancer risk of 1×10^{-6}). To calculate a maximum allowable concentration, the maximum allowable daily dose was multiplied by a typical body weight and divided by a typical breathing or water ingestion rate.

2. The *maximum allowable release rate* to air or water was then determined, by calculating the release rate that would give an air or water concentration equal to the maximum allowable air concentration or water concentration. Release rates were calculated using conservative environmental dispersion or dilution factors.

3. The *maximum allowable release rate* in g s^{-1} was then converted to a maximum allowable release rate in kg y^{-1}. This quantity was assumed to be the threshold inventory quantity for the material.

Quantitative screening using a two-level screening approach**S** each level used a different set of assumptions to calculate potential doses and screening-level risk indices; the goal of this approach is to identify those contaminants that produced doses or health risks to exposed individuals or populations that are clearly below established minimum levels of concern (called a Level I Screen) and identify those contaminants that produced doses or health risks to exposed individuals or populations that are likely to have been above the established minimum levels of concern (called a Level II or Refined Level I Screen).

Both the Level I and Level II screening calculations used mathematical equations for calculation of dose and risk through multiple exposure pathways. These equations relate dose to the exposure point concentration and the magnitude of intake. Pathway equations used in the screening assessment are presented in Appendix B. The equations included all pathways potentially significant for the contaminants in question; exposure pathways evaluated included inhalation, ground exposure (for radionuclides), ingestion of soil or sediment, vegetable ingestion, and ingestion of meat, milk, and/or fish.

Parameter values used to calculate dose were selected based on historical knowledge of the Oak Ridge area, literature review, and professional judgment**S** parameter values used are given in Appendix C. The parameters that were varied between screening levels included lifestyle factors such as intake rates, time spent outdoors, etc. Contaminant-specific transfer factors and toxicity values were kept constant for both levels of screening. Different exposure durations and averaging times were assumed for radionuclides, carcinogenic chemicals, and noncarcinogenic chemicals. For radionuclides and carcinogenic chemicals, exposure durations of 50 years and 10 years were used in the Level I and Refined Level I screening, respectively. For carcinogenic chemicals, the risk was calculated in terms of the total intake averaged over the estimated lifetime, assumed to be 70 years, to give a lifetime average daily intake. For radionuclides, the risk was calculated in terms of the total cumulative dose, and an averaging time was not needed. For noncarcinogenic materials, an exposure duration and averaging time of one year were used.

The Level I screen was designed to estimate the dose or risk to a "maximally exposed" reference individual who should have received the highest exposure and thus would have been most at-risk. This level incorporated conservative exposure parameter values (such as intake rates) not expected to lead to an underestimate of risk to any real person in the population of interest. For Level I screening, each screening-level risk estimate ("screening index") was compared to the appropriate risk-based decision guide[1] as follows:

- If the screening index for the maximally exposed individual was *below* the decision guide, it was concluded that further study of the contaminant can be deferred until time and resources permit further study, because risks to members of the general population would be even lower. Continued expenditure of time and resources on that contaminant is not justified as long as there are more important situations to be studied.

- If the screening index for the maximally exposed individual was *above* the decision guide, it was concluded that the contaminant should be further evaluated in refined Level I screening or in Level II screening.

Refined Level I and Level II screens are designed to estimate the dose or risk to a more typical individual in the population of interest than was addressed in Level I screening. They incorporated reasonable average or more typical values for the exposure parameter values. It was assumed that the Level II screening value underestimated the dose or risk for the most highly exposed individual, although the dose or risk may be overestimated for the general population. For Refined Level I or Level II screening, each screening index was compared to the appropriate decision guide as follows:

---

[1] The risk-based decision guides established by the ORHASP were as follows: For radionuclides and carcinogenic chemicals, the decision guide was a lifetime excess cancer incidence of 1 in 10,000 (10^{-4}). For noncarcinogenic chemicals, the decision guide was a Hazard Index of 1.0. The Hazard Index is equal to the contaminant dose divided by its Reference Dose (RfD), where the RfD is defined as a dose of a chemical that is not expected to cause adverse noncarcinogenic health effects over a lifetime of daily exposure..

- If the screening index was *above* the decision guide, it was concluded that the contaminant should be given high priority for detailed study, because it is likely that some individuals received exposures or doses high enough to warrant further investigation.

- If the screening index was *below* the decision guide, the contaminant was deferred for further study to later phases of the project, after the highest priority contaminants are evaluated.

The "Refined Level I" screening evaluations described in this report were considerably less conservative than the Level I evaluations they followed. As described in Appendix C, for example, many of the exposure parameter values used in the dose and risk calculations are less conservative (more realistic or more typical) than the values of the same parameters used in Level I screening. A good example would be the assumed exposure duration for carcinogens, which is 50 years in Level I screening and 10 years in Level II screening. The "Refined Level I" evaluations described in this report used the Level II exposure parameters.

While a general goal in refined screening is to reduce or eliminate sources of conservative bias, it is not always feasible or advisable to eliminate all conservative bias, or easily determined when a sufficient level of realism has been achieved. In the refined screening evaluations described in this report, some degree of conservatism was retained, particularly in the estimation of contaminant concentrations in environmental media of interest. One important reason for this is that there were very few measurements of the contaminants of concern made in the environment during the (pre-1970s) periods when levels of many contaminants in the environment were likely the highest. Measurements in process streams or effluents are even more rare. Because of the paucity of information for some vital components of the risk assessment process, some conservatism was retained in the estimation of exposure point concentrations for the Refined Level I assessments to ensure that exposures were not underestimated for significant portions of the potentially exposed populations. Because of this, the second-level assessments are called Refined Level I assessments rather than Level II assessments.

Methods with which Contaminants of Potential Concern were Addressed

Following is a summary of the methods that were used to address the potential health significance of each contaminant that was evaluated within the Task 7 study.

- **Arsenic–** releases from K-25 and Y-12 steam plants were estimated using reported arsenic contents of the coal burned at each plant, usage rates of the coal, and an USEPA emission factor for arsenic. Air concentrations were estimated at Union/Lawnville and Scarboro, based on dispersion modeling and the empirical dispersion factor for Y-12 releases to Scarboro. Surface water exposures were evaluated based on the highest concentrations measured in Poplar Creek near the mouth of the Clinch River (for K-25 releases) and in McCoy Branch (for Y-12 releases). Exposures from arsenic in soil/sediment were evaluated based on levels measured in a sediment core collected at Poplar Creek Mile 1.0 and on measurements in sediments of McCoy Branch. Concentrations of arsenic in vegetables, meat, milk, and fish were estimated based on concentrations in air, water, and soil used with biotransfer and bioconcentration factors for arsenic from scientific literature. Level I and Refined Level I screening assessments were performed for arsenic for cancer (from inhalation and oral exposure) and non-cancer health effects (from oral exposure).

- **Asbestos–** The potential off-site significance of asbestos used in ORR facilities was an open question from the Feasibility Study. During the Dose Reconstruction, the project team summarized available information on the use and disposal of asbestos on the ORR. This report presents that information and discusses some factors that affect the potential for off-site releases and exposures in nearby communities. Health Studies investigators found no indication that the presence of asbestos at ORR facilities extended beyond various building materials, including thermal insulation of pipes, ducts, and vessels. The demolition of buildings and removal of asbestos containing materials would potentially lead to short-term increases of airborne asbestos in the immediate vicinity of these operations, but they would be expected to have a limited potential to affect asbestos concentrations to off-site receptors. The project team did not identify any specific asbestos-related exposure events or activities that are believed to have been associated with community exposure. In the absence of such "focal" events, it is unlikely that asbestos-related activities at ORR have resulted in off-site exposures beyond what might be expected from other sources in the community.

- **Beryllium Compounds–** were evaluated based on 1980 Y-12 stack monitoring data for total beryllium and the empirical dispersion factor for Y-12 releases to Scarboro. Beryllium exposures from surface waters were evaluated based on the maximum concentration measured in the EFPC remedial investigation. The maximum beryllium concentration measured in soil in the EFPC remedial investigation was also used. Levels in meat, milk, vegetables, and fish were estimated using the selected concentrations in air, soil, and water with biotransfer and bioconcentration factors from the scientific literature. Level I screening assessments were performed for beryllium for cancer and non-cancer health effects via inhalation and oral exposure. A Refined Level I screening was performed for beryllium as a carcinogen, as the Level I screening index exceeded the decision guide.

- **Boron carbide, boron nitride, yttrium boride, titanium boride, rubidium nitrate, Triplex Coating, carbon fibers, glass fibers, four-ring polyphenyl ether–** were evaluated qualitatively and found to have been used in very small quantities or in forms unlikely to have resulted in off-site releases. These materials were formerly classified *per se*, that is their presence on the ORR was classified in the past.

- **Copper–** was evaluated based on airborne concentrations measured at the most-affected on-site air sampler, adjusted according to the ratio of dispersion model results at that sampler to those for the reference location at Union/Lawnville. Surface water exposures were evaluated based on the highest concentration reported from a special monitoring project in the Clinch River; it was measured just downstream of the K-25 Site. The highest mean concentration of copper in sediment from the Clinch River was used to evaluate soil/sediment exposure pathways; it was measured just downstream of the mouth of Poplar Creek. Levels in meat, milk, vegetables, and fish were estimated using the selected concentrations in air, soil, and water with biotransfer and bioconcentration factors from the scientific literature. Level I and Refined Level I screening assessments were performed for copper for non-cancer health effects from inhalation and oral exposure. Screening indices for inhalation exposure were evaluated based on a derived RfD based on the ACGIH Threshold Limit Value® for occupational exposure to copper in air.

- **Hexavalent Chromium–** was evaluated on a Reservation-wide basis using a composite of environmental concentration estimates from K-25 and Y-12 releases. Air exposures were based on modeling of Cr(VI) emission and drift from six gaseous diffusion process cooling towers at the K-25 Site to the reference location at Union/Lawnville. Surface water exposures were evaluated based on the maximum reported Cr(VI) concentrations in EFPC, which were measured in 1969. For a more realistic, Refined Level I analysis, the maximum Cr(VI) level measured in Poplar Creek was used. The average concentration of total chromium in soil in the EFPC remedial investigation was used in the screening; in the Level I screen, this concentration was assumed to be all Cr(VI), while in the Refined Level I screening it was assumed that one-sixth of the total chromium was Cr(VI). Levels in meat, milk, vegetables, and fish were estimated using the selected concentrations in air, soil, and water with biotransfer and bioconcentration factors from the scientific literature. Level I and Refined Level I screening assessments were performed for hexavalent chromium for cancer (from inhalation exposure) and non-cancer health effects (from inhalation and oral exposure).

- **Lead–** In the Feasibility Study's screening evaluation, lead ranked second after mercury in terms of potential noncarcinogenic health hazards to off-site populations. To be consistent with other materials screened in the feasibility study, this ranking was established using a provisional USEPA noncarcinogenic RfD for lead. Currently, however, the USEPA recommends evaluating lead exposures using the USEPA Integrated Exposure Uptake Biokinetic (IEUBK) model, which predicts blood lead concentrations in children.

No data describing measured air concentrations of lead at the location of the nearest off-site population (the Scarboro community) were identified by the project team during the Dose Reconstruction. Airborne lead concentrations at this location were estimated based on information on background concentrations of lead in air prior to the mid-1970s, when air concentrations began to decline due to discontinuing of lead use in gasoline. The project team used the highest measured surface water concentration reported for EFPC in the screening for the surface water pathways. The project team used the highest soil/sediment concentration from available environmental investigations in the screening analyses for the soil/sediment pathways; it was from the EFPC remedial investigation. To adjust for the possibility of higher surface soil concentrations of lead in past years, the 95% upper confidence limit (UCL) concentration was multiplied by a factor of 3.5, yielding an adjusted surface soil concentration for use in the screening. The maximum measured concentration in EFPC fish was also used. The project team calculated concentrations of lead in vegetation, meat, and milk using biotransfer factors that characterize the transfer of lead from other media, including air, water, and soil to these food products. The IEUBK model was used with the above concentration estimates to estimate blood lead concentrations in children. These estimated blood lead concentrations were then compared to an acceptable blood lead guidance concentration developed by the Centers for Disease Control and Prevention (CDC).

- **Lithium–** was evaluated based on limited stack sampling from two lithium processing buildings, used with the empirical dispersion factor for airborne releases from Y-12 to Scarboro. Surface water exposures were evaluated based on the highest quarterly average lithium concentration measured in EFPC. This concentration was not inconsistent with concentrations estimated by the project team based on documented lithium losses from process buildings and the average EFPC flow rate. Exposures from lithium in soil and sediment were evaluated based on the highest lithium concentration measured in soil in the EFPC floodplain. Levels in meat, milk, vegetables, and fish were estimated using the selected concentrations in air, soil, and water with biotransfer and bioconcentration factors from the scientific literature. Level I and Refined Level I screening assessments were performed for lithium for non-cancer health effects from inhalation and oral exposure. Screening indices for inhalation exposure were evaluated based on a derived RfD based on 10 percent of the maximum daily therapeutic maintenance dose of lithium carbonate used to control mania, a manifestation of manic-depressive illness.

- **Neptunium-237–** Airborne releases of Np-237 from K-25 were estimated based on the reported mass of recycled uranium received annually, estimated Np-237 concentrations in that uranium (estimated based on the alpha activity limit on what would be accepted), and a release fraction based on estimated uranium releases divided by reported uranium inventories at K-25. A similar process was used to estimate Y-12 air releases, with the release fraction based on reported inventory differences for natural uranium (that is, estimates of quantities lost or unaccounted for based on material accountability records). Air concentrations were estimated at Union/Lawnville and Scarboro based on dispersion modeling and the empirical dispersion factor for Y-12 releases to Scarboro. Concentrations of Np-237 in the Clinch River were estimated based on reported releases of Np-237 and transuranic radionuclides from the K-25 Site and correction for dilution by the median flow rate of the Clinch. Waterborne releases from the Y-12 Plant were estimated as three quarters of the loss that was estimated based on natural uranium inventory differences. Releases were diluted by the lower bound of measured flow rates in EFPC. Exposures from Np-237 in soil and sediment were evaluated based on the highest sediment concentration reported in the Clinch River in 1981 and the highest sediment concentration reported in the EFPC remedial investigation. Levels in meat, milk, vegetables, and fish were estimated using the selected concentrations in air, soil, and water with biotransfer and bioconcentration factors from the scientific literature. A Level I screening assessment was performed for Np-237 as a carcinogen due to internal radiation exposure after inhalation and oral exposure.

- **Nickel–** Airborne concentrations of nickel near the K-25 Site were estimated based on measurements made with an air sampler located about 300 feet east of gaseous diffusion barrier production building K-1037 in the mid-1970s, corrected according to the ratio of dispersion model results for that location compared to that for the off-site reference location at Union/Lawnville. Surface water exposures were evaluated based on the highest mean concentration reported for the Clinch River as part of routine monitoring in 1975, and soil/sediment exposures were evaluated based on the highest mean sediment concentration reported for the Clinch River just upstream of the mouth of Poplar Creek in 1976. Levels in meat, milk, vegetables, and fish were estimated using the selected concentrations in air, soil, and water with biotransfer and bioconcentration factors from the scientific literature. Level I and Refined Level I screening assessments were performed for lithium for cancer from inhalation exposure and non-cancer health effects from oral exposure.

- **Niobium–** was evaluated using the threshold quantity approach, using a reference dose derived from an LD_{50} (the dose lethal to 50% of those exposed) in test animals, an empirically derived dispersion factor for airborne releases carried from the Y-12 Plant to Scarboro, and estimated average EFPC flow rates. While production rates and release estimates for niobium remain classified, information reviewed in Y-12 Plant quarterly reports and obtained in interviews indicated that usage rates of niobium never exceeded the threshold release rates to air or water that were calculated.

- **Plutonium–** In the Feasibility Study, airborne plutonium releases were estimated for plutonium separation at the X-10 Chemical Processing Pilot Plant, for radioactive barium/lanthanum processing, and for ruptures of fuel "slugs" in the Clinton Pile. For screening in the Feasibility Study, quantities of plutonium present in Clinton Pile fuel slugs were estimated based on documented rates of "product" formation. The records that documented these rates of formation did not, however, specify the isotopic composition of the plutonium "product" formed, so the project team assumed the plutonium formed and in part released to the environment was plutonium-239 (^{239}Pu). At the close of the Feasibility Study, the potential ramifications of this assumption were identified by the project team as an area that should be further addressed. During the Dose Reconstruction, the project team calculated the plutonium content of the fuel slugs for eleven isotopes of plutonium using the ORIGEN2.1 computer code. The results indicate that ^{239}Pu comprised at least 99.9 percent of the plutonium present in Clinton Pile fuel slugs, therefore assuming the plutonium "product" was ^{239}Pu did not introduce significant inaccuracy into the Feasibility Study screening of past airborne releases of plutonium from the X-10 Site.

- **Technetium-99–** Airborne releases of Tc-99 from K-25 were estimated based on reported releases from a 1978 material balance report and from routine environmental reports in later years. Releases from the Y-12 Plant were estimated based on documentation of quantities of recycled uranium received at Y-12, the estimated concentration of Tc-99 in that uranium, and the release fraction based on reported inventory differences for natural uranium. Air concentrations were estimated at Union/Lawnville and Scarboro based on dispersion modeling and the empirical dispersion factor for Y-12 releases to Scarboro. Surface water exposures were evaluated based on the highest surface water concentration for the Clinch River (reported in 1992), and the highest concentration reported for EFPC (reported in 1993 from a sample collected near the junction of Bear Creek Road and Scarboro Road). Exposures from soil/sediment pathways were evaluated based on the highest Tc-99 concentration in sediment reported for the Clinch River (from routine sampling in the 1970s) and the highest concentration reported for EFPC in a 1984 TVA study. Levels in meat, milk, vegetables, and fish were estimated using the selected concentrations in air, soil, and water with biotransfer and bioconcentration factors from the scientific literature. Level I and Refined Level I screening assessments were performed for Tc-99 as a carcinogen due to internal radiation exposure after inhalation and oral exposure.

- **Tellurium–** was evaluated qualitatively. The project team found that its short duration of use and the method in which it was used made it unlikely that tellurium was released in quantities sufficient to pose an off-site health hazard.

• **Tetramethylammoniumborohydride [TMAB, $(CH_3)_4NBH_4$]–** was evaluated using the threshold quantity approach because inventory quantities remain classified. TMAB was formerly classified *per se.*

• **Tritium–** was evaluated based on deuterium inventory differences (quantities "lost" or unaccounted for in deuterium processing) and the peak documented tritium concentration in the deuterium that was processed at Y-12 from heavy water received from Savannah River. A release estimate obtained from these data was used with the International Atomic Energy Agency (IAEA) method for tritium dose assessment assuming all of the tritium that escaped was released to EFPC over a 40-year period. The resulting Level I screening index was well below the project's decision guide of 1×10^{-4} added lifetime cancer risk.

• **Zirconium–** was evaluated using the threshold quantity approach, using a reference dose derived from an ACGIH Threshold Limit Value® for occupational exposure, the empirically derived dispersion factor for air releases from Y-12 to Scarboro, and estimated average EFPC flow rates. While production information for zirconium remains classified, it was reviewed by project team members and clearly indicates that quantities of zirconium at Y-12 in any given year were less than the threshold release rates to air or water that were calculated.

Results of Task 7 Screening

The results of the screening analyses of materials that were quantitatively evaluated are shown in Table ES-1. For each of the 13 assessments depicted, the table identifies:

- The identity of the contaminant and its source (K-25 Site or Y-12 Plant in most cases).

- The identity of the reference location for which concentrations, doses, and screening indices were estimated. These reference locations were selected as the areas where the highest off-site exposures likely occurred.

- The calculated cancer screening index for materials evaluated as carcinogens. These values were estimated by multiplying the total dose of a chemical by its cancer potency slope factor, or the radiation dose from a radionuclide times a risk factor of 7.3% Sv^{-1}.

  Results are presented for a Level I evaluation, and for a refined Level I analysis where applicable. Values are shown in bold when they exceed the decision guide in use on the project (that is, the screening index is 1×10^{-4} or greater).

- The non-cancer screening index for materials associated with toxic effects other than cancer. These values were in most cases calculated by dividing the dose of a chemical by its USEPA reference dose. Cases where different approached had to be used (for chemicals without established reference doses) are identified in the "Notes" column.[1]

In cases where doses were compared to reference doses for both inhalation and ingestion, the screening index that represents the largest fraction of (or multiple of) the applicable reference dose is provided. In each case described here, the highest screening indices resulted from comparing doses from ingestion to the oral reference dose.

Results are presented for the Level I evaluation, and for a Refined Level I analysis where applicable. Values are shown in bold when they exceed the decision guide in use on the project (that is, the screening index is 1 or greater).

- Notes are provided to indicate where non-standard approaches had to be used in an assessment. Notes are also provided, for non-carcinogens, to describe the relationship between the reference dose and a relevant toxicologic reference value, such as a NOAEL or LOAEL. In cases where screening indices indicate potential doses above the applicable reference dose, it is important to know how much separation there is between the reference dose and the NOAEL or LOAEL (that is, how much of a safety factor there is) in order to be able to evaluate the potential for health effects.

Examination of the results in Table ES-1 shows that:

- For carcinogens, 3 of 10 analyses ended with the Level I screening (Np-237 from K-25, Np-237 from Y-12, and tritium from Y-12). In other words, the initial, most conservative screening calculations for these materials yielded results that were below the decision guide in use on the project.
- For the other seven assessments of carcinogens, refined screening was performed. Of these refined assessments, two yielded results that were still above the applicable decision guide (arsenic from K-25 and arsenic from Y-12). The other five were below the decision guide with refined screening (beryllium from Y-12, chromium(VI) from the ORR, nickel from K-25, technetium-99 from K-25 and technetium-99 from Y-12).
- For non-carcinogens, 1 of 8 analyses ended with the Level I screening (beryllium from Y-12). In other words, the initial, most conservative screening calculations for beryllium yielded results that were below the decision guide in use on the project.

---

[1] Reference doses were derived by the project team for niobium (from an LD_{50} in mice), lithium (from the therapeutic dose of lithium carbonate), and copper and zirconium (from ACGIH Threshold Limit Values®, TLVs®).

Table ES-1: Summary of Task 7 Screening Results for Materials Evaluated Quantitatively

Material	Cancer Screening Index	Non-cancer Screening Index	Notes
Arsenic from K-25 *exposure for Union/Lawnville*	**Level I = 3.8×10^{-2}** **Refined Level I = 8.9×10^{-4}**	**Level I = 120** **Refined Level I = 13**	The NOAEL (from a human study) is a factor of 3 above the RfD. A non-cancer Screening Index above 3 could indicate exposures above the NOAEL.
Arsenic from Y-12 *exposure for Scarboro*	**Level I = 1.8×10^{-2}** **Refined Level I = 2.6×10^{-4}**	**Level I = 41** **Refined Level I = 4.0**	The NOAEL (from a human study) is a factor of 3 above the RfD. A non-cancer Screening Index above 3 could indicate exposures above the NOAEL.
Beryllium from Y-12 *exposure for Scarboro*	**Level I = 4.0×10^{-4}** Refined Level I = 1.3×10^{-5}	Level I = 0.066	The NOAEL (based on a rat study) is a factor of 100 above the RfD. A non-cancer Screening Index above 100 could indicate exposures above the NOAEL.
Chromium(VI) from the ORR *estimated based on Union/Lawnville air levels and EFPC/Poplar Creek water and soil/sediment data*	**Level I = 1.3×10^{-4}** Refined Level I = 1.0×10^{-5}	**Level I = 9.7** Refined Level I = 0.55	The NOAEL (based on a rat study) is a factor of 800 above the RfD. A non-cancer Screening Index above 800 could indicate exposures above the NOAEL.
Copper from K-25 *exposure for Union/Lawnville*		**Level I = 2.4** Refined Level I = 0.13	The LOAEL (from human studies of gastrointestinal effects) is a factor of 2 above the RfD. A non-cancer Screening Index above 2 could indicate exposures above the LOAEL.
Lead Releases from Y-12 *based on levels in EFPC water and soil/sediment, estimated average urban air levels prior to the 1970s*		**Level I = 5.2 to 6.7** **Refined Level I = 1.8 to 2.3**	These Screening Indices are based on the calculated range of blood lead levels divided by the CDC/USEPA action level of 10 µg dL^{-1} . Non-cancer Screening Indices above 1 could indicate exposures above the CDC action level.

Table ES-1: Summary of Task 7 Screening Results for Materials Evaluated Quantitatively

Material	Cancer Screening Index	Non-cancer Screening Index	Notes
Lithium from Y-12 *exposure for Scarboro*		**Level I = 2.3** Refined Level I = 0.29	The derived RfD is a factor of 10 below the normal therapeutic dose. Lithium toxicity can occur at doses close to therapeutic levels. A non-cancer Screening Index above 10 could indicate exposures above the therapeutic dose.
Neptunium-237 from K-25 *exposure for Union/Lawnville*	Level I = 7.3×10^{-6}		
Neptunium-237 from Y-12 *exposure for Scarboro*	Level I = 6.8×10^{-6}		
Nickel from K-25 *exposure for Union/Lawnville*	**Level I = 1.1×10^{-4}** Refined Level I = 6.1×10^{-6}	**Level I = 12** Refined Level I = 0.75	The NOAEL (from a rat study) is a factor of 250 above the RfD. A non-cancer Screening Index above 250 could indicate exposures above the NOAEL.
Technetium-99 from K-25 *exposure for Union/Lawnville*	**Level I = 3.0×10^{-2}** Refined Level I = 1.8×10^{-5}		
Technetium-99 from Y-12 *exposure for Scarboro*	**Level I = 2.7×10^{-4}** Refined Level I = 2.8×10^{-5}		
Tritium from Y-12 Heavy Water *exposure for Scarboro*	Level I = 1.6×10^{-6}		

For the other seven assessments of non-carcinogens, refined screening was performed. Of these refined assessments, three yielded results that were still above the applicable decision guide (arsenic from K-25, arsenic from Y-12, and lead from the Y-12 Plant). The other four were below the decision guide with refined screening (chromium(VI) from the ORR, copper from K-25, lithium from Y-12, and nickel from K-25).

Several materials were quantitatively evaluated in Task 7 that do not appear in Table ES-1. These are the materials that were evaluated using the threshold quantity approach, a method that does not yield numerical screening indices. Materials that fall in this category include tetramethyl-ammoniumborohydride (TMAB), niobium, and zirconium. Based on evaluation of the limited information available on these materials, it was determined that the quantities of each that were present at the Y-12 Plant were not likely great enough to have posed off-site health hazards.

Conclusions of the Task 7 Study

Based on the qualitative and quantitative screening performed under Task 7 of the Oak Ridge Dose Reconstruction project, it was possible to separate materials into classes based on their apparent importance in terms of potential off-site health hazards. This classification process was to a great degree dependent on the information that is available concerning past uses and releases of the materials of interest. In the course of Task 7 work, it was not possible to perform extensive directed searches for records relevant to each Task 7 material to the extent that was possible for the operations and contaminants studied in detail under Tasks 1, 2, 3, 4, and 6. For some materials, very little historical information is available. As a result, it was necessary to make a significant number of conservative assumptions for some materials to ensure that potential doses were not underestimated. If, in the future, more extensive document searching is performed, some of the conclusions reached in the screening evaluations described herein might well change.

Based on the qualitative and quantitative screening, five materials used at the K-25 Site and 14 materials (or classes of materials) used at the Y-12 Plant are judged to not warrant further study related to their potential for off-site health effects. These materials are identified in the second column of Table ES-2. The materials named to this category were placed there because either:

1) Quantitative screening of the most conservative nature (Level I screening) yielded screening indices that fell below the guides in use on the project;

2) Application of a threshold quantity approach demonstrated that not enough of the material was present to have posed an off-site health hazard; or

3) In qualitative evaluation of available information by project team members, it became obvious that quantities used, forms used, and/or manners of usage were such that off-site releases could not have been sufficient to have posed off-site health hazards.

Based on quantitative screening, three materials used at the K-25 Site, three materials (or classes of materials) used at the Y-12 Plant, and one material used at all sites were identified as potential candidates for further study. These materials are:

- From the K-25 Site:
  - copper powder
  - nickel
  - technetium-99
- From the Y-12 Plant:
  - beryllium compounds
  - lithium compounds
  - technetium-99
- From the ORR:
  - chromium(VI)

These materials were identified as potential candidates for further study because quantitative screening of the most conservative nature (Level I screening) yielded screening indices that exceeded the decision guide in use on the project. As shown in Table ES-2, some of these materials exceeded the decision guides as carcinogens, while others exceeded published or derived reference doses for materials that cause effects other than cancer. When less conservative, "refined" screening was performed for each of these materials, results in each case fell below the decision guides.

Based on quantitative screening, one material used at the K-25 Site and two used at the Y-12 Plant were identified as high priority candidates for further study. These materials are:

- From the K-25 Site:
  - arsenic
- From the Y-12 Plant:
  - arsenic
  - lead

These materials were identified as high priority candidates for further study because less conservative, "refined" quantitative screening yielded screening indices that exceeded the decision guide in use on the project. As shown in Table ES-2, arsenic achieved this status as both a carcinogen and as a non-carcinogen, while lead achieved this status as a non-carcinogen.

For the non-carcinogenic contaminants with screening indices that exceeded 1 in refined screening, it is important to evaluate the relationship between the reference dose and toxicologic reference levels such as the NOAEL or LOAEL. The importance of a screening index above 1 varies from one material to the next, because the amount of separation between the reference dose and the dose at which health effects have been shown to occur varies significantly. For this project, the materials for which this type of evaluation is most critical are arsenic and lead. Following are summaries of the relationships between screening indices for these materials and applicable NOAELs or action levels.

Table ES-2: Categorization of Evaluated Materials Based on Screening Results

CONTAMINANT SOURCE	NOT CANDIDATES FOR FURTHER STUDY (Level I result less than the decision guide)	POTENTIAL CANDIDATES FOR FURTHER STUDY (Refined Level I result less than the decision guide)	HIGH PRIORITY FOR FURTHER STUDY (Refined Level I result greater than the decision guide)
K-25 SITE	Neptunium-237 (cancer) Evaluated qualitatively: carbon fibers, four-ring polyphenyl ether, glass fibers, Triplex coating	Arsenic (cancer) Arsenic (non-cancer) Copper powder (non-cancer) Nickel (cancer) Nickel (non-cancer) Technetium-99 (cancer)	Arsenic (cancer) Arsenic (non-cancer)
Y-12 PLANT	Beryllium Compounds[1] (non-cancer) Neptunium-237 (cancer) Niobium[2] (non-cancer) Tetramethylammoniumborohydride (TMAB) Tritium (cancer) Zirconium[3] (non-cancer) Evaluated qualitatively: boron carbide, boron nitride, rubidium nitrate, rubidium bromide, tellurium, titanium boride, yttrium boride, zirconium	Arsenic (cancer) Arsenic (non-cancer) Beryllium Compounds (cancer) Lead (non-cancer) Lithium Compounds[4] (non-cancer) Technetium-99 (cancer)	Arsenic (cancer) Arsenic (non-cancer) Lead (non-cancer)
THE OAK RIDGE RESERVATION *(ALL COMPLEXES)*		Chromium(VI) (cancer) Chromium(VI) (non-cancer)	

---

[1] Forms of beryllium used include beryllium hydride, beryllium deuteride, beryllium metal, beryllium oxide, niobium beryllide, tantalum beryllide.

[2] Niobium was used as part of “mulburry” alloy (depleted uranium, niobium, and zirconium) and “binary” alloy (depleted uranium and niobium).

[3] Zirconium was used as part of “mulburry” alloy (depleted uranium, niobium, and zirconium).

[4] Forms of lithium used include lithium chloride, lithium deuteride, lithium fluoride, lithium hydride, and lithium tetraborate.

- For arsenic, the NOAEL (from a human study) is a factor of three above the oral reference dose. Non-cancer screening indices above 3 could indicate that exposures above the NOAEL occurred. Because the screening indices from the refined assessments are 13 and 4.0 for K-25 and Y-12, respectively, it is possible that doses above the NOAEL were experienced. At the same time, it is impossible to say if health effects occurred or not.

- For lead, it has been reported that adverse health effects can occur in children at blood lead concentrations as low as 10 µg dL^{-1}; this is the action level set by CDC in 1991. Non-cancer screening indices for lead in Table ES-1 were based on the ranges of blood lead concentrations calculated with the IEUBK model, divided by the CDC action level of 10 µg dL^{-1}. Because the screening indices from the refined assessment for lead range between 1.8 and 2.3, it is possible that doses above action level were received by some children. It is less likely that any adults received doses that exceeded the OSHA standard of 40 µg dL^{-1}.

Some of the materials evaluated in this project have very limited toxicologic information available. For example, very little is known about potential effects from exposure to lithium at levels that can be expected to occur in the environment. Reference doses for niobium, lithium, copper, and zirconium are not available from the USEPA. For the purposes of this study, reference doses were derived from studies of lethal doses in mice (niobium), from ACGIH Threshold Limit Values® for the workplace (copper and zirconium), and from therapeutic doses used in humans (for lithium). If better toxicologic data become available for these materials, the analyses described herein would likely benefit from evaluation with that new information. This is particularly true for lithium, which was evaluated based on a "derived" reference dose equal to the lithium equivalent of one-tenth of the dose of lithium carbonate used in humans to control mania. Because lithium toxicity can occur at doses close to the therapeutic dose, the lack of information concerning effects of exposure to lithium at environmental levels is an important data gap.

1.0 INTRODUCTION

The variety and complexity of past operations and materials used on the Oak Ridge Reservation (ORR; see Figure 1-1), both radionuclides and hazardous chemicals, place the ORR among the most complex sites in the world. Several well-publicized occurrences of environmental contamination raised concerns among some members of the public regarding potential health hazards to people who have lived near the ORR. In 1991, the U.S. Department of Energy (DOE) and the State of Tennessee Department of Health (TDH) entered into a Health Studies Agreement. Among the goals of the agreement was to assemble a panel charged with designing a study to evaluate the feasibility of conducting a dose reconstruction of past releases from Oak Ridge facilities**S** in effect, an independent investigation of the potential for heath effects from past Oak Ridge operations. The Oak Ridge Dose Reconstruction Feasibility Study was conducted from 1992 to 1993. In it, investigators took an intense and comprehensive but relatively quick “look through the key hole” at past Oak Ridge operations and performed screening calculations to identify those operations and materials that warranted investigation in terms of potential off-site exposures.

At the close of the Feasibility Study, TDH and the Oak Ridge Health Agreement Steering Panel (ORHASP) recommended that dose reconstructions be conducted for radioactive iodine releases from X-10 radioactive lanthanum processing, mercury releases from Y-12 lithium enrichment, polychlorinated biphenyls (PCBs) in the environment near Oak Ridge, and radionuclides released from White Oak Creek to the Clinch River. They called for the study to also include systematic searching of historical records, an evaluation of the quality of historical uranium effluent monitoring data, and additional screening of some materials that could not be evaluated during the Feasibility Study. In addition, several questions raised during the Feasibility Study that could not be answered at the time (regarding assumptions made in assessing X-10 plutonium releases and the potential for off-site exposure to asbestos) were slated for evaluation during the dose reconstruction project.

The Oak Ridge Dose Reconstruction began in late 1994. The project was designed to develop detailed estimates of historical doses and health risks from the selected contaminants to people who lived in off-site areas near the ORR, and included seven tasks, as follows:

Task 1**S** Iodine-131 Releases from X-10 Radioactive Lanthanum Processing
Task 2**S** Mercury Releases from Y-12 Lithium Enrichment
Task 3**S** PCBs in the Environment Near Oak Ridge
Task 4**S** Radionuclides Released from White Oak Creek to the Clinch River
Task 5**S** Systematic Search of Historical Records
Task 6**S** Evaluation of the Quality of Uranium Monitoring Data and Screening-level Evaluation of Potential Off-site Health Hazards
Task 7**S** Evaluation of Additional Potential Materials of Concern

This report presents the methods and results of the Task 7 screening of additional potential materials of concern. The Task 7 investigation included quantitative, screening-level evaluations of 10 materials or classes of materials and less detailed evaluations of 18 others.

The purposes of this report are to:

- Present additional information regarding questions raised in the Feasibility Study that could not be answered at the time,
- Describe the methods used to evaluate the potential importance of additional materials,
- Describe steps taken to identify and evaluate materials present on the ORR that were classified by their mere presence (termed to be classified "*per se*") or that had classified aspects of use, in terms of their potential to pose off-site health hazards, and
- Present screening-level evaluations of the significance of a number of materials in terms of their potential to pose off-site health hazards.

Subsequent sections of this document are organized as follows:

Section 2.0, Methods Used for ScreeningS presents an overview of the methods used by the project team to screen materials, to evaluate their importance in terms of their potential to pose off-site health hazards.

Section 3.0, Issues Remaining from the Feasibility StudyS addresses several issues that could not be fully addressed in the Feasibility Study, including evaluation of the possible off-site health risks associated with asbestos on the ORR, and further evaluation of the composition of plutonium formed and in part released to the environment from the ORR.

Section 4.0, Materials Screened Using Methods Other than the Standard Task 7 ApproachS describes the methods used and results of screening for materials that were screened using methods other than the standard Task 7 screening approach, including evaluation of materials with classified aspects, tritium, and lead.

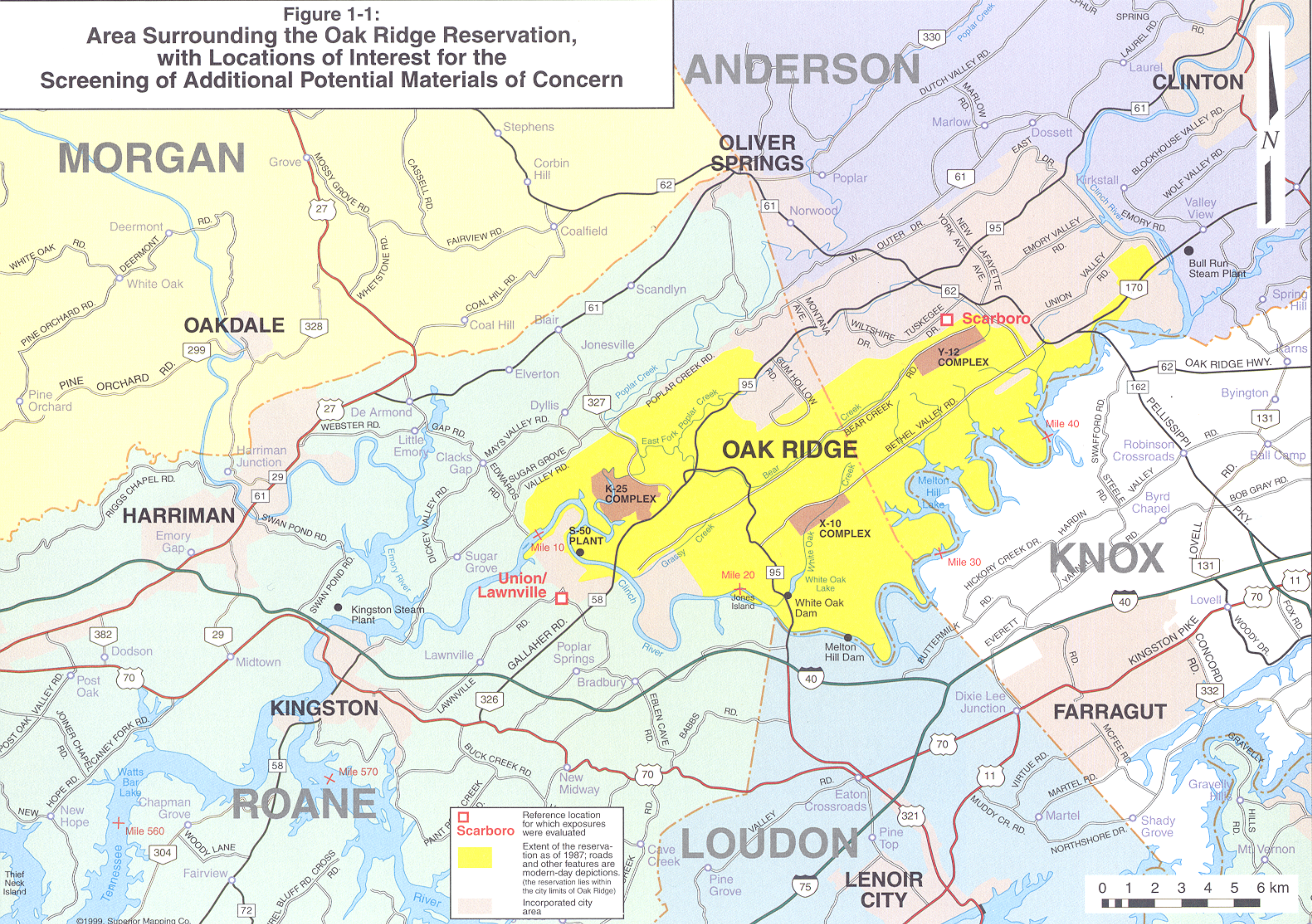
Section 5.0, Materials Screened with the Task 7 MethodologyS discusses the results of screening for materials that were screened using the Task 7 screening approach.

Section 6.0, Summary of Task 7 ResultsS presents a table summarizing the risk-based screening indices calculated using the Task 7 screening approach for materials of interest at the ORR.

Section 7.0, ConclusionsS presents recommendations for further analysis of the materials evaluated in the screening process.

Section 8.0, ReferencesS provides the references used in this assessment.

Figure 1-1: Area Surrounding the Oak Ridge Reservation, with Locations of Interest for the Screening of Additional Potential Materials of Concern



This page intentionally left blank

2.0 METHODS USED FOR SCREENING

The purpose of screening in the Oak Ridge Dose Reconstruction was to permit attention and resources to be focused on the most important contaminants and to avoid dilution of resources by identifying situations that are obviously of only minor importance. Three different methods were used by the Task 7 project team to evaluate the importance of materials in terms of their potential to pose off-site health hazards. The method selected to evaluate a given material was dependent on the quantity of the material present on-site, the form and manner in which the material was used, and the availability of environmental monitoring and release data, as well as whether the material was classified *per se* (i.e., its mere presence on the ORR remained classified). Fortunately, before this project was completed, the presence of any material on the ORR (at the site level) could be publicly revealed.

Briefly, the methods used to screen materials were as follows:

- *Qualitative screening***S** all materials identified as having been used on the ORR were subject to qualitative screening; for some materials, the project team determined that based on evaluation of quantities used, forms used, and/or manners of usage, it was unlikely that off-site releases of the material could have been sufficient to pose an off-site health hazard; these materials were not subject to quantitative screening.

- *Threshold quantity approach***S** for some materials, information describing on-site inventories or release quantities was insufficient to conduct a quantitative screening using the approach described below; for these materials, a threshold inventory quantity was calculated, and materials present on-site at quantities below this value were assumed to have little likelihood of being released off-site in quantities that could pose a health hazard.

- *Quantitative screening* using a two-level screening approach**S** each level used a different set of assumptions to calculate potential doses and screening-level risk indices; the goal of this approach is to identify those contaminants that produced doses or health risks to exposed individuals or populations that are clearly below established minimum levels of concern (Level I Screen) and identify those contaminants that produced doses or health risks to exposed individuals or populations that are likely to have been above the established minimum levels of concern (Level II or Refined Level I Screens).

This section presents an overview of each screening method. Screening assessments for specific materials conducted using these methods are presented in later sections of this report.

2.1 Qualitative Screening

All materials identified by the project team as historically being present on the ORR were subject to qualitative screening. For some materials used on the ORR, the project team determined that *quantities used*, *forms used*, and/or *manners of usage* were such that it is unlikely that off-site releases were sufficient to have posed health hazards. These materials were given low priority for further evaluation in the screening process.

In performing the qualitative evaluation of materials used on the ORR, it was important that project team members have a broad understanding of the nature of past ORR operations and the characteristics of materials used in these operations. All of the Task 7 project team members involved in the qualitative screening were familiar with the comprehensive review of historical ORR operations performed in the earlier Dose Reconstruction Feasibility Study. In addition, members of the team had extensive technical training in environmental toxicology and/or health physics as well as several years of experience in health risk assessment for toxic chemicals and radionuclides in a wide variety of applications and industries, including nuclear weapons complexes in Colorado, Idaho, and Tennessee. This knowledge allowed team members to make defensible judgements regarding the likelihood that material releases could have posed off-site health hazards.

Quantities Used– Some materials listed on hazardous material inventories were present on the ORR in very small quantities, below quantities required for materials that played an important role in production activities. Small-quantity materials included chemicals and radionuclides used as calibration standards or check sources for laboratory instruments or analytical methods. For many materials, the quantities present were so small that it was apparent they could not have posed an off-site health hazard; these materials were given low priority for further investigation.

Evaluation of small-inventory materials was in some cases supported by threshold inventory quantities estimated for other materials. Once a threshold inventory quantity was calculated for a material, below which off-site health hazard would not be likely (see Section 2.2), that information could by analogy be helpful in evaluating other materials with similar physical characteristics and behaviors in the environment. For example, if the threshold inventory quantity for Compound A was 100,000 kg, and Compound B had similar characteristics of behavior but was known to be less toxic than Compound A (possibly indicated by a higher USEPA reference dose or a lower cancer slope factor), it is unlikely that inventory quantities of Compound B below 100,000 kg would have led to off-site health hazards.

Forms Used– Some materials were given low priority for further investigation because the physical forms in which they were used were not conducive to off-site release. For example, carbon fibers and glass fibers were received at the K-25 site as premanufactured filaments, which were likely wound on spools. These fibers were used in construction of rotors used in the centrifuge method of uranium enrichment, in a process by which they were wound on a spool and a plastic binder applied to form the tall, cylindrical rotors.

Manners of Usage– Some materials were given low priority for further investigation because they were used in manners that made off-site release unlikely. For example, liquids, gases, or powders that were kept sealed in cylinders or processed in containment systems that included multiple barriers against release were unlikely to have been released to the off-site environment in significant quantities. Placement of a material in this category required verification of the apparent effectiveness of the containment measures through review of historical records, interviews of active or retired workers, and/or analysis of material accountability records. Other materials that could fall into this category include those that were transformed after receipt into forms that were not toxic or of minimal toxicity.

2.2 The Threshold Quantity Approach

The quantities of many materials present at ORR complexes were so small that, after dispersion, dilution, and loss between the release point and the location of off-site exposure, it was unlikely that these materials would have been present in concentrations sufficient to have posed a health risk; these materials were identified as low priority for further investigations of potential off-site health risks.

Accurate estimates of inventory quantities of materials used at the Oak Ridge complexes are often not available, or in some cases not publicly releasable. It is typically much easier to determine, based on historical records or interviews of active or retired workers, if inventory quantities of a material were below a calculated threshold quantity. In many cases, while workers may be hesitant to make a guess of quantities that were actually present, they can often much more confidently state that quantities on hand were clearly below a stated threshold quantity. In addition, statements of the nature that quantities of a material on hand were "over two orders of magnitude" below the stated threshold quantity can often be publicly released, while inventory quantities often cannot.

For such materials, project investigators used conservative assumptions to calculate a "threshold quantity" below which a material was highly unlikely to have posed a risk to human health through off-site releases. Threshold quantities were calculated using the following approach:

1. The *maximum allowable air concentration or water concentration* of a material was calculated based on the maximum allowable daily dose (assumed to be equivalent to the noncarcinogenic reference dose (RfD) or the dose that would produce a cancer risk of 1×10^{-6}). To calculate a maximum allowable concentration, the maximum allowable daily dose was multiplied by a typical body weight and divided by a typical breathing rate or water ingestion rate.

2. The *maximum allowable release rate* to air or water was then determined, by calculating the release rate that would give an air or water concentration equal to the maximum allowable air concentration or water concentration. Release rates were calculated using conservative environmental dispersion or dilution factors.

3. The *maximum allowable release rate* in g s^{-1} was then converted to a maximum allowable release rate in kg y^{-1}. This quantity was assumed to be the threshold inventory quantity for the material.

2.3 The Standard Task 7 Screening Method

For materials with more detailed information on quantities used, manners of usage, or quantities released, a more comprehensive quantitative two-level screening approach was used to evaluate the likelihood that these materials were released off-site in quantities that could have posed a health hazard to off-site populations. The results of the screening analyses described below were used to identify materials as low, medium, and high priority for further study.

2.3.1 Purposes of the Standard Screening Method

A two-level screening approach, described in more detail below and in a separate project report (ChemRisk 1996), was used to:

- *Identify those contaminants that produced doses or health risks to exposed individuals or populations that are clearly below established minimum levels of concern (Level I Screen)*— these materials were assigned a low priority for further study; and

- *Identify those contaminants that produced doses or health risks to exposed individuals or populations that are likely to have been above the established minimum levels of concern (Refined Level I or Level II Screen)*— these materials were assigned the highest priority for detailed study.

Each level of the two-level screening approach used a different set of assumptions to calculate potential doses and screening-level risk indices. The results of the screening calculations were then compared to minimum levels of concern (risk-based decision criteria) established by the Oak Ridge Health Agreement Steering Panel (ORHASP).

2.3.2 The Task 7 Approach to Screening Calculations

The Level I screen was designed to estimate the dose or risk to a “maximally exposed” reference individual who should have received the highest exposure and thus would have been most at-risk. This level incorporated conservative exposure parameter values (e.g., intake rates) not expected to lead to an underestimate of risk to any real person in the population of interest. If doses and risk indices from the Level I Screen were below the risk-based decision guides, then it was assumed that risks to essentially all members of a population, including the maximally exposed individual, would be below minimum levels of concern, and a Refined Level I or Level II screen for this material was not conducted.

In this report, the screening evaluations that “Refined Level I” screening evaluations were performed described in this report were considerably less conservative than the Level I evaluations they followed. As described in Appendix C, for example, many of the exposure parameter values used in the dose and risk calculations are less conservative (more realistic or more typical) than the values of the same parameters used in Level I screening. A good example would be the assumed exposure duration for carcinogens, which is 50 years in Level I screening and 10 years in Level II screening. "Refined Level I" evaluations described in this report used the Level II exposure parameters.

While a general goal in refined screening is to eliminate sources of conservative bias, it is not always feasible or advisable to eliminate all conservative bias, or easily determined when a sufficient level of realism has been achieved. In the refined screening evaluations described in this report, some degree of conservatism was retained, particularly in the estimation of contaminant concentrations in environmental media of interest. One important reason for this is that there were very few measurements of the contaminants of concern made in the environment during the (pre-1970s) periods when levels of many contaminants in the environment were likely the highest. Measurements in process streams or effluents are even more rare. Because of the paucity of information for some vital components of the risk assessment process, some conservatism was retained in the estimation of exposure point concentrations for the Refined Level I assessments to ensure that exposures were not underestimated for significant portions of the potentially exposed populations. Because of this, the second-level assessments are called Refined Level I assessments rather than Level II assessments.

Refined Level I and Level II screens are designed to estimate the dose or risk to a more typical individual in the population of interest. They incorporated reasonable average or more typical values for the exposure parameter values. It was assumed that the Level II screening value underestimated the dose or risk for the most highly exposed individual, although the dose or risk may be overestimated for the general population. If doses and risk indices from the Level II Screen were below the risk-based decision criteria, then it was assumed that risks to most members of a population would be below minimum levels of concern, and this material was given a low priority for further evaluation.

If doses and screening-level risk estimates ("screening indices") from the Refined Level I or Level II Screen were above the risk-based decision criteria, then the material was identified as a high priority material for further evaluation.

Both the Level I and Level II screening calculations used mathematical equations for calculation of dose and risk through multiple exposure pathways. These equations relate dose to the exposure point concentration and the magnitude of intake. For example, the dose of a chemical through a given pathway (in milligrams per kilogram of body weight per day) was estimated using the following general equation:

$$Dose\ (mg\ kg^{-1}\ d^{-1}) = \frac{C \times U \times EF \times ED}{BW \times AT}$$

Where:

C = Concentration of a chemical at the exposure point (mg kg^{-1}, mg L^{-1}, or mg m^{-3})

U = Intake rate (breathing rate [$m^3\ d^{-1}$], drinking rate [$L\ d^{-1}$], or ingestion rate [$kg\ d^{-1}$])

EF = Exposure frequency ($d\ y^{-1}$)

ED = Exposure duration (y)

BW = Body weight (kg)

AT = Averaging time (d)

Pathway equations used in the screening assessment are presented in Appendix B. The equations included all pathways potentially significant for the contaminants in question; exposure pathways evaluated included inhalation, ground exposure (for radionuclides), ingestion of soil or sediment, vegetable ingestion, and ingestion of meat, milk, and/or fish.

Parameter values used to calculate dose were selected based on historical knowledge of the Oak Ridge area, literature review, and professional judgment**S** parameter values used are given in Appendix C. In general, more conservative parameter values were used in the Level I screen than in the Level II screen, in order to provide upper-bound estimates of exposure. The parameters that were varied between screening levels included lifestyle factors such as intake rates, time spent outdoors, etc. Contaminant-specific transfer factors (i.e., factors describing the transfer of a contaminant to milk or meat or uptake of a contaminant from soil into vegetation) and toxicity values were kept constant for both levels of screening. In general, the potential toxicity of chemicals was evaluated using USEPA reference doses (RfDs) for noncarcinogenic effects and slope factors for carcinogenic effects**S** these toxicity values take into consideration the most sensitive health effects endpoints for the specified target individuals. For radionuclides, risks were evaluated using ICRP dose conversion factors.

Different exposure durations and averaging times were assumed for radionuclides, carcinogenic chemicals, and noncarcinogenic chemicals. For radionuclides and carcinogenic chemicals, exposure durations of 50 years and 10 years were used in the Level I and Refined Level I screening, respectively. For carcinogenic chemicals, the risk was calculated in terms of the total intake averaged over the estimated lifetime, assumed to be 70 years, to give a lifetime average daily intake. For radionuclides, the risk was calculated in terms of the total cumulative dose, and an averaging time was not needed. For noncarcinogenic materials, an exposure duration and averaging time of one year were used.

2.3.3 Reference Locations for Screening Calculations

For each ORR complex that used materials screened in Task 7 (that is, the K-25 and Y-12 complexes), a reference location for exposure assessment was selected based on mathematical modeling of the dispersion of airborne releases from that complex and evaluation of sampling/ monitoring data for surface water, soil, or sediment. In general, the goal was to identify the residential area likely to have experienced the highest average concentrations of airborne and waterborne contaminants.

The following reference locations, shown in Figure 1-1, served as the primary locations of interest for the Task 7 screening assessments:

- *The Scarboro Community*– The Scarboro community was selected as the reference location for materials released from Y-12 because of the community's close proximity to the northern portions of the facility (approximately 1 km to the north of the Y-12 facility). Although Scarboro is located on the opposite side of Pine Ridge, airborne uranium monitoring data collected near the Scarboro community center indicate that Pine Ridge is not a perfect barrier to airborne contaminants from Y-12, and that some Y-12 releases have been transported to Scarboro (ChemRisk 1997). The closest water body to Scarboro is East Fork Poplar Creek (EFPC), which runs along the south side of the Y-12 complex and passes north of the Scarboro community further downstream. EFPC does not flow through Scarboro, but is within walking distance for Scarboro residents. Interviews indicate that some Scarboro residents have played, fished, and caught turtles in EFPC.

- *The Union/Lawnville Community*– For releases from the K-25 Complex, the Union/ Lawnville community was selected as the reference location. This community is located approximately 4.5 km south-southwest of the K-25 Site. Based on air dispersion modeling performed as part of the Task 6 assessment of K-25 uranium releases, as well as assessment of areas around the K-25 complex that were inhabited during years of past operations, this area was selected as most representative of maximum exposures to airborne contaminants released from K-25 operations. The location of this community is defined by Union Church, which is located on Lawnville Road approximately 1 km north of Gallaher Road. The surface water body closest to Union/Lawnville is the Clinch River, which is approximately 1.5 km northeast of Union Church.

2.3.4 Estimates of Environmental Concentrations used for Screening

Exposure point concentrations used in the Level I and II screening were typically based on available release information (source terms) and/or measured environmental concentrations. In general, for both the Level I and Refined Level I screening, upper bound exposure point concentrations were used because of the uncertainty associated with actual releases for most materials evaluated in Task 7. For example, doses and risks were typically calculated using the upper bound (e.g., 95th percentile or maximum) measured or modeled exposure point concentration at the location of the nearest downwind or downstream population center. Use of upper bound exposure point concentrations ensures that calculated doses were not underestimated.

In general, for a given contaminant and a given medium, the maximum concentration at or near the surface water location of interest for each plant site was selected for use in the screening evaluation. These locations represent the nearest location downstream of the plant facilities where people could have realistically come into contact with surface water.

Assumptions used to model dispersion of airborne releases to the Scarboro and Union/Lawnville communities are described below.

2.3.4.1 Air Concentrations at the Scarboro Community

Because of the complex terrain surrounding the Y-12 facility, a classical air dispersion modeling approach would over-estimate air concentrations at Scarboro resulting from releases of a contaminant from Y-12. For example, the ISCST3 model uses a flat terrain approach and would not account for the attenuation and redirection of wind flow away from Scarboro caused by the ridge-and-valley terrain. Although algorithms for complex terrain are available for the ISCST3 model, it is questionable if these algorithms could account for the abrupt change in topography. Further, the relative altitude of the Scarboro Community below the top of Pine Ridge further complicates the dispersion characteristics. Concentrations of contaminants in air at Scarboro due to direct airborne releases from Y-12 were therefore estimated using an empirical approach based on the ratio between measurements of airborne uranium at Scarboro and estimates of uranium releases from Y-12 developed by the project team (ChemRisk 1997). It is assumed that the relationship between contaminant concentrations in air at Scarboro and release rates of those contaminants from Y-12 is the same as the relationship between uranium air concentrations and release rates.

Empirical χ/Q values used to estimate airborne contaminant concentrations at Scarboro, based on uranium releases from Y-12, were developed as follows:

$$\text{Empirical}\ \chi/Q\ (s\ m^{-3}) = \frac{\text{Uranium Air Concentration Measured at Scarboro}\ (pCi\ m^{-3})}{\text{Uranium Release Rate}\ (pCi\ s^{-1})}$$

Empirical χ/Q’s were calculated for calendar years 1986 through 1995 (the years of uranium sampler operation at Scarboro) for two uranium istopes— $^{234/235}U$ and ^{238}U. Statistical analyses of the annual χ/Q values yields the summary statistics presented in Table 2-1. Although tests for conformance of the data set with various distributions were inconclusive, for this application, the data were treated as if normally distributed. For estimating airborne contaminant concentrations at Scarboro due to direct releases from Y-12, the empirical χ/Q value corresponding to the 95% upper confidence limit of the mean was used (3.1×10^{-7} s m^{-3}, which will be rounded to 3×10^{-7} s m^{-3}).

Table 2-1: Statistical Analysis of Empirical χ/Q Values for Y-12 Uranium Releases

Statistic	**Empirical χ/Q (s m^{-3})**
Mean	2.2×10^{-7}
Standard deviation	2.3×10^{-7}
95th UCL of the mean	3.1×10^{-7}
Maximum	6.8×10^{-7}
Minimum	3.5×10^{-8}
Data points	20

Contaminant concentrations at Scarboro were then estimated as follows:

$$\mathit{Concentration\ at\ Scarboro}\ \left(\frac{\mu g}{m^3}\right) = \mathit{Emission\ Rate}\ \left(\frac{\mu g}{s}\right) \times 3\times10^{-7}\ \left(\frac{s}{m^3}\right)$$

2.3.4.2 Air Concentrations at the Union/Lawnville Community

The reference location for screening of airborne releases from the K-25 Site was Union/Lawnville, about 2.5 miles southwest of the K-25 Building. The project team did not locate any air concentration data for Union/Lawnville. Therefore, air concentrations at that area were modeled assuming a unit release (1 g s^{-1}) from the center of the K-25 Site. From the Task 6 assessment, the *X*/Q value for Union/Lawnville (more specifically at UTM coordinates X= 733000 and Y= 3976000) assuming a unit release was 7.4×10^{-7} s m^{-3}.

Contaminant concentrations at Union/Lawnville were then estimated as follows:

$$\mathit{Concentration\ at\ Union/Lawnville}\ \left(\frac{\mu g}{m^3}\right) = \mathit{Emission\ Rate}\ \left(\frac{\mu g}{s}\right) \times 7.4\times10^{-7}\ \left(\frac{s}{m^3}\right)$$

2.3.5 Decision Guides Used to Support Decision Making

As described briefly above, the results of the screening calculations were compared to minimum levels of concern (risk-based decision guides) established by the Oak Ridge Health Agreement Steering Panel (ORHASP). For radionuclides and carcinogenic chemicals, the decision guide was a lifetime excess cancer incidence of 1 in 10,000 (10^{-4}). For noncarcinogenic chemicals, the decision guide was a Hazard Index of 1.0. The Hazard Index is equal to the contaminant dose divided by its RfD, where the RfD is defined as a dose of a chemical that is not expected to cause adverse noncarcinogenic health effects over a lifetime of daily exposure.

For Level I screening, the screening value was compared to the appropriate decision guide as follows:

- If the screening estimate of risk to the maximally exposed individual was *below* the decision guide, it was concluded that further study of the contaminant can be deferred until time and resources permit further study, because risks to members of the general population would be even lower. Continued expenditure of time and resources on that contaminant is not justified as long as there are more important situations to be studied.

- If the screening estimate of risk to the maximally exposed individual was *above* the decision guide, it was concluded that the contaminant should be further evaluated in refined Level I screening or in Level II screening.

For Refined Level I screening, the screening value was compared to the appropriate decision guide as follows:

- If the screening value was *above* the decision guide, it was concluded that the contaminant should be given high priority for detailed study, because it is likely that some individuals received exposures or doses high enough to warrant further investigation.

- If the screening value was *below* the decision guide, the contaminant was deferred for further study to later phases of the project, after the highest priority contaminants are evaluated.

3.0 ISSUES REMAINING FROM THE FEASIBILITY STUDY

Task 7 of the Oak Ridge Dose Reconstruction included a component for additional evaluation of issues that could not be fully addressed in the Feasibility Study. These issues included evaluation of the possible off-site health risks associated with asbestos on the ORR, and further evaluation of the composition of plutonium formed and in part released to the environment from the ORR. This section describes the evaluations conducted by the project team to address these issues.

3.1 Asbestos on the Oak Ridge Reservation

Asbestos was historically used at the ORR in a variety of applications, most notably in the insulation of process components, buildings, and residences. Elsewhere, considerable public health concerns have arisen about the possibility of asbestos exposure in schools, homes, and the workplace. More recent scientific evidence has shown that the risks from asbestos in such buildings may have been greatly overestimated. The following section summarizes available information on the use and disposal of asbestos on the ORR and discusses factors that affected the potential for off-site releases and exposures in nearby communities.

3.1.1 Asbestos Use and Disposal Practices at the ORR

The activities that serve as the principal contributors to environmental asbestos typically include manufacturing of asbestos products (such as insulation, gaskets, automotive brake linings and other friction materials, textiles, and cement pipe) and mining, milling, and quarrying operations (ATSDR 1995). None of these activities were performed on the ORR. Less significant sources of asbestos release to the ambient environment include frictional wear of brake linings and remedial activities associated with asbestos-containing buildings. Any asbestos released from the ORR would have been associated with these kinds of uses, not as a result of asbestos-related manufacturing. The environmental concentrations that result from such uses and activities as have occurred on the ORR may be comparable to the levels of asbestos that are naturally present in the environment. Asbestos is a naturally occurring mineral found in rock formations, such as serpentine[1].

Numerous documents maintained by ChemRisk contain references to asbestos use and disposal at the ORR. The project team did not identify any uses of asbestos in production at the three plants. It is believed that use of asbestos at the ORR was primarily limited to building construction materials and systems. Most of the asbestos information is related to disposal sites at the ORR where asbestos-containing waste materials were disposed. This information does not begin to appear in the site documentation until the late 1970s.

---

[1] A mineral or rock consisting essentially of a hydrous magnesium silicate, usually having a dull green color and often a mottled appearance.

K-25 On-Site Disposal

At the K-25 Site, asbestos-containing classified equipment was disposed of in the K-1070-B Area, originally created by filling a wetlands area adjacent to a small creek that flowed into Poplar Creek (Goddard et al. 1991). The K-1070-B Area, also referred to as the Old Classified Burial Ground, was used from the early 1950s to 1976 (Goddard et al. 1991). Asbestos was also disposed of at the K-1070 C/D Classified Burial Ground at K-25; use of this disposal site was discontinued in 1989 (SAIC 1995). 2,318 kg of asbestos-covered lead pipe was placed in storage at K-25 in 1991 (Kornegay et al. 1992). Metal scrap and asbestos from pipe insulation was stored in the 30 acre K-770 Scrap Metal Yard on the east bank of the Clinch River, approximately 2,000 feet upstream of its confluence with Poplar Creek (Goddard et al. 1991). The Scrap Metal Yard was operated from the 1960s to 1991.

In 1990, 75 m^3 of radiological asbestos/beryllium waste was generated at K-25 [and probably was disposed of on-site] (Kornegay et al, 1991).

X-10 On-Site Disposal

In 1987, 600 and 34,000 kg of radiological and nonradiological asbestos material, respectively, was generated at ORNL (Rogers et al. 1988a). Another report indicates that in 1987, 575 kg radiological asbestos waste was buried at ORNL SWSA-6 (Rogers et al. 1988b). According to a flow diagram for hazardous waste sources (Rogers et al. 1988a), asbestos material, resulting from facility renovation and demolition at ORNL, was disposed of at SWSA-6 if contaminated with radioactive substances, or at the Y-12 plant centralized sanitary landfill if not contaminated with radioactive substances.

According to (Rogers et al. 1988a), more than two asbestos spills occurred at ORNL in 1987; the nature of these spills was not characterized. Waste materials, including asbestos, generated at Y-12 were disposed of at the ORNL Burial Site (MMES 1984). In 1990, 6,600 kg radiological asbestos waste was generated at ORNL [and was probably disposed of on-site] (Kornegay et al. 1991).

Y-12 On-Site Disposal

Over 400,000 kg of asbestos-bearing materials were placed in on-site burial at Y-12 between 1978 and 1985 (Fee 1986). Between 1979 and 1985, 193,817 ft^3 of asbestos were removed from K-25 and deposited in the Y-12 Burial Grounds (Goodpasture and Rogers 1986). Another report indicates, that between 1978 and 1984, approximately 390 tons (64 tons y^{-1}) of asbestos and asbestos-containing materials were disposed of at the Y-12 Asbestos Disposal Pits, located approximately 2.5 miles west on Bear Creek Road from the main portal of the Y-12 Plant. The source of this asbestos waste was plant maintenance and demolition activities (MMES 1984). Waste asbestos, containerized in plastic bags or drums, was placed in unlined trenches and covered with a minimum of two feet of soil. The disposal area is posted and on restricted-access government land. There are no residents located within a mile of the disposal pits. There are no records for burial of asbestos insulation prior to 1979 (MMES 1984).

During fiscal year 1986, 139,000 kg of asbestos was disposed of in the Y-12 Centralized Sanitary Landfill II and 172,000 kg of asbestos was disposed of in the Y-12 Bear Creek Burial Grounds (Oakes et al. 1987). According to Kornegay et al. 1991, the Y-12 Plant Centralized Sanitary Landfill II is the state-approved disposal site for nonradioactive asbestos waste.

In 1987, 3,700,000 kg of uncontaminated and 270,000 kg of uranium-contaminated asbestos/beryllium oxide material was generated at the Y-12 Site (Rogers et al. 1988a). During 1987, 31 m^3 of nonradiological asbestos waste was generated at the ORGDP and probably went to the Y-12 Landfill (Rogers et al. 1988a). Another report indicates, that in 1987, 33,518 kg nonradiological asbestos waste was sent from ORNL to Y-12 for on-site disposal (Rogers et al. 1988b).

In 1990, 217,000 kg of nonradiological asbestos waste and 84,638 kg of radiological asbestos waste was generated at Y-12 (Kornegay et al. 1991). In 1990, 1,332 m^3 of nonradiological asbestos/beryllium waste was generated at K-25 [and probably went to the Y-12 Landfill] (Kornegay et al. 1991). In 1990, 15,237 kg of nonradiological asbestos waste was generated at ORNL [and probably went to the Y-12 Landfill] (Kornegay et al. 1991). Also during 1990, two environmental releases of asbestos were reported at Y-12 (Kornegay et al. 1991). No details of the releases were provided.

In 1991, 88 m^3 of nonradiological asbestos was sent from K-25 to the Y-12 Landfill (Kornegay et al. 1992). In 1991, 19,229 kg asbestos was sent from ORNL to the Y-12 Sanitary Landfill (Kornegay et al. 1992).

Appendix D summarizes the information identified by the project team regarding disposal of asbestos-containing materials at the ORR.

Community Housing

In 1993, the City of Oak Ridge studied the types of asbestos-containing materials in housing units built by the government during the Manhattan Project. Fifteen housing units, selected to be representative of the 5000-6000 housing units built by the federal government in the 1940s in the City of Oak Ridge, were tested for asbestos-containing materials. The purpose of the survey was to identify typical building materials used in these units and to determine whether any of these materials contained asbestos. Asbestos was commonly used in thermal insulation and as a reinforcement material in products such as floor tile, shingles, and paint. To decide whether to renovate this housing to create more affordable housing, the City and the Oak Ridge Housing Development Corporation (HDC) wanted to factor in the potential cost of asbestos removal. 248 total samples were collected from the 15 units. Asbestos-containing materials were generally identified in insulation, cemesto (fiber board with cement-asbestos bonded to each side) exterior walls, linoleum, floor tile, and siding shingles in 14 of the 15 housing units. The asbestos in these materials ranged from 2% in floor tile to 45% in thermal insulation. The results were presented to the public during a seminar on asbestos in the home sponsored by the Oak Ridge Environmental Quality Board on September 12, 1993. No remedial activities were conducted on the asbestos-containing materials in the houses, and no environmental samples of the interiors or exteriors of the houses were taken (Peer 1993).

3.1.2 Off-Site Asbestos Monitoring

Only four documents identified by the project team make any reference to off-site releases or off-site monitoring for asbestos. However, the information that is available suggests that off-site monitoring has been performed in at least one investigation of asbestos releases from cooling tower operations. However, data associated with this investigation have not been located.

A Union Carbide memorandum dated November 29, 1976 indicated the need to "establish the base-line information needed for monitoring asbestos fibers in K-25 plant potable water and in streams." This memorandum also noted that development of an analytical method for monitoring asbestos fibers in the Paducah recirculating water systems was recently completed. In a Union Carbide memorandum dated January 4, 1977, it was reported that samples of ORGDP potable water and Clinch River water samples have been analyzed for asbestos fibers by transmission electron microscopy, and that a routine asbestos-in-water analysis support capability now exists. In a Union Carbide memorandum dated March 29, 1977, it was reported that a second survey for asbestos in water at the ORGDP area found 3 million fibers L^{-1} in one Poplar Creek sample. All other samples were below the detection limit. The memorandum also indicated that an extensive sampling program would be conducted in August 1977 to obtain baseline data prior to the installation of synthetic asbestos-containing tower fill material in a new cooling tower. This material was used as contact media for water to trickle over to lose heat (Lay 1997). A Union Carbide memorandum dated April 12, 1978 reported that for the first time, no asbestos fibers were found during quarterly monitoring of asbestos in surface water samples (Union Carbide 1978). In the same memorandum, a request was made to determine airborne concentrations of asbestos fibers downwind from cooling towers.

3.1.3 Toxicity of Asbestos

The health effects reportedly associated with human exposure to asbestos have only been documented for certain occupational activities, specifically those where substantial exposure has occurred. For example, studies of workers involved in the manufacture of asbestos-containing brake linings, insulation, and other materials have inferred a link between asbestos exposure and various respiratory diseases including lung cancer and mesothelioma (ATSDR 1995). In these situations, workers had been exposed to extremely high levels of asbestos. As a result, OSHA occupational exposure limits to asbestos are 0.1 fiber per cubic centimeter. Little is known about the potential health effects associated with the significantly lower levels of exposure that might be expected to result from asbestos present in the ambient environment. Although some have argued that exposure to a single asbestos fiber could lead to chronic health effects such as lung cancer or mesothelioma, such effects from environmental exposure have not been reported. Asbestos is a naturally occurring mineral found in rock formations, such as serpentine.

3.1.4 Potential for Significant Off-Site Asbestos Release and Community Exposure

The amount of information currently available (based on a review of documents maintained by ChemRisk) regarding historical asbestos releases from ORR is very limited, and is not sufficient for quantitatively reconstructing doses to off-site receptors. However, based on limited information about the historical uses and disposal practices at ORR, and consideration of background levels commonly found in the environment, it is possible to discern whether asbestos releases from ORR have resulted in toxicologically-significant exposures.

The Phase I Dose Reconstruction Feasibility Study found no indication that the presence of asbestos at ORR facilities extended beyond various building materials, including thermal insulation of pipes, ducts, and vessels. The demolition of buildings and removal of asbestos containing materials would potentially lead to short-term increases of airborne asbestos in the immediate vicinity of these operations, but they would be expected to have a limited potential to affect asbestos concentrations to off-site receptors. The estimation of potential off-site exposures resulting from such short-term activities is difficult if not impossible without actual off-site monitoring data of asbestos airborne concentrations because of a number of unique characteristics of asbestos.

Asbestos fibers, unlike many substance that are discrete entities definable by a fixed chemical structure, comprise a group of materials that are less easily defined. Asbestos has a broad range of chemical compositions, crystalline structures, sizes, shapes, and properties. The term "asbestos" is not a mineral name, but a commercial-industrial term applied to a group of naturally occurring mineral silicate fibers of the serpentine and amphibole groups. Estimation of environmental concentrations of asbestos fibers in soil, water, and air is difficult because of their varying size, and standardized sampling and analytic techniques. Spatial and temporal asbestos exposure patterns cannot be easily defined in the environment. Settling and deposition of asbestos depends on air movement, particle dimensions, and precipitation events. The appropriateness of current particulate air dispersion models for dust, in describing asbestos fiber entrainment and deposition has not been demonstrated.

3.1.5 Conclusions

In summary, investigations performed to date at ORR have failed to identify any specific asbestos-related exposure events or activities that are believed to have been associated with community exposure. In the absence of such "focal" events, it is unlikely that asbestos-related activities at ORR have resulted in off-site exposures beyond what might be expected from other sources in the community.

3.2 Isotopic Composition of Plutonium Releases from X-10

In the Oak Ridge Dose Reconstruction Feasibility Study (DRFS), airborne plutonium releases were estimated for the following X-10 operations:

- Plutonium separation at the X-10 Chemical Processing Pilot Plant in 1944 and 1945,
- Radioactive barium/lanthanum processing from 1944 through 1956, and
- Ruptures of fuel “slugs” in the Clinton Pile (after 1957 called the Oak Ridge Graphite Reactor).

These operations all involved plutonium formed as a result of neutron capture and subsequent beta decay in the aluminum clad, natural uranium fuel slugs used in the Clinton Pile. For screening in the Dose Reconstruction Feasibility Study, quantities of plutonium present in Clinton Pile fuel slugs were estimated based on documented rates of “product” formation (ChemRisk 1993). The records that documented these rates of formation did not, however, specify the isotopic composition of the plutonium “product” formed.

For purposes of the Dose Reconstruction Feasibility Study screening, the project team assumed the plutonium formed and in part released to the environment was plutonium-239 (^{239}Pu). At the close of the Feasibility Study, the potential ramifications of this assumption were identified by the project team as an area that should be further addressed. Because of this recommendation, plutonium was included on the list of materials to be evaluated further in Task 7. This section describes calculations performed by the project team to address the composition of the plutonium of interest.

3.2.1 Methods

The mass contents and fractions of total plutonium were calculated for eleven isotopes of plutonium present in irradiated Clinton Pile fuel slugs. These calculations were performed using the ORIGEN2.1 computer code for three irradiation periods (50, 100, and 150 days) and 11 decay periods (ranging from time of discharge to 100 days later).

Details of the ORIGEN calculations are presented in Appendix E.

3.2.2 Results

Selected results of the ORIGEN calculations are presented in Tables 3-1 and 3-2. To illustrate the range of fractions of total plutonium that ^{239}Pu comprised, results are presented for four cases: at the time of discharge from the reactor following 50 days of irradiation in the reactor, at 100 days of decay following 50 days of irradiation in the reactor, at the time of discharge from the reactor following 150 days of irradiation in the reactor, and at 100 days of decay following 150 days of irradiation in the reactor.

Table 3-1: Calculated Plutonium Content of 258 X-10 Slugs (grams)

Plutonium Isotope	50 d Irradiation, at Discharge	50 d Irradiation, 100 d Decay	150 d Irradiation, at Discharge	150 d Irradiation, 100 d Decay
Pu-236	3.0×10^{-13}	3.0×10^{-13}	3.5×10^{-12}	3.4×10^{-12}
Pu-237	2.1×10^{-12}	4.6×10^{-13}	1.4×10^{-11}	3.1×10^{-12}
Pu-238	2.8×10^{-7}	3.3×10^{-7}	3.6×10^{-6}	3.8×10^{-6}
Pu-239	1.5	1.6	4.7	4.8
Pu-240	4.4×10^{-4}	4.4×10^{-4}	4.2×10^{-3}	4.2×10^{-3}
Pu-241	1.2×10^{-7}	1.2×10^{-7}	3.5×10^{-6}	3.5×10^{-6}
Pu-242	1.6×10^{-11}	1.6×10^{-11}	1.4×10^{-9}	1.4×10^{-9}
Pu-243	9.9×10^{-18}	0.0	9.2×10^{-16}	0.0
Pu-244	3.5×10^{-22}	3.5×10^{-22}	1.2×10^{-19}	1.2×10^{-19}
Pu-245	5.6×10^{-29}	0.0	2.0×10^{-26}	0.0
Pu-246	3.7×10^{-35}	3.7×10^{-35}	1.0×10^{-31}	1.0×10^{-31}

Table 3-2: Calculated Percent of Total Plutonium Content

Plutonium Isotope	50 d Irradiation, at Discharge	50 d Irradiation, 100 d Decay	150 d Irradiation, at Discharge	150 d Irradiation, 100 d Decay
Pu-239	**99.97**	**99.97**	**99.91**	**99.91**
Pu-240	**0.03**	**0.03**	**0.09**	**0.09**
Pu-236, -237, -238, -242, -243, -244, -245, -246	0.00	0.00	0.00	0.00

3.2.3 Conclusions

The results of the plutonium inventory calculations indicate that ^{239}Pu comprised at least 99.9 percent of the plutonium present in Clinton Pile fuel slugs. These results indicate that assuming the plutonium "product" was ^{239}Pu did not introduce significant inaccuracy into the Feasibility Study screening of past airborne plutonium releases from the X-10 Site.

4.0 MATERIALS SCREENED USING METHODS OTHER THAN THE STANDARD TASK 7 APPROACH

As described briefly in Section 2.0, a number of materials were screened using methods other than the standard, two-level Task 7 screening approach. These materials included those with classified aspects for which limited information was available on uses, release rates, or off-site concentrations, as well as lead and tritium.

4.1 Evaluation of Materials with Classified Aspects

This section identifies the materials evaluated by the Task 7 project team that were formerly classified *per se* or that have classified aspects of use, and describes the methods used to evaluate the likelihood that these materials could have been released off-site in quantities sufficient to cause health risks to the public. This portion of the Oak Ridge Dose Reconstruction is a major cornerstone of the credibility of this independent study. For many years, the facilities on the ORR and at other DOE sites were operated behind cloaks of secrecy. A good deal of public distrust resulted, and some people believe that systems for protection of classified information have been used to conceal information about activities or materials that could pose off-site health hazards. It is important to note that members of the project team with the required security clearances had free access to information about the identified materials with classified names or aspects of use, and they performed a wide-ranging review of past Oak Ridge operations. As described below, a number of different approaches were used to evaluate the potential importance of materials that had classified names or aspects of use. In addressing public concerns, what the investigators *did not find* in this part of the study is almost as important as what they *did* find in the project's other investigations. Following are descriptions of the qualitative and quantitative approaches that were used to evaluate materials with classified aspects, with sample calculations included.

4.1.1 Introduction

A portion of the work in the Dose Reconstruction Feasibility Study was devoted to identification of materials whose mere presence on the ORR remained classified (i.e., they were classified *per se*) and collection and evaluation of available information concerning the uses and potential releases of such materials. During the Feasibility Study, members of the project team had complete access to classified and unclassified information relevant to uses of the materials of concern. Based upon this review, none of the materials that were classified *per se* were identified as warranting high priority for detailed investigation in a dose reconstruction study.

During the Oak Ridge Dose Reconstruction, the DOE declassified the presence of all simple alloys, compounds, substances, or isotopes present at any DOE site. Consequently, any material that was present at the K-25, X-10, and/or Y-12 sites can now be publicly revealed, although in many cases, specific locations, quantities, associations with certain programs, and other aspects of use remain classified. In the Dose Reconstruction Task 7 investigation, a number of materials formerly classified *per se* or having classified aspects of use were evaluated for screening of potential off-site health significance.

The following materials that were formerly classified *per se* and/or have classified aspects of use were evaluated in the Task 7 investigation. Materials that were formerly classified *per se* are marked with an asterisk. Except where noted otherwise, these materials were associated with the Y-12 complex.

beryllium hydride*
beryllium deuteride*
beryllium metal
beryllium oxide
“binary” alloy (depleted uranium and niobium)
boron carbide*
boron nitride*
carbon fibers (K-25 complex)*
copper powder (K-25 complex)
four-ring polyphenyl ether (K-25 complex)*
glass fibers (K-25 complex)*
lead
lithium chloride
lithium deuteride
lithium fluoride*
lithium hydride
lithium tetraborate*
“mulberry” alloy (depleted uranium, niobium, and zirconium)
nickel (K-25 complex)
niobium
niobium beryllide*
rubidium nitrate*
rubidium bromide*
tantalum beryllide*
tellurium
tetramethylammoniumborohydride [TMAB, $(CH_3)_4NBH_4$]*
titanium boride*
Triplex coating (K-25 complex)*
tritium
yttrium boride*
zirconium

These materials were evaluated in Task 7 using a variety of approaches based on manners of usage at the ORR, quantities used, and material toxicity. In general, materials with classified aspects evaluated in Task 7 fall into three categories:

- those that were used in very small quantities;
- those that were used in large quantities for very short periods; and
- those that were used in large quantities for longer periods.

The methods used by the project team to evaluate these materials are described in Sections 4.1.2 through 4.1.5.

4.1.2 Materials Used in Very Small Quantities

Based on evaluation of available information regarding applications of materials, the project team determined that the following materials listed in Section 4.1.1 were used in very small quantities or in forms unlikely to have resulted in off-site releases, and were not likely to have resulted in off-site releases sufficient to have posed health hazards:

boron carbide
boron nitride
yttrium boride
titanium boride
rubidium nitrate
Triplex coating (K-25 complex)
carbon fibers (K-25 complex)
glass fibers (K-25 complex)
four-ring polyphenyl ether (K-25 complex)

Information reviewed by the project team in Y-12 Plant quarterly reports and/or K-25 records is consistent with statements that these materials were used in small quantities. The following are examples of applications of materials in this category (Baylor 1997):

- *Boron carbide* was used in an unclassified application as an abrasive, and was used in research and development as a potential replacement for another material. It was not selected for use in large quantities.

- *Boron nitride* was used in an unclassified application as a crucible material in laboratory quantities.

- *Yttrium boride* and *titanium boride* were used in research and development of a process that was not selected for use in production.

- *Rubidium bromide* and *rubidium nitrate* were used in very small quantities in a research and development program at Y-12.

- *Carbon fibers* and *glass fibers* were received at the K-25 Site as premanufactured filaments, which were likely wound on spools. They were used in construction of rotors used in the centrifuge method of uranium enrichment, in a process by which they were wound around the tall, cylindrical rotors and a plastic binder was used. Because of the forms in which these materials were present, off-site release was highly unlikely.

Materials in this category were not quantitatively screened.

4.1.3 Materials Used in Large Quantities for Short Periods

Some materials with classified aspects were used in large quantities, but over very limited periods of time. The project team determined that tellurium fell into this category**S** the short duration of use and method of use of tellurium make it unlikely that the material was released in quantities sufficient to pose an off-site health hazard. Consequently, this material was identified as low priority for further evaluation.

4.1.4 Materials Used in Large Quantities over Long Periods for Which Monitoring Data were Available

Several other materials with classified aspects were used in relatively large quantities over significant periods of time. For some of these materials, data are available describing concentrations of the key element in the material (e.g., beryllium, copper, lead, lithium, nickel, tritium) in effluents or in the nearby environment. For example, a number of compounds of beryllium and lithium were used at the Y-12 Plant. Compounds or forms of beryllium and lithium identified by the project team as used at the Y-12 Plant are as follows:

Compounds of Beryllium	*Compounds of Lithium*
beryllium deuteride	lithium chloride
beryllium hydride	lithium deuteride
beryllium metal	lithium fluoride
beryllium oxide	lithium hydride
niobium beryllide	lithium tetraborate
tantalum beryllide	

These materials were quantitatively screened based on measured concentrations of beryllium or lithium in the environment and estimates of releases of beryllium and lithium. Similarly, materials containing copper, lead, nickel, and tritium were quantitatively screened based on measured concentrations of the key element. Screening evaluations of off-site releases of beryllium, copper, lead, lithium, and nickel, and tritium are described in Section 5.

4.1.5 Materials Used in Large Quantities over Long Periods Evaluated Using the Threshold Quantity Approach

For materials that were formerly classified *per se* and were used in large quantities over long periods of time, the project team calculated “threshold inventory quantities” based on conservative assumptions about the amount of these materials that may have been released and transported off-site in air or water and conservative assumptions about the amount of these materials in the off-site environment to which individuals may have been exposed. For these materials, the threshold quantity approach was used instead of the Task 7 screening method because source term information and environmental measurements were extremely limited. This information was combined with information on the toxicity of these materials and assumptions about levels of acceptable risk (i.e., a cancer risk of 10^{-6} for carcinogens). This threshold quantity approach assumes that if a material was present on-site in quantities below the threshold, it is highly unlikely that the material posed an off-site health hazard. This method addresses only two environmental exposure pathways (water ingestion and inhalation); however, the exposure parameter assumptions used to calculate threshold quantities are extremely conservative and should provide conservative estimates of exposure despite the exclusion of other pathways. The threshold inventory quantity approach is described in greater detail in Appendix F.

The project team calculated a threshold inventory quantity for tetramethylammoniumborohydride (TMAB), a material that was formerly classified *per se*. Quantities of TMAB used and its specific application(s) remain classified. On-site quantities of this material were below the threshold inventory quantity; therefore, it is highly unlikely that TMAB posed an off-site health hazard.

To illustrate the approach described above, the following two sections present the assumptions used to calculate threshold quantities of niobium and zirconium.

4.1.5.1 Niobium

From 1969 to 1975, niobium was used at Y-12 in an alloy called "mulberry" consisting of 90% depleted uranium, 7.5% niobium, and 2.5% zirconium. In 1975, production of mulberry ceased and was replaced by an alloy known as "binary" consisting of 94% depleted uranium and 6% niobium. Mulberry production records remain classified.

Niobium was purchased from an outside vendor in ¼-inch plates. Castings of mulberry were made in the Y-12 Building-9998 H1 Foundry and transferred to Alpha 5, where they were machined (using a type of sawing), skull cast arc melted, machined again, and then remelted in a vacuum arc furnace. All three of the different types of equipment used to produce mulberry were vented to the same stack in Alpha 5. Per interviews with Y-12 personnel, most airborne releases occurred during the sawing activities.

<u>Niobium Toxicity Data</u>

Toxicity data for niobium are extremely limited. The USEPA has not established regulatory limits or guidance for chronic niobium exposure. In the absence of chronic toxicity information, the project team derived a reference dose (RfD) for chronic exposure by dividing the dose of niobium reported in the literature to be lethal to 50% of a test animal population by an uncertainty factor of 100,000 (Layton 1987). The oral LD_{50} for niobium chloride in mice is 829 mg kg^{-1} (Sax 1989). Thus:

$$RfD_{derived} = \frac{829\ mg\ kg^{-1}}{100{,}000} = 8.3 \times 10^{-3}\ mg\ kg^{-1}\ d^{-1}$$

The derived RfD is assumed to reflect the maximum allowable daily dose for chronic exposure.

Niobium in Air

For exposure to niobium in air, the maximum allowable daily dose is assumed to be equivalent to the maximum allowable air concentration multiplied by an individual's breathing rate divided by the individual's body weight:

$$MADD\ (mg\ kg^{-1}\ d^{-1}) = \frac{MAAC\ (mg\ m^{-3}) \times BR\ (m^{3}\ d^{-1})}{BW\ (kg)}$$

Where:

MADD = Maximum allowable daily dose
MAAC = Maximum allowable air concentration
BR = Breathing rate
BW = Body weight

The maximum allowable air concentration can thus be calculated by rearranging the equation:

$$MAAC\ (mg\ m^{-3}) = \frac{MADD\ (mg\ kg^{-1}\ d^{-1}) \times BW\ (kg)}{BR\ (m^{3}\ d^{-1})}$$

For niobium:

$$MAAC\ (mg\ m^{-3}) = \frac{(8.3{\times}10^{-3}\ mg\ kg^{-1}\ d^{-1}) \times (70\ kg)}{20\ m^{3}\ d^{-1}}$$

$$MAAC = 0.029\ mg\ m^{-3}$$

The maximum allowable release rate (kg y^{-1}) of niobium to air that would produce the maximum allowable air concentration can then be calculated based on the relationship between a unit release rate (1 g s^{-1}) from the site and the reference air concentration calculated for that release rate at the off-site location of interest, as follows:

$$\frac{Max.\ allowable\ RR\ (kg\ y^{-1})}{MAAC\ (mg\ m^{-3})} = \frac{RR\ (kg\ y^{-1})}{RC\ (mg\ m^{-3})}$$

Where:

Max. allowable RR = Maximum allowable release rate to attain the MAAC
MAAC = Maximum allowable air concentration
RR = Unit release rate of 1 g s^{-1} or 31,536 kg y^{-1}
RC = Reference air concentration from release of 1 g s^{-1}

As discussed in Section 2.3.4.1, the reference air concentration at the Scarboro Community modeled based on a unit release of 1 g s^{-1} from the center of the Y-12 site is 3.1×10^{-4} mg m^{-3}.

Thus, for niobium:

$$\frac{Max.\ allowable\ RR\ (kg\ y^{-1})}{0.029\ mg\ m^{-3}} = \frac{31{,}536\ kg\ y^{-1}}{3.1\times10^{-4}\ mg\ m^{-3}}$$

$$Max.\ allowable\ RR = 3.0\times10^{6}\ kg\ y^{-1}$$

Based on interviews with plant personnel (Tindell and Wood 1997) and reviews of plant quarterly reports, the usage rate of niobium never exceeded the calculated maximum allowable release rate.

Niobium in Water

A similar calculation can be made to estimate the maximum allowable release rate of niobium to water.

$$MADD\ (mg\ kg^{-1}\ d^{-1}) = \frac{MAWC\ (mg\ L^{-1}) \times IR\ (L\ d^{-1})}{BW\ (kg)}$$

Where:

MADD = Maximum allowable daily dose
MAWC = Maximum allowable water concentration
IR = Ingestion rate
BW = Body weight

The maximum allowable water concentration can thus be calculated by rearranging the equation:

$$MAWC\ (mg\ L^{-1}) = \frac{MADD\ (mg\ kg^{-1}\ d^{-1}) \times BW\ (kg)}{IR\ (L\ d^{-1})}$$

For niobium:

$$MAWC\ (mg\ L^{-1}) = \frac{(8.3\times10^{-3}\ mg\ kg^{-1}\ d^{-1}) \times (70\ kg)}{2\ (L\ d^{-1})}$$

$$MAWC = 0.29\ mg\ L^{-1}$$

This water concentration can then be related to the maximum allowable release rate to water by the equation:

$$Max.\ Allowable\ RR\ (kg\ d^{-1}) = MAWC\ (mg\ L^{-1}) \times 10^{-6}\ kg\ mg^{-1} \times EFPC\ Flow\ (L\ d^{-1})$$

Where:

Max. allowable RR	=	Maximum release rate to attain the MAWC
MAWC	=	Maximum allowable water concentration
EFPC Flow	=	Daily flow rate of East Fork Poplar Creek

A lower bound estimate of the average EFPC flow rate is 8 million gallons per day, or 3.0×10^{7} L d^{-1}.

Thus, for niobium:

$$Max.\ Allowable\ RR\ (kg\ d^{-1}) = (0.29\ mg\ L^{-1}) \times (10^{-6}\ kg\ mg^{-1}) \times (3\times10^{7}\ L\ d^{-1})$$

$$Max.\ Allowable\ RR\ (kg\ d^{-1}) = 8.7\ kg\ d^{-1}\ \ or\ \ 3{,}200\ kg\ y^{-1}$$

Production and release estimates for niobium remain classified; however, the release estimate falls below the calculated maximum allowable release rate for niobium to EFPC.

4.1.5.2 Zirconium

From 1969 to 1975, mulberry was made up of 90% depleted uranium, 7.5% niobium, and 2.5% zirconium**.** Production of mulberry containing zirconium ceased during the fourth quarter of 1975. Mulberry production records remain classified.

Zirconium Toxicity Data

Although the USEPA does not provide regulatory guidance for zirconium, the ACGIH has established a Threshold Limit Value®, Time Weighted Average (TLV-TWA) of 5 mg m^{-3}. In order to establish a threshold quantity for zirconium, an inhalation RfD was derived from the TLV. The derived RfD was calculated by converting the TLV air concentration, which is based on an 8 hour exposure, to an air concentration for a continuous exposure. This was accomplished using the following equation:

$$CEAC = TLV \times IRCF \times EFCF$$

Where:

CEAC = Continuous exposure air concentration;
TLV = Threshold Limit Value (TLV) (equal to 5 mg m^{-3} for zirconium);
IRCF = Inhalation rate conversion factor for converting an 8-hour occupational exposure to continuous exposure (equal to 0.5; 10 m^3/20 m^3); and,
EFCF = Exposure frequency conversion factor for converting occupational exposure to continuous exposure (equal to 0.54 [(250 d/365 d) × (40 y/70 y)]).

Solving the equation using the zirconium TLV:

$$CEAC = 5\ mg\ m^{-3} \times 0.5 \times 0.57 = 1.4\ mg\ m^{-3}$$

The derived RfD is then calculated using the equation:

$$RfD_{der} = \frac{CAEC \times BR}{BW \times SF}$$

Where:

RfD_{der} = Derived RfD;
BR = Breathing rate, 20 m^3 d^{-1};
BW = Body weight, 70 kg; and
SF = Safety factor for sensitive subpopulations, 10.

Solving for the $\mathrm{RfD_{der}}$:

$$RfD_{der} = \frac{1.4\ mg\ m^{-3} \times\ 20\ m^{3}\ d^{-1}}{70\ kg\ \times\ 10} = 0.040\ mg\ kg^{-1}\ d^{-1}$$

This $\mathrm{RfD_{der}}$ was used in the screening analysis.

Zirconium in Air

Using the same methods described for niobium, the maximum allowable air concentration was calculated to be 0.14 mg m^{-3}. From this value, the calculated maximum allowable release rate in air was 1.4×10^{7} kg y^{-1}. Based on classified production information, it is clear that the maximum allowable release rate is greater than the zirconium quantities at Y-12 in any given year.

Zirconium in Water

The calculated maximum allowable water concentration for zirconium was 1.4 mg L^{-1}. From this value, the calculated maximum allowable release rate for zirconium in water was 1.5×10^{4} kg y^{-1}. This release rate is also greater than the zirconium quantities at Y-12 during any given year.

4.1.6 Conclusions of Analysis of Materials with Classified Aspects

The threshold quantity screening methodology indicates that the quantities of materials evaluated herein that may have been released to the air or water from ORR complexes were not likely great enough to pose hazards to off-site populations. It is important to note that there are several limitations to these analyses, including the lack of chronic toxicity data for the materials of concern and the paucity of effluent and environmental sampling data. Despite these limitations, the method used to calculate acceptable daily intakes and threshold inventory quantities is generally considered extremely conservative. In the future, if chronic toxicity data become available, these assessments could be refined to incorporate those data.

The evaluations that were performed of materials with classified aspects of use did not indicate that any of the materials warrant detailed investigation with regard to potential off-site health hazards. Furthermore, the project team saw no indication that systems for control and protection of classified information have been used for the purpose of concealing information about activities or materials that could pose off-site health hazards.

4.2 Lead Releases from the Oak Ridge Reservation

In the Feasibility Study's screening-level evaluation of materials released from the ORR, lead ranked second after mercury in terms of potential noncarcinogenic health hazards to off-site populations. In order to ensure consistency with other materials screened in the feasibility study, this ranking was established using a provisional USEPA noncarcinogenic reference dose (RfD) for lead. Currently, however, the USEPA recommends evaluating lead exposures using the USEPA Integrated Exposure Uptake Biokinetic (IEUBK) model, which predicts blood lead concentrations in children, rather than comparing predicted exposure rates to the RfD.

Because current USEPA guidelines for evaluating lead exposure advocate use of the IEUBK model, the Oak Ridge Health Agreement Steering Panel recommended a rescreening of lead in Phase II of the Oak Ridge Dose Reconstruction Project, to determine if historical releases of lead warrant more detailed investigation. This section summarizes the screening-level evaluation of potential lead exposures to people living near the ORR using the IEUBK model. Since children are particularly sensitive to neurological effects of lead, this assessment evaluates exposures to children.

4.2.1 Lead Use at Y-12

Lead has been used in several processes at Y-12, including the production of nuclear weapons components from the mid-1940s through 1992. Much information regarding the use of lead at Y-12 is classified. Processes in which lead was used included the following:

- Vacuum casting;
- Arc melting;
- Powder compaction;
- Rolling;
- Forming; and
- Machining.

Other uses of lead at Y-12 included:

- Application of lead-based paints; and
- Use of lead shielding for radiation purposes.

Some of these uses may have resulted in the release of lead to air, surface water, and soil. For example, liquid wastes containing lead and other elements were released from Y-12 to storm sewers. These releases were probably greatest from 1959 through 1970 when production was highest (ORNL 1995). Lead may also have been released to air as particulates from process stacks and plant ventilation systems.

The project team did not locate quantitative information on the amount of lead released from Y-12. This assessment relies primarily on data from recent environmental investigations and information on historical levels of lead in the environment resulting from use of leaded gasoline.

4.2.2 Concentrations of Lead in the Environment near the ORR

Lead may be present in the environment from both natural and human-related sources. The following sections describe levels of lead in the environment that result from both natural and anthropogenic (human-related) sources, and concentrations that have been measured in the environment near the ORR.

4.2.2.1 Background Levels

Natural sources of lead are expected to contribute only a very small fraction to total lead in the biosphere (USEPA 1986a). The single largest source of lead in air is vehicle exhaust (ATSDR 1997c). Other anthropogenic sources of lead include lead-based paints and releases from iron and steel production, smelting operations, municipal waste incinerators, and lead-acid battery manufacturers (ATSDR 1997c).

Adults and children in the United States are exposed to lead in air, food, water, and dust on a daily basis (USEPA 1986a). Typically, the route through which adults and older children receive the largest lead intake is through foods, with reported estimates of dietary lead intake in rural areas in the mid-1980s ranging from 35 to 55 µg d^{-1} (USEPA 1986a). This level of exposure is referred to as the “baseline exposure” for the American population, because it is unavoidable except by drastic changes in lifestyle or by regulation of lead in foods or ambient air.

Young children receive a significant proportion of their daily lead exposure through intake of contaminated dusts during normal hand-to-mouth activity (USEPA 1986a; ATSDR 1997c). As much as 45% of baseline intake of lead by children is estimated to result from consumption of 0.1 g dust per day (ATSDR 1997c). Ingestion of flaking paint or weathered powdered paint may also contribute significantly to lead exposure of children in older houses (USEPA 1986a; ATSDR 1997c).

The project team identified several sources of data on background concentrations of lead, including concentrations measured in areas of Eastern Tennessee assumed to not be impacted by ORR activities. Data on background concentrations of lead are summarized briefly by media below.

- *Air* - Concentrations of lead in air in remote areas are reported to range from 0.00005 to 0.0005 µg m^{-3} (USEPA 1986a). Prior to the mid-1970s, lead concentrations in urban air were significantly higher than at present, due to the combustion of leaded gasoline in vehicles. Air concentrations in urban air in 1975 averaged approximately 1.2 µg m^{-3}. Since then, air concentrations have shown a downward trend, coincident with reductions of lead in gasoline. By 1993, typical annual average air concentrations in urban areas had dropped to approximately 0.1 µg m^{-3} (USEPA 1994).

- *Surface Water* - Present concentrations of lead in natural waters due to human activities range from 0.02 to 1.0 µg L^{-1}, as much as 50-fold higher than true background concentrations. Surface waters receiving urban effluent may have concentrations of 50 µg L^{-1} and higher. The major source of lead contamination of drinking water is the distribution system itself, particularly in older urban areas. Most drinking water contains 0.007 to 0.011 µg g^{-1} lead (USEPA 1986a).

Lead was not detected in any surface water samples collected in Norris Reservoir during 1992 (detection limit 1 µg L^{-1}; Cook et al. 1992). Norris Reservoir is located on the Clinch River above the ORR, and was identified as the "reference location" for the Clinch River Remedial Investigation.

- *Soil/Sediment* - Soil is considered the major sink for lead releases, with soil residency half-times of decades. Soils adjacent to roads that have been traveled since 1930 may be enriched in lead content by to much as 10,000 mg kg^{-1}, and soils adjacent to houses with exterior lead-based paints may have lead concentrations greater than 10,000 mg kg^{-1} (USEPA 1986a). Concentrations of lead in dusts deposited on or near heavily traveled traffic arteries in major US cities have been reported up to 8,000 mg kg^{-1} and higher. In residential areas, exterior dust lead levels in areas contaminated only by atmospheric lead are approximately 1,000 mg kg^{-1} or lower.

  Background concentrations of lead in Tennessee soils in non-urban areas are reported to range from nondetect to 70 mg kg^{-1}, with a mean concentration of 24 mg kg^{-1} (Dragun and Chiasson, 1991). In 1973 and 1982, the Tennessee Valley Authority (TVA) measured concentrations of several metals, including lead, in sediments just upstream of a number of TVA dams. Reservoirs evaluated included Fort Loudoun Reservoir on the Tennessee River upstream of the ORR. The lead concentration in Fort Loudoun Reservoir sediments was 62 mg kg^{-1} (dry weight) in 1973 and 75 mg kg^{-1} in 1982 (TVA 1986). During the same years, lead concentrations in sediments at Watts Bar Dam, downstream from the ORR, were 10 mg kg^{-1} and 53 mg kg^{-1}, respectively.

- *Meat, Fish, Vegetables, and other Food Items* - Lead intake in foods is typically higher in areas with high atmospheric lead concentrations, due to transfer of lead from soil to food crops or direct deposition onto crops. In the mid-1980s, typical levels of lead in leafy vegetables ranged from 0.011 to 0.65 mg kg^{-1} (ATSDR 1997c). Concentrations in meat, fish, and poultry ranged from 0.002 to 0.16 mg kg^{-1}, while concentrations in dairy products ranged from 0.003 to 0.083 mg kg^{-1} (ATSDR 1997c). It is likely that lead concentrations in leafy vegetables were higher in the past due to higher air concentrations, particularly in vegetables grown near roadways. Concentrations in processed, canned foods were also likely higher in the past due to use of lead-soldered cans (USEPA 1986a).

  In the mid-1980s, TVA measured lead concentrations in fish from Tennessee waterways during the Instream Contaminant Study (TVA 1986). The average concentration of lead in Tennessee fish state-wide was 0.70 mg kg^{-1}. The average concentration in spotted bass from Norris Reservoir was 0.22 mg kg^{-1} (range 0.02 to 0.27 mg kg^{-1}).

4.2.2.2 Environmental Concentrations near the ORR

Data describing lead concentrations in air, soil, sediment, water, and other environmental media near Y-12 are discussed in the following sections. Exposure point concentrations used by the project team in the screening level dose assessment are also described.

4.2.2.2.1 Lead Concentrations in Air

The project team did not locate any stack releases data for lead. In the 1970s, concentrations of lead and other elements were measured in rain water near the Y-12 Plant. Although lead concentrations were elevated, the authors attributed this elevation to auto releases tied to use of tetraethyl lead and tetramethyl lead as gasoline additives (Anden et al. 1975).

Two studies of lead concentrations in outside air on or near the K-25 Plant site were undertaken by ORGDP staff beginning in the mid-1970s. From 1973 to 1980, data were collected weekly at four locations on the K-25 site, approximately north, south, east, and west of the center of the plant (Weber and White 1977). Individual sample results were tabulated by ORGDP staff (ORGDP 1981b). These data were obtained and statistically evaluated by the project team. The highest average concentrations were measured at the East sampling location between 1977 and 1979. Airborne lead levels at the East station ranged from <0.0018 to 10 µg m^{-3} (Weber and White 1977).

The other sampling program consisted of collecting and analyzing a limited number of atmospheric samples from five different regions of East Tennessee, in several directions from K-25. Locations of sample collection were Claxton, North Knox, West Knox, Townsend, and Sugar Grove. Lead concentrations at these five sites during a one week period in December 1976 were obtained by the project team. Airborne lead concentrations ranged from nondetect (<0.0001 µg m^{-3}) to 0.817 µg m^{-3}. The highest concentration (0.817 µg m^{-3}) was measured at Townsend on December 9-10. The average concentration during this period was 0.15 (± 0.19) µg m^{-3}.

<u>Off-Site Air Concentration Used in the Screening Assessment</u>

No data describing measured air concentrations of lead at the location of the nearest off-site receptor (the Scarboro community) were identified by the project team. Concentrations at this receptor were estimated based on information on background concentrations of lead in air prior to the mid-1970s, when air concentrations began to decline due to discontinuing of lead use in gasoline. Per USEPA, the average background air concentration of lead in urban areas prior to the mid-1970s was 1.2 µg m^{-3}. This air concentration was used in the screening dose calculations to evaluate exposure via air pathways.

4.2.2.2.2 Lead Concentrations in Surface Water

Y-12 staff routinely measured lead in Y-12 liquid effluent beginning at least as early as the 1960s. Occasional detects of 0.15 to 0.20 mg L^{-1} were reported in the 1960s; concentrations were generally below 0.10 mg L^{-1} (UCC 1964). In the mid-1970s, concentrations were generally below 0.050 mg L^{-1}. These measurements reflect lead concentrations prior to mixing and dilution with surface water.

Results from routine analyses of lead in surface water near Y-12 are reported in the annual environmental monitoring reports beginning in 1971 (UCC 1972). Oak Ridge staff collected surface water grab samples weekly in EFPC near Y-12 and at locations further downstream on the Clinch River (at Melton Hill Dam, at the ORGDP sanitary water intake, at the ORGDP recirculating water intake, and at Center's Ferry near Kingston) and composited them monthly for analysis for a number of materials, including lead. Beginning in 1973, data are also reported for samples collected in two locations in Poplar Creek (upstream of K-25 and near the confluence of Poplar Creek with the Clinch River). Statistics presented in the annual reports include the minimum, maximum, mean, standard deviation of sample concentrations, and the number of samples collected. Maximum concentrations in EFPC between 1971 and 1982 ranged from <0.002 to 0.4 mg L^{-1}. The highest concentration (0.4 mg L^{-1}) was measured in 1974 (UCC 1975). Average concentrations during that year, and all other years for which data are reported, were <0.02 mg L^{-1}.

In addition to this routine monitoring program, lead concentrations were also measured in surface water during several special monitoring programs. These programs include:

- *The Tennessee Valley Authority (TVA) Instream Contaminant Study (1984)*– Although the focus of this program was evaluating the transport of mercury in surface water and sediment downstream of Y-12, samples collected in May 1984 were also analyzed for other trace metals, including lead. Surface water sample locations included one in EFPC at the outfall of New Hope Pond, one in Poplar Creek upstream of K-25, and two in the Clinch River above and below the Poplar Creek/Clinch River confluence (TVA 1985a). The reported lead concentration at the outfall of New Hope Pond and at the Poplar Creek location was 0.002 mg L^{-1}. Concentrations in the Clinch River were below the detection limit (<0.001 mg L^{-1}). The TVA report presents individual sample results.

- *The Clinch River Remedial Investigation (1989 and 1990)*– DOE initiated this program in 1989 to investigate the transport, fate, and distribution of waterborne contaminants released from the ORR to the Clinch and Tennessee Rivers (Cook et al. 1992). They collected surface water, sediment, and fish samples from 10 reaches, six of which are potentially affected by releases from the ORR, and four which serve as reference or background areas. Sample locations included Poplar Creek and the Clinch River. Lead was detected in surface water samples from only one location, the Kingston city park at Clinch River Mile (CRM) 0.5, at 0.006 mg L^{-1}. The LMES OREIS database presents individual sample results (LMES OREIS 1997).

- *The East Fork Poplar Creek Floodplain Remedial Investigation (EFPC RI) (1990 and 1991)*– During the EFPC RI, investigators collected surface water, sediment, and floodplain soil samples at several locations in and along EFPC. Lead was not detected in any surface water samples. The RI/FS report describes these data (SAIC 1994).

Off-Site Surface Water Concentration Used in the Screening Assessment

The project team used the highest measured surface water concentration reported for EFPC during the routine monitoring program of Y-12 effluent (0.2 mg L^{-1}) in the screening assessment for the surface water pathways. A higher concentration (0.4 mg L^{-1}) was measured near Y-12 in 1974; however, this concentration exceeded concentrations in effluent, and average concentrations during that year were <0.02 mg L^{-1}). Therefore, that single high value was considered to be anomalous. Although EFPC was not used as a drinking water source, the project team assumed for purposes of the screening level assessment that consumption of EFPC water occurred.

4.2.2.2.3 Lead Concentrations in Soil/Sediment

The project team identified limited data on lead concentrations in surface soil near the Y-12 site. Recently, investigators conducted several studies to determine concentrations of contaminants in soil and sediment near the ORR that included lead in the suite of metals evaluated. These included the 1984 TVA Instream Contaminant Study and the 1990-1991 SAIC EFPC-Sewer Line Beltway Remedial Investigation. These sampling programs and others conducted near Y-12 are described below.

- *A survey of sediments in streams surrounding the K-25 Plant by ORGDP staff (1985)*– ORGDP staff collected surface sediment samples at 180 locations in the Clinch River, Poplar Creek, EFPC, and tributaries to Poplar Creek, to identify locations where contaminants were entering surface water systems (Ashwood et al. 1986). Lead concentrations were measured in some of these samples. Lead concentrations ranged from 5 to 140 mg kg^{-1}. Ashwood et al. present individual sample results.

- *The Tennessee Valley Authority (TVA) Instream Contaminant Study (1984)*– In addition to surface water samples, TVA collected surface sediment samples and analyzed them for a variety of contaminants including lead. They collected samples from the length of EFPC, three locations in Poplar Creek, and five locations in the Clinch River (TVA 1985b,c). Concentrations in EFPC ranged from 36 to 130 mg kg^{-1} with an average concentration of 73 mg kg $^{-1}$(standard deviation = 28 mg kg $^{-1}$, n = 18). Concentrations in Norris Reservoir ranged from 58 to 67 mg kg^{-1}. The TVA report presents individual sample results.

- *The Clinch River Remedial Investigation (1989 and 1990)*– In addition to surface water and fish samples, investigators collected sediment samples from 10 reaches including Poplar Creek, the Clinch River, and Watts Bar Reservoir. The maximum lead concentration in Poplar Creek sediment was 44.3 mg kg^{-1}. The maximum concentration in the Clinch River was 69.9 mg kg^{-1}. The LMES Oak Ridge Environmental Information System (OREIS) database includes individual sample results (LMES OREIS 1997).

- *The East Fork Poplar Creek Floodplain Remedial Investigation (EFPC RI) (1990 and 1991)*– Investigators collected sediment and floodplain soil samples at several locations in and along EFPC, including a total of 184 16-inch soil/sediment cores from the EFPC floodplain. Individual soil/sediment cores were composited (blended) prior to analysis. The maximum soil/sediment core lead concentration was 984 mg kg^{-1}; this sample was collected near a shooting range and contained visible lead shot. The highest concentration exclusive of this result was 625 mg kg^{-1}. The OREIS database includes individual sample results (LMES OREIS 1997).

- *The Oak Ridge Associated Universities (ORAU) assessment of City of Oak Ridge soil (1984-1985)*– Although the focus of the ORAU investigation was measurement of mercury in surface soils at locations throughout the City of Oak Ridge, ORAU analyzed a number of samples for additional metals, including lead. Lead concentrations measured in Oak Ridge Civic Center soils in March 1984 ranged from 40 to 110 mg kg^{-1} (ORAU 1984). Concentrations near the Southfield Apartments ranged from 84 to 100 mg kg^{-1}.

Off-Site Soil/Sediment Concentration Used in the Screening Assessment

The project team used the highest soil/sediment concentration reported in the above investigations in the screening analyses for the soil/sediment pathways. The highest reported concentrations were measured during the EFPC RI. Lead concentrations measured in composited samples from the top depth interval (0 - 16 inches bgs) ranged from 5.2 to 625 mg kg^{-1}, exclusive of the sample collected near the shooting range assumed to have been impacted by lead shot. The mean concentration was 50 mg kg^{-1} (standard deviation 54 mg kg^{-1}). The 95% upper confidence limit (UCL) about the mean was 57 mg kg^{-1}. For purposes of the screening assessment, the 95% UCL was used to evaluate exposures to lead in soil.

As indicated above, sample cores collected in this program were composited prior to analysis. Because the history of contaminant release from Y-12 varied over the years of operation, it is likely that lead concentrations in surface soils at some time in the past were higher than they are at present. Surface soils with higher concentrations may subsequently have been covered by less contaminated soils deposited during flooding events. However, results from the analysis of the composited soil/sediment cores essentially reflect the concentration of lead averaged over the 0-16-inch depth interval. Therefore, the 95% UCL based on the composited samples was adjusted to account for the possibility of historically higher surface soil concentrations.

In 1992, SAIC conducted a "Vertical Integration Study" in which they collected five 16-inch long soil cores from four floodplain locations. SAIC divided the soil cores into 1-inch intervals, and analyzed each interval for a number a metals. Although lead was not included in the analyses, uranium concentrations were measured. Because uranium does not degrade significantly in the environment, it is likely that concentrations measured at the different depth intervals reflect the historical record of releases from Y-12 and subsequent deposition in the floodplain. For uranium, the highest 1-inch interval concentration in each core was 2 to 5 times higher (mean = 3.5 times higher) than the concentration in the corresponding composited core sample. Therefore it is assumed that at some point in the past, uranium concentrations in the surface soil were about 3.5 times higher than concentrations in the composited cores. To adjust for the possibility of historically higher surface soil concentrations of lead, the 95% UCL concentration was multiplied by a factor of 3.5, yielding an adjusted surface soil concentration for use in the screening assessment of 200 mg kg^{-1}. This calculation assumes that peak lead releases correlated with peak uranium releases.

4.2.2.2.4 Lead Concentrations in Fish

Investigators measured lead concentrations in fish from waterways near the ORR during several of the programs in which surface water and sediment/soil samples were collected (e.g., the TVA Instream Contaminant Study and the Clinch River Remedial Investigation). The earliest data on lead concentrations in fish identified by the project team were collected by Loar et al. (1981a) in 1979. Data describing lead concentrations in fish include the following:

- *A biological sampling program to evaluate the effects of ORNL operations on aquatic biota in the White Oak Creek watershed (1979)*– ORNL staff measured concentrations of lead and other metals in fish from a number of locations including CRM 19 and 22 (the confluence of the Clinch River and Poplar Creek is at CRM 12) (Loar et al., 1981a). The Loar et al. report presents mean concentrations. Mean concentrations in Clinch River fish ranged from 0.009 to 0.061 mg kg^{-1}.

- *The Tennessee Valley Authority (TVA) Instream Contaminant Study (1984)*– TVA collected fish from several locations including EFPC at EFPCM 13.8, the Clinch River at CRM 2, 6, and 11, and Poplar Creek at PCM 0.2 (TVA 1985d). Lead concentrations in EFPC fish ranged from nondetect (<0.02 mg kg^{-1}) to 0.29 mg kg^{-1}. Slightly higher concentrations were measured at Melton Hill Dam, upstream of the junction of Poplar Creek with the Clinch River. The maximum concentration measured at Melton Hill Dam was 0.71 mg kg^{-1}. The TVA reports present individual sample results.

- *A TVA fish tissue screening study of Eastern Tennessee reservoirs (1987)*– TVA collected fish from several Tennessee reservoirs and analyzed them for numerous contaminants, including lead, to assess the general level of contamination in Tennessee reservoirs (TVA 1989). Sample locations included CRM 20 in Watts Bar Reservoir and CRM 24 in Melton Hill Reservoir. Lead concentrations in fish from these locations ranged from 0.03 to 0.04 mg kg^{-1}. The TVA report presents individual sample results.

- *The Clinch River Remedial Investigation (1989 and 1990)*– Investigators collected fish from a number of locations including CRM 0.5 and CRM 9.5 (Cook et al. 1992). Lead was not detected in any of the samples (detection limit 0.28 to 0.61 mg kg^{-1}). The LMES OREIS database presents individual sample results (LMES OREIS 1997).

Off-Site Fish Concentration Used in the Screening Assessment

As indicated, concentrations of lead in fish downstream from Y-12 were not measured prior to the late 1970s. The maximum measured concentration in EFPC fish (measured in 1984) was 0.29 mg kg^{-1} (TVA 1985d). This concentration is consistent with background concentrations measured in areas not influenced by releases from the ORR (e.g., the maximum concentration measured in fish from Norris Reservoir was 0.27 mg kg^{-1}; TVA 1986). Concentrations of lead in EFPC fish during earlier years were likely higher if water concentrations were higher.

For purposes of the screening analyses, concentrations of lead in fish were calculated using a bioconcentration factor (BCF) that describes the transfer of lead from water to fish. The BCF used in this assessment is 49 (mg kg^{-1})/(mg L^{-1}) (USEPA 1986b). Use of this BCF and the water concentration described in Section 4.2.2.2.2 (0.20 mg L^{-1}) predicts a fish concentration of 9.8 mg kg^{-1}. This concentration likely significantly overestimates the maximum lead concentration in fish.

4.2.2.2.5 Lead Concentrations in other Food Items

The project team did not identify any data describing concentrations of lead in vegetation, meat, or milk near the ORR.

Off-Site Vegetation, Meat, and Milk Concentrations Used in the Screening Assessment

For purposes of the screening analyses, the project team calculated concentrations of lead in vegetation, meat, and milk using biotransfer factors that characterize the transfer of lead from other media, including air, water, and soil. The biotransfer factors used in this assessment are listed in Table 4-1.

Table 4-1: Biotransfer and Bioconcentration Factors for Lead

Parameter	Value	Source
Soil to Vegetables (B_{veg})	0.009 (mg kg^{-1} wet)/(mg kg^{-1} dry)	Baes et al. 1984
Soil to Pasture ($B_{pasture}$)	0.045 (mg kg^{-1} dry)/(mg kg^{-1} dry)	Baes et al. 1984
Biotransfer to Milk (F_m)	0.00025 d L^{-1}	Ng et al. 1977
Biotransfer to Meat (F_f)	0.0003 d kg^{-1}	Baes et al. 1984
Bioconcentration in Fish (*BCF*)	49 (mg kg^{-1})/(mg L^{-1})	USEPA 1986b

4.2.3 Toxicity Assessment

The following sections discuss health effects that have been associated with lead exposure through different exposure pathways, and summarize the toxicity criteria used in the screening analysis to assess the possible significance of lead exposures near the ORR.

4.2.3.1 Health Effects from Lead Exposure

Investigators have associated lead exposure with a variety of adverse effects in humans and animals, including neurological, cardiovascular, and carcinogenic effects and effects on blood cell development. The most sensitive endpoints to low-level lead exposure are neurobehavioral deficits and growth retardation in young children and hypertension in middle-aged men (ATSDR 1997c). Symptoms of nervous system damage range from subtle decreases in intelligence, demonstrated by neurological test scores, to clinically evident alterations of brain structure. Effects on the nervous system are generally considered irreversible (ATSDR 1997c). Effects on heme synthesis also occur at very low levels, leading to a decrease in the number and lifespan of red blood cells (ATSDR 1997c). In pregnant women, lead may also be passed to the fetus. Effects on the fetus may include premature birth, low birth weight, and effects on mental development (ATSDR 1997c).

Studies in animals indicate that chronic oral exposure to very high doses of lead salts may cause an increased frequency of tumors of the kidney (IRIS 1997). For this reason, USEPA has classified lead as a probable human carcinogen (Group B2), based on sufficient information from animal studies but inadequate information in humans (IRIS 1997). However, the USEPA has not established a cancer slope factor for lead.

A great deal of data exists on lead dose-response relationships in humans. Investigators generally describe lead intake in terms of internal exposure (i.e., measured or predicted blood lead concentrations) rather than external exposure. In part, this is because lead may be taken in from multiple sources through both inhalation and ingestion routes, such that measurement of lead intake through a single route does not accurately reflect total exposure. Effects of lead exposure are the same regardless of the route of entry into the body (e.g., inhalation or ingestion) and have been correlated with blood lead concentrations (ATSDR 1997c).

4.2.3.2 The USEPA Integrated Exposure Uptake Biokinetic (IEUBK) Model

The USEPA does not currently support using a reference dose (RfD, presented as milligrams of intake per kilogram of body weight per day) to evaluate exposure to lead because (1) there is a lack of data on dose-response relationships for lead using external exposure measurements, (2) the toxicokinetics (i.e., absorption, distribution, and excretion) of lead vary significantly based on individual-specific factors, including age, health, nutritional state, body burden, and exposure duration, and (3) no thresholds have been demonstrated for many of the non-cancer effects of lead that have been observed in infants and young children. Alternatively, the USEPA recommends comparing measured or estimated blood lead concentrations to dose-response relationships established from blood lead concentrations. The USEPA has developed a pharmacokinetic model, the Integrated Exposure Uptake Biokinetic (IEUBK) model, that estimates blood lead concentrations in children (ages 6 months to 7 years) from exposures to lead in air, soil, dust, diet, drinking water, and paint (USEPA 1994a). These estimated blood lead concentrations are then compared to an acceptable blood lead guidance concentration developed by the Centers for Disease Control (CDC).

In 1991, the CDC reduced the action level (the regulatory concentration of concern) for lead in blood from 25 µg of lead per deciliter of whole blood (µg dL^{-1}) to 10 µg dL^{-1}. The action level was developed from analysis of correlations between blood lead concentrations and several blood chemistry and behavioral indices, including aminolevulinic acid dehydratase (ALA-D) activity, vitamin D, and pyrimidine metabolism, neuro-behavioral indices, growth, and blood pressure. Since there is no widely accepted theoretical basis for an absence of a threshold for many of the health effects associated with lead exposure, CDC recommends that children with blood lead concentrations above the action level be medically evaluated and their environment scrutinized. Historically, higher levels of blood lead have been considered acceptable (25 µg dL^{-1} in 1985; 30 µg dL^{-1} in 1978; 40 µg dL^{-1} in 1970; and 60 µg dL^{-1} in 1965). Reduction in the action level to 10 µg dL^{-1} was based on new data indicating adverse health effects at blood lead concentrations in children "...at least as low as 10 micrograms per deciliter..." (CDC 1991). The CDC (1991) further noted that establishing an action level below 10 µg dL^{-1} has practical limitations, including that quantification of blood lead concentrations below 10 µg dL^{-1} may be inaccurate and imprecise. For adults, the primary blood lead criterion is the OSHA standard of 40 µg dL^{-1} (OSHA 1978). Exceedence of this blood lead concentration requires immediate removal from exposure.

Typically, per USEPA policy, lead exposure at a site is not considered significant if the 95th percentile of the population blood lead concentration distribution determined by the IEUBK model is at or below 10 µg dL^{-1}. The IEUBK model calculates the 95th percentile blood lead concentration assuming that blood lead concentrations in the exposed population are lognormally distributed with a geometric standard deviation (GSD) of 1.60. This GSD is based on several studies, including the Urban Soil Lead Abatement Demonstration Project and a study in Butte, Montana (USEPA 1994a).

4.2.4 Exposure Assessment

In the current assessment, the project team calculated concentrations of lead in the blood following, to the extent possible, the guidelines developed in the Task 7 Screening methodology for a Level I and II screening (ChemRisk 1996). Rather than calculating average daily doses in mg kg^{-1} d^{-1}, however, the project team used the IEUBK model to predict blood lead concentrations in µg lead per deciliter (dL) of whole blood. As described above, the IEUBK model predicts blood lead concentrations in children ages 6 months to 7 years from intake of lead through four media: air (inhalation), soil and dust (ingestion), water or fluids (ingestion), and diet (ingestion). The model allows the user to input concentrations of lead in each of these media, as well as some receptor-specific parameters such as inhalation rate. The user cannot adjust other factors such as age, body weight, and total food consumption. Further, the model does not calculate concentrations of lead in vegetables, meat, milk, or fish due to uptake from air, soil, or water– the user must calculate concentrations in these dietary media separately and input the media concentrations directly into the model. If site- or population-specific information is not available, the model provides default assumptions.

Consistent with the Task 7 Screening methodology for a Level I and Level II screening (ChemRisk 1996), the project team calculated blood lead concentrations assuming contributions to lead uptake from the exposure pathways listed in Table 4-2.

Appendix G describes in detail the exposure assumptions input into the IEUBK model to predict blood lead concentrations. To the extent possible, these assumptions are consistent with the Level I and Level II screening assumptions described in Task 7 Screening Method report (ChemRisk 1996). However, because the IEUBK model predicts annual average blood lead concentrations in specific age groups of children (ages 6 months to 7 years), the project team used IEUBK age-specific default parameters for some of the exposure assumptions rather than Task 7 Level I and II screening assumptions for children, which are generally based on a child approximately 6 years of age. Briefly, the exposure assumptions input into the IEUBK model are as follows:

C Air– The project team used default IEUBK model parameter values for inhalation of lead in air. These included age-specific inhalation rates ranging from 2 to 7 m^3 d^{-1} and a lung absorption fraction of 32% (USEPA 1994a). As described in Section 4.2.2.2.1, an ambient air concentration of 1.2 µg m^{-3} was assumed.

- *Drinking Water/ Milk*– Set to default, the IEUBK model calculates lead intake from ingestion of tap water but does not consider intake from contaminated milk. For purposes of this evaluation, the project team assumed that the total fluid intake was equal to the IEUBK age-specific default parameter values for ingestion of tap water (approximately 0.6 L d^{-1}) *plus* the Task 7 screening model value for ingestion of milk by children (1.0 L d^{-1} for Level I and 0.5 L d^{-1} for Refined Level I). For Level I, the concentration of lead in fluids was assumed to be 0.16 mg L^{-1}, calculated assuming that approximately 50% of total fluid consumption is milk with a lead concentration of 0.22 mg L^{-1}, 25% is contaminated water with a lead concentration of 0.2 mg L^{-1}, and 25% is "uncontaminated" water with a background lead concentration of 0.004 mg L^{-1}. For Refined Level I, the concentration of lead in fluids was assumed to be 0.071 mg L^{-1}, calculated assuming that

Table 4-2: Exposure Pathways Evaluated in the Screening Analysis for Lead

Pathway	Exposure Component Evaluated in the IEUBK Model
Air Pathways	
Air to Humans– Direct Inhalation	Air
Air to Livestock/Game to Humans– Beef Consumption	Diet
Air to Dairy Cattle to Humans– Milk Consumption	Water (Fluids)
Air to Vegetables to Humans– Vegetable Consumption	Diet
Air to Pasture to Livestock/Game to Humans– Beef Consumption	Diet
Air to Pasture to Dairy Cattle to Humans– Milk Consumption	Water (Fluids)
Water Pathways	
Water to Humans– Water Ingestion	Water (Fluids)
Water to Livestock/Game to Humans– Beef Consumption	Diet
Water to Dairy Cattle to Humans– Milk Consumption	Water (Fluids)
Water to Fish to Humans– Fish Consumption	Diet
Soil Pathways	
Soil to Air to Humans– Inhalation of Resuspended Dust	Air
Soil to Humans– Soil Ingestion	Soil
Soil to Livestock/Game to Humans– Beef Consumption	Diet
Soil to Dairy Cattle to Humans– Milk Consumption	Water (Fluids)
Soil to Vegetables to Humans– Vegetable Consumption	Diet
Soil to Pasture to Livestock/Game to Humans– Beef Consumption	Diet
Soil to Pasture to Dairy Cattle to Milk to Humans– Milk Consumption	Diet

approximately 50% of total fluid consumption is milk with a lead concentration of 0.040 mg L^{-1}, 25% is contaminated water with a lead concentration of 0.2 mg L^{-1}, and 25% is "uncontaminated" water with a background lead concentration of 0.004 mg L^{-1}. Assumptions used to calculate the lead concentration in milk are described in Appendix G. The IEUBK model default value for the bioavailability of lead in water in the gastrointestinal tract (50%) was used.

- *Soil/Dust*– The project team used default IEUBK model parameter values for ingestion of lead in soil and/or dust. These included age-specific ingestion rates ranging from 0.085 to 0.135 g d^{-1}, along with the assumption that soil ingestion comprises 45% of total soil and dust ingestion. The ratio of dust lead concentration to soil lead concentration was set to 0.70 (default), and the ratio of dust lead concentration to outdoor air concentration was set to 100 µg g^{-1} dust per µg m^{-3} air (default, USEPA 1994a). The IEUBK default value for the bioavailability of lead in soil/dust in the gastrointestinal tract (30%) was used. As described in Section 4.2.2.2.3, a soil concentration of 200 mg kg^{-1} was assumed.

- *Diet*– Reasonable upper bound exposures to lead in vegetables, beef, and fish were evaluated based on a consideration of the fraction of home-grown/home-caught foods to total intake, and calculated concentrations of lead in home-grown/home-caught foods assuming transfer of lead from air, soil, and water. The concentration of lead in fish was assumed to be 9.8 mg kg^{-1}, calculated by multiplying the BCF for fish by the lead water concentration (Section 4.2.2.2.4). For Levels I and II, the concentrations of lead in homegrown vegetables were assumed to be 6.6 mg kg^{-1} and 2.5 mg kg^{-1}, respectively, calculated assuming root uptake of lead from soil and uptake of lead from air. For Levels I and II, the concentrations of lead in beef and game animals was assumed to be 0.23 mg kg^{-1} and 0.050 mg kg^{-1}, respectively, calculated based on uptake of lead into animals by ingestion of soil, contaminated pasture grass, and contaminated water and inhalation of lead in air. Assumptions used to calculate the lead concentration in meat are described in Appendix G.

  For the Level I and II screens, 60% and 23%, respectively, of total vegetable consumption was assumed to be associated with lead-contaminated vegetables from the site. For the meat ingestion pathway, 56% and 23% of total meat consumption was assumed to be associated with lead-contaminated beef/game from the site (it was assumed that 75% of all meat consumed is beef/game) and 20% and 5% of total meat consumption was assumed to be associated with lead-contaminated fish from the site (it was assumed that 25% of all meat consumed is fish). The IEUBK default value for the bioavailability of lead in food in the gastrointestinal tract (50%) was used.

Exposure point concentrations used in the Level I and II screen are summarized in Table 4-3.

Table 4-3: Summary of Exposure Point Concentrations for Lead

Environmental Medium	Concentration
Air	0.0012 mg m^{-3}
Soil/ Sediment/ Dust	200 mg kg^{-1}
Water	0.2 mg L^{-1}
Total Fluids (milk and water)	0.16 mg L^{-1} (a)/ 0.071 mg L^{-1} (b)
Vegetables	6.6 mg kg^{-1} (a)/ 2.5 mg kg^{-1} (b)
Meat (beef and game)	0.23 mg kg^{-1} (a)/ 0.050 mg kg^{-1} (b)
Fish	9.8 mg kg^{-1}

a Level I value
b Refined Level I value

4.2.5 Risk Characterization

Using the IEUBK model, the project team calculated blood lead concentrations potentially associated with the screening level estimates of lead uptake, and compared 95th percentiles of the population blood lead concentration distributions to the CDC/USEPA risk-based decision criterion for lead in blood of 10 µg/dL. Per USEPA policy, lead exposure at a site is not considered significant if the 95th percentile of population blood lead distribution determined by the model is at or below 10 µg dL^{-1}.

For the Level I screening, the predicted 95th percentile blood lead concentration for children aged 6 months to 7 years associated with exposures to lead near the ORR ranges from 52.4 to 66.8 µg dL^{-1}. For the Refined Level I screening, the predicted 95th percentile blood lead concentration for children aged 6 months to 7 years from exposures to lead near the ORR ranges from 18.1 to 23.3 µg dL^{-1}. For the Level I screening, the dominant contributors are diet (contributing approximately 66% of the total dose) and milk/water (contributing approximately 20% of the total dose). For the Refined Level I screening, the dominant contributors are also diet and milk/water.

4.6.6 Conclusions

Blood lead concentrations estimated using the IEUBK model combined with the Task 7 screening methodology exceed the CDC/USEPA risk-based decision criterion for lead in blood of 10 µg dL^{-1}. However, it is likely that use of default parameters to predict lead concentrations in vegetables and meat/milk (e.g., deposition rates, biotransfer factors) significantly overestimate concentrations in these media, particularly since available data on lead concentrations in the environment near the ORR suggest that lead concentrations were consistent with historical background concentrations resulting from use of lead in gasoline. Therefore, further evaluation of blood lead concentrations that may have resulted from exposure to lead from the ORR may not be warranted.

4.3 Y-12 Tritium Releases from Heavy Water Received from Savannah River

A screening analysis was performed during the Oak Ridge Dose Reconstruction Feasibility Study for tritium releases from radioisotope production at the X-10 Site. Other possible sources of tritium releases not reviewed during the Feasibility Study were deuterium gas production and lithium deuteride recovery operations at the Y-12 Plant.

Y-12 manufactured deuterium gas from heavy water (D_2O) supplied by the Savannah River Plant. Heavy water is water enriched in the deuterium isotope of hydrogen (^2H). In the heavy water production process, the radioactive tritium isotope of hydrogen (^3H) is also produced, resulting in some contamination of the heavy water by tritiated water. In the deuterium gas production performed at Y-12, the tritium was carried along with the deuterium resulting in deuterium gas that was also contaminated with tritium.

Deuterium gas produced from the heavy water was used at Y-12 in lithium deuteride production, placed into weapon components, and shipped off site. When the weapons were retired, the weapons components were returned to Y-12 and the lithium and deuterium were recovered. The recovered deuterium was converted to “half-heavy” water (HDO) and subsequently returned to the Savannah River Plant.

4.3.1 Deuterium Processing at Y-12

Savannah River supplied Y-12 with heavy water for the production of deuterium gas beginning in February 1956 (Union Carbide Corp. 1956 [Y-1013, Quarterly Report for Jan.-Mar. 1956]) and continuing until the program was shut down in 1995. The Y-12 deuterium processing plant operated on an intermittent basis, often gearing up for a few days or weeks of production to replenish deuterium supplies (Union Carbide Corporation 1956-1975). Generally, the amount of deuterium produced during a production campaign would be enough to last several months or years, depending on production schedules.

At Y-12, the heavy water was electrolyzed to obtain deuterium, which in turn was used in the production of lithium deuteride for thermonuclear weapons. Tritium was reportedly carried along with the deuterium in each phase of the production (Bogard 1983). The electrolysis process took place in a sealed system, with oxygen venting being the only expected release to the atmosphere (Richesin 1992). Because deuterium was considered a special nuclear material (SNM), conservation and accountability of the material was extremely important. Two types of controls to capture deuterium/tritium were part of the deuterium gas facility— a mist separator and a catalytic recombiner. Figure 4-1 is a flow chart of the Y-12 deuterium gas facility.

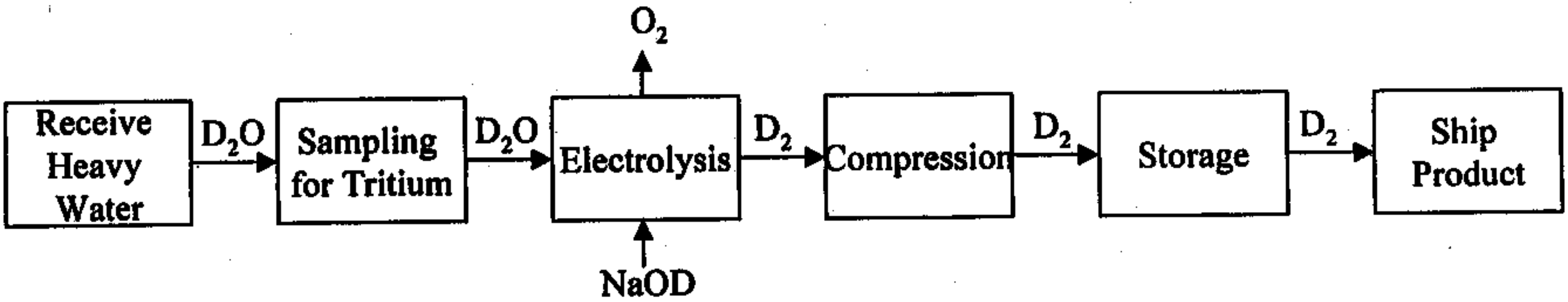


Figure 4-1 Flow Chart of Y-12 Deuterium Gas Processing

When retired weapon parts were returned to Y-12, lithium and deuterium were recovered from the lithium deuteride (Headrick 1997). As in deuterium production, tritium was reportedly carried along with deuterium in this recovery process. The lithium and deuterium were recovered through hydrogen generation and gas burning. The gas burning, coupled with an ion exchange process, converted the deuterium to half-heavy water that was returned to Savannah River. When the weapon parts were returned to Y-12, the quantities of tritium remaining were less than when the parts were produced, due to the radioactive decay of tritium with a half-life of 12.6 years.

4.3.2 Data Availability

Limited information is available regarding quantities of tritium at the Y-12 site. The project team located the following types of data that were relevant to this screening-level assessment:

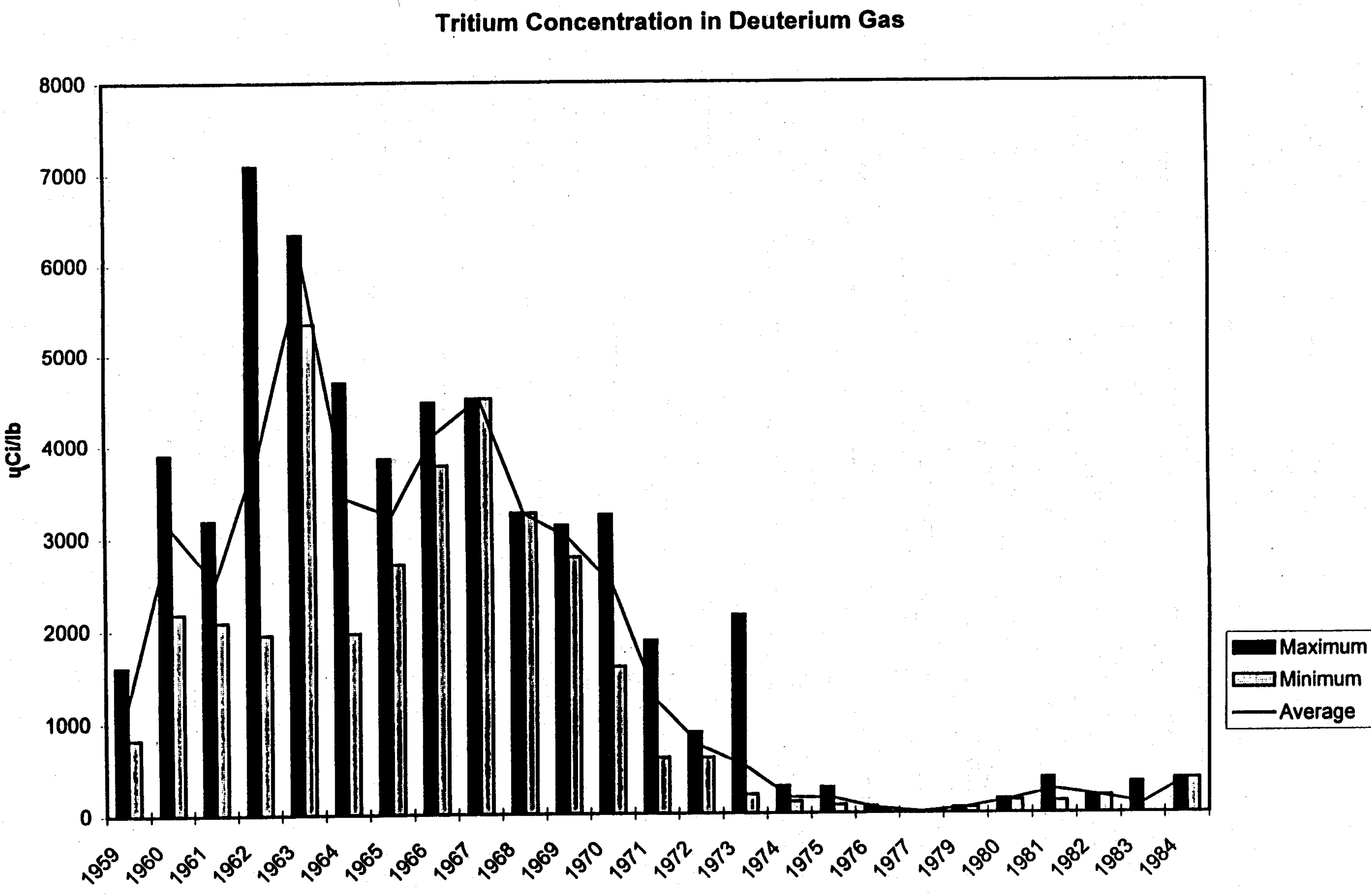
- The quantity of deuterium handled at Y-12 from 1956 to 1995 are available in a classified version of the report by Owings (1995).

- Although the total quantity of deuterium received at Y-12 is classified, deuterium “inventory difference” values are unclassified. The deuterium “inventory difference” (formerly called Material Unaccounted For, or MUF) represents the amount of deuterium “lost” or unaccounted for in the deuterium processing. This inventory difference information will be discussed in Section 4.3.3 (Owings 1995).

- Tritium concentrations in the heavy water shipped to Y-12 from Savannah River were provided to the project team for 1983 through 1994 (Martin Marietta 1995). No data of this type are available prior to 1983. The tritium concentration in the heavy water over 1983-1994 ranged from 0.01 to 749 FCi L^{-1}, with a mean of 120 FCi L^{-1}. This information could be used in a screening analysis for tritium if heavy water inventory information were available. Unfortunately, the project team did not locate heavy water inventory information.

- Tritium concentrations in the deuterium gas produced at Y-12 are available for 1959 through 1984. Concentrations in the gas ranged from 4 FCi lb^{-1} to 7,092 FCi lb^{-1}, with an average of 1500 FCi lb^{-1}. Tritium concentrations in the deuterium gas from 1959 through 1984 are graphed in Figure 4-2. These data indicate that tritium levels peaked in 1963 and again in 1967, then decreased through the 1960s and early 1970s, when they leveled off and averaged less than 500 FCi L^{-1}. This information, along with the deuterium “inventory difference”, was used in this screening analysis (Section 4.3.3) (Lockheed-Martin 1996).

Figure 4-2


Tritium Concentration in Deuterium Gas
μCi/lb
8000
7000
6000
5000
4000
3000
2000
1000
0
1959
1960
1961
1962
1963
1964
1965
1966
1967
1968
1969
1970
1971
1972
1973
1974
1975
1976
1977
1979
1980
1981
1982
1983
1984
Maximum
Minimum
Average

Reports of Accidental Releases

The project team also reviewed the Y-12 Plant quarterly reports for information about any accidents that might have involved the release of heavy water or deuterium gas. Two accidents were identified in the Quarterly Reports (Union Carbide Corporation 1956-1975). The first accident occurred on March 27, 1960, and was the result of a fault in the rupture plugs of a vent line. The accident reportedly involved the loss of 15 kg of deuterium gas. The second accident occurred on December 7, 1960, and was caused by the overturn of a trailer in the deuterium storage area. The overturn damaged the top seven cylinders containing deuterium, and resulted in a reported release of 31.8 kg of deuterium gas. These two releases appear to have been included in the deuterium "inventory difference" record.

In the early 1990s, Y-12 plant personnel compiled information relating to tritium contamination of the heavy water supplied by Savannah River. One section of this compilation notes the search for any accidental releases or spills of the heavy water. The author notes (Richesin 1992):

> "...A search of the Quality Event Reports, Quality Incident Reports, Unusual Occurrence Reports, and Occurrence reports indicates no record of spills [on the Y-12 Plant] of heavy water. R.K. Barnett, who has worked in the lithium processing areas for the past 37 years, also has no recollection of any such spills. Please note there was a spill of half heavy water [HDO] in a truck in October of 1991. This incident was reported in Occurrence Report #MMES-91-4340 Y12-91-1819."

This information indicates that there were no large, uncontrolled releases of deuterium or heavy water that were not accounted for. However, it should be noted that, during the early years of deuterium gas production, heavy water was not likely accounted for in the same careful manner as deuterium gas. Spills of heavy water may not have been considered reportable.

4.3.3 Tritium Quantities Related to Deuterium Handled at the Y-12 Site

Limited information regarding the amount of tritium present at the Y-12 site was located by the project team. Since the concentration of the tritium in the deuterium gas is known (for 1959-1984), it would be possible to calculate the amount of tritium that was present from the deuterium inventory receipts. Because the deuterium inventory is classified, it was not possible to release a calculation based on the deuterium inventory. However, Y-12 plant classification personnel did agree to allow the deuterium inventory difference information to be released as unclassified. The deuterium inventory difference represents the amount of deuterium "lost" or unaccounted for in the deuterium processing. This project team assumed that the amount of tritium released would be proportional to the amount of deuterium "lost". This assumption is based on the knowledge that the tritium follows deuterium in the processing, and the processing does not involve steps that would remove the tritium.

Reported deuterium inventory differences for fiscal years 1955 through 1995 are presented in Table 4-4 (Owings 1995).

The reported total inventory difference of deuterium for the 1955 to 1995 time period was 15,843 kg or 34,855 lb. For this analysis, the deuterium inventory difference was assumed to represent the deuterium lost to the environment. The amount of tritium released through deuterium processing can be calculated by coupling the deuterium inventory difference with the known tritium concentration in the deuterium gas. As mentioned previously, the peak tritium concentration for the 1959 to 1984 time period was 7,092 μCi lb^{-1}. If all of the deuterium was lost, and the maximum tritium concentration of was present in the deuterium, then the total tritium lost to the environment would equal:

$$(35{,}000\ lb) \times (7092\ \mu Ci\ lb^{-1}) \times (1\times10^{-6}\ Ci\ \mu Ci^{-1}) \\ = 250\ Ci$$

This value was used to calculate the dose to an off-site individual. The IAEA method, described in Appendix H, was used to calculate the dose to the individual (IAEA 1996).

Table 4-4: Reported Deuterium Inventory Differences at Y-12

Fiscal Year	Inventory Difference (kg)	Fiscal Year	Inventory Difference (kg)
1955	2070	1975	99
1956	1642	1976	155
1957	1480	1977	210
1958	871	1978	156
1959	459	1979	99
1960	592	1980	120
1961	906	1981	142
1962	987	1983	251
1963	1093	1984	175
1964	800	1985	254
1965	491	1986	105
1966	497	1987	121
1967	344	1988	47
1968	290	1989	64
1969	172	1990	19
1970	265	1991	11
1971	210	1992	31
1972	178	1993	39
1973	141	1994	38
1974	204	1995	15
Total: 15,843 kg (~35,000 lbs)			

4.3.4 Level I Screening Analysis

In the Level I risk screening calculation, the peak tritium concentration (7,092 µCi lb^{-1}) in the deuterium gas was used to calculate the total tritium activity, and the environmental release was assumed to occur over the 40 years that the deuterium was handled or stored at the site. For the screening calculation, the following assumptions were used:

- 100% of the tritium was released to the water. This pathway is unlikely to occur, because the deuterium was stored as a gas. However, deuterium arrived at the plant-site in a liquid form (heavy water). The assumption that all tritium was released to water is also extremely conservative from a dosimetry standpoint, as tritiated water is considered a greater health risk than releases to the atmosphere.

- The fraction of consumed water that was contaminated for a resident along East Fork Poplar Creek (EFPC) was 1.0.

- The 250 Ci was released to EFPC over a 40-year period.

- The flow of EFPC was 30 million liters per day (4.4×10^{11} L over 40 y).

The IAEA dose equation was therefore applied as follows:

$$D_T^{\max} = [0 + (C_W)_n^{\max} \times (f_W)_n]\ g$$

The concentration in the off-site water would be equal to:

$$(C_W)_n^{\max} = \frac{(250\ Ci) \times (3.7\times10^{10}\ Bq\ Ci^{-1})}{4.4\times10^{11}\ L}$$

Solving the IAEA dose equation:

$$D_T^{\max} = [0 + (21\ Bq\ L^{-1}) \times (1.0)] \times (2.6\times10^{-8}\ Sv\ y^{-1} per\ Bq\ L^{-1})$$

$$D_T^{\max} = 5.5\times10^{-7}\ Sv\ y^{-1}$$

The resulting dose rate was 5.5×10^{-7} Sv y^{-1}. Over 40 years, this would have resulted in a total dose of 2.2×10^{-5} Sv. To convert this dose to health risk, ICRP recommends a value of 7.3% Sv^{-1} (ICRP 1990). This value combines ICRP's 5%, 1%, and 1.3% values for fatal cancer, non-fatal cancer, and severe hereditary effects, respectively. Thus, a Screening Index related to excess cancer risk was calculated as follows:

$$Screening\ Index = (2.2\times10^{-5}\ Sv) \times (0.073\ Sv^{-1})$$

$$Screening\ Index = 1.6\times10^{-6}$$

As this Level I Screening Index for Y-12 tritium releases is below the Oak Ridge Health Studies decision guide of 1×10^{-4}, a Refined Level I screening calculation will not be presented.

This page intentionally left blank.

5.0 MATERIALS SCREENED WITH THE STANDARD TASK 7 METHODOLOGY

The following materials were screened using the two-level screening approach described in Section 2.3:

- arsenic
- beryllium
- copper
- hexavalent chromium
- lithium
- neptunium-237
- nickel
- technetium-99

This section presents the results of the screening evaluations for each material. For each material, the following information is provided:

- Summary of use of the material on the ORR;
- Available release and environmental monitoring data for the material, and identification of exposure point concentrations for different environmental media;
- Discussion of the material's toxicity;
- Assessment of potential exposures to the material;
- Risk characterization of the potential health hazards resulting from exposure to the material due to releases from the ORR; and
- Comparison of screening level risk indices to risk-based decision guides established by ORHASP.

5.1 Arsenic Releases from the Oak Ridge Reservation

In the Dose Reconstruction Feasibility Study (ChemRisk 1993), arsenic was identified as a "contaminant that could not be quantitatively evaluated for any medium" because no environmental measurements of arsenic were located by the project team in Phase I. During the Dose Reconstruction, efforts were made to locate information concerning the presence, use, and environmental concentrations of arsenic at the ORR.

This section documents the sources of arsenic at ORR and the potential for adverse health effects that may have been associated with releases of arsenic to off-site locations. A review of historical operations at the ORR indicated that the only likely source of arsenic releases was from activities related to coal combustion. Therefore, investigations of coal combustion at ORR power generation facilities and associated fly ash disposal practices, and directed searches for environmental monitoring data for arsenic were conducted.

5.1.1 Arsenic Releases from Power Generation Operations at the ORR

Arsenic is a naturally occurring metallic element found in coal. Because the ORR complexes were large consumers of electrical power, several coal-fired steam plants were constructed and operated at each of the three ORR facilities. The storage and use of coal and the disposal of ash from coal burning operations likely resulted in the release of arsenic to air, surface water, soil, and sediments. For example, burning of coal containing naturally occurring arsenic can result in direct arsenic releases to air. Coal storage practices can result in arsenic releases to soil, surface water, and sediments when, for example, coal piles are not equipped with runoff treatment systems. Disposal of fly ash (a solid waste product formed from noncombustible components and incomplete combustion of coal particles) in pits, quarries, or unlined disposal facilities can also result in arsenic contamination of soil, surface water, and sediments.

Several power generation facilities have operated at the ORR from 1944 to the present. These facilities include:

- K-25 K-701 Power Station/Boiler House (1944-1962);
- K-25 K-1501 Steam Plant (1944-present);
- Y-12 Steam Plant- Building 9401-1 (1944-1956);
- Y-12 Steam Plant- Building 9401-2 (1944-1956);
- Y-12 Steam Plant- Building 9401-3 (1956-present); and,
- X-10 Steam Plant.

The K-701 Power Station and the Y-12 Building 9401-3 Steam Plant were the largest of the six power generating facilities. At K-25, the K-701 Power Station was much larger than the K-1501 Steam Plant, which was built to provide energy for heating buildings. The K-701 Power Station provided energy to run the gaseous diffusion processes at K-25. At Y-12, the 34,000 ft^2 Y-12 Building 9401-3 Steam Plant was designed to replace two older 12,000 ft^2 Y-12 steam plants in Buildings 9401-1 and 9401-2 (Thomason and Associates 1996), and is reported to have a larger capacity than the other two added together (Choat 1996). In this screening assessment, arsenic releases are estimated only for the two largest power generation facilities.

5.1.1.1 K-701 Power Station

The K-25 Power Station facilities were constructed during World War II to provide electricity for uranium enrichment activities at the K-25 site. The K-701 Power Station was located near the former S-50 Plant, adjacent to the Clinch River. The power station operated from May 1944 to October 1962 and was coal-fired, but at times was supplemented with fuel oil (Pesci 1995). Power generation was discontinued in 1962, after it was decided to use TVA-provided power only (MMES 1988). Power-generating equipment was later removed and sold, and the building was torn down in September 1995 (USDOE 1995a).

Coal Burning

Over its 18 years of operation, the K-701 Power Station burned a total of 5.4×10^9 kg (1.2×10^{10} pounds) of coal (Pesci 1995), or an annual average of 3×10^8 kg (6.6×10^8 pounds) per year.

Table 5-1 shows the mean arsenic content of coal from several different sources.

Table 5-1: Mean Arsenic Content of Coal

Source/ Type of Coal	Mean Arsenic Content (mg kg^{-1})	Reference
United States	14	USEPA (1974)
Appalachian	22-27	USEPA (1984b), USGS (1976)
"Usual coal content"	25	IARC (1980)
Bituminous	20.3 (range 0.02 to 357)	USEPA (1986c)
Coal burned at Y-12 steam plant (1976-1977)	47 (range 22 to 65)	UCCND (1983a)

As shown, the mean arsenic content of Y-12 coal ranged from about 1.7 to 3.4 times that of the reported mean arsenic content of coal reported for other parts of the United States.

It is assumed that coal burned at the K-25 powerhouse facilities was similar to coal burned at Y-12. If it is assumed that 3×10^{8} kg of coal were burned per year at the K-701 Power Station, and that the coal had an arsenic content of 47 mg kg^{-1}, then the total mass of arsenic available to be released to the environment per year from direct air releases and/or fly ash disposal was 14,100 kg.

Fly Ash Disposal

Fly ash from coal burning at the K-701 Power Station was collected and transported by pipeline to the K-720 fly ash pile, which covered approximately 20 acres. Runoff and leachate from the pile was not controlled, and drained directly to Poplar Creek at about Poplar Creek Mile (PCM) 1.0.

5.1.1.2 Y-12 Steam Plant (Building 9401-3)

Construction of a new steam plant (Building 9401-3) to meet the increased electricity requirements of the Alpha-5 and Alpha-4 lithium separation operations at Y-12 began on January 19, 1954 and was completed in June 1956. The two older and smaller steam plants, Buildings 9401-1 and 9401-2, were subsequently dismantled, and the buildings used for other purposes (Thomason and Associates 1996). The Building 9401-3 Y-12 steam plant operated from 1956 to present and was originally designed to burn coal. However, natural gas was burned at various times during its operating history (Alpha-5 Plant Chronology 1953-1954; Murray 1956; UCCND 1956; UCCND 1983b).

Coal Burning

Per UCCND (1983), 2.4×10^8 lb (1.1×10^8 kg) of coal were burned per year at Y-12. This value is corroborated by several other reports:

- A 1970 report (Schmitt 1970) discusses an investigation of the fly-ash disposal system at Y-12. The report states that the Y-12 steam plant generated approximately 100,000 lbs of steam per hour in the summer and approximately 600,000 lbs per hour in the winter (Schmitt 1970; UCCND 1983a). The report further states that one pound of coal was required per 10 pounds of steam generated. Assuming that the steam plant generated an average of 350,000 lb of steam per hour on an annual basis (the mean of the summer and winter values) and that the plant operated 24 hours per day, the estimated volume of coal burned was 35,000 lbs per hour, or 3.1×10^8 lbs (1.4×10^8 kg) per year.

- Quarterly summaries of coal burned at Y-12 from 1956 to 1959 are also available (Turner et al. 1991)—the reported volumes of coal burned during 1956, 1957, 1958, and 1959 were 1.7×10^8 kg, 0.73×10^8 kg, 0.50×10^8 kg, and 0.64×10^8 kg, respectively (UCCND 1956, 1957, 1958, 1959b).

Assuming 1.1×10^8 kg of coal were burned per year (UCCND 1983a) containing 47 mg kg^{-1} of arsenic, then 5,200 kg per year of arsenic were available for potential release to the environment.

Fly Ash Disposal

Fly ash from coal burning at the Y-12 Steam Plant was mixed with water and pumped as a slurry over the crest of Chestnut Ridge, located at the south perimeter of the Y-12 Plant, where it flowed by gravity to an excavated earthen retention basin and dam. The retention basin covers 20 acres and was completed in 1955 (Schmitt 1970). Originally, an earthen dam or dike was constructed across the McCoy Branch watershed to provide sedimentation for the ash slurry before discharge into McCoy Branch. However, this ash retention impoundment reached capacity in the early 1960s (Pesci 1995). The overflow from the retention basin went through a channel along McCoy Branch for about a half-mile to Rogers Quarry, an abandoned, water-filled limestone quarry, where the ash solids and sluice water were separated by sedimentation (Schmitt 1970; Turner et al.1986). In 1989, a bypass line was constructed to carry the slurry directly to the quarry from the steam plant (USDOE 1995b). Fly ash disposal in Rogers Quarry was stopped in the early 1990s due to environmental concerns. After 1990, the fly ash was taken to the Chestnut Ridge Landfill (Wilburn 1997).

A study to characterize the coal ash slurry discharge and the chemical quality of the McCoy Branch was conducted from March through May 1986. The results of the investigation indicated that arsenic concentrations in the effluents discharged to McCoy Branch ranged between 0.20 and 0.22 mg L^{-1} (Turner et al. 1986).

5.1.2 Concentrations of Arsenic in the Environment near the ORR

Arsenic may be present in the environment from both natural and human-related sources. The following sections describe typical natural, or background, levels of arsenic in the environment, and concentrations that have historically been measured in the environment near the ORR.

5.1.2.1 Background Levels

Arsenic is found widely in nature and most abundantly in sulfide ores (HSDB 1999). Several sources of data on background concentrations of arsenic were identified by the project team. These included general background concentrations and near-site background concentrations measured in areas of Eastern Tennessee assumed not to be impacted by ORR activities.

Data on background concentrations of arsenic identified by the project team are summarized briefly by media below.

- *Air* - Mean concentrations of arsenic in air in the United States usually range from <1 to 3 ng m^{-3} in remote areas and from 20 to 30 ng m^{-3} in urban areas (Davidson et al. 1985; USEPA 1982a; IARC 1980; and NAS 1977, as cited in ATSDR 1993). Large cities generally have higher arsenic air concentrations than smaller cities due to releases from coal-fired power plants, but maximum 24-hour concentrations generally are less than 100 ng m^{-3} (IARC 1980, as cited in ATSDR 1993).

- *Surface Water* - The median arsenic concentration in surface water reported in the USEPA's STORET database was 0.003 mg kg^{-1} (USEPA 1982b, as cited in ATSDR 1993). Surveys of arsenic concentrations in rivers and lakes indicate that most values are below 0.010 mg kg^{-1}, although concentrations in individual samples may range up to 1 mg kg^{-1} (NAS 1977; Page 1981; Smith et al. 1987; and Welch et al. 1988, as cited in ATSDR 1993).

  Arsenic was not detected in any of the surface water samples collected in Norris Reservoir during the Clinch River Remedial Investigation (with a detection limit of 0.0013 mg L^{-1}; USDOE 1996). Norris Reservoir was identified in the CRRI as the "reference location," indicating that it was considered unaffected by releases from the ORR. The mean concentration of arsenic detected in Poplar Creek upstream of mile 5.5 (the point where EFPC flows into Poplar Creek) measured during the mid-1990s was 0.0012 mg L^{-1} (reported in the OREIS database, LMES 1997). The mean concentration of arsenic in the Clinch River upstream of mile 48 was 0.0018 mg L^{-1} (LMES 1997).

- *Soil/Sediment* - Arsenic concentrations in background soils range from about 1 to 40 mg kg^{-1}, with a mean value of about 5 mg kg^{-1} (Beyer and Cromartie 1987; Eckel and Langley 1988; USEPA 1982a; and NAS 1977, as cited in ATSDR 1993). In aquatic systems, arsenic concentrations in sediment are often higher than those in water. Most sediment arsenic concentrations reported for U.S. lakes, rivers and streams range from 0.1 to 4,000 mg kg^{-1} (Heit et al. 1984; NAS 1977; and Welch et al. 1988, as cited in ATSDR 1993).

The mean concentration of arsenic detected in sediments in Poplar Creek upstream of mile 5.5 measured during the mid-1990s was 3.3 mg kg^{-1} (reported in the OREIS database, LMES 1997). The mean concentration of arsenic in Clinch River sediment upstream of mile 48 was 3.9 mg kg^{-1} (LMES 1997).

- *Food* - The highest arsenic concentrations in food are found in marine products. Mean levels in fish and shellfish are usually about 4-5 mg kg^{-1}, but may be as high as 170 mg kg^{-1} (NAS 1977, as cited in ATSDR 1993). Typical U.S. dietary levels of arsenic in meat, fish, and poultry are about 0.14 mg kg^{-1} (Gartrell et al. 1986, as cited in ATSDR 1993). Arsenic is frequently found in plants, often as a result of pesticide treatment (NAS 1977, as cited in ATSDR 1993). Levels of arsenic in tobacco averaged 1.5 mg kg^{-1} (USEPA 1984a, as cited in ATSDR 1993).

  The mean concentration of arsenic detected in fish from Poplar Creek upstream of mile 5.5 measured during the mid-1990s was 0.053 mg kg^{-1} (reported in the OREIS database, LMES 1997). The mean concentration of arsenic in Clinch River fish collected upstream of mile 48 was 0.085 mg kg^{-1} (LMES 1997).

5.1.2.2 Environmental Concentrations on or near the ORR

Environmental samples (including sediment, surface water, and fish) have been analyzed for arsenic as part of a number of environmental investigations conducted at the ORR and in the surrounding areas. In addition, data on arsenic concentrations emitted to air from burning of coal were identified. Available data describing arsenic concentrations in air, surface water, soil, sediment, and other environmental media on or near the ORR are discussed in the following sections. In addition, determination of exposure point concentrations used in the screening assessment is described.

5.1.2.2.1 Arsenic Concentrations in Air

No stack releases data were located for arsenic. Consequently, USEPA emission factors (USEPA 1989b) were used to estimate arsenic air releases from the K-701 Power Station and the Y-12 Steam Plant. USEPA reports an emission factor for uncontrolled releases of arsenic from a pulverized wet bottom bituminous coal-fired boiler of 538 lb of arsenic per 10^{12} Btu (USEPA 1989b). Because the arsenic content of coal burned at the Y-12 Plant appears to have been about 2.3 times higher than the national average for bituminous coal, this emission factor was multiplied by 2.3. Therefore, it was assumed that the release of arsenic was 1,237 lb per 10^{12} Btu.

Air Emission Calculation for the K-701 Power Station

1.19×10^{10} lbs of coal were burned over 18 years (Pesci 1995). Assuming a constant burn rate, the average amount of coal burned per year was 6.61×10^{8} lbs.

For purposes of this assessment, arsenic releases from coal burning were estimated based on the coal's caloric value and the assumption that arsenic was released at a rate of 1,237 lb per 10^{12} Btu. According to Marks' Standard Handbook for Mechanical Engineers (1996), the caloric value of West Virginia coal (bituminous) is 14,040 Btu lb^{-1}, and the caloric value of Kentucky coal is 11,680 Btu lb^{-1} (Marks 1996). For purposes of this assessment, the caloric value of the coal burned at the ORR was assumed to be 14,040 Btu lb^{-1}.

The annual caloric content of the coal burned at K-25 was calculated as follows:

$$\frac{14{,}040\ Btu}{1\ lb\ coal} = \frac{x\ Btu}{6.61\times10^{8}\ lbs\ of\ coal}$$

$$x = 9.2\times10^{12}\ Btu\ y^{-1}$$

Releases of arsenic were then estimated as follows:

$$\frac{1{,}237\ lbs\ Arsenic}{10^{12}\ Btu} \times (9.2 \times 10^{12}\ Btu\ y^{-1}) = 11{,}380\ lbs\ arsenic\ per\ year$$

$$= 0.16\ g\ s^{-1}$$

Off-Site Air Concentrations of Arsenic from K-25(K-701) Releases

At Union/Lawnville, the χ/Q corresponding to a unit release of a material from K-25 was estimated to be 7.4×10^{-7} s m^{-3} in Task 6 modeling (see Section 2.3.4.2). The air concentration of arsenic at Union/Lawnville was then estimated by multiplying this χ/Q by the arsenic release rate for the K-701 Power Station:

$$Lawnville\ air\ concentration = (7.4\times10^{-7}\ s\ m^{-3}) \times (0.16\ g\ s^{-1})$$

$$= 1.2 \times 10^{-7}\ g\ m^{-3} = 120\ ng\ m^{-3}$$

Arsenic Air Release Calculation for the Y-12 Steam Plant

Y-12 burned 120,000 tons[1] of coal per year (UCCND 1983a). This is equal to 2.2×10^{8} lbs per year. The annual caloric content of the coal burned at Y-12 was calculated as follows:

$$\frac{14{,}040\ Btu}{1\ lb\ coal} = \frac{x}{2.2\times10^{8}\ lbs\ y^{-1}}$$

$$x = 3.1\times10^{12}\ Btu\ y^{-1}$$

Releases of arsenic were then estimated as follows:

$$\frac{1{,}237\ lbs\ Arsenic}{10^{12}\ Btu} \times (3.1 \times 10^{12}\ Btu\ y^{-1}) = 3{,}834\ lbs\ arsenic\ y^{-1} = 0.055\ g\ s^{-1}$$

Off-Site Air Concentrations of Arsenic from Y-12 Steam Plant (9401-3) Releases

At Scarboro, the χ/Q corresponding to a unit release of a material from Y-12 was estimated to be 3×10^{-7} s m^{-3} in Task 6 modeling (see Section 2.3.4.1). The air concentration of arsenic at Scarboro was estimated by multiplying this χ/Q by the arsenic release rate for the Y-12 Steam Plant:

$$Scarboro\ air\ concentration = 3.1\times10^{-7}\ s\ m^{-3} \times 0.055\ g\ s^{-1}$$

$$= 1.7\times10^{-8}\ g\ m^{-3} = 17\ ng\ m^{-3}$$

5.1.2.2.2 Arsenic Concentrations in Surface Water

Arsenic concentrations in surface water are typically very low. Reported water solubility of arsenic in coal ash is 1% for ash from three power plants in the UK and 4% for U.S. ash (Alloway 1990; EPRI 1981, as cited in MMES 1988).

---

[1] In this assessment, a ton is taken to represent a short ton, which is equal to 2,000 pounds.

In 1994, surface water samples were collected near the K-720 ash pile at K-25 between PCM 0.1 and PCM 1.4 and analyzed for arsenic. The average arsenic concentration of the 38 samples was 0.001 mg L^{-1} (LMES 1997). However, recent concentrations of arsenic in surface water are not likely to be representative of historical surface water arsenic concentrations, because the K-710 Power Station was shut down in 1962 and ash was no longer stored at the K-720 ash pile. Surface water samples collected at PCM 0.3 in 1977-78 were located by the project team in a 1981 report (Loar et al. 1981a). The mean arsenic concentration of the samples was below the limit of detection of 0.01 mg L^{-1}. The maximum concentration was 0.02 mg L^{-1}.

At Y-12, the McCoy Branch connects Rogers Quarry to the Clinch River. Surface water samples were collected between McCoy Branch Mile (MBM) 0.2 and MBM 0.4 in 1994. The 95% UCL on the mean of the 46 samples was 0.0024 mg L^{-1} (LMES 1997).

The Bull Run Steam Plant ash pile is located between CRM 46.5 and CRM 48. Surface water samples collected downstream between CRM 41.5 and CRM 46.2 in 1994 and analyzed for arsenic had a mean concentration of 0.001 mg L^{-1} (N=34) (LMES 1997). Low arsenic concentrations downstream of Bull Run might reflect the fact that as of 1983, dry ash was collected from the Bull Run stacks rather than placing wet ash slurry in fly ash ponds and returning the water to the river (Beeles 1997).

Off-Site Surface Water Concentration Used in the Screening Assessment

The highest surface water concentration reported in Poplar Creek near the mouth of the Clinch River was used by the project team in the screening assessment to evaluate exposures to arsenic in surface water due to releases from K-25. The highest reported surface water concentration, measured during 1977-1978, was 0.02 mg L^{-1}.

The 95% UCL on the mean surface water concentration reported in the McCoy Branch was used in the screening assessment to evaluate exposures to arsenic in surface water due to releases from Y-12. Data collected in the McCoy Branch are assumed to reflect concentrations to which off-site individuals may have been exposed. The 95% UCL, based on samples collected in 1994, was 0.0024 mg L^{-1}.

5.1.2.2.3 Arsenic Concentrations in Soil/Sediment

No soil arsenic measurements from the McCoy Branch area south of Y-12, or from the K-25 site near the K-701 Power Station, were located by the project team. The nearest soil arsenic measurements located by the project team were taken at an unidentified property in the EFPC floodplain (#564) in 1984, and ranged from 6 to 16 mg kg^{-1} (Hibbitts 1984-87). Consequently, for purposes of this screening evaluation, measurements of arsenic in sediments were used to represent soil arsenic concentrations.

Sediment samples were taken near the K-720 ash pile at K-25 between PCM 0.5 and PCM 1.5 in 1994 and analyzed for arsenic. The average arsenic level in the 13 samples was 24 mg kg^{-1} (LMES

1997). However, by 1994, arsenic released from the K-720 ash pile would have been buried under layers of sediment because the K-710 Power Station was shut down in 1962 and ash was no longer stored at the K-720 ash pile. Arsenic concentrations measured at depth in sediments were determined to be most representative of historical soil concentrations of arsenic. A sediment core collected at PCM 1.0 in 1985 contained 55 mg kg^{-1} arsenic at 76-80 cm below ground surface (bgs). It is assumed that samples collected at this depth correspond to ash deposited in the early 1960s (Ashwood et al. 1986).

Sediment samples were collected in the McCoy Branch between MBM 0.2 and MBM 0.4 in 1991 and 1994. The 95% UCL on the mean of the 32 samples was 22 mg kg^{-1} (LMES 1997).

The Bull Run Steam Plant ash pile is located between CRM 46.5 and CRM 48. Sediment samples collected downstream at CRM 44 in 1990 and analyzed for arsenic had a mean concentration of 5 mg kg^{-1} (LMES 1997). Low arsenic concentrations downstream of Bull Run in 1990 might reflect the fact that as of 1983, dry ash was collected from the stacks at Bull Run instead of wet ash slurry being placed in fly ash ponds and the water returned to the river (Beeles 1997).

Off-Site Soil/Sediment Concentration Used in the Screening Assessment

The arsenic concentration measured at a depth of 76-80 cm bgs in the sediment core collected at PCM 1.0 in 1985 (55 mg kg^{-1}) was used by the project team in the screening assessment to evaluate exposures to arsenic in soil and sediment due to releases from K-25.

The 95% UCL on the mean sediment concentration reported in the McCoy Branch was used in the screening assessment to evaluate exposures to arsenic in soil and sediment due to releases from Y-12. The 95% UCL, based on samples collected in 1994, was 22 mg kg^{-1}.

5.1.2.2.4 Arsenic Concentrations in Food Items

Arsenic concentrations in fish from waterways near the ORR were measured during several sampling programs. Identified data describing arsenic concentrations in fish include the following:

C At K-25, recent analyses of fish tissues from the Clinch River Study (LMES 1997) were identified by the project team. Fish tissues were collected between PCM 1.0 and PCM 1.4 and analyzed for arsenic in 1989 and 1993 (LMES 1997). The average arsenic concentration was 0.12 mg kg^{-1}. These recent arsenic measurements are not representative of historical fish concentrations, as the K-710 Power Station was shut down in 1962 and was no longer the source of arsenic releases to surface water at K-25.

C No measurements of arsenic in fish tissues from McCoy Branch south of the Y-12 Plant or downstream of the Bull Run steam plant were located by the project team.

No measurements of arsenic in meat, milk or vegetables were identified by the project team.

Off-Site Fish, Vegetation, Meat, and Milk Concentrations Used in the Screening Assessment

For the purposes of the screening analyses, concentrations of arsenic in vegetation, meat, and milk were calculated using biotransfer factors that describe the transfer of arsenic from other media, including air, water, and soil. The biotransfer factors used in this assessment are listed in Table 5-2. Concentrations of arsenic in fish were calculated using a bioconcentration factor (BCF), shown in Table 5-2, that describes the transfer of arsenic from water to fish.

Table 5-2: Biotransfer and Bioconcentration Factors for Arsenic

Parameter	Value	Source
Soil to Vegetables (B_{veg})	0.08 (mg kg^{-1} wet)/(mg kg^{-1} dry)	NCRP 1996
Soil to Pasture ($B_{pasture}$)	0.20 (mg kg^{-1} dry)/(mg kg^{-1} dry)	NCRP 1996
Biotransfer to Milk (F_m)	0.0001 d L^{-1}	NCRP 1996
Biotransfer to Meat (F_f)	0.020 d kg^{-1}	NCRP 1996
Bioconcentration in Fish (BCF)	400 (mg kg^{-1})/(mg L^{-1})	ATSDR 1993

5.1.3 Toxicity Assessment

The main routes of exposure of individuals in the general population to arsenic are via ingestion of food and water and in tobacco for smokers (Marcus and Rispin 1988). Food and Drug Administration (FDA) surveys indicate an average arsenic intake in the United States of approximately 50 µg d^{-1}. The concentration of arsenic in fish and seafood can be ten times higher than in other foods (FDA 1982-1984). Despite a limited data base, there is some evidence of a nutritional requirement for arsenic in humans (USEPA 1988), with an essential daily intake from 12 to 50 µg d^{-1} (Marcus and Rispin 1988).

Arsenic may be present in the environment in several different forms or species. Trivalent arsenites (As(III)) tend to be somewhat more toxic than pentavalent arsenates (As(V)). However, the differences in the relative potency are reasonably small (about 2-3 fold), often within the bounds of uncertainty regarding no observable adverse effect levels (NOAELs) or lowest observed adverse effect levels (LOAELs) from animal studies. In addition, arsenic may undergo reduction or oxidation to different species, both in the environment and in the body. In many cases of human exposure, especially those involving intake from water or soil, the precise chemical speciation is not known (ATSDR 1993).

The following sections describe data characterizing the toxicity of arsenic through different routes of exposure, and summarize the toxicity criteria used to evaluate exposure to arsenic in the screening assessment.

5.1.3.1 Inhalation Exposure

The primary health effect associated with exposure to airborne arsenic is irritation of the skin and mucous membranes (ATSDR 1993). Several case reports and epidemiological studies in human populations indicate that inhalation of inorganic arsenic can lead to neurological injury, including peripheral neuropathy of sensory and motor neurons (numbness, loss of reflexes, muscle weakness), and frank encephalopathy (hallucinations, agitation, emotional lability, memory loss). The effects tend to diminish after exposure ceases, but some effects may persist. However, available data are not sufficient to define a level of concern for the neurological effects of inhaled arsenic (ATSDR 1993).

There is convincing evidence from a large number of epidemiological studies that inhalation exposure to inorganic arsenic increases the risk of lung cancer. Most worker exposure studies involve exposure to airborne arsenic trioxide dust at copper smelters, but increased incidences of lung cancer have also been observed at chemical plants where exposure was primarily to arsenate. Many of the studies provide only qualitative evidence of an association between duration and/or level of arsenic exposure and risk of lung cancer, but several studies provide sufficient exposure data to permit quantification of cancer risk. In general, the data indicate an approximately linear increase in relative risk (the frequency of lung cancer in the exposed group divided by the frequency of lung cancer in the control group) as a function of increasing cumulative exposure (ATSDR 1993).

The USEPA has assigned arsenic a weight-of-evidence classification of Group A through inhalation exposure, indicating sufficient evidence of carcinogenicity in humans. The USEPA established an inhalation slope factor for arsenic of 15 $(\text{mg kg}^{-1}\ \text{d}^{-1})^{-1}$ (USEPA 1999).

5.1.3.2 Oral (Ingestion) Exposure

Most studies indicate that animals are less sensitive to the toxic effects of arsenic than humans. Inorganic arsenic is recognized as potentially toxic to humans through ingestion, and large doses (600 $\mu\text{g kg}^{-1}\ \text{d}^{-1}$ or higher) can be lethal. Classic symptoms of acute oral exposure to arsenic are gastrointestinal irritation, anemia, neuropathy, skin and vascular lesions, and hepatic (kidney) or renal (liver) lesions (ATSDR 1993; Seiler et al. 1988). These symptoms are based on reports of suicidal or homicidal arsenic ingestion, because reports of acute arsenic poisoning arising from environmental exposure are rare (Franzblau and Lilis 1989).

Chronic oral exposure to arsenic has been reported to result in gastrointestinal effects such as nausea, vomiting and diarrhea. Long-term exposure may also lead to anemia, leukopenia, and eosinophilia, as well as peripheral vascular disease.

The USEPA has assigned inorganic arsenic a Group A weight-of-evidence classification for ingestion exposure, indicating sufficient evidence of carcinogenicity in humans. A multistage model was used by the USEPA in 1988 to derive a unit risk value for ingestion of inorganic arsenic, based on studies of populations in Taiwan exposed to inorganic arsenic in drinking water (Tseng et al. 1968). By 1966, Tseng et al. had surveyed 40,421 persons in 37 villages for skin cancer, who were exposed to inorganic arsenic from shallow drinking water wells. Concentrations of arsenic in water from these wells ranged from 1.0 $\mu g\ L^{-1}$ to 182 $\mu g\ L^{-1}$. The control population consisted of 7,500 persons with an age distribution similar to that of the study population. This study is the largest available on human arsenic exposure to date. The study demonstrated an association between arsenic exposure and the development of skin cancer; however, the study has several weaknesses and uncertainties, including poor nutritional status of the exposed populations, their genetic susceptibility, and their exposure to inorganic arsenic from non-water sources, that limit the study's usefulness in risk estimation. Dietary inorganic arsenic was not considered nor was the potential confounding by contaminants other than arsenic in drinking water (USEPA 1999). In addition, there is concern of the applicability of extrapolating data from Taiwanese to the U.S. population because of different background rates of cancer, possibly genetically determined, and differences in diet other than arsenic (e.g., low protein and fat and high carbohydrate) (USEPA 1999).

Based on the Taiwanese studies, the USEPA calculated a lifetime cancer risk range for inorganic arsenic between 1×10^{-3} and 2×10^{-3}, based on oral exposure of 1.4 $\mu g\ kg^{-1}\ d^{-1}$ (USEPA 1988). These risk estimates assume linearity at low doses and may overestimate risk if a threshold for arsenic induced skin cancer exists (USEPA 1988; Petito and Beck 1990). Using these data, the USEPA established an oral slope factor for inorganic arsenic of 1.5 $(mg\ kg^{-1}\ d^{-1})^{-1}$ (USEPA 1999).

Several other studies have shown a relationship between ingestion of inorganic arsenic in drinking water and increased incidence of fatal organ cancers in humans (USEPA 1999). There has not been consistent demonstration of carcinogenicity of arsenic administered through different routes in test animals. The meaning for non-positive data for carcinogenicity of inorganic arsenic is uncertain, the mechanism of action in causing human cancer is not known, and rodents may not be a good model for arsenic carcinogenicity testing (USEPA 1999).

Epidemiologically, there are no reports of increased disease incidence related to consumption of high levels of arsenic via seafood ingestion. Arsenobetaine $[(CH_3)As^+CH_2COOH^-]$, an organic form of arsenic, is typically the predominant form of arsenic found in fish, and often comprises virtually all of total arsenic in fish tissue. Arsenobetaine is very stable and highly resistant to degradation by reagents used for digestion of organic material. Therefore, it seems reasonable to expect that degradation in the human gut does not occur. Research has shown that between 70-85% of all arsenobetaine ingested by humans is absorbed systemically and excreted unmetabolized within 5 days (Vahter et al. 1983). In studies involving human volunteers ingesting arsenobetaine, it was the only detected arsenical species in urine, indicating essentially no biotransformation (Vahter et al. 1983; Cannon et al. 1983; Kaise et al. 1985). In short, there is no evidence to suggest that humans might degrade arsenobetaine to inorganic arsenic *in vivo*.

Arsenobetaine has never been tested for carcinogenicity in an animal bioassay, yet it is assumed by USEPA to be noncarcinogenic. This is due to the fact that other methylated derivatives (i.e., mono and dimethyl) have been tested and not found to be carcinogenic. Jongen et al. (1985) found that arsenobetaine was not genotoxic *in vitro*. Specifically, it is well known that the toxicity of inorganic arsenic is due to its ability to react with sulfhydryl groups on proteins and DNA. When arsenic becomes successively methylated, the reactive sites become "capped," rendering the arsenic compound essentially, biologically nonreactive. Hence, lack of reactivity of the mono-, di, and trimethylated forms of arsenic strongly suggests that arsenobetaine would be almost inert toxicologically, as the available data suggest.

The USEPA established an oral RfD for noncarcinogenic effects of chronic exposure to inorganic arsenic of 0.0003 mg kg^{-1} d^{-1} (USEPA 1999). The RfD is based on the appearance of blackfoot disease, a hyperpigmentation of the skin with possible vascular disease, reported in the studies by Tseng et al. (1968) and Tseng (1977). The exposure to arsenic was through drinking water.

5.1.3.3 Toxicity Criteria Used in the Screening Assessment

Toxicity criteria used in the screening analyses to evaluate exposures to arsenic at Oak Ridge are presented in Table 5-3. As discussed above, studies suggest that the organic form of arsenic found in fish may be essentially nontoxic. However, for purposes of this screening assessment, the USEPA oral slope factor and oral RfD for inorganic arsenic were used to evaluate possible cancer risk and noncarcinogenic health effects from ingestion of arsenic in fish.

Table 5-3: Toxicity Criteria for Arsenic Used in the Screening Analyses

Exposure Route	Toxicity Criteria/Source	Value
Inhalation (Cancer)	USEPA inhalation slope factor	15 $(mg\ kg^{-1}\ d^{-1})^{-1}$
Ingestion (Cancer)	USEPA oral slope factor	1.5 $(mg\ kg^{-1}\ d^{-1})^{-1}$
Ingestion (Non-cancer)	USEPA oral RfD	3.0×10^{-4} mg kg^{-1} d^{-1}

5.1.4 Exposure Assessment

Following the Task 7 Screening methodology for Level I and Level II screening (ChemRisk 1996), doses were calculated for exposure to arsenic in air, surface water, and soil/sediment. Exposures to arsenic through ingestion of fish, vegetables, meat, and milk were evaluated using equations (presented in Appendix B) and biotransfer factors (discussed in Section 5.1.2.2.4) that describe the uptake of arsenic from air, surface water, and soil/sediment into these media. A summary of the concentrations for each medium that were used in the screening assessment is presented in Table 5-4.

Table 5-4: Summary of Environmental Concentrations of Arsenic used in Screening

Environmental Medium	Concentration	Source of Value	Typical Background[1]
K-25			
Air	0.12 µg m^{-3}	Calculated for K-25 coal use, dispersion. modeling to Union/Lawnville	<0.001 to 0.003 µg m^{-3} rural 0.020 to 0.030 µg m^{-3} urban
Surface Water	0.02 mg L^{-1}	Maximum surface water concentration in Poplar Creek (1977-78)	0.0012 mg L^{-1} (mean) above PCM 5.5; 0.0018 mg L^{-1} (mean) above CRM 48
Soil/Sediment	55 mg kg^{-1}	Measured at 76-80 cm bgs in sediment core from PCM 1.0 (1985)— assumed to represent early 1960s	Soil: 1-40 mg kg^{-1} Sed: 3.3 mg kg^{-1} (mean) above PCM 5.5; 3.9 mg kg^{-1} (mean) above CRM 48
Fish	8.0 mg kg^{-1}	Calculated based on water concentration shown above and a BCF of 400	usually about 4 to 5 mg kg^{-1}
Y-12			
Air	0.017 µg m^{-3}	Calculated for coal burning at Y-12 and air dispersion modeling to Scarboro	same as above
Surface Water	0.0024 mg L^{-1}	95% UCL of concentrations measured in surface water from McCoy Branch (1994)	same as above
Soil/Sediment	22 mg kg^{-1}	95% UCL of concentrations measured in sediment from McCoy Branch (1994)	same as above
Fish	0.96 mg kg^{-1}	Calculated based on water concentration shown above and a BCF of 400	same as above

[1] For references, see Section 5.1.2.1.

5.1.4.1 Screening Level Estimates of Dose

The total average daily doses of arsenic calculated for the inhalation and ingestion (oral) exposure routes for arsenic released from K-25 and Y-12 are summarized in Table 5-5. The doses calculated for individual pathways and the contribution of each pathway to the total dose are summarized in Appendix I.

Table 5-5: Arsenic Doses Calculated in the Level I and Refined Level I Screening (mg kg^{-1} d^{-1})

	Lifetime Average Daily Inhalation Dose- (Carcinogen)	**Lifetime Average Daily Oral Dose- (Carcinogen)**	**Average Daily Oral Dose- (Noncarcinogen)**
K-25			
Level I	9.9×10^{-6}	2.6×10^{-2}	3.6×10^{-2}
Refined Level I	1.9×10^{-6}	5.7×10^{-4}	4.0×10^{-3}
Y-12			
Level I	1.5×10^{-6}	1.2×10^{-2}	1.2×10^{-2}
Refined Level I	8.2×10^{-8}	1.7×10^{-4}	1.2×10^{-3}

5.1.5 Risk Characterization

Cancer and non-cancer screening indices were calculated based on the screening estimates of lifetime average daily dose and average daily dose, respectively, for arsenic released from K-25 and from Y-12. The results of the screening risk characterization are presented below, and compared to risk-based decision guides established by the Oak Ridge Health Agreement Steering Panel (ORHASP).

5.1.5.1 Cancer Screening Indices Associated with Inhalation and Ingestion of Arsenic Released from K-25

The following equation was used to calculate the cancer screening indices for inhalation and ingestion of arsenic:

$$\text{Cancer Screening Index} = \text{Lifetime Average Daily Dose}\ (mg\ kg^{-1}\ d^{-1}) \times \text{Slope Factor}\ (mg\ kg^{-1}\ d^{-1})^{-1}$$

At Level I, the cancer screening index for inhalation of arsenic released from K-25 is:

$$\text{Cancer Screening Index} = 9.9\times10^{-6}\ (mg\ kg^{-1}\ d^{-1}) \times 15\ (mg\ kg^{-1}\ d^{-1})^{-1} = 1.5\times10^{-4}$$

and the cancer screening index for ingestion of arsenic released from K-25 is:

$$\text{Cancer Screening Index} = 2.6\times10^{-2}\ (mg\ kg^{-1}\ d^{-1}) \times 1.5\ (mg\ kg^{-1}\ d^{-1})^{-1} = 3.8\times10^{-2}$$

The cancer screening indices for inhalation and ingestion of arsenic were then summed to give a K-25 Level I cancer screening index for both routes of exposure (3.8×10^{-2}). Because the Level I cancer screening index for arsenic exceeded 1×10^{-4}, the decision guide established by ORHASP for evaluating the need for further study of carcinogens, a Refined Level I screening was conducted.

At Refined Level I, the cancer screening index for inhalation of arsenic released from K-25 is:

$$\mathit{Cancer\ Screening\ Index} = 1.9\times10^{-6}\ (mg\ kg^{-1}\ d^{-1}) \times 15\ (mg\ kg^{-1}\ d^{-1})^{-1} = 2.9\times10^{-5}$$

and the cancer screening index for ingestion of arsenic released from K-25 is:

$$\mathit{Cancer\ Screening\ Index} = 5.7\times10^{-4}\ (mg\ kg^{-1}\ d^{-1}) \times 1.5\ (mg\ kg^{-1}\ d^{-1})^{-1} = 8.6\times10^{-4}$$

The cancer screening indices for inhalation and ingestion of arsenic were then summed to give a K-25 Refined Level I cancer screening index for both routes of exposure (8.9×10^{-4}).

5.1.5.2 Non-cancer Screening Indices Associated with Ingestion of Arsenic Released from K-25

The following equation was used to calculate non-cancer screening indices for ingestion of arsenic:

$$\mathit{Non\text{-}cancer\ Screening\ Index} = \frac{\mathit{Average\ Daily\ Dose}\ (mg\ kg^{-1}\ d^{-1})}{\mathit{Reference\ Dose}\ (mg\ kg^{-1}\ d^{-1})}$$

At Level I, the non-cancer screening index for ingestion of arsenic released from K-25 is:

$$\mathit{Non\text{-}cancer\ Screening\ Index} = \frac{3.6\times10^{-2}\ mg\ kg^{-1}\ d^{-1}}{3.0\times10^{-4}\ mg\ kg^{-1}\ d^{-1}} = 120$$

Because the Level I non-cancer screening index for arsenic released from K-25 was greater than 1.0, the decision guide established by ORHASP for evaluating the need for further study of noncarcinogens, a Refined Level I screening was conducted.

At Refined Level I, the non-cancer screening index for ingestion of arsenic released from K-25 is:

$$\mathit{Noncancer\ Screening\ Index} = \frac{4.0\times10^{-3}\ mg\ kg^{-1}\ d^{-1}}{3.0\times10^{-4}\ mg\ kg^{-1}\ d^{-1}} = 13$$

5.1.5.3 Cancer Screening Indices Associated with Inhalation and Ingestion of Arsenic Released from Y-12

The following equation was used to calculate the cancer screening indices for inhalation and ingestion of arsenic:

$$\mathit{Cancer\ Screening\ Index} = \mathit{Lifetime\ Average\ Daily\ Dose}\ (mg\ kg^{-1}\ d^{-1}) \times \mathit{Slope\ Factor}\ (mg\ kg^{-1}\ d^{-1})^{-1}$$

At Level I, the cancer screening index for inhalation of arsenic released from Y-12 is:

$$\mathit{Cancer\ Screening\ Index} = 1.5{\times}10^{-6}\ (mg\ kg^{-1}\ d^{-1}) \times 15\ (mg\ kg^{-1}\ d^{-1})^{-1} = 2.2{\times}10^{-5}$$

and the cancer screening index for ingestion of arsenic released from Y-12 is:

$$\mathit{Cancer\ Screening\ Index} = 1.2{\times}10^{-2}\ (mg\ kg^{-1}\ d^{-1}) \times 1.5\ (mg\ kg^{-1}\ d^{-1})^{-1} = 1.8{\times}10^{-2}$$

The cancer screening indices for inhalation and ingestion of arsenic were then summed to give a Y-12 Level I cancer screening index for both routes of exposure ($1.8{\times}10^{-2}$). Because the Level I cancer screening index for arsenic exceeded $1{\times}10^{-4}$, the decision guide established by ORHASP for evaluating the need for further study of carcinogens, a Refined Level I screening was conducted.

At Refined Level I, the cancer screening index for inhalation of arsenic released from Y-12 is:

$$\mathit{Cancer\ Screening\ Index} = 8.2{\times}10^{-8}\ (mg\ kg^{-1}\ d^{-1}) \times 15\ (mg\ kg^{-1}\ d^{-1})^{-1} = 1.2{\times}10^{-6}$$

and the cancer screening index for ingestion of arsenic released from Y-12 is:

$$\mathit{Cancer\ Screening\ Index} = 1.7{\times}10^{-4}\ (mg\ kg^{-1}\ d^{-1}) \times 1.5\ (mg\ kg^{-1}\ d^{-1})^{-1} = 2.6{\times}10^{-4}$$

The cancer screening indices for inhalation and ingestion of arsenic were then summed to give a Y-12 Refined Level I cancer screening index for both routes of exposure ($2.6{\times}10^{-4}$).

5.1.5.4 Non-cancer Screening Indices Associated with Ingestion of Arsenic Released from Y-12

The following equation was used to calculate non-cancer screening indices for ingestion of arsenic:

$$\mathit{Noncancer\ Screening\ Index} = \frac{\mathit{Average\ Daily\ Dose}\ (mg\ kg^{-1}\ d^{-1})}{\mathit{Reference\ Dose}\ (mg\ kg^{-1}\ d^{-1})}$$

At Level I, the non-cancer screening index for ingestion of arsenic released from Y-12 is:

$$Noncancer\ Screening\ Index = \frac{1.2 \times 10^{-2}\ mg\ kg^{-1}\ d^{-1}}{3.0 \times 10^{-4}\ mg\ kg^{-1}\ d^{-1}} = 41$$

Because the Level I non-cancer screening index for arsenic released from Y-12 was greater than 1.0, the decision guide established by ORHASP for evaluating the need for further study of noncarcinogens, a Refined Level I screening was conducted.

At Refined Level I, the non-cancer screening index for ingestion of arsenic released from Y-12 is:

$$Noncancer\ Screening\ Index = \frac{1.2 \times 10^{-3}\ mg\ kg^{-1}\ d^{-1}}{3.0 \times 10^{-4}\ mg\ kg^{-1}\ d^{-1}} = 4.0$$

5.1.5.5 Comparison of Screening Indices to Decision Guides

The cancer and non-cancer screening indices calculated using the Level I and Refined Level I screening methodologies are presented in Table 5-6 for releases from K-25 and Y-12. These screening indices are compared to the risk-based decision guides established by ORHASP for screening of contaminants released from the ORR.

Table 5-6: Results of Level I and Refined Level I Screening of Arsenic

	Cancer Screening Index	**Exceeds Decision Guide?** [a]	**Non-cancer Screening Index**	**Exceeds Decision Guide?** [b]
K-25				
Level I	3.8×10^{-2}	Yes	120	Yes
Refined Level I	8.9×10^{-4}	Yes	13	Yes
Y-12				
Level I	1.3×10^{-2}	Yes	41	Yes
Refined Level I	2.6×10^{-4}	Yes	4.0	Yes

[a] For carcinogenic chemicals, a lifetime excess cancer risk of 10^{-4} was established by ORHASP as a decision guide for evaluating the need for further study (ChemRisk 1996)

[b] For noncarcinogenic chemicals, a hazard index of 1.0 was established by ORHASP as a decision guide for evaluating the need for further study (ChemRisk 1996).

5.1.6 Conclusions

In this screening analysis, exposures of off-site residents to arsenic in air and water released from the K-25 Power Station and from the Y-12 steam plant led to screening indices that are above the decision guides in use on the project. It should be noted that, even in Refined Level I screening, the analysis maintains considerable conservatism. In the Refined Level I methodology used in this assessment, somewhat different (less conservative) exposure parameters were used than in the Level I methodology; however, the environmental concentrations, biotransfer factors, and cancer potency slope factors remained the same. The biotransfer factors and the environmental concentrations each add considerable conservatism to the analysis, and the oral slope factor is based on a Taiwaneese study that has several weaknesses noted earlier. The NCRP Report 123 (NCRP 1996) biotransfer factors used in this analysis were developed for use in screening assessments, and are at the upper end of the range of biotransfer factors found in the literature. The environmental concentrations used in these analyses were also at the upper end of the range of values located by the project team.

5.2 Beryllium Releases from the Y-12 Site

In the Dose Reconstruction Feasibility Study, beryllium was evaluated as a potential material of concern for both the K-25 and Y-12 sites. The screening indices for beryllium as a carcinogen for the sites were 2×10^{-7} and 1×10^{-7}, respectively. In Phase II of the Oak Ridge Health Studies, the project team located new information on beryllium uses and releases that included stack monitoring data for the Y-12 plant. Although beryllium was not specifically called out as a material of concern for Phase II, this new information warranted a second look at potential off-site risks from airborne releases of beryllium.

5.2.1 Beryllium Use at Y-12

The use of beryllium began at Y-12 in the early 1950s. A 1952 article in the journal *American Machinist* (Case 1952) describes the "finish machining, deep hole drilling, and milling" of beryllium at Y-12. Additionally, beryllium was used for fuels and recycled through an arc melting process. In a memo dated January 29, 1959 (Ramsey 1959). The author, replying to a request from the Atomic Energy Commission, states that beryllium use as an oxide or fluoride for fuels was fairly limited until 1957, after which the beryllium use increased substantially. The use of beryllium in production in various chemical forms continued through the early 1990s.

5.2.2 Concentrations of Beryllium the Environment near the ORR

Beryllium may be present in the environment from both natural and human-related sources. The following sections describe typical natural, or background, levels of beryllium in the environment, and concentrations that have historically been measured in the environment near the ORR.

5.2.2.1 Background Levels

Beryllium is found naturally in soils, and may be present in air from a number of natural sources including soil particles, volcanoes, and forest fires. Several sources of data on background concentrations of beryllium were identified by the project team. These include general background concentrations and near-site background concentrations measured in areas of Eastern Tennessee assumed to not be impacted by ORR activities. Data on background concentrations of beryllium identified by the project team are summarized briefly by media below.

- *Air* - The average concentration of beryllium in air in the United States is 0.03 ng m^{-3} but the median concentration in cities is 0.2 ng m^{-3} (ATSDR 1997a). The project team did not locate background measurements for beryllium in air in the East Tennessee region.

- *Surface Water* - Beryllium concentrations in 15 major US river basins ranged from 0.01 to 1.22 µg L^{-1} , with a mean of 0.19 µg L^{-1} (IARC 1980). Beryllium was not detected in any of the surface water samples collected in Norris Reservoir during the

Clinch River Remedial Investigation, in which the detection limit was 19 µg L^{-1} (Cook et al. 1992). Norris Reservoir was identified in the CRRI as the "reference location."

- *Soil/Sediment* - Background concentrations of beryllium in Tennessee soils are reported to range from less than the detection limit to 2.0 mg kg^{-1} (Dragun and Chiasson 1991). Background concentrations up to 15 mg kg^{-1} are reported for other areas of the United States (Dragun and Chiasson 1991).

  In 1973 and 1982, the Tennessee Valley Authority (TVA) measured concentrations of several metals, including beryllium, in sediments just upstream of a number of TVA dams. Reservoirs evaluated included several on the Tennessee River upstream of the ORR including Fort Loudoun Reservoir, and several on different river systems including Douglas Reservoir on the French Broad River and Wilson Reservoir on Fleet Hollow Embayment. Beryllium concentrations in sediments of reservoirs assumed not to be impacted by ORR activities ranged from <0.7 to 1.9 mg kg^{-1} (dry weight) (TVA 1986).

- *Food* - Beryllium has been detected in milk at concentrations of 0.02 ppm in ash (HSDB 1999). Concentrations have also been measured in plants: concentrations in potatoes, tomatoes, and head lettuce were 0.17 mg kg^{-1}, 0.24 mg kg^{-1}, and 0.33 mg kg^{-1}, respectively (HSDB 1999). No data were identified on beryllium concentrations in fish.

5.2.2.2 Environmental Concentrations on or near the ORR

Available data describing beryllium concentrations in air, soil, sediment, surface water, and other environmental media on or near the ORR are discussed in the following sections. In addition, derivation of exposure point concentrations used in the screening assessment is described.

5.2.2.2.1 Beryllium Concentrations in Air

Plant records indicate that some form of beryllium stack and ambient air monitoring was occurring as early as 1952 (Case 1952; UCCND 1954-1959). However, no early beryllium stack data were located. In 1980, at the request of DOE (Marciante 1980), Y-12 plant personnel sampled beryllium effluent points during peak processing periods (Wing 1980). Average release rates determined from these analyses are presented in Table 5-7.

Table 5-7: Measured Beryllium Release Rates for Y-12 Buildings

Building	Stack No.	Release Rate (Fg d^{-1})	Release Rate (g s^{-1})
9201-5	3	1600	1.9×10^{-8}
9201-5	4	690	8.0×10^{-9}
9201-5	6	660	7.7×10^{-9}
9201-5	7	56	6.5×10^{-10}
9201-5	11	410	4.8×10^{-9}
9201-5	12	3100	3.6×10^{-8}
9201-5	16	1300	1.5×10^{-8}
9202	1	230	2.7×10^{-9}
9202	2	580	6.7×10^{-9}
9202	3	1500	1.7×10^{-8}
9202	4	930	1.1×10^{-8}
9202	5	1900	2.2×10^{-8}
9202	6	190	2.2×10^{-9}
9995	--	460	5.3×10^{-9}

Off-Site Air Concentration Used in the Screening Assessment

The stack release rates from Table 5-7 were converted to g s^{-1}, summed, and multiplied by the empirically derived relative concentration at Scarboro (3×10^{-7} s m^{-3}). This dispersion factor was determined in the Task 6 study of uranium releases (ChemRisk 1997). The resulting beryllium concentration at Scarboro was 4.7×10^{-8} Fg m^{-3}.

5.2.2.2.2 Beryllium Concentrations in Surface Water

Although there is evidence that liquid effluents from the Y-12 plant to EFPC may have been monitored for beryllium as early as the 1970s, associated data were not located by the project team. Results from routine analyses of beryllium in EFPC are reported in the annual environmental monitoring reports beginning in 1987. Statistics presented in the annual reports include minimum, maximum, and mean of sample concentrations, along with the number of samples collected. Maximum concentrations flowing into New Hope Pond between 1987 and 1991 ranged from <0.1 Fg L^{-1} to 1Fg L^{-1}.

In addition to this routine monitoring program, beryllium concentrations were also measured in surface water during several special monitoring programs, including:

- *The Tennessee Valley Authority (TVA) Instream Contaminant Study (1984)–* Although the focus of this program was evaluating the transport of mercury in surface water and sediments downstream of Y-12, samples were also analyzed for beryllium. Surface water sample locations included one In EFPC at the outfall of New Hope Pond, one in Poplar Creek upstream of K-25, and two in the Clinch River both above and below the Poplar Creek/ Clinch River confluence (TVA 1985a). Beryllium was not detected in any of the samples, while the detection limit was 1 Fg L^{-1}. Data from individual samples are presented in the TVA reports.

- *The East Fork Poplar Creek Floodplain Remedial Investigation (EFPC RI) (1990 and 1991)–* Surface water samples were collected at several locations in and along EFPC. Beryllium concentrations ranged from <1 Fg L^{-1} to 2 Fg L^{-1}. Data for individual samples are presented in the LMES OREIS database (LMES OREIS 1997).

Off-Site Surface Water Concentration Used in the Screening Assessment

The maximum measured water concentration, 2.0 Fg L^{-1} from the EFPC RI, was used in this analysis.

5.2.2.2.3 Beryllium Concentrations in Soil/Sediment

Results of routine historical measurements of beryllium concentrations in soil and sediment near the Y-12 Plant were not located by the project team. Beryllium concentrations in soil and sediments were, however, measured as part of several special monitoring programs. These programs include:

- *The Clinch River Remedial Investigation (1989 and 1990)–* Sediment samples were collected from 10 different areas of the Clinch River and Poplar Creek. Beryllium concentrations ranged from 0.1 to 2.1 mg kg^{-1}. Data for individual samples are presented in the LMES OREIS database (LMES OREIS 1997).

- *The East Fork Poplar Creek Floodplain Remedial Investigation (EFPC RI) 1990 and 1991)–* Sediment and floodplain soil samples were collected at several location in and along EFPC. Sediment concentrations ranged from 0.47 to 2.7 mg kg^{-1}. Soil concentrations ranged from 0.24 to mg kg^{-1} . Data for individual samples are presented in the LMES OREIS database (LMES OREIS 1997).

Off-Site Soil/Sediment Concentration Used in the Screening Assessment

The maximum beryllium soil concentration, 2.7 mg kg^{-1} from the EFPC RI, was used by the project team in the screening assessment for the soil/sediment exposure pathways.

5.2.2.2.4 Beryllium Concentrations in Fish

Beryllium concentrations in fish in waterways near the ORR were measured during the TVA Instream Contaminant Study and the Clinch River Remedial Investigation. Identified data describing beryllium concentrations in fish include the following:

- *The Tennessee Valley Authority (TVA) Instream Contaminant Study (1984)–* Fish were collected from several locations in the Clinch River and Poplar Creek, including CRM 2, 6, and 11 and PCM 0.2 (TVA 1985d). Data for individual samples are presented in the TVA reports. The maximum reported beryllium concentration was 50 μg kg^{-1}.

- *A TVA fish tissue screening study of Eastern Tennessee reservoirs (1987)–* Fish were collected and analyzed for a variety of contaminants, including beryllium, to assess the general level of contamination in Tennessee reservoirs (TVA 1989). Sample locations included CRM 20, in Watts Bar Reservoir, and CRM 24, in Melton Hill Reservoir. Beryllium was not detected in any of the Clinch River samples, while the detection limit was 0.02 mg kg^{-1}. Data from individual samples are presented in the TVA report.

- *The Clinch River Remedial Investigation (1989 and 1990)–* Fish were collected at a number of locations along the Clinch River and Poplar Creek. The maximum reported beryllium concentration in Poplar Creek was 16 μg kg^{-1}. The maximum reported beryllium concentration in the Clinch River was 6 μg kg^{-1}.

Off-Site Surface Water Concentration Used in the Screening Assessment

For the purposes of this screening analysis, concentrations of beryllium in fish were calculated using a measured water concentration and a bioconcentration factor (BCF) that describes the transfer of beryllium from water to fish. The BCF used in this assessment was 100 (NCRP 1996). Use of this BCF and the water concentration described in Section 5.2.2.2.2 (2.0 μg L^{-1}) yields a fish concentration of 200 μg kg^{-1}.

5.2.2.2.5 Beryllium Concentrations in Other Food Items

No measurements of beryllium in meat, milk, or vegetables were identified by the project team. The screening exposure model predicts meat, milk, and vegetable beryllium concentrations using biotransfer factors and air and soil beryllium concentrations. The biotransfer factors used in this assessment are listed in Table 5-8.

Table 5-8: Biotransfer and Bioconcentration Factors for Beryllium

Parameter	Value	Source
Soil to Vegetables (B_{veg})	0.004 (mg kg^{-1} wet)/(mg kg^{-1} dry)	NCRP 1996
Soil to Pasture ($B_{pasture}$)	0.1 (mg kg^{-1} dry)/(mg kg^{-1} dry)	NCRP 1996
Biotransfer to Milk (F_m)	2.0×10^{-6} d L^{-1}	NCRP 1996
Biotransfer to Meat (F_f)	0.005 d kg^{-1}	NCRP 1996
Bioconcentration in Fish (BCF)	100 (mg kg^{-1})/(mg L^{-1})	NCRP 1996

5.2.3 Toxicity Assessment

Exposure to airborne beryllium has been shown to produce adverse respiratory effects in both humans and animals. Based on evidence from animal studies, the USEPA considers beryllium to be a probable human carcinogen. A summary of what is known about the toxicity of beryllium is provided in this section.

Acute nonspecific chemical pneumonitis has resulted from acute exposures to soluble beryllium salts (ATSDR 1997a). Exposures to beryllium-containing dusts and high-fired beryllium oxides (less soluble forms of beryllium) have produced chronic lung disease (ATSDR 1997a), characterized by formation of non-cancerous granulomas in the lung. Both the chronic form of beryllium disease (berylliosis) and chemical pneumonitis have been produced from exposures ranging from less than 2.0 to 1000 Fg m^{-3} (ATSDR 1997a). Human epidemiologic studies on beryllium workers have largely been criticized for their lack of control on confounding factors such as smoking (USEPA 1997). However, beryllium has been shown to induce lung cancer via inhalation in rats and monkeys. Based on evidence from animal studies, the USEPA considers beryllium to be a probable human carcinogen (Group 2B) and has set an inhalation slope factor for evaluating cancer risk at 8.4 (mg kg^{-1} d^{-1})$^{-1}$ (USEPA 1999).

Based on rabbit studies in which intravenous injection of beryllium induced bone cancer, the USEPA also considers beryllium to be a carcinogen through ingestion. The USEPA (1999) has set an oral slope factor for evaluation of cancer risk at 4.3 (mg kg^{-1} d^{-1})$^{-1}$.

The USEPA (1999) has also established an oral RfD for beryllium of 2.0×10^{-3} mg kg^{-1} d^{-1}. The RfD is based on a drinking water study with rats in which the No Observed Adverse Effect Level (NOAEL) was determined to be 0.54 mg kg^{-1}. An uncertainty factor of 100 was added to reflect a factor of 10 each for interspecies conversion (rat6human) and for the protection of sensitive human subpopulations.

Toxicity criteria used in the screening analyses to evaluate exposures to beryllium at Oak Ridge are presented in Table 5-9.

Table 5-9: Toxicity Criteria for Beryllium Used in the Screening Analyses

Exposure Route	Toxicity Criteria/Source	Value
Inhalation (Cancer Risk)	USEPA Inhalation Slope Factor	8.4 $(mg\ kg^{-1}\ d^{-1})^{-1}$
Inhalation (Non-cancer)	USEPA Inhalation Reference Dose	5.7×10^{-6} $mg\ kg^{-1}\ d^{-1}$
Ingestion (Cancer Risk)	USEPA Oral Slope Factor	4.3 $(mg\ kg^{-1}\ d^{-1})^{-1}$
Ingestion (Non-cancer)	USEPA Oral Reference Dose	2.0×10^{-3} $mg\ kg^{-1}\ d^{-1}$

5.2.4 Exposure Assessment

Following the Task 7 Screening methodology for Level I and Level II screening (ChemRisk 1996), doses were calculated for exposure to beryllium in air, soil/sediment, and surface water. Exposures to beryllium through ingestion of fish, vegetables, meat, and milk were evaluated using equations (presented in Appendix B) and biotransfer factors (discussed in Section 5.2.2.2.4) that describe the uptake of beryllium from air, surface water, and soil/sediment into these media. A summary of the concentrations for each medium that were used in the screening is presented in Table 5-10.

Doses were calculated separately to support evaluation of potential excess cancer risks and non-cancer effects from inhalation and ingestion of beryllium. The total average daily doses of beryllium calculated for the inhalation and ingestion (oral) exposure routes are summarized in Table 5-11. The doses calculated for individual pathways and the contribution of each pathway to the total dose are summarized in Appendix I.

Table 5-10: Summary of Environmental Concentrations of Beryllium used in Screening

Environmental Medium	Concentration	Source of Value	Typical Background[1]
Air	4.7×10^{-8} µg m^{-3}	Calculated based on release rates and χ/Q value	U.S. average 3.0×10^{-5} µg m^{-3}
Soil/Sediment	2.7 mg kg^{-1}	Maximum sediment concentration measured in EFPC	Less than detection limit to 2.0 mg kg^{-1}
Surface Water	0.002 mg L^{-1}	Maximum concentration measured in EFPC	1.0×10^{-5} to 1.2×10^{-3} mg L^{-1}; mean 1.9×10-4 mg L^{-1}
Fish	0.2 mg kg^{-1}	Calculated based on the water concentration shown above and a BCF of 100.	Not available

[1] For references, see Section 5.2.2.1.

Table 5-11: Beryllium Doses Calculated in the Level I and Refined Level I Screen (mg kg^{-1} d^{-1})

	Lifetime Average Daily Inhalation Dose- (Carcinogen)	Lifetime Average Daily Oral Dose- (Carcinogen)	Average Daily Inhalation Dose- (Noncarcinogen)	Average Daily Oral Dose- (Noncarcinogen)
Level I	1.4×10^{-8}	9.4×10^{-5}	2.0×10^{-8}	1.3×10^{-4}
Refined Level I	8.2×10^{-10}	2.9×10^{-6}	5.8×10^{-9}	2.1×10^{-5}

5.2.5 Risk Characterization

Cancer and non-cancer screening indices were calculated based on the screening estimates of lifetime average daily dose and average daily dose, respectively, for beryllium released from Y-12. The results of the screening risk characterization are presented below, and compared to risk-based decision guides established by the Oak Ridge Health Agreement Steering Panel (ORHASP).

5.2.5.1 Cancer Screening Indices Associated with Inhalation and Ingestion of Beryllium

The following equation was used to calculate the cancer screening indices for inhalation and ingestion of beryllium:

$$Cancer\ Screening\ Index = Lifetime\ Average\ Daily\ Dose\ (mg\ kg^{-1}\ d^{-1}) \times Slope\ Factor\ (mg\ kg^{-1}\ d^{-1})^{-1}$$

At Level I, the cancer screening index for inhalation of beryllium released from Y-12 is:

$$Cancer\ Screening\ Index = 1.4\times10^{-8}\ (mg\ kg^{-1}\ d^{-1}) \times 8.4\ (mg\ kg^{-1}\ d^{-1})^{-1} = 1.2\times10^{-7}$$

and the cancer screening index for ingestion of beryllium released from Y-12 is:

$$Cancer\ Screening\ Index = 9.4\times10^{-5}\ (mg\ kg^{-1}\ d^{-1}) \times 4.3\ (mg\ kg^{-1}\ d^{-1})^{-1} = 4.0\times10^{-4}$$

The cancer screening indices for inhalation and ingestion of beryllium were then summed to give a Level I cancer screening index for both routes of exposure (4.0×10^{-4}). Because the Level I cancer screening index for beryllium exceeded 1×10^{-4}, the decision guide established by ORHASP for evaluating the need for further study of carcinogens, a Refined Level I screening was conducted.

At Refined Level I, the cancer screening index for inhalation of beryllium released from Y-12 is:

$$Cancer\ Screening\ Index = 8.2\times10^{-10}\ (mg\ kg^{-1}\ d^{-1}) \times 8.4\ (mg\ kg^{-1}\ d^{-1})^{-1} = 6.9\times10^{-9}$$

and the cancer screening index for ingestion of beryllium released from Y-12 is:

$$Cancer\ Screening\ Index = 2.9\times10^{-6}\ (mg\ kg^{-1}\ d^{-1}) \times 4.3\ (mg\ kg^{-1}\ d^{-1})^{-1} = 1.3\times10^{-5}$$

The cancer screening indices for inhalation and ingestion of beryllium were then summed to give a Refined Level I cancer screening index for both routes of exposure (1.3×10^{-5}).

5.2.5.2 Non-cancer Screening Indices Associated with Inhalation and Ingestion of Beryllium

The following equation was used to calculate non-cancer screening indices for ingestion of beryllium:

$$Noncancer\ Screening\ Index = \frac{Average\ Daily\ Dose\ (mg\ kg^{-1}\ d^{-1})}{Reference\ Dose\ (mg\ kg^{-1}\ d^{-1})}$$

At Level I, the non-cancer screening index for ingestion of beryllium released from Y-12 is:

$$Noncancer\ Screening\ Index = \frac{2.0\times10^{-8}\ mg\ kg^{-1}\ d^{-1}}{5.7\times10^{-6}\ mg\ kg^{-1}\ d^{-1}} = 0.0035$$

and the non-cancer screening index for ingestion of beryllium is:

$$Noncancer\ Screening\ Index = \frac{1.3\times10^{-4}\ mg\ kg^{-1}\ d^{-1}}{2.0\times10^{-3}\ mg\ kg^{-1}\ d^{-1}} = 0.065$$

The non-cancer screening indices for inhalation and ingestion of beryllium were then summed to give a Level I non-cancer screening index for both routes of exposure (0.074). Because the Level I non-cancer screening index for beryllium released from Y-12 was less than 1.0, the decision guide established by ORHASP for evaluating the need for further study of noncarcinogens, a Refined Level I screening was not conducted.

5.2.5.3 Comparison of Screening Indices to Decision Guides

The cancer and non-cancer screening indices for beryllium calculated using the Level I and Refined Level I screening methodologies are presented in Table 5-12. These screening indices are compared to the risk-based decision guides established by ORHASP for screening of contaminants released from the ORR.

Table 5-12: Results of Level I and Refined Level I Screening of Beryllium

	Cancer Screening Index	Exceeds Decision Guide? [a]	Non-cancer Screening Index	Exceeds Decision Guide? [b]
Level I	4.0×10^{-4}	Yes	0.065	No
Refined Level I	1.3×10^{-5}	No	Not necessary	Not necessary

a For carcinogenic chemicals, a lifetime excess cancer risk of 10^{-4} was established by ORHASP as a decision guide for evaluating the need for further study (ChemRisk 1996).

b For noncarcinogenic chemicals, a hazard index of 1.0 was established by ORHASP as a decision guide for evaluating the need for further study (ChemRisk 1996).

It is important to note that the Task 7 screening methodology does not include a factor for bioavailability of beryllium when ingested. Beryllium is poorly absorbed through the gastrointestinal tract, and it is likely that the Level I ingestion cancer screening index would also fall below the Task 7 decision guide if bioavailability were considered.

5.2.6 Conclusions

Based on this analysis, exposure of off-site residents to beryllium in air, soil, and water due to releases from the Y-12 Plant does not appear to warrant high priority for detailed investigation in the Oak Ridge Health Studies. It is important to note that this analysis is limited by the lack of beryllium concentration data in the soil and water. In the future, if such information becomes available, this analysis could be refined.

5.3 Copper Releases from the K-25 Site

In the Dose Reconstruction Feasibility Study, copper did not undergo a quantitative screening because no information was located indicating that copper had been released in significant quantities from the ORR. However, environmental data gathered in Phase II of the Oak Ridge Health Studies indicate that copper may have been released to the environment from the K-25 Site. Because of these findings, copper was identified as a material that warranted a quantitative screening evaluation.

5.3.1 Copper Use at K-25

Copper powder was used at the K-25 Site from the late 1940s to around 1981. At this time, it is not possible to publicly identify the locations of use of the copper powder, the manner of its use, or quantities that were used. However, the inability to release information in these areas does not interfere with the ability of the project team to perform a screening evaluation of the health significance of copper releases.

5.3.2 Concentrations of Copper in the Environment near the ORR

Copper may be present in the environment from both natural and human-related sources. The following sections describe typical natural, or background, levels of copper in the environment, and concentrations that have historically been measured in the environment near the ORR.

5.3.2.1 Background Levels

Copper is found naturally in soils and may be present in air from a number of natural sources, including windblown dust and volcanic eruptions (ATSDR 1996). Several sources of data on background concentrations of copper were identified by the project team. These include general background concentrations and near-site background concentrations measured in areas of Eastern Tennessee assumed to not be impacted by ORR activities.

Data on background concentrations of copper identified by the project team are summarized briefly by media below.

- *Air* - Background concentrations of copper in air in rural areas are reported to range from 0.003 to 0.28 µg m^{-3} (ATSDR 1996). In urban areas, background concentrations are reported to range from 0.003 to 5.1 µg m^{-3} (ATSDR 1996). In general, concentrations of copper in air are higher in industrialized areas.

- *Surface Water* - Typical background concentrations of copper in surface water range from 0.5 to 1000 µg L^{-1}, with a median of 10 µg L^{-1} (ATSDR 1996). Copper was not detected in any of the surface water samples collected in Norris Reservoir during the Clinch River Remedial Investigation (with a detection limit of 4.4 µg L^{-1}; USDOE 1996). Norris Reservoir was identified in the CRRI as the “reference location,” indicating that it was considered unaffected by releases from the ORR.

- *Soil/Sediment* - Background concentrations of copper in Tennessee soils are reported to range from 10 to 50 mg kg^{-1}, with a mean of 22 mg kg^{-1} (Dragun and Chiasson 1991). In 1973 and 1982, the Tennessee Valley Authority (TVA) measured concentrations of several metals, including copper, in sediments just upstream of a number of TVA dams. Reservoirs evaluated included several on the Tennessee River upstream of the ORR (including Fort Loudoun Reservoir) and several on different river systems (including Douglas Reservoir on the French Broad River and Wilson Reservoir on Fleet Hollow Embayment). Copper concentrations in sediments of reservoirs assumed not to be impacted by ORR activities ranged from 10 to 63 mg kg^{-1} (TVA 1986).

- *Fish* - In the CRRI (USDOE 1996), fish tissue samples collected from Norris Reservoir had copper concentrations ranging from less than the limit of detection to 0.61 mg kg^{-1}. In 1987 TVA measured concentrations of several metals, including copper, in fish from Norris Reservoir. Copper concentrations in fish flesh ranged from less than the limit of detection (0.2 mg kg^{-1}) to 0.4 mg kg^{-1}.

5.3.2.2 Environmental Concentrations on or near the ORR

The earliest environmental samples for copper identified by the project team were collected in 1973, when routine monitoring for copper in air was initiated. Available data describing copper concentrations in air, surface water, soil, sediment, and other environmental media on or near the ORR are discussed in the following sections. In addition, determination of exposure point concentrations used in the screening assessment is described.

5.3.2.2.1 Copper Concentrations in Air

No stack monitoring data for copper were located by the project team. However, from 1973 to 1980, K-25 personnel collected air samples from four locations on the K-25 Site. Individual sample results were tabulated by plant staff (ORGDP 1981b). Results for 1976 through 1979 were obtained and statistically evaluated by the project team. One of the samplers consistently indicated higher average concentrations than the others during the period examined. Measured copper concentrations (weekly averages) from that sampler in 1978 (the year within the data set with the highest average concentration) ranged from 0.035 to 7.7 µg m^{-3}, with an average of 0.75 µg m^{-3} and a standard deviation of 1.2.

Off-site Air Concentration Used in the Screening Assessment

The reference location for screening of airborne copper releases from the K-25 Site was Union/Lawnville (Section 2.3.4.2). Concentrations at Union/Lawnville were estimated based on air concentrations measured at the location of the most affected on-site sampler described above. It was assumed that the relationship between the copper concentrations measured at the sampler and concentrations at Union/Lawnville was the same as the relationship between the air concentration modeled to the sampler from the center of the K-25 Site (based on a unit release of 1 g s^{-1}) and the air concentration modeled to Union/Lawnville from the same release rate. From the Task 6 assessment, the modeled air concentration at the sampler from a unit release from the center of the K-25 Site was 1.1×10^{-6} s m^{-3} (ChemRisk 1997). The corresponding modeled air concentration at Union/Lawnville was 7.4×10^{-7} s m^{-3}. Based on these two values, the ratio of the modeled air concentration at Union/Lawnville to the modeled on-site concentration was as follows:

$$Ratio = \frac{modeled\ off\text{-}site\ (Union/Lawnville)}{modeled\ on\text{-}site\ (near\ the\ sampler\ with\ highest\ results)} = \frac{7.4\times10^{-7}\ s\ m^{-3}}{1.1\times10^{-6}\ s\ m^{-3}} = 0.67$$

The concentration of airborne copper at Union/Lawnville was calculated by multiplying the above ratio by the 95% upper confidence limit (UCL; 1.1 µg m^{-3}) of the airborne copper concentration measured at the sampler in 1978, the year when the highest average concentration was measured (0.75 µg m^{-3}):

$$Union/Lawnville\ air\ concentration = 1.1\ \mu g\ m^{-3} \times 0.67 = 0.74\ \mu g\ m^{-3}$$

This calculated concentration was used in the screening calculations to evaluate potential off-site exposures via air pathways.

5.3.2.2.2 Copper Concentrations in Surface Water

No measurements of copper concentrations in liquid effluent from the K-25 Site were identified by the project team. Copper concentrations were measured in surface water during two special monitoring programs. These programs were:

- *The Tennessee Valley Authority (TVA) Instream Contaminant Study (1984)*— Although the focus of this program was evaluating the transport of mercury in surface water and sediment downstream of Y-12, samples were also analyzed for copper. Surface water sample locations included two in the Clinch River both above and below the Poplar Creek confluence (TVA 1985a). Copper concentrations were all less than the detection limit of 5 µg L^{-1}. Data from individual samples are presented in the TVA reports.

- *The Clinch River Remedial Investigation (1989 and 1990)*— This program was initiated by DOE in 1989 to address the transport, fate, and distribution of waterborne contaminants released from the ORR to the Clinch and Tennessee Rivers (USDOE 1996). Surface water and sediment samples were collected from 10 reaches, six of which are potentially affected by releases from the ORR, and four of which serve as reference or background areas. Sample locations included Poplar Creek and the

Clinch River. The maximum copper concentration detected in the Clinch River was 83 µg L^{-1}.

Off-Site Surface Water Concentration Used in the Screening Assessment

The highest surface water concentration reported for the Clinch River was used by the project team in the screening assessment for surface water pathways. Data collected in the Clinch River are assumed to reflect concentrations to which off-site individuals may have been exposed.

The highest reported surface water concentration was measured as part of the Clinch River Remedial Investigation. The value, measured just downstream of the K-25 Site, was 83 µg L^{-1}.

5.3.2.2.3 Copper Concentrations in Soil/Sediment

Surface sediment samples were collected semiannually at several locations beginning in the mid-1970s, including K-25 Site holding ponds, Poplar Creek, and the Clinch River. Samples were collected semiannually at up to 12 sites on Poplar Creek, both above and below K-25 discharge points, and up to two sites on the Clinch River. Summary data (annual averages) from this program are presented in the annual environmental monitoring reports (UCC 1976, 1977, 1978, 1979, 1980, 1981, 1982, 1983). Average sediment concentrations in the Clinch River between 1975 and 1982 ranged from 11 to 65 mg kg^{-1}. The maximum Clinch River concentration was measured in 1979 just downstream of the mouth of Poplar Creek.

In addition to the routine monitoring program, copper concentrations were also measured in sediments/soils as part of several special monitoring programs. These programs include:

- *A survey of sediments in streams surrounding the K-25 Site (1985)*— Surface sediment samples were collected by K-25 staff at 180 locations in the Clinch River, Poplar Creek, EFPC, and tributaries to Poplar Creek, to identify locations where contaminants were entering surface water systems (Ashwood et al. 1986). Copper concentrations were measured in some of these samples, including four locations in the Clinch River both upstream and downstream of the mouth of Poplar Creek. Data for individual samples are presented in the Ashwood et al. report. Copper concentrations in the Clinch River ranged from 1 to 38 mg kg^{-1}.

- *The Clinch River Remedial Investigation (1989 and 1990)*— In addition to surface water, sediment samples were collected from the 10 reaches (USDOE 1996). Sample locations included Poplar Creek and the Clinch River. The maximum sediment copper concentration in the Clinch River was 54 mg kg^{-1}.

Off-Site Sediment Concentration Used in the Screening Assessment

The highest mean sediment copper concentration reported for the Clinch River was used by the project team in the screening analyses for soil/sediment exposure pathways. The highest reported copper concentrations were measured as part of the routine sediment sampling presented in the annual environmental reports. The highest mean, from sampling site CS-1 just downstream of the mouth of Poplar Creek in 1976, was 65 mg kg^{-1}. This concentration exceeded concentrations reported by other studies.

5.3.2.2.4 Copper Concentrations in Food Items

Copper concentrations in fish in waterways near the ORR were measured during several of the same programs in which surface water and sediment/soil samples were collected. The earliest data on copper concentrations in fish identified by the project team were collected by the TVA in 1984. Identified data describing copper concentrations in fish include the following:

- *The Tennessee Valley Authority (TVA) Instream Contaminant Study (1984)*— Fish were collected in several locations in the Clinch River and Poplar Creek, including CRM 2,6, and 11 and PCM 0.2 (TVA 1985d). Data for individual samples are presented in the TVA reports. The maximum reported copper concentration was 1.4 mg kg^{-1}.

- *A fish tissue screening study of Eastern Tennessee reservoirs (1987)*— Fish were collected and analyzed for a variety of contaminants, including copper, to assess the general level of contaminants in Tennessee reservoirs (TVA 1989). Sample locations included CRM 20, in Watts Bar Reservoir, and CRM 24, in Melton Hill Reservoir. Copper was not detected in any of the Clinch River samples (detection limit 0.2 mg kg^{-1}). Data from individual samples are presented in the TVA report.

- *The Clinch River Remedial Investigation (1989, 1990, and 1994)*— Fish were collected at a number of locations including the Clinch River just downstream from K-25. The maximum concentration in the fish tissue was 2.8 mg kg^{-1}.

Off-Site Fish, Vegetation, Meat, and Milk Concentrations Used in the Screening Assessment

For the purposes of the screening analyses, concentrations of copper in vegetation, meat, and milk were calculated using biotransfer factors that describe the transfer of copper from other media, including air, water, and soil. The biotransfer factors used in this assessment are listed in Table 5-13. Concentrations of copper in fish were calculated using a bioconcentration factor (BCF), shown in Table 5-13, that describes the transfer of copper from water to fish.

Table 5-13: Biotransfer and Bioconcentration Factors for Copper

Parameter	Value	Source
Soil to Vegetables (B_{veg})	0.05 (mg kg^{-1} wet)/(mg kg^{-1} dry)	NCRP 1996
Soil to Pasture ($B_{pasture}$)	0.8 mg kg^{-1} dry)/(mg kg^{-1} dry)	NCRP 1996
Biotransfer to Milk (F_m)	0.002 d L^{-1}	NCRP 1996
Biotransfer to Meat (F_f)	0.01 d kg^{-1}	NCRP 1996
Bioconcentration in Fish (BCF)	100 (mg kg^{-1})/(mg L^{-1})	ATSDR 1996

5.3.3 Toxicity Assessment

The following sections describe data characterizing the toxicity of copper through different routes of exposure, and summarize the toxicity criteria used to evaluate exposure to copper in the screening procedure.

5.3.3.1 Inhalation Exposure

Occupational exposure studies have shown that health effects associated with exposure to copper dust or fumes in air include irritation of the upper respiratory tract, metallic or sweet taste, nausea, and metal fume fever (a 24 - 48 hour illness characterized by chills, fever, aching muscles, dryness in the mouth and throat, and headache) (ACGIH 1996, ATSDR 1996). No animal or human data on the carcinogenicity of copper following inhalation are available (ATSDR 1996).

Based on metal fume fever effects and respiratory irritation, the National Institute for Occupational Safety and Health (NIOSH) and the American Conference of Governmental Industrial Hygienists (ACGIH) set occupational exposure limits (TLVs® - Threshold Limit Values) for an 8 hour period at 1 mg m^{-3} (ATSDR 1996). The USEPA has not developed a reference dose (RfD) for copper in air. In order to evaluate copper in air for this screening assessment, an inhalation RfD was derived from the occupational exposure limit of 1 mg m^{-3}. The derived RfD was calculated by converting the TLV® air concentration, which is based on an 8 hour exposure, to an air concentration for a continuous exposure. This was accomplished using the following equation:

$$CEAC = TLV \times IRCF \times EFCF$$

Where:

CEAC = Continuous exposure air concentration;
TLV = Threshold Limit Value (TLV®) (equal to 1 mg m^{-3} for copper);

IRCF = Inhalation rate conversion factor for converting an 8-hour occupational exposure to continuous exposure (equal to 0.5; 10 m^3/20 m^3); and,
EFCF = Exposure frequency conversion factor for converting occupational exposure to continuous exposure (equal to 0.54 [(250 d/365 d) × (40 y/50 y)]).

Solving the equation using the copper TLV®:

$$CEAC = 1\ mg\ m^{-3} \times 0.5 \times 0.54 = 0.27\ mg\ m^{-3}$$

The derived RfD is then calculated using the equation:

$$RfD_{der} = \frac{CEAC \times BR}{BW \times SF}$$

Where:

RfD_{der} = derived RfD;
BR = Breathing rate, 20 $m^3\ d^{-1}$;
BW = Body weight, 70 kg; and
SF = Safety factor for sensitive subpopulations, 10.

Solving for the RfD_{dv}:

$$RfD_{der} = \frac{0.27\ mg\ m^{-3} \times 20\ m^3\ d^{-1}}{70\ kg \times 10} = 0.0077\ mg\ kg^{-1}\ d^{-1}$$

This RfD_{der} was used in the screening analysis.

5.3.3.2 Oral (Ingestion) Exposure

Chronic ingestion studies in which copper-containing compounds were administered to rats or mice through the diet at concentrations up to 1,000 mg kg^{-1} did not produce evidence of a carcinogenic effect (ATSDR 1996; IRIS 1998). No human data on the carcinogenicity of copper following ingestion are available (ATSDR 1996).

Effects on the gastrointestinal system have been shown to be a target for chronic ingestion of copper. Humans consuming high levels of copper in tap water for about 1.5 years reported nausea, vomiting, and abdominal pain. The measured water concentration was 7.8 mg L^{-1} (ATSDR 1996).

The USEPA Maximum Contaminant Level Guide (MCLG) for copper in water of 1.3 mg L^{-1} was used as the basis for the USEPA's 1992 non-carcinogenic reference dose (RfD) for ingestion of copper. The MCLG is based on a NOAEL of 5.3 mg d^{-1} observed in mice. An uncertainty factor of of 2, a water consumption rate of 2 L d^{-1}, and a body weight of 70 kg were applied to yield an RfD of 0.037 mg kg^{-1} d^{-1} (USEPA 1992). In 1999 documentation (USEPA 1999), the USEPA no longer provides an RfD for ingestion of copper. As such, this analysis used the RfD published in 1992.

5.3.3.3 Toxicity Criteria Used in the Screening Assessment

Toxicity criteria used in the screening analyses to evaluate exposures to copper at Oak Ridge are presented in Table 5-14.

Table 5-14: Toxicity Criteria for Copper Used in the Screening Analyses

Exposure Route	Toxicity Criteria/Source	Value
Inhalation (Non-cancer)	Derived from TLV®	7.7×10^{-3} mg kg^{-1} d^{-1}
Ingestion (Non-cancer)	USEPA Oral Reference Dose (USEPA 1992)	3.7×10^{-2} mg kg^{-1} d^{-1}

5.3.4 Exposure Assessment

Following the Task 7 Screening methodology for Level I and Level II screening (ChemRisk 1996), doses were calculated for exposure to copper in air, surface water, and soil/sediment. Exposures to copper through ingestion of fish, vegetables, meat, and milk were evaluated using equations (presented in Appendix B) and biotransfer factors (discussed in Section 5.3.2.2.4) that describe the uptake of copper from air, surface water, and soil/sediment into these media. A summary of the concentrations for each medium that were used in the screening is presented in Table 5-15.

The total average daily doses of copper calculated for the inhalation and ingestion (oral) exposure routes for releases from K-25 are summarized in Table 5-16. The doses calculated for individual pathways and the contribution of each pathway to the total dose are summarized in Appendix I.

Table 5-15: Summary of Environmental Concentrations of Copper used in Screening

Environmental Medium	Concentration	Source of Value	Typical Background[1]
Air	0.00074 mg m^{-3} (0.74 µg m^{-3})	Calculated from 95% UCL value for year of highest measured concentrations in on-site air samplers (1978) and air dispersion modeling to Union/Lawnville.	0.003 to 0.28 µg m^{-3} rural, 0.003 to 5.1 µg m^{-3} urban.
Surface Water	0.083 mg L^{-1}	Highest Clinch River Remedial Investigation program sample (1989-90).	0.0005 to 1 mg L^{-1} median = 0.010 mg L^{-1}
Soil/Sediment	65 mg kg^{-1}	Highest average concentration from routine sediment sampling (Clinch River near mouth of Poplar Creek, 1979)	10 to 50 mg kg^{-1}, mean = 22 mg kg^{-1} (soil); 10 to 63 mg kg^{-1} (sediment)
Fish	8.3 mg kg^{-1}	Calculated based on the water concentration shown above and a BCF of 100.	<0.2 to 0.61 mg kg^{-1}

[1] For references, see Section 5.3.2.1

Table 5-16: Copper Doses Calculated in the Level I and Refined Level I Screening (mg kg^{-1} d^{-1})

	Average Daily Inhalation Dose- (Noncarcinogen)	Average Daily Oral Dose- (Noncarcinogen)
Level I	8.5×10^{-5}	8.9×10^{-2}
Refined Level I	2.5×10^{-5}	4.9×10^{-3}

5.3.5 Risk Characterization

Non-cancer screening indices for copper were calculated based on the screening estimates of average daily dose associated with the inhalation and oral exposure pathways. The results of the screening characterization are presented below, and compared to decision guides established by the Oak Ridge Health Agreement Steering Panel (ORHASP).

5.3.5.1 Non-cancer Screening Indices Associated with Inhalation and Ingestion of Copper

The following equation was used to calculate the non-cancer screening indices for inhalation and ingestion of copper:

$$Noncancer\ Screening\ Index = \frac{Average\ Daily\ Dose\ (mg\ kg^{-1}\ d^{-1})}{Reference\ Dose\ (mg\ kg^{-1}\ d^{-1})}$$

At Level I, the non-cancer screening index for inhalation of copper is:

$$Noncancer\ Screening\ Index = \frac{0.000085\ mg\ kg^{-1}\ d^{-1}}{0.02\ mg\ kg^{-1}\ d^{-1}} = 0.0043$$

and the non-cancer screening index for ingestion of copper is:

$$Noncancer\ Screening\ Index = \frac{0.089\ mg\ kg^{-1}\ d^{-1}}{0.037\ mg\ kg^{-1}\ d^{-1}} = 2.4$$

The non-cancer screening indices for inhalation and ingestion of copper were then summed to give a Level I non-cancer screening index for both routes of exposure (2.4). Because the Level I non-cancer screening index for ingestion of copper exceeded 1.0, the decision guide established by ORHASP for evaluating the need for further study of noncarcinogens, a Refined Level I screening was conducted.

At Refined Level I, the non-cancer screening index for ingestion of copper is:

$$Noncancer\ Screening\ Index = \frac{0.0049\ mg\ kg^{-1}\ d^{-1}}{0.037\ mg\ kg^{-1}\ d^{-1}} = 0.13$$

5.3.5.2 Comparison of Screening Indices to Decision Guides

The non-cancer screening indices calculated using the Level I and Refined Level I screening methodologies are presented in Table 5-17, and compared to the risk-based decision guides established by ORHASP for screening of contaminants released from the ORR.

Table 5-17: Results of Level I and Refined Level I Screening of Copper

	Non-cancer Screening Index	**Exceeds Decision Guide?**[a]
Level I	2.4	Yes
Refined Level I	0.13	No

[a] For noncarcinogenic chemicals, a hazard index of 1.0 was established by ORHASP as a decision guide for evaluating the need for further study (ChemRisk 1996).

5.3.6 Conclusions

In this screening analysis, exposures of off-site residents to copper in air and water released from the K-25 Site led to screening indices that are below the decision guides in use on the project. It should be noted that, even in Refined Level I screening, the analysis maintains considerable conservatism. In the Refined Level I methodology used in this assessment, somewhat different (less conservative) exposure parameters were used than in the Level I methodology; however, the environmental concentrations and biotransfer factors remained the same. Both the biotransfer factors and the environmental concentrations add considerable conservatism to the analysis. The NCRP Report 123 (NCRP 1996) biotransfer factors used in this analysis were developed for use in screening assessments, and are at the upper end of the range of biotransfer factors found in the literature. The environmental concentrations used in these analyses were also at the upper end of the range of values located by the project team. Given the conservatism built into this analysis, the screening indicates that historical copper releases do not warrant a high priority for further evaluation.

5.4 Hexavalent Chromium Releases from the Oak Ridge Reservation

In the Oak Ridge Dose Reconstruction Feasibility Study, chromium was screened using environmental data for total chromium and the assumption that all of the chromium was trivalent. Hexavalent chromium [Cr(VI)] was identified as a material for which more information was needed to conduct a screening. It is known that Cr(VI) compounds were used as corrosion inhibitors in cooling systems and cooling towers at X-10, Y-12, and K-25. Each site was a source of Cr(VI) releases from cooling towers via drift to the air and blowdown to surface waters.

Environmental measurements of Cr(VI) corresponding to releases from the individual complexes are scarce and incomplete. In order to screen the Cr(VI) compounds, the project team collected information about the cooling towers used at each site. The screening of Cr(VI) releases from the ORR was performed using a composite of conservative concentration estimates corresponding to releases from K-25 and Y-12. As is described in the text that follows, airborne concentrations of Cr(VI) in the ORR area as a whole were estimated based on K-25 cooling tower releases and dispersion modeling to Union/Lawnville, and waterborne concentrations of Cr(VI) near the ORR were estimated based on Cr(VI) levels measured in EFPC and in Poplar Creek. Although Y-12 and X-10 also had a number of cooling towers, this analysis focused on airborne releases from the cooling towers at K-25 because these towers used 10 to 20 times the amount of cooling water that the towers at X-10 and Y-12 used. The waterborne Cr(VI) measurement used for initial screening was also conservative, in that it was the highest measurement reported from an area near where public access was possible.

5.4.1 Hexavalent Chromium Use at the ORR

Cooling towers reduce the temperature of water in process cooling systems, so that the water can be recycled in the industrial process. The process cooling water systems at K-25 employed what is termed an "open recirculating" water system**S** a system in which water recirculating through cooling towers dissipates the heat from the diffusion cascade. In this type of system, water is lost from the system through evaporation, "drift," and "blowdown." Drift is water lost from the cooling tower as liquid droplets entrained in exhaust air. Blowdown is water that is deliberately purged from the cooling tower to avoid the accumulation of dissolved solids (Jallouk 1974). Water must be pumped into the system to "makeup" for the water loss. The water in the cooling tower must be treated to protect the system piping and heat exchangers from corrosive attack, prevent excessive scale formation on the heat transfer surfaces, and prevent growth of algae.

While the original design of the K-25 cooling towers called for the use of Calgon (sodium hexametaphosphate) for corrosion control, this method was not successful and plant personnel soon began testing other materials to control corrosion (Byrnes 1947). The use of Cr(VI) in the water treatment program can be traced to the initial testing of Betz dianodic (a mixture of a chromate salt, zinc, and phosphate) in the G Loop of the K-25 cooling towers in early 1956 (Fowlkes et al. 1959). At that time, the K-25 utilities group was testing several different methods in each of the four (A, C, E and G) cooling tower recirculating water loops (for six cooling towers). After about a year of testing, the Betz dianodic treatment proved most effective, and after June 11, 1957, the treatment was begun in the C and E loops. In May of 1958, the chromate treatment was begun in the A loop (Fowlkes et al. 1959). The use of Cr(VI) in the gaseous diffusion cooling towers continued until the plant was shut down in 1984. The makeup water requirement for the K-25 gaseous diffusion cascade was 18 million gallons per day. The entire system was capable of circulating 380 million gallons of water per day (UCCND 1959b).

Y-12 and X-10 also used Cr(VI) in their cooling towers. However, the cooling towers at these sites were much smaller. At Y-12, approximately 1 million gallons of makeup water per day was required for all of the 22 cooling towers. Y-12 began using Cr(VI) in its cooling towers in 1963 (Dykstra 1970) and stopped using Cr(VI) in most of the cooling systems in 1974. X-10 required approximately 1 million gallons of makeup water per day for the High Flux Isotope Reactor (HFIR) cooling system and about 1 million gallons per day from smaller cooling towers around the site. The history of chromate usage at X-10 was not located; however, it is known that chromate was used in the HFIR cooling system.

5.4.2 Concentrations of Hexavalent Chromium in the Environment near the ORR

Cr(VI) may be present in the environment from both natural and human-related sources. The following sections describe typical natural, or background, levels of Cr(VI) in the environment, and concentrations that have historically been measured in the environment near the ORR.

5.4.2.1 Background Levels

Chromium exists in the environment primarily in two forms: trivalent chromium [Cr(III)], which is a naturally occurring essential element, and hexavalent chromium [Cr(VI)], which occurs naturally in the rare mineral crocoite but is mostly produced from human activities. At high doses, Cr(VI) can cause adverse health effects. In the environment, Cr(VI) is converted naturally to Cr(III) in the presence of reducing agents, particulary in acidic environments (Kerger 1996). As a result, very few measurements of background concentrations of Cr(VI) in the environment were located. However, several sources of data on background concentrations of chromium were identified by the project team. These include general background concentrations and near-site background concentrations measured in areas of Eastern Tennessee assumed to not be impacted by ORR activities.

Data on background concentrations of chromium identified by the project team are summarized briefly by media below.

- *Air* - Background concentrations of total chromium in air in remote areas are reported to range from 0.000005 to 0.0026 mg m^{-3} (ATSDR 1997b). Background level of Cr(VI) have not been reported in the literature; however, Falerios (1992) measured the airborne Cr(VI) concentrations at industrial sites contaminated by chromite ore-processing residue. The concentrations ranged from 0.00013 to 0.11 mg m^{-3}, with a mean of 0.0099 mg m^{-3}.

- *Surface Water* - No data regarding background concentrations of Cr(VI) in surface water were located. Research suggests Cr(VI) in surface water is reduced fairly quickly to trivalent chromium (ATSDR 1997b). Typical background concentrations of chromium in surface water average between <1 and 30 mg L^{-1} (ATSDR 1997b). Chromium was not detected in either of the two surface water samples collected in Norris Reservoir during the Clinch River Remedial Investigation (detection limit 9.8 µg L^{-1}; Cook et al. 1992). Norris Reservoir was identified in the CRRI as the "reference location," indicating that it was considered unaffected by releases from the ORR.

- *Soil/Sediment* - No data regarding background concentrations of Cr(VI) were located by the project team. Background concentrations of chromium in Tennessee soils are reported to range from 30 to 200 mg kg^{-1} (Dragun and Chiasson 1991). Background concentrations up to 2,000 mg kg^{-1} are reported for other areas of the United States (Dragun and Chiasson 1991).

  In 1973 and 1982, the Tennessee Valley Authority (TVA) measured concentrations of several metals, including chromium, in sediments just upstream of a number of TVA dams. Reservoirs evaluated included several on the Tennessee River upstream of the ORR (including Fort Loudoun Reservoir) and several on different river systems (including Douglas Reservoir on the French Broad River and Wilson Reservoir on Fleet Hollow Embayment). Chromium concentrations in sediments of reservoirs assumed not to be impacted by ORR activities ranged from 5 to 50 mg kg^{-1} (dry weight) (TVA 1986).

5.4.2.2 Environmental Concentrations on or near the ORR

The earliest environmental samples for Cr(VI) that were identified by the project team were collected in the mid-1960s, when routine monitoring for Cr(VI) in surface water near each ORR site was initiated. No earlier environmental data were identified by the project team. Available data describing Cr(VI) concentrations in air, surface water, soil, sediment, and other environmental media on or near the ORR are discussed in the following sections. In addition, determination of exposure point concentrations used in the screening assessment is described.

Unlike some of the other Task 7 screening assessments, the evaluation of Cr(VI) exposures was performed using a composite of concentration estimates from K-25 and Y-12 releases. The project team confirmed that hexavalent chromium was released in cooling tower blowdown (to surface waters) and cooling tower drift (to the atmosphere) from the K-25, X-10, and Y-12 complexes. Environmental measurements of Cr(VI) corresponding to releases from the individual complexes are scarce and incomplete. Because of this, airborne concentrations of Cr(VI) in the ORR area as a whole were estimated based on K-25 cooling tower releases and dispersion modeling to Union/Lawnville. Because the cooling towers at K-25 were much larger than those at Y-12 or X-10, that is a conservative assumption.

In a similar manner, waterborne concentrations of Cr(VI) in the ORR area were estimated for Level I screening based on Y-12 releases as measured in EFPC at the point where it exited New Hope Pond. Because waterborne Cr(VI) concentrations measured there were the highest of any documented measurement location near where public access was possible, this again is a conservative assumption. For the more realistic Refined Level I screening, a Cr(VI) concentration measured in Poplar Creek was used. While the Level I screening has an added degree of conservatism due to the reduced likelihood that individuals would be concurrently exposed to the air concentration estimated for Union/Lawnville and the water concentration estimated for EFPC, it should be remembered that elevated air concentrations were likely experienced across the general area, and that the Union/Lawnville value is simply being used as a conservative surrogate for the concentrations across the ORR as a whole. In the Refined Level I assessment, it is more realistic to assume that individuals could be exposed to the air concentration estimated for Union/Lawnville and the water concentration measured in Poplar Creek. Details of the methods used to estimate environmental concentrations of Cr(VI) follow.

5.4.2.2.1 Hexavalent Chromium Concentrations in Air

Monitoring data for total chromium in air were located by the project team beginning in 1973; however, no air monitoring data for Cr(VI) were located. To estimate Cr(VI) concentrations at Union/Lawnville, the off-site reference location used in the screening assessments to evaluate airborne releases from K-25 (see Section 2.3.4.2), the ISCST3 air dispersion model was employed. The ISCST3 model was used to model Cr(VI) releases from the six gaseous diffusion process cooling towers at K-25 (K-801-H, K-802-H, K-832-H, K-861, K-892-G, and K-892-H).

The following is a list of assumptions used in the ISCST3 modeling runs:

- Although each cooling tower contained a number of individual cells (release sources), it was assumed that the center of each cooling towers was the discharge point of cooling water drift.

- One year of site-specific meteorology measurements for the K-25 facility was used to determine annual average air concentrations of Cr(VI).

- The maximum drift fraction, 0.12%, measured in the 1974 study for the K-861-H cooling tower, was applied to the five other cooling towers (K-801-H, K-802-H, K-832-H, and K-892G&H).

- The drift release rates (R, g s^{-1}) for K-801-H, K-802-H, K-832-H, K-861-H, and K-892-G&H were calculated using the equation:

$$R = n \times Q \times D \times C \times F$$

Where:

n = Number of cooling tower units (unitless);
Q = Flow rate (L min^{-1});
D = Drift fraction (L L^{-1});
C = Cr(VI) concentration (mg L^{-1}), equivalent to the chromate concentration in mg L^{-1} multiplied by the molar fraction of Cr(VI) in chromate (52 mg Cr(VI) per 116 mg CrO_4); at Oak Ridge the chromate concentration in the cooling tower makeup water was approximately 20 mg L^{-1} (Cromer 1969, Zimmerman 1969), thus the Cr(VI) concentration was 9 mg L^{-1}; and,
F = 1.67×10^{-5} g min mg^{-1} s^{-1}, conversion factor to convert milligrams to grams and minutes to seconds.

Table 5-18 summarizes the parameter values and drift release rates calculated for each cooling tower.

Table 5-18: Calculations of Cooling Tower Cr(VI) Release Rates

Cooling Tower Number	Number of Units	Flow Rate, Q (L min^{-1})	Drift Fraction, D	Cr(VI) Concentration, C (mg L^{-1})	Unit Conversion Factor, F	Drift Release Rate (g s^{-1})
K-801-H	1	76,000	0.0012	9	1.67×10^{-5}	1.4×10^{-2}
K-802-H	1	42,000	0.0012	9	1.67×10^{-5}	7.6×10^{-3}
K-832-H	1	42,000	0.0012	9	1.67×10^{-5}	7.6×10^{-3}
K-861-H	1	174,000	0.0012	9	1.67×10^{-5}	3.1×10^{-2}
K-892-G&H	1	605,000	0.0012	9	1.67×10^{-5}	1.1×10^{-1}

- Drift release rate studies demonstrate that the droplet size fraction below 100 µm for humid conditions and below 300 µm for arid conditions will evaporate, leaving chromate particulates in air. The mass fraction of droplets below 100 µm is in the range of 20 to 35% and the mass fraction of droplets below 300 µm is in the range of 40 to 65%. For the purpose of this modeling, humid conditions were assumed to represent the K-25 site. For this analysis, the release rate of Cr(VI) particulate was conservatively assumed to be 60% of the drift release rate shown in Table 5-18.

- The release heights for K-801-H, K-802-H, K-861-H, K-892-G&H cooling towers were 19.2 m. The release height for K-832-H was 19.8 m.

- The exit velocity (8.21 m s^{-1} for K-801-H, K-802-H, K-832-H, and K-861-H, 6.66 m s^{-1} for K-892-G&H), temperature (309.15 K), and stack diameter (6.2 m for K-801-H, K-802-H, K-832-H, and K-892-G&H; 7.0 for K-861-H) were obtained from Jallouk (1974).

- The chromate concentration in the cooling tower water was 20 mg L^{-1} (Cromer 1969, Zimmerman 1969).

- No dry depletion of the plume was assumed to occur.

- The Flat Terrain option was chosen for the model run.

Using the above assumptions, the model predicted that the Cr(VI) concentration at Union/Lawnville due to drift releases from the K-25 cooling towers was 5.0×10^{-5} mg m^{-3}. This predicted Cr(VI) air concentration is a conservative estimate for the off-site reference location, because the reduction in air concentration due to dry and wet deposition of particulates is not accounted for in the modeling. Other studies of Cr(VI) releases from cooling towers located within the K-25 facility indicated that the measured Cr(VI) deposition flux rate reduced from 500 µg m^{-2} h^{-1} at approximately 25 m from the cooling tower to approximately 10 µg m^{-2} h^{-1} at 1500 m from the cooling tower (Jallouk 1974). These data suggest that there is a 98% reduction of airborne Cr(VI) particulate at 1500 m downwind from the source.

Off-Site Air Concentration Used in the Screening Assessment

The calculated Cr(VI) concentration at Union/Lawnville of 5.0×10^{-5} mg m^{-3} was used in this screening assessment. Representation of plume depletion in the air dispersion modeling of Cr(VI) releases was not possible within the scope of this screening analysis. A reduction factor of the nature found by Jallouk (1974) was not applied, due to concerns regarding the transferability of the results of that study to the assessments at hand and the lack of certainty regarding whether peak concentrations were actually measured at the 1500 m distance.

The lack of correction for plume depletion in the dispersion calculations is a particularly conservative factor in the screening of cancer risk from Cr(VI) exposure, which according to current thinking comes only via inhalation. For Cr(VI) as a non-carcinogen, exposure pathways associated with airborne Cr(VI) contribute very little to the total dose to the reference population, even with plume depletion ignored (see Appendix I).

5.4.2.2.2 Hexavalent Chromium Concentrations in Surface Water

The cooling water that is blown down to prevent the build up of dissolved solids in the cooling water is typically discharged to a nearby stream. At K-25, about 1 million gallons per day of blowdown was discharged through a holding pond, through a limestone neutralizing bed, and then to Poplar Creek.

Summary level Cr(VI) monitoring data of Poplar Creek downstream of K-25 indicate that the maximum Cr(VI) concentration was 0.05 mg L^{-1} (Dykstra 1970). The maximum concentration in the Clinch River downstream of Poplar Creek was 0.01 mg L^{-1} (Dykstra 1970). A 1970s document that was written when the plant was upgrading many systems estimated that when the K-25 plant was running again, the maximum Cr(VI) concentration in the Clinch River would be 0.05 mg L^{-1} (Dykstra 1970).

At Y-12, a maximum of 0.7 million gallons of blowdown was discharged directly to EFPC in the late 1960s. The total amount of Cr(VI) added to the cooling systems was estimated to be 700 pounds. Based on the usage information, Dykstra (1970) performed a theoretical calculation of the chromate concentration leaving New Hope Pond. This calculation estimated an average Cr(VI) concentration of 0.28 mg L^{-1} in EFPC at the point where it exited New Hope Pond. This concentration compares favorably with the measured concentrations in EFPC. Concentrations in EFPC were measured in two separate sampling programs during 1969 and 1973. The 1969 program consisted of sampling the water in EFPC at the point where it exited New Hope Pond; samples were collected at routine intervals and composited on a monthly basis. The Cr(VI) concentration ranged from 0.12 to 0.25 mg L^{-1} with an average of about 0.18 mg L^{-1} (Cromer 1969).

The 1973 sampling program was carried out during a five day period in November while the plant was changing over from chromate to nonchromate treatments. The cooling towers were taken out of service one at a time and all chromate-treated water was dumped to the sewers. Samples of EFPC were taken during the dumping procedure and analyzed for chromium and Cr(VI). The Cr(VI) concentrations ranged from 0.03 to 0.173 mg L^{-1} (DeMonbrum 1975).

At X-10, there were four cooling towers that used a Cr(VI) treatment system (Bolton 1971). Summary level sampling data were located for the 1962 to 1969 time period. During this time period, the average Cr(VI) concentrations at White Oak Dam ranged from 0.01 to 0.13 mg L^{-1} (Bolton 1971). The maximum Cr(VI) concentrations at White Oak Dam in 1968 and 1969 were 0.35 mg L^{-1} and 0.42 mg L^{-1}, respectively (Bolton 1971).

Off-Site Water Concentration Used in the Screening Assessment

Although maximum Cr(VI) concentrations at the White Oak Dam outfall were greater than the EFPC concentrations, EFPC is more accessible to off-site populations. Therefore, the maximum measured Cr(VI) concentration in EFPC of 0.25 mg L^{-1} from the 1969 sampling was used in Level I screening analysis to reflect concentrations to which off-site individuals may have been exposed.

For the more realistic Refined Level I screening analysis, the maximum Cr(VI) level of 0.05 mg L^{-1} measured in Poplar Creek downstream of K-25 was used. This concentration is more representative of average or typical exposures than the Level I value, as it reflects dilution to levels that were more likely to have been experienced by individuals in areas where recreational fishing is more common than in EFPC.

5.4.2.2.3 Hexavalent Chromium Concentrations in Soil and Sediment

Cr(VI) is not stable in soil or sediment and generally reduces to its less toxic trivalent form. As a result, no measurements for Cr(VI) in soil or sediments were located. The available monitoring data only included analyses for total chromium in the soils and sediments surrounding the ORR. A summary of these data is provided below.

Surface sediment samples were collected semiannually at several locations beginning in the mid-1970s, including K-25 site holding ponds, Poplar Creek, and the Clinch River. Results from routine analyses of Cr(VI) in sediment near K-25 are reported in the annual environmental monitoring reports beginning in 1975 (UCC 1976). Samples were collected semiannually at up to 12 sites on Poplar Creek, both above and below K-25 plant discharge points, and up to two sites on the Clinch River. Summary data (i.e., annual averages) from this program are presented in the annual environmental monitoring reports. Average sediment concentrations in the Clinch River between 1975 and 1982 ranged from 14 to 244 mg kg^{-1}. The maximum Clinch River concentration was measured in 1979 at Clinch River mile 10 at Brashear Creek, about 2 miles downstream of the confluence of Poplar Creek and the Clinch River.

In addition to the routine monitoring program, chromium concentrations were also measured in sediments/soils as part of several special monitoring programs. These programs include:

- *A survey of sediments in streams surrounding the K-25 Plant by ORGDP staff (1985)*– Surface sediment samples were collected by ORGDP staff at 180 locations in the Clinch River, Poplar Creek, EFPC, and tributaries to Poplar Creek, to identify locations where contaminants were entering surface water systems (Ashwood et al. 1986). Chromium concentrations were measured at some of these sampling locations, including three locations in the Clinch River both upstream and downstream of the Poplar Creek/ Clinch River confluence. Data for individual samples are presented in the Ashwood et al. report. Concentrations in the Clinch River ranged from 1 to 51 mg kg^{-1}.

- *The Tennessee Valley Authority (TVA) Instream Contaminant Study (1984)*– In addition to surface water samples, sediment samples were also collected and analyzed for a variety of contaminants including chromium. Surface sediment sample locations included the length of EFPC, three locations in Poplar Creek, and five locations in the Clinch River (TVA 1985b, c). Concentrations ranged from 9 to 25 mg $kg^{-1.}$ Data for individual samples are presented in the TVA reports.

- *The Clinch River Remedial Investigation (1989 and 1990)*– Sediment samples were collected from 10 reaches. Sample locations included Poplar Creek and the Clinch River. The maximum sediment concentration in the Clinch River was 18 mg kg^{-1}. Data for individual samples are presented in the LMES OREIS database (LMES OREIS 1997).

C *The East Fork Poplar Creek Floodplain Remedial Investigation (EFPC RI) (1990 and 1991)*– Sediment and floodplain soil samples were collected at several locations in and along EFPC. Maximum chromium concentrations in sediment and soil were 68 mg kg^{-1} and 117 mg kg^{1}, respectively. The average chromium concentration in soil was 66 mg kg^{-1}. Data for individual samples are presented in the LMES OREIS database (LMES OREIS 1997).

Off-Site Soil Concentration Used in the Screening Assessment

The average soil concentration of 66 mg kg^{-1} total chromium from the EFPC RI was used in the specification of soil concentrations for this screening analysis. This concentration is within the reported range of background concentrations of chromium in Tennessee soils of 30 to 200 mg kg^{-1} (Dragun and Chiasson 1991). For Level I screening, this soil concentration was assumed to be all Cr(VI). The very conservative nature of this assumption is offset somewhat by the use of an average value from the EFPC RI study rather than the maximum reported value (117 mg kg^{-1}).

For the more realistic Refined Level I screening, Cr(VI) was assumed to be a minor component of the total chromium measured. For planning purposes, the USEPA uses an assumption that Cr(VI) comprises one-sixth of total chromium (USEPA 1998). This fraction (about 17%) is based on measurements of Cr(III) and Cr(VI) in the air of an operating chromate production plant. In the Refined Level I screening analysis, Cr(VI) was assumed to be present at 11 mg kg^{-1}, which is 17% of the average total chromium measured in EFPC soil.

5.4.2.2.4 Hexavalent Chromium Concentrations in Food Items

No data were identified by the project team describing measured concentrations of Cr(VI) in fish, vegetables, meat, or milk. It is likely that Cr(VI) taken into fish, vegetables, meat, or milk will be reduced to trivalent chromium.

Off-Site Fish, Vegetation, Meat, and Milk Concentrations Used in the Screening Assessment

For purposes of the screening analyses, concentrations of Cr(VI) in vegetation, meat, and milk were calculated using biotransfer factors that describe the transfer of Cr(VI) from other media, including air, water, and soil. The biotransfer factors used in this assessment are listed in Table 5-19. Concentrations of Cr(VI) in fish were calculated using a BCF, shown in Table 5-19, that describes the transfer of chromium from water to fresh water fish. The BCF used in this assessment for fish is 200 (mg kg^{-1})/(mg L^{-1}) (NCRP 1996). Use of this BCF and the water concentrations described in Section 5.4.2.2.2 (0.25 and 0.05 mg L^{-1}) predicts fish concentrations of 50 and 10 mg kg^{-1}.

Since these fish concentrations were estimated based on measurements of Cr(VI) in surface waters near the ORR, they should reflect the reduction of Cr(VI) to Cr(III) that took place in the environment between the times of release and sample analysis. The estimated concentrations do not, however, reflect any further reduction of Cr(VI) to Cr(III) that occurred within the fish.

Table 5-19: Biotransfer and Bioconcentration Factors for Hexavalent Chromium

Parameter	Value	Source
Soil to Vegetables (B_{veg})	0.01 (mg kg^{-1} wet)/(mg kg^{-1} dry)	NCRP 1996
Soil to Pasture ($B_{pasture}$)	0.1 (mg kg^{-1} dry)/(mg kg^{-1} dry)	NCRP 1996
Biotransfer to Milk (F_m)	0.002 d L^{-1}	NCRP 1996
Biotransfer to Meat (F_f)	0.003 d kg^{-1}	NCRP 1996
Bioconcentration in Fish (*BCF*)	200 (mg kg^{-1})/(mg L^{-1})	NCRP 1996

5.4.3 Toxicity Assessment

Cr(VI) is considered to be noncarcinogenic through most routes of exposure, including ingestion of Cr(VI) in food products and/or soil and dermal contact. Inhalation of Cr(VI) by workers involved in chromate production has been associated with an increased incidence of lung cancer. The following sections describe the toxicity of Cr(VI) through different routes of exposure, and summarize the toxicity criteria used to evaluate exposure to Cr(VI) in the screening analyses.

5.4.3.1 Inhalation Exposure

Numerous epidemiological studies have reported an increased incidence of respiratory cancer in occupational cohorts of chromate production workers exposed to high levels of airborne Cr(VI). Although many of these studies did not differentiate between Cr(III) and Cr(VI) levels in air, it has generally been assumed that Cr(VI) was responsible for the apparent carcinogenic activity of inhaled chromium. The rationale is that Cr(III) compounds have not been shown to be carcinogenic in animal studies and that Cr(III) can be genotoxic *in vitro* only at extremely high exposure levels or in cells with phagocytic activity (ATSDR 1997b).

The epidemiological study by Mancuso (1975) has been used for quantitative risk assessment. This study is a follow-up of the health status of chromate production workers using vital statistics of employees (Mancuso and Heuper, 1951). Individuals included in the study worked for more than one year at the Painesville, Ohio chromate production plant during 1931-1949. The percentage of deaths due to lung cancer among chromate workers (18.2%) was significantly different ($p<0.01$) than deaths due to lung cancer among males in the county where the plant was located (1.2%). Although the study is acknowledged to have many shortcomings, it has been judged by the USEPA to be the best available study for use in quantitative risk assessment. Based on this study, the USEPA (USEPA 1999) has established an inhalation slope factor for Cr(VI) of 42 (mg kg^{-1} d^{-1})$^{-1}$.

In addition to an inhalation slope factor for carcinogenic effects of Cr(VI), the USEPA has established an inhalation RfD for Cr(VI) in airborne particulates of 2.9×10^{-5} mg kg^{-1} d^{-1}. The RfD is based on studies showing lower respiratory effects in rats exposed to airborne chromium particulates (Glaser et al. 1985, 1990).

5.4.3.2 Oral (Ingestion) Exposure

Following oral administration, Cr(VI) has not been reported to cause health effects in animals except at fairly high doses (Finley 1996). The lack of oral toxicity at lower doses is believed to be due, in part, to the fact that the reductive conditions of the stomach convert ingested Cr(VI) to Cr(III) prior to systemic absorption (Finley 1996). A chronic ingestion study in which chromate salts were administered to rats in drinking water at concentrations up to 25 mg L^{-1} did not produce evidence of a carcinogenic response (IRIS 1998). This 25 mg L^{-1} no observed adverse effect level (NOAEL), equivalent to a daily dose of 2.4 mg Cr(VI) kg^{-1} d^{-1}, was used as the basis for the USEPA's noncarcinogenic RfD for ingestion of Cr(VI) of 3.0×10^{-3} mg kg^{-1} d^{-1}.

5.4.3.3 Toxicity Criteria Used in the Screening Assessment

Toxicity criteria used to evaluate exposures to Cr(VI) at Oak Ridge are presented in Table 5-20.

Table 5-20: Toxicity Criteria for Cr(VI) Used in the Screening Analyses

Exposure Route	Toxicity Criteria/Source	Value
Inhalation (Cancer)	USEPA Inhalation Slope Factor	42 (mg kg^{-1} d^{-1})$^{-1}$
Inhalation (Non-cancer)	USEPA Inhalation RfD	3.0×10^{-5} mg kg^{-1} d^{-1}
Ingestion (Non-cancer)	USEPA Oral RfD	3.0×10^{-3} mg kg^{-1} d^{-1}

5.4.4 Exposure Assessment

Following the Task 7 Screening methodology for Level I and Level II screening (ChemRisk 1996), doses were calculated for exposure to Cr(VI) in air, surface water, and soil/sediment. Exposures to Cr(VI) through ingestion of fish, vegetables, meat, and milk were evaluated using equations (presented in Appendix B) and biotransfer factors (discussed in Section 5.4.2.2.4) that describe the uptake of Cr(VI) from air, surface water, and soil/sediment into these media. A summary of the concentrations for each medium that were used in the screening is presented in Table 5-21.

Table 5-21: Summary of Environmental Concentrations of Cr(VI) used in Screening

Environmental Medium	**Concentration**	**Source of Value**	**Typical Background** (For references, see Section 5.4.2.1)
Air	5.0×10^{-5} mg m^{-3} (0.05 µg m^{-3})	Modeled from K-25 cooling towers to Union/Lawnville.	0.005 to 2.6 µg m^{-3}, total Cr- rural
Surface Water, Level I Screen	0.25 mg L^{-1}	Maximum concentration measured in EFPC in 1969 study.	<1 to 30 mg L^{-1}, total Cr
Surface Water, Refined Level I Screen	0.05 mg L^{-1}	Maximum concentration measured in Poplar Creek before 1970.	
Soil/Sediment, Level I Screen	66 mg kg^{-1}	Average total Cr concentration measured in EFPC Floodplain RI (assumed to be all Cr(VI))	Soil: 30 to 200 mg kg^{-1}, total Cr Sediment: 5 to 50 mg kg^{-1}, total Cr
Soil/Sediment, Refined Level I Screen	11 mg kg^{-1}	Average total Cr concentration measured in EFPC Floodplain RI (assumed to be 17% Cr(VI))	
Fish, Level I Screen	50 mg kg^{-1}	Calculated based on above surface water concentrations and a BCF of 200	Not available
Fish, Refined Level I Screen	10 mg kg^{-1}		

Doses were calculated separately for inhalation (carcinogenic pathway) and ingestion (noncarcinogenic pathway) of Cr(VI). The total average daily doses of Cr(VI) calculated for the inhalation and ingestion (oral) exposure routes for Cr(VI) released from K-25 are summarized in Table 5-22. Doses calculated for individual pathways and the contribution of each pathway to total dose are shown in Appendix I.

Table 5-22: Cr(VI) Doses Calculated in the Level I and Refined Level I Screening (mg kg^{-1} d^{-1})

	Lifetime Average Daily Inhalation Dose- (Carcinogen)	**Average Daily Inhalation Dose- (Noncarcinogen)**	**Average Daily Oral Dose- (Noncarcinogen)**
K-25			
Level I	3.1×10^{-6}	6.2×10^{-6}	2.9×10^{-2}
Refined Level I	2.4×10^{-7}	1.7×10^{-6}	1.6×10^{-3}

5.4.5 Risk Characterization

Cancer and non-cancer screening indices were calculated based on the screening estimates of lifetime average daily dose and average daily dose, respectively, for Cr(VI) released from K-25. The results of the screening risk characterization are presented below, and compared to risk-based decision guides established by the Oak Ridge Health Agreement Steering Panel (ORHASP).

5.4.5.1 Cancer Screening Indices Associated with Inhalation of Hexavalent Chromium

The following equation was used to calculate the cancer screening index for inhalation of Cr(VI):

$$Cancer\ Screening\ Index = Lifetime\ Average\ Daily\ Dose\ (mg\ kg^{-1}\ d^{-1}) \times Slope\ Factor\ (mg\ kg^{-1}\ d^{-1})^{-1}$$

At Level I, the cancer screening index for inhalation of Cr(VI) is:

$$Cancer\ Screening\ Index = 3.1\times10^{-6}\ (mg\ kg^{-1}\ d^{-1}) \times 42\ (mg\ kg^{-1}\ d^{-1})^{-1} = 1.3\times10^{-4}$$

Because the Level I cancer screening index for inhalation of Cr(VI) exceeded 1×10^{-4}, the decision guide established by ORHASP for evaluating the need for further study of carcinogens, a Refined Level I screening was conducted.

At Refined Level I, the cancer screening index for inhalation of Cr(VI) is:

$$Cancer\ Screening\ Index = 2.4\times10^{-7}\ (mg\ kg^{-1}\ d^{-1}) \times 42\ (mg\ kg^{-1}\ d^{-1})^{-1} = 1.0\times10^{-5}$$

5.4.5.2 Non-cancer Screening Indices Associated with Inhalation and Ingestion of Hexavalent Chromium

The following equation was used to calculate the non-cancer screening indices for inhalation and ingestion of Cr(VI):

$$Noncancer\ Screening\ Index = \frac{Average\ Daily\ Dose\ (mg\ kg^{-1}\ d^{-1})}{Reference\ Dose\ (mg\ kg^{-1}\ d^{-1})}$$

At Level I, the non-cancer screening index for inhalation of Cr(VI) is:

$$\text{Noncancer Screening Index} = \frac{6.2 \times 10^{-6}\ mg\ kg^{-1}\ d^{-1}}{3.0 \times 10^{-5}\ mg\ kg^{-1}\ d^{-1}} = 0.21$$

and the non-cancer screening index for ingestion of Cr(VI) is:

$$\text{Noncancer Screening Index} = \frac{2.9 \times 10^{-2}\ mg\ kg^{-1}\ d^{-1}}{3.0 \times 10^{-3}\ mg\ kg^{-1}\ d^{-1}} = 9.7$$

The non-cancer screening indices for inhalation and ingestion of Cr(VI) were then summed to give a Level I non-cancer screening index for both routes of exposure (10).

Because the Level I non-cancer screening index for inhalation of Cr(VI) was less than the decision guide of 1.0 for evaluating the need for further study of noncarcinogens, a Refined Level I screening for inhalation was not conducted. However, because the Level I non-cancer screening index for ingestion of Cr(VI) exceeded 1.0, a Refined Level I screening for ingestion was conducted.

At Refined Level I, the non-cancer screening index for ingestion of Cr(VI) is:

$$\text{Noncancer Screening Index} = \frac{1.6 \times 10^{-3}\ mg\ kg^{-1}\ d^{-1}}{3.0 \times 10^{-3}\ mg\ kg^{-1}\ d^{-1}} = 0.55$$

5.4.5.3 Comparison of Screening Indices to Decision Guides

The cancer and non-cancer screening indices calculated using the Level I and Refined Level I screening methodologies are presented in Table 5-23, and compared to the risk-based decision guides established by ORHASP for screening of contaminants released from the ORR.

Table 5-23: Results of Level I and Refined Level I Screening of Cr(VI)

	Cancer Screeening Index	**Exceeds Decision Guide? [a]**	**Non-cancer Screeening Index**	**Exceeds Decision Guide? [b]**
Level I	1.3×10^{-4}	Yes	9.7	Yes
Refined Level I	1.0×10^{-5}	No	0.55	No

a) For carcinogenic chemicals, a lifetime excess cancer risk of 10^{-4} was established by ORHASP as a decision guide for evaluating the need for further study (ChemRisk 1996)

b) For noncarcinogenic chemicals, a hazard index of 1.0 was established by ORHASP as a decision guide for evaluating the need for further study (ChemRisk 1996).

5.4.6 Conclusions

Based on the conservative Level I screening analyses, exposures of off-site residents to Cr(VI) in surface water and entrained in cooling tower drift released from K-25 and Y-12 cooling tower operations led to screening indices that are above the decision guides in use on the project. Under these guidelines, historical releases of Cr(VI) warrant further evaluation with regard to potential cancer and non-cancer health effects. With a non-cancer screening index of 9.7, however, it is unlikely that adverse health effects would have occurred, given the fact that there is a safety factor of 800 between the ingestion RfD and the NOAEL on which it was based. However, because this NOAEL was based on a rat study, conclusions about the safety of the estimated doses in humans should be made cautiously.

The Level I analyses were indeed designed to be conservative, due to the significant uncertainties encountered in the assessment of off-site exposures from chromium releases. The most important area of uncertainty related to this assessment is the fraction of total chromium that remains in hexavalent form after release to the environment. Because Cr(III) is considerably less toxic than Cr(VI), the rate of reduction of Cr(VI) to Cr(III) after release to the air as a result of cooling tower drift or to surface waters as a result of cooling tower blowdown or spills plays a critical role in the determination of whether health effects will likely occur.

The Refined Level I analyses were considerably less conservative than the initial analyses, in that they incorporated estimates of more typical environmental concentrations and exposure parameters. For example, chromium measured in soil was assumed to be 17% Cr(VI) instead of 100%, and surface water concentrations were estimated based on the maximum measured Cr(VI) in Poplar Creek water instead of the maximum concentration measured in EFPC close to the Y-12 Plant. The fact that the non-cancer screening index from the refined analysis is below 1 and the refined cancer screening index is below 1×10^{-4} indicates that Cr(VI) does not warrant high priority for further evaluation of the potential for adverse health effects among those who were exposed.

If releases of Cr(VI) from ORR complexes are studied further, the following areas of conservatism in the screening analyses described herein should be considered for refinement:

- Estimation of chromium levels in fish and other foods based on measurements or other methods more refined than use of bioconcentration factors (because over 80% of the total dose from ingestion of Cr(VI) in the refined screening comes through fish consumption, associated concentrations and exposure parameters are particularly important);

- Estimation of Cr(VI) fraction of total chromium levels in environmental media and in relevant food products;
- Reflection of the dilution that occurred between points where waterborne Cr(VI) measurements were made and locations where the largest numbers of individuals were likely exposed; and
- Representation of the reduction of airborne chromium levels by deposition in transit between the release points and points of potential off-site exposure.

5.5 Lithium Releases from the Y-12 Site

In the Dose Reconstruction Feasibility Study, lithium was not initially specified as a material to be evaluated in Phase II of the Oak Ridge Health Studies. However, as a result of Task 5 document searches, new information about lithium became available that warranted a rescreening of lithium as part of the Task 7 screening assessment. This section summarizes available information on past off-site releases of lithium from the ORR and presents a screening-level evaluation of potential doses that could have been received by people living near the ORR.

5.5.1 Lithium Use at Y-12

Lithium was historically handled at the ORR in lithium isotope separation, chemical, and component fabrication operations. In the early 1950s, the United States launched a crash program to produce ^{6}Li-enriched lithium deuteride for use in the more powerful and efficient thermonuclear weapons. The Y-12 Plant was given the assignment to develop, design, construct, and operate a production process to produce enriched ^{6}Li. The majority of ^{6}Li separation at Y-12 occurred in Buildings 9201-5 and 9201-4 from 1955-1963. Earlier pilot production operations were conducted in Building 9204-4 from 1952-1955. Several other Y-12 and X-10 buildings were used for pilot scale development studies of various lithium separation processes between 1950 and 1953 (UCCND 1983a).

The desired final product of lithium separation operations was lithium deuteride containing more of the ^{6}Li isotope than the 7.5% found in natural lithium. During and after isotopic enrichment, lithium was transferred from the amalgam (mercury) phase to an aqueous phase and converted to lithium hydroxide (UCCND 1983a). The product of lithium separation operations was stored in a liquid form as lithium chloride. Subsequent chemical and fabrication operations converted enriched lithium chloride to the metal and finally to the deuteride. Pulverized lithium deuteride was shaped by isostatic pressing, machined, canned in stainless steel, and assembled into thermonuclear weapon components (UCNC 1957). Figure 5-1 is a flow chart of the lithium chemical and fabrication operations conducted at Y-12. These operations were conducted primarily in Building 9204-2, although they were initially developed in Building 9204-4 (Baylor 1997).

Lithium chemical and fabrication operations consisted of multiple steps that processed enriched lithium in three different physical forms– aqueous solution, powder, and solid. The neutralization and evaporation steps handled lithium in a liquid form. Air releases from these steps were likely negligible due to the low vapor pressure of aqueous lithium solutions (Baylor 1997).

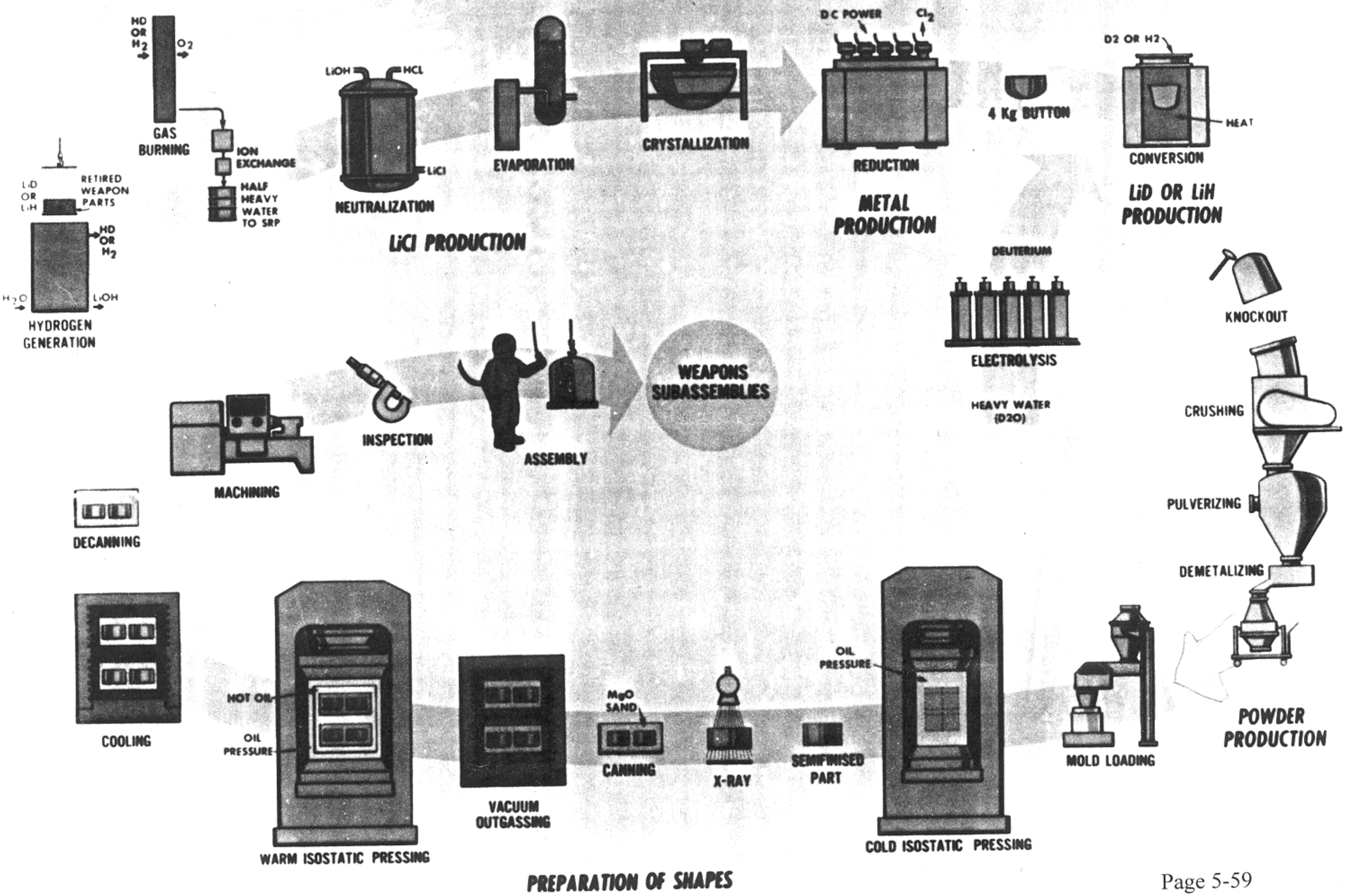
The crystallization and metal production steps (see Figure 5-1) produced a solid form of lithium and are known to have generated lithium dust. While the crystallization vessel was initially an open container and likely released dust into the room air, in later years it was a closed container. The metal reduction cell was always a closed system, but small amounts of dust could have become airborne during loading and unloading. It is reported that the process operators did not want to wear respirators because lithium dust around the edge of a respirator caused caustic facial burns. Therefore, generation of lithium dust was minimized in these operations as much as possible so that respirators did not have to be worn (Baylor 1997).

Conversion of enriched lithium metal to lithium hydride or deuteride was accomplished by reacting it with deuterium or hydrogen in a reactor that was also a closed system. The lithium hydride or deuteride was then "knocked out" of the reactor vessel and crushed using hammers. This operation was done in a ventilated hood. It is reported that filters were located in the top of the hood to trap the lithium dust generated from this step. The "knocked out" material was then pulverized to a powder, blended in a closed “dry box” (containing very dry air or an inert gas) and loaded into molds for pressing. The mold loading operation is known to have generated lithium dust in early years, but was a closed system in later years (Baylor 1997).

The lithium deuteride/hydride was canned in stainless steel for pressing operations and therefore generated no lithium dust once canned. Machining operations are known to have generated lithium dust, but were conducted in a closed dry box. Dust from machining operations was reportedly collected and recycled back into the production process at the hydrogen generation step shown in Figure 5-1. The economic value of the lithium deuteride/hydride after isotope separation was significantly higher than the cost of lithium feed material, and losses of the valuable product were therefore minimized. The material was decanned from its stainless steel container prior to machining and for the remaining production steps. Although it has been reported that the production processes on the solid lithium material did not generate significant amounts of surface dust (Baylor 1997), this analysis will rely on air sampling data from the production areas.

Figure 5-1

LITHIUM CHEMICAL AND FABRICATION OPERATIONS



Usage of lithium is documented in the Y-12 Plant Quarterly reports starting in 1953. The peak usage of lithium feed material (lithium hydroxide) occurred between 1955 and 1958, and was reported to range between 5 million and 9 million pounds per quarter. In 1959-60, lithium hydroxide usage dropped to 4 million pounds per quarter, and in 1961-62 usage dropped further to 2 million pounds per quarter. From October of 1962 to June of 1963, the Alpha 5 plant used about 1 million kg of feed to produce ^{7}Li, sometimes referred to as marble. Production of lithium isotopes stopped in 1963 (UCCND 1950-1970). The project team located some room air sampling data for chemical and fabrication operations in Building 9204-2 in 1955 and 1956, and some stack sampling data for salt (most likely lithium chloride) drumming operations in Building 9201-4 in 1955.

The form of lithium released to air was particulate dust. The lithium released to water would likely have remained solubilized in the water due to its high water solubility and low vapor pressure.

5.5.2 Concentrations of Lithium in the Environment near the ORR

Lithium may be present in the environment from both natural and human-related sources. The following sections describe natural, or background, levels of lithium in the environment, and concentrations that have historically been measured in the environment near the ORR.

5.5.2.1 Background Levels

Lithium is widely distributed in nature. Trace amounts are present in many minerals, most rocks and soils, and many natural waters (HSDB 1999). Several sources of data on background concentrations of lithium were identified by the project team. Data on background concentrations of lithium identified by the project team are summarized briefly by media below.

- *Air* - No data on background concentrations of lithium in air were identified by the project team. However, studies of the atmospheric loading of lithium to lakes in the Great Lakes Basin Ecosystem showed annual loadings of up to 225 metric tons (HSDB 1999). Because of the ionic nature of lithium, volatilization from water or soil surfaces is not an important environmental fate process (HSDB 1999). However, lithium has been detected in fly ash from coal boilers at a concentration of 152 mg kg^{-1}, and in exhaust from leaded and unleaded automobile fuel (HSDB 1999).

- *Surface Water* - Typical background concentrations of lithium in river water in the eastern United States range between 0.0003 and 0.002 mg L^{-1} (Walker and Blase 1960).

- *Soil/Sediment* - Background concentrations of lithium in Tennessee soils are reported to range from 15 to 93 mg kg^{-1} (Dragun and Chiasson 1991). Background concentrations up to 140 mg kg^{-1} are reported for other areas of the United States (Dragun and Chiasson 1991).

C *Fish* - Concentrations of lithium in ocean fish are reported to range between 0.007 to 0.11 mg kg^{-1}, wet weight (HSDB 1999). No data on lithium concentrations in fresh water fish were identified.

5.5.2.2 Environmental Concentrations on or near the ORR

Available data describing lithium concentrations in air, surface water, soil, and sediment on or near the ORR are discussed in the following sections. In addition, determination of exposure point concentrations used in the screening assessment is described.

5.5.2.2.1 Lithium Concentrations in Air

Several letter reports regarding air and stack sampling in two lithium processing buildings, 9204-2 and 9201-4, were found by the project team. Three of the reports present lithium air concentrations measured during various chemical and fabrication operations conducted in Building 9204-2 during a two-month period in 1955. Two reports describe sampling conducted on a stack that exhausted air from a salt drumming operation in Building 9201-4 during one month in 1955 (LaFrance 1955a,b, 1956). Based on data from these reports, the average lithium air concentration in Building 9204-2 was calculated at 154.1 µg m^{-3}, and the average stack release rate of lithium in Building 9201-4 was 0.56 g s^{-1}, which corresponds to a loss of about 3,200 lbs of lithium per month. Lithium release rates from Building 9204-2 were calculated by multiplying the building air concentration (µg m-3) by the air flow rate from the building (m^3 s^{-1}). Estimated lithium release rates from Buildings 9204-2 and 9201-4 are presented in the Table 5-24.

Table 5-24: Lithium Release Rates from Two Y-12 Process Buildings

Building	Average Indoor Lithium Concentration (Fg m^{-3})	Air Flow Rate (m^3 s^{-1})	Release Rate (Fg s^{-1})
9204-2	150	28	4,200 [a]
9201-4	NA	NA	560,000
Total			**564,200**

[b] Calculated by multiplying the average indoor lithium concentration by the stack flow rate.
N/A = Not applicable (stack release rate was measured).

Off-Site Air Concentration Used in the Screening Assessment

The reference location for screening of airborne lithium releases from the Y-12 Site was the Scarboro Community, the location of the nearest population center to Y-12. No measurements of airborne lithium concentrations at Scarboro were identified by the project team. Therefore, concentrations at Scarboro were estimated based on stack release rates and the empirical χ/Q discussed in Section 2.3.4.1:

$$Scarboro\ Air\ Concentration\ (\frac{\mu g}{m^3}) = Release\ Rate\ (\frac{\mu g}{s}) \times Empirical\ \chi/Q\ (\frac{s}{m^3})$$

$$Scarboro\ Air\ Concentration = 564{,}200\ \mu g\ s^{-1} \times (3{\times}10^{-7}\ s\ m^{-3}) = 0.17\ \mu g\ m^{-3}$$

The calculated concentration of airborne lithium at the Scarboro resident location (0.17 µg m^{-3}) was used to estimate an inhalation dose of lithium that may have resulted from releases of lithium to air from Y-12 buildings 9204-2 and 9201-4.

5.5.2.2.2 Lithium Concentrations in Surface Water

Quarterly and monthly average lithium concentrations based on daily EFPC water samples collected at the Y-12 site boundary are reported in Y-12 Plant Quarterly reports for the period October 1955 through December 1959. This period corresponds with the peak lithium usage period discussed in Section 5.5.1. Quarterly average flow rates for EFPC are also reported. The peak monthly average lithium concentration in water discharged to EFPC was 17 mg L^{-1} in the last quarter of 1955. After that period, lithium concentrations show a steady decline to a low of 1.2 mg L^{-1} in the last quarter of 1959 (UCCND 1950-70).

Three documents regarding estimates of lithium losses to water during lithium separation operations were identified by the project team:

- *Alloy and Solvent Loss Study for Alpha-5* (October 1957) reports an average monthly lithium loss from Building 9201-5 for July through September 1957 of 2,869 pounds (UCCND 1957).
- *Monthly Tabulations of Sump Losses* (April 1959) reports an average monthly lithium loss from Building 9201-5 for April 1957 through March 1959 of 2,423 pounds (UCCND 1959a).

- *A Preliminary Study of the Recovery of Lithium and Mercury Losses* (January 1958) reports an average monthly lithium loss from Buildings 9201-5 and 9201-4 between August 1957 and January 1958 of 7,810 pounds per month, plus 13.2 pounds per day (409 pounds per month) from acid washing of mercury, for a total average monthly loss of lithium of 8,219 pounds (UCCND 1958).

Assuming a total average lithium loss from Buildings 9201-5 and 9201-4 of 8,219 pounds per month (the maximum reported loss of lithium), an estimate of the lithium concentration in EFPC was calculated. The concentration in EFPC was calculated by dividing the average monthly lithium release by a lower-bound estimate of the average EFPC flow rate of 8 million gallons per day:

$$Li\ Conc = \frac{\dfrac{8{,}219\ lb\ mo^{-1}}{30\ d\ mo^{-1}} \times 455\ g\ lb^{-1} \times 1000\ mg\ g^{-1}}{8\times10^{6}\ gal\ d^{-1} \times 3.75\ L\ gal^{-1}} = 4.2\ mg\ L^{-1}$$

The resulting concentration, 4.2 mg L^{-1}, is within the range of values reported by the plant in the Y-12 Quarterly Reports for the same time period (late 1957; 3.5-7.6 mg L^{-1}).

The 1957 and 1958 lithium loss studies describe steps taken to minimize losses of lithium from Y-12 lithium separation operations and provide suggestions to bring about further reductions. It appears that some of these methods to minimize lithium losses may have been implemented between 1955 and 1959, since the concentrations of lithium released to EFPC show a steady decline during this period (UCCND 1950-70). Lithium was a valuable product in the 1950s and 1960s, and any process losses were undesirable; for example, the value of the ^{6}Li solution lost in a 1965 spill was approximately $118 per pound (USAEC 1965). The peak quarterly average EFPC concentration of 17 mg L^{-1} reported in the Quarterly Reports occurred in the last quarter of 1955, prior to implementation of steps to minimize lithium losses.

A study released in 1960 presented measured total lithium and ^{7}Li/^{6}Li concentrations in surface water samples collected upriver and downriver from the Y-12 plant in 1958 and 1959. Lithium losses were reported to be a mixture of ^{7}Li tails (waste product), ^{6}Li product, and normal lithium feed material as evidenced by the observed variation in the ^{7}Li/^{6}Li ratio of daily effluent measurements. A four-week study of lithium concentrations in EFPC in October and November 1957 showed a daily variation in concentration from 0.07 to 12.5 mg L^{-1}. Results indicated that the lithium concentrations varied widely, and Y-12 was estimated to contribute 15 pounds of lithium per hour to surface water (Walker and Blase 1960). This loss of 15 lbs per hour is equal to a loss of about 10,800 lbs per month and would correspond to an EFPC concentration of 5.5 mg L^{-1}.

On January 15, 1965, an estimated 1,140 kg (2,500 pounds) of an aqueous solution of lithium hydroxide was spilled from an evaporator storage tank in Building 9201-5 at the Y-12 Plant (USAEC 1965). None of the material could be recovered. Runoff into EFPC would have been included in lithium concentrations measured at the Y-12 site boundary and reported in the Y-12 Plant Quarterly Reports.

Off-Site Surface Water Concentration Used in the Screening Assessment

The highest quarterly average lithium concentration measured in EFPC was used by the project team in the screening analyses for surface water pathways. The highest concentration, 17 mg L^{-1}, was measured during the last quarter of 1955. Soon after that, steps were taken to minimize losses of lithium from Y-12 lithium separation operations, and concentrations in EFPC decreased.

5.5.2.2.3 Lithium Concentrations in Soil and Sediment

Measurements of lithium concentrations in soils from areas around the Y-12 Plant are sparse. Results from only 25 soil samples were located. These samples were collected in February of 1993, and were limited to the area surrounding Upper East Fork Poplar Creek. Lithium concentrations in soil samples ranged from 14 mg kg^{-1} to 44.4 mg kg^{1}, with an average of 25 mg kg^{1} (standard deviation 7.4). By comparison, lithium concentrations in Tennessee soils have been reported to range from 15 to 93 mg kg^{-1}, with a mean of 30 mg kg^{-1} (standard deviation 23) (Dragun 1991).

Off-Site Soil Concentration Used in the Screening Assessment

The highest lithium concentration measured in soil in the EFPC floodplain (44.4 mg kg^{-1}) was used by the project team in the screening analyses for soil/sediment exposure pathways.

5.5.2.2.4 Lithium Concentrations in Food Items

No data on lithium concentrations in fish, vegetation, meat, or milk near the ORR were identified by the project team.

Off-Site Fish, Vegetation, Meat, and Milk Concentrations Used in the Screening Assessment

For the purposes of the screening analyses, concentrations of lithium in vegetation, meat, and milk were calculated using biotransfer factors that describe the transfer of lithium from other media, including air, water, and soil. The biotransfer factors used in this assessment are listed in Table 5-25. Concentrations of lithium in fish were calculated using a bioconcentration factor (BCF), shown in Table 5-25, that describes the transfer of lithium from water to fish.

Table 5-25: Biotransfer and Bioconcentration Factors for Lithium

Parameter	Value	Source
Soil to Vegetables (B_{veg})	0.001 (mg kg^{-1} wet)/(mg kg^{-1} dry)	NCRP 1996
Soil to Pasture ($B_{pasture}$)	0.1 mg kg^{-1} dry)/(mg kg^{-1} dry)	NCRP 1996
Biotransfer to Milk (F_m)	0.05 d L^{-1}	NCRP 1996
Biotransfer to Meat (F_f)	0.02 d kg^{-1}	NCRP 1996
Bioconcentration in Fish (BCF)	1 (mg kg^{-1})/(mg L^{-1})	NCRP 1996

5.5.3 Toxicity Assessment

The following section describes data characterizing the toxicity of lithium and summarizes the toxicity criterion used to evaluate ingestion exposure to lithium in the screening assessment. No dose-response data on the toxicity of inhaled lithium were located by the project team; therefore, the same toxicity criterion was used for both the ingestion and inhalation exposure pathways.

5.5.3.1 Ingestion and Inhalation Exposures

Lithium toxicity can occur at doses close to therapeutic levels used medically to treat sufferers of manic depression. Early signs of lithium toxicity are diarrhea, vomiting, tremor, and mild ataxia (muscular incoordination, drowsiness, and muscular weakness). Clinical signs of toxicity at higher levels are giddiness, ataxia, blurred vision, ringing in the ears, and increased output of dilute urine. Blood serum levels of lithium are closely related to toxicity, and are used to monitor doses in patients taking lithium therapeutically. Lithium is primarily excreted in urine, with insignificant excretion in feces. The half-life of elimination of lithium is approximately 24 hours. Lithium is also excreted in human milk, and it is recommended that mothers taking lithium not nurse. Although teratogenicity has been demonstrated in submammalian species, no evidence of lithium-induced teratogenicity has been seen in rats, rabbits, or monkeys. However, data from lithium birth registries suggest an increase in cardiac and other anomalies. Therefore, it is recommended that lithium be withdrawn for the first trimester of pregnancy unless it is determined that this would seriously endanger the mother (PDR 1995).

The USEPA has not established a toxicity criterion for exposure to lithium. Consequently, an RfD was derived based on the therapeutic dose. The maximum daily maintenance dose of lithium carbonate (to control mania, a manifestation of manic-depressive illness) is 1,200 mg (PDR 1995). This equates to 280 mg of lithium, since by weight lithium is 19% of lithium carbonate. Assuming an individual weighs 70 kg, this daily intake would correspond to a dose of 3.3 mg kg^{-1} d^{-1} . Application of a safety factor of 10 would yield a derived RfD as follows:

$$RfD_{dv} = \frac{1{,}200\ mg\ d^{-1} \times 0.19}{70\ kg \times 10} = 0.33\ mg\ kg^{-1}\ d^{-1}$$

This dose was used as an RfD to evaluate exposures to lithium through both the inhalation and ingestion pathways.

5.5.3.2 Toxicity Criteria Used in the Screening Assessment

Toxicity criteria used in the screening analyses to evaluate exposures to lithium at Oak Ridge are presented in Table 5-26.

Table 5-26: Toxicity Criteria for Lithium Used in the Screening Analyses

Exposure Route	Toxicity Criteria/Source	Value
Inhalation (Non-cancer)	Derived from therapeutic dose	3.3×10^{-1} mg kg^{-1} d^{-1}
Ingestion (Non-cancer)	Derived from therapeutic dose	3.3×10^{-1} mg kg^{-1} d^{-1}

5.5.4 Exposure Assessment

Following the Task 7 Screening methodology for Level I and Level II screening (ChemRisk 1996), doses were calculated for exposure to lithium in air, surface water, and soil/sediment. Exposures to lithium through ingestion of fish, vegetables, meat, and milk were evaluated using equations (presented in Appendix B) and biotransfer factors (discussed in Section 5.5.2.2.4) that describe the uptake of lithium from air, surface water, and soil/sediment into these media. A summary of the concentrations for each medium that were used in the screening is presented in Table 5-27.

The total average daily doses of lithium calculated for the inhalation and ingestion (oral) exposure routes for releases from Y-12 are summarized in Table 5-28. The doses calculated for individual pathways and the contribution of each pathway to the total dose are summarized in Appendix I.

Table 5-27: Summary of Environmental Concentrations of Lithium Used in Screening

Environmental Medium	Concentration	Source of Value	Typical Background[1]
Air	0.17 μg m^{-3}	Calculated from Y-12 stack release rate and ÷/Q for Scarboro	No data located
Surface Water	17 mg L^{-1}	Highest quarterly average lithium concentration in EFPC (last quarter 1955)	0.0003 to 0.002 mg L^{-1}
Soil/Sediment	44 mg kg^{-1}	Maximum concentration measured in EFPC floodplain soil	15 to 93 mg kg^{-1}
Fish	17 mg kg^{-1}	Calculated based on water concentration shown above and a BCF of 1	No data located

[1] For references, see Section 5.5.2.1

Table 5-28: Lithium Doses Calculated in the Level I and Refined Level I Screening (mg kg^{-1} d^{-1})

	Average Daily Inhalation Dose- (Noncarcinogen)	Average Daily Oral Dose- (Noncarcinogen)
Level I	2.0×10^{-5}	7.7×10^{-1}
Refined Level I	5.7×10^{-6}	9.7×10^{-2}

5.5.5 Risk Characterization

Non-cancer screening indices were calculated based on the screening estimates of average daily dose for lithium released from Y-12. The results of the screening risk characterization are presented below, and compared to risk-based decision guides established by the Oak Ridge Health Agreement Steering Panel (ORHASP).

5.5.5.1 Non-cancer Screening Indices Associated with Inhalation and Ingestion of Lithium

The following equation was used to calculate the non-cancer screening indices for inhalation and ingestion of lithium:

$$\mathit{Noncancer\ Screening\ Index} = \frac{\mathit{Average\ Daily\ Dose}\ (mg\ kg^{-1}d^{-1})}{\mathit{Reference\ Dose}\ (mg\ kg^{-1}d^{-1})}$$

At Level I, the non-cancer screening index for inhalation of lithium is:

$$\mathit{Noncancer\ Screening\ Index} = \frac{0.00002\ mg\ kg^{-1}d^{-1}}{0.33\ mg\ kg^{-1}d^{-1}} = 6.0\times10^{-5}$$

and the non-cancer screening index for ingestion of lithium is:

$$\mathit{Noncancer\ Screening\ Index} = \frac{0.77\ mg\ kg^{-1}d^{-1}}{0.33\ mg\ kg^{-1}d^{-1}} = 2.3$$

The non-cancer screening indices for inhalation and ingestion of lithium were then summed to give a Level I non-cancer screening index for both routes of exposure (2.3).

Because the Level I non-cancer screening index for inhalation of lithium was less than 1.0, the decision guide established by ORHASP for evaluating the need for further study of noncarcinogens, a Refined Level I screening was not conducted. Because the Level I non-cancer screening index for ingestion exceeded 1.0, a Refined Level I screening for ingestion of lithium was conducted.

At Refined Level I, the non-cancer screening index for ingestion of lithium is:

$$\mathit{Noncancer\ Screening\ Index} = \frac{0.097\ mg\ kg^{-1}d^{-1}}{0.33\ mg\ kg^{-1}d^{-1}} = 0.29$$

5.5.5.2 Comparison of Screening Indices to Decision Guides

The non-cancer screening indices calculated using the Level I and Refined Level I screening methodologies are presented in Table 5-29, and compared to the risk-based decision guides established by ORHASP for screening of contaminants released from the ORR.

Table 5-29: Results of the Level I and Refined Level I Screening of Lithium

	Non-cancer Screening Index	**Exceeds Decision Guide?** [a]
Level I	2.3	Yes
Refined Level I	0.29	No

a For noncarcinogenic chemicals, a hazard index of 1.0 was established by ORHASP as a decision guide for evaluating the need for further study (ChemRisk 1996).

5.5.6 Conclusions

The Level I evaluation of lithium releases indicates that off-site exposures may warrant further investigation. The Refined Level I evaluation indicates that past lithium releases do not warrant high priority in any further investigations of ORR releases.

It is important to note that the screening analyses presented here are limited by the lack of information regarding the chronic toxicity of lithium at environmental levels.

5.6 Neptunium-237 Releases from the Oak Ridge Reservation

Neptunium-237 (Np-237) was not evaluated in the Dose Reconstruction Feasibility Study quantitative screening because not enough information was located to conduct a screening (ChemRisk 1993). However, documents located in Phase II of the Oak Ridge Health Studies contain information concerning the mass of the recycled uranium received at the plant sites. Uranium recycle receipt information, combined with conservative estimates of the Np-237 concentrations in recycled uranium and release fractions estimated in Task 6, allowed for calculation of conservative Np-237 source terms and screening indices.

5.6.1 Neptunium-237 Use at the ORR

The Np-237 isotope was introduced to the K-25 and Y-12 sites as part of uranium recycling programs. Beginning in 1953, K-25 received recycled uranium from commercial reactors, the Savannah River and Hanford production reactors, and the Paducah Gaseous Diffusion Plant. In 1953, Y-12 also began receiving recycled uranium from Savannah River and the Idaho Chemical Processing Plant (Egli et al. 1985). Savannah River and Idaho processed the spent fuel from the reactors in order to either recover plutonium or enriched uranium. The processing resulted in uranium that was contaminated with trace quantities of transuranics and fission products. The uranium was then either sent to K-25 for enrichment or to Y-12 for parts fabrication.

5.6.2 Concentrations of Neptunium-237 in the Environment near the ORR

Neptunium-237 is present in the environment due to nuclear weapons testing (global fallout) and nuclear fuel reprocessing. The following sections describe typical background levels of Np-237 in the environment, and concentrations that have historically been measured in the environment near the ORR.

5.6.2.1 Background Levels

No information regarding background levels of Np-237 in air, water, or soil were located by the project team. Although Np-237 does not occur naturally, one might expect very low levels to be present in the environment primarily as a result of fallout from nuclear weapons testing.

5.6.2.2 Environmental Concentrations on or Near the ORR

Historical environmental data for Np-237 for the Oak Ridge Reservation are extremely limited; data do not begin to appear in site literature until the mid-1970s and available data are limited to Np-237 concentrations in soil and sediment. This section describes available environmental data, as well as information on Np-237 releases to air and water. In addition, determination of exposure point concentrations used in the screening assessment is described.

5.6.2.2.1 Neptunium-237 Concentrations in Air

No historical stack monitoring or ambient air monitoring data for Np-237 were located by the project team. Therefore, Np-237 source terms for K-25 and Y-12 were estimated based on information about uranium recycle material sent to each plant. The following discussions describe the methodology used to estimate airborne Np-237 releases from the K-25 and Y-12 sites.

Air Releases of Neptunium-237 from K-25

Very little information was found regarding the release or quantity of Np-237 present at the K-25 site. In the 1988 DOE Historical Release Report, DOE reported releases of Np-237 in liquid wastes for the 1979 to 1983 time period. However, DOE did not provide estimates for air releases of Np-237. Consequently, Np-237 releases were calculated by estimating the total annual Np-237 activity at K-25 in a given year and the fraction of the Np-237 that was released.

The total annual Np-237 activity released from K-25 to air in a given year was estimated using the following three-step process:

1. *Calculate the mass of recycled uranium received annually at K-25* from outside sources, based on data on recycled uranium received from individual sources or data on the concentration of technetium in the recycled uranium, and the total mass of technetium received;
2. *Calculate the Np-237 activity received annually at K-25* based on the mass of recycled uranium received annually and the specific activity of Np-237;
3. *Calculate the Np-237 activity released to air per year* based on the uranium release fraction and the assumption that the Np-237 release fraction was equivalent to the uranium release fraction.

These steps are described in greater detail below.

Calculate the mass of recycled uranium received annually at K-25

K-25 received recycled uranium from four sources: Paducah Gaseous Diffusion Plant, Hanford, the Savannah River Plant, and commercial reactors. The project team located information on the mass of recycled uranium that K-25 received annually from Paducah from 1953 to 1977 (Table 5-30), and on the total mass of technetium received in recycled uranium at K-25 from Savannah River, Hanford, and commercial reactors from 1953 to 1977 (Table 5-31). A material balance report prepared by ORGDP staff to evaluate the amount of Tc-99 entering and leaving K-25 (ORGDP, no date) indicates that the concentration of technetium in recycled uranium received at K-25 from government reactors (Savannah River and Hanford) was estimated at 5 g per ton (or about 7 ppm on U basis), and that the average concentration of technetium in recycled uranium from commercial reactors was 0.08 ppm. The report notes that all recycled uranium from Savannah River and Hanford had been received by 1962.

Table 5-30: Recycled Uranium Sent to K-25 from the Paducah Gaseous Diffusion Plant

Time Period	Recycled Uranium Received from Paducah (kg)
1953-1962	3.3×10^7
1963-1965	7.2×10^6
1966-1971	1.5×10^7
1972	3.5×10^6
1973	2.0×10^6
1974	2.2×10^6
1975	1.8×10^5
1976-1977	4.7×10^6
Total	$\mathbf{6.7\times10^7}$

Table 5-31: Mass of Technetium Received at K-25 from all Sources

Source of Technetium	Mass of Technetium Received (kg)
Paducah Gaseous Diffusion Plant	117
Savannah River Plant	56
Hanford	30
Commercial Reactors	<0.04
Total	**203**

From the information regarding the total mass of technetium received at K-25 and the concentration of technetium in the recycled uranium from Savannah River, Hanford, and commercial reactors, it was possible to estimate the mass of recycled uranium that K-25 received from these sources using the equation:

$$U\ (kg) = \frac{Tc-99\ (mg)}{Tc-99\ (mg\ kg^{-1}\ U)}$$

For example, if the mass of technetium received from Savannah River was 5.6×10^7 mg (Table 5-31) and the estimated technetium concentration in recycled uranium from this source was 7 mg kg^{-1}, then the estimated mass of recycled uranium received from Savannah River was:

$$U_{Savannah}\ (kg) = \frac{5.6 \times 10^7\ mg}{7\ mg\ kg^{-1}}$$

$$U_{Savannah} = 8.0 \times 10^6\ kg$$

Using the same approach, the estimated total masses of recycled uranium received from Hanford from 1953 to 1962 was 4.3×10^6 kg. The total estimated mass of recycled uranium received from Paducah, Savannah River, and Hanford was then divided evenly over the representative years to produce estimates of the annual average mass of recycled uranium received per year. For each year, estimates of the annual average mass of recycled uranium received from each site were then summed, to produce an estimate of the mass of recycled uranium received by the K-25 Plant from all sources during a given year (Table 5-32).

Calculate the Np-237 activity received annually at K-25

Estimates of the mass of recycled uranium received annually at K-25 were multiplied by the estimated Np-237 concentration in the recycled uranium to arrive at an annual estimate of total Np-237 activity at the K-25 site. Np-237 concentrations were calculated based on the uranium upper alpha activity limit of 200,000 dpm g^{-1} (Egli et al. 1985). Use of this value assumes that all of the alpha activity in the recycled uranium is Np-237. This assumption likely significantly overestimates the Np-237 activity present at K-25, since alpha activity in uranium is a result of the uranium, plutonium, and thorium as well as neptunium.

Table 5-32: Estimated Mass of Recycled Uranium Received at K-25 from All Sources

Year	Mass Recycled Uranium Received at K-25 (10^6 kg)			
	Paducah	**Savannah River**	**Hanford**	**Total**
1953	3.31	0.80	0.43	4.53
1954	3.31	0.80	0.43	4.53
1955	3.31	0.80	0.43	4.53
1956	3.31	0.80	0.43	4.53
1957	3.31	0.80	0.43	4.53
1958	3.31	0.80	0.43	4.53
1959	3.31	0.80	0.43	4.53
1960	3.31	0.80	0.43	4.53
1961	3.31	0.80	0.43	4.53
1962	3.31	0.80	0.43	4.53
1963	2.41	0	0	2.41
1964	2.41	0	0	2.41
1965	2.41	0	0	2.41
1966	2.41	0	0	2.41
1967	2.41	0	0	2.41
1968	2.41	0	0	2.41
1969	2.41	0	0	2.41
1970	2.41	0	0	2.41
1971	2.41	0	0	2.41
1972	3.45	0	0	3.45
1973	1.96	0	0	1.96
1974	2.23	0	0	2.23
1975	0.18	0	0	0.18
1976	2.37	0	0	2.37
1977	2.37	0	0	2.37

The alpha activity limit, 200,000 dpm g^{-1}, was converted to nCi g^{-1} as follows:

$$\frac{200{,}000\ dpm\ g^{-1}}{3.7\times10^{10}\ dps\ Ci^{-1}} \times 60\ s\ \mathrm{min}^{-1} \times 1\times10^{9}\ nCi\ Ci^{-1}$$

$$= 90.1\ nCi\ g^{-1}$$

The Np-237 activity at K-25 was then estimated based on the specific activity of Np-237 as follows:

$$Np{-}237\ Activity = U\ (g) \times Np{-}237\ (nCi\ g^{-1})$$

Thus, in 1953, the Np-237 activity in recycled uranium at K-25 was:

$$Np{-}237\ Activity = 4.53\times10^{9}\ g \times 90.1\ nCi\ g^{-1}$$

$$Np{-}237\ Activity = 4.08\ Ci$$

Calculate the Np-237 activity released to air per year

The next step in defining the Np-237 source term was to estimate the fraction of total Np-237 that was released to the atmosphere. Release fractions for Np-237 were based on uranium release fractions presented in the Oak Ridge Health Studies Task 6 uranium report. The Task 6 uranium report provides estimates of atmospheric releases of uranium from K-25 for each year. Uranium release fractions were calculated by dividing the mass of uranium released by the total mass of uranium at the site during that year (USDOE 1995c):

$$U\ Release\ Fraction = \frac{Annual\ U\ Release\ Mass}{Annual\ U\ Total\ Mass}$$

An example of this calculation for 1953 is shown below:

$$U\ Release\ Fraction\ (1953) = \frac{1{,}287\ kg}{4{,}749{,}238\ kg} = 0.00027$$

The uranium release fraction was calculated for each year from 1953 to 1977. For these years, the calculated release fraction varied between 0.001% and 0.027%. The estimated K-25 Np-237 activity inventory for each year was then multiplied by the uranium release fraction for that year to give an estimate of the Np-237 activity that was released to air:

$$Np{-}237\ Release\ Activity = Np{-}237\ Activity \times U\ Release\ Fraction$$

The results of the analysis of Np-237 releases to air from the K-25 facility are provided in Appendix J.

Air Releases of Neptunium-237 from Y-12

The project team located data on the total amount of recycled uranium received at Y-12 from the Idaho Chemical Processing Plant and the Savannah River Plant for each year from 1953 to 1984 (Appendix K) (Egli et al. 1985). Similar to the K-25 calculations, the uranium upper alpha activity limit of 200,000 dpm g^{-1} was used to estimate the maximum Np-237 activity inventory at the Y-12 site (Egli et al. 1985).

Np-237 releases to air from Y-12 were estimated by calculating a release fraction from the inventory differences for natural uranium reported in Owings (1995). The calculated natural uranium release fraction based on inventory differences was 1.0×10^{-3} (or 0.1%). As the inventory difference value does not distinguish between releases to either air or water, the project team relied upon its knowledge of the uranium processing at Y-12 to estimate the fraction of the inventory difference that might have been released to air and water. In this analysis, it was assumed that one quarter of the 0.1% inventory difference was released to the air, while three quarters was released to water. The estimated release fraction to air (0.025%) was then multiplied by the Y-12 Np-237 activity inventories to result in yearly release estimates to air. The results of this analysis are presented in Appendix K.

Off-Site Air Concentrations Used in the Screening Assessment

Np-237 air concentrations at the location of the Union/Lawnville subdivision resulting from airborne Np-237 releases from K-25 were estimated using the X/Q described in Section 2.3.4.2 (ChemRisk 1997). Annual average release rates (in Ci s^{-1}) were multiplied by the Union/Lawnville X/Q (7.4×10^{7} s m^{-3}) to give estimated Np-237 concentrations at the Union/Lawnville location for each year from 1953-1995. To simplify the analysis, the average Np-237 concentration at Union/Lawnville, 3.8×10^{-4} pCi m^{-3}, for the 1953-1995 time period was used in the screening assessment to estimate exposures to airborne Np-237 released from K-25 during these years.

Np-237 air concentrations at the Scarboro Community resulting from airborne Np-237 releases from Y-12 were estimated using the χ/Q described in Section 2.3.4.1 (ChemRisk 1997). Release rates (in Ci s^{-1}) were multiplied by the Scarboro χ/Q (3×10^{-7} s m^{-3}) to give estimated Np-237 concentrations at the Scarboro location for each year from 1953-1995. To simplify the analysis, the average Np-237 concentration at Scarboro, 6.6×10^{-7} pCi m^{-3}, for the 1953-1995 time period was used in the screening assessment to estimate airborne Np-237 released from Y-12 during these years.

5.6.2.2.3 Neptunium-237 Concentrations in Surface Water

No historical data on neptunium concentrations in or releases to Poplar Creek, EFPC, or the Clinch River prior to the 1970s were identified by the project team. More recent summary-level data were located in the Historical Radionuclide Releases report (USDOE 1988), the ORR annual environmental monitoring reports, and the East Fork Poplar Creek Sewer Line Beltway Study (SAIC 1993). Data on Np-237 releases to surface water from K-25 and Y-12 are described in the following sections.

Water Releases of Neptunium-237 from K-25

Estimates of waterborne neptunium releases from K-25 were provided in the DOE K-25 Historical Radionuclide Releases report (USDOE 1988) and the ORR annual environmental monitoring reports. The Historical Radionuclide Releases report provides Np-237 waterborne release estimates from K-25 for 1979 to 1983 and the annual reports provide waterborne release estimates for transuranics from the ORR for 1973 to 1986. The reported Np-237 and transuranic releases to water are provided in Table 5-33.

An estimate of the fraction of transuranic releases from the ORR that was Np-237 released from K-25 was calculated by dividing the reported releases of Np-237 from K-25 during 1980 to 1983 by the reported releases of transuranics from the entire site during these same years (Table 5-34). Np-237 release estimates from K-25 for 1973 to 1978 were then calculated by multiplying the largest calculated fraction (0.056) by the transuranic release estimates for 1973 to 1978 (Table 5-35).

Table 5-33: Reported Np-237 and Transuranic Releases to Water

Year	Np-237 Release to Water from K-25 (Ci)[a]	Transuranics Release to Water from ORR (Ci)[b]
1973	NA	0.08
1974	NA	0.02
1975	NA	0.02
1976	NA	0.01
1977	NA	0.03
1978	NA	0.03
1979	0.0015	NA
1980	0.0014	0.04
1981	0.0021	0.043
1982	0.0019	0.034
1983	0.0004	0.048

NA	Not available
a	Reported in the Historical Radionuclide Releases report (USDOE 1988)
b	Reported in the Annual Environmental Monitoring Reports (MMES 1973-1983)

Table 5-34: Calculation of Np-237 Fraction of Transuranic Releases

Year	Np-237 Release to Water from K-25 (Ci)	Transuranic Release to Water from ORR (Ci)	Np-237 Fraction of Transuranics
1980	0.0014	0.04	0.035
1981	0.0021	0.043	0.049
1982	0.0019	0.034	0.056
1983	0.0004	0.048	0.0083

Table 5-35: Estimated Np-237 Releases to Water from K-25 for 1973 to 1978

Year	Transuranic Release to Water from ORR (Ci)	Assumed Np-237 Fraction of Transuranics	Estimated Np-237 Release to Water from K-25 (Ci)
1973	0.08	0.056	0.0045
1974	0.02	0.056	0.0011
1975	0.02	0.056	0.0011
1976	0.01	0.056	0.00056
1977	0.03	0.056	0.0017
1978	0.03	0.056	0.0017

For the years that no data were available regarding transuranic or Np-237 releases to water (1953-1972 and 1984-1995), annual Np-237 releases to water were assumed to be equal to the 95% UCL of measured and estimated Np-237 releases for 1973 to 1983, 0.0022 Ci. This is thought to be a representative value because 1973 to 1983 was a period of active equipment decontamination and barrier replacement. Much of the Np-237 was known to deposit out on the equipment surfaces and would not have been released to the environment until the equipment was taken off-line and decontaminated.

Water Releases of Neptunium-237 from Y-12

As with Np-237 air releases from Y-12, very little data regarding Np-237 releases to off-site surface waters from Y-12 were located. Therefore, an approach similar to the Y-12 source term estimates for Np-237 in air was used to estimate Np-237 releases to off-site surface waters. As stated previously, the calculated natural uranium release fraction based on inventory differences was 1.0×10^{-3} (or 0.1%). This value does not distinguish between releases to air or water. The project team estimated the fraction of the inventory difference that might have gone to air and water, based on knowledge of uranium processing (Task 6). In this analysis, it was assumed that one quarter of the 0.1% inventory difference was released to air, while three quarters was released to water. This value (0.075%) was then multiplied by the Y-12 Np-237 activity inventories to result in yearly release estimates to water (presented in Appendix K).

Off-Site Surface Water Concentrations Used in the Screening Assessment

Concentrations of Np-237 in the Clinch River near the K-25 site were calculated based on the total annual releases. Surface water dilution was approximated by dividing the estimated release rate by the Clinch River flow rate. This method is inherently conservative and does not account for settling and dispersion. The flow rate for the Clinch River was obtained from early USGS data summaries for rivers (Surface Water Supply of the United States, USGS). A median flow rate of 4,500 $ft^3\ s^{-1}$ (1.1×10^{10} L d^{-1}) was estimated from data available for 1953 to 1961. This flow rate was measured at the right bank of the Clinch River, 0.6 miles downstream of Beaver Creek, 2.5 miles due south of Scarboro. Dividing the release rate by the flow rate generates a surface water concentration in Ci L^{-1}. The overall annual average concentration of 5.1×10^{-4} pCi L^{-1}, based on all years of release data, was used for the screening assessment.

A similar method was used to estimate Np-237 concentrations in EFPC as a result of releases from the Y-12 site. The flow rate for EFPC was assumed to be 8 million gallons per day (3.3×10^{-6} L d^{-1})**S** this flow rate was at the lower bound of measured flow rates in EFPC. The overall annual average Np-237 concentration of 2.0×10^{-2} pCi L^{-1} was determined in a similar fashion as that used to estimate K-25 surface water concentrations.

5.6.2.2.3 Neptunium-237 Concentrations in Soil/Sediment

Three sources of off-site soil and sediment data were located for Np-237. Summary level data for the 1990s were located in the Annual Environmental Surveillance Reports (MMES 1993-1995) and the East Fork Poplar Creek Sewer Line Beltway Study (SAIC 1993). Additionally, raw data were located in a compilation of Clinch River and Poplar Creek bottom sediment data for 1975 to 1981(MMES 1981). A summary of the soil and sediment data follows.

- *Beginning in the mid-1970s, surface sediment samples were collected semiannually at several locations in Poplar Creek and the Clinch River*. Np-237 concentrations in sediments in the Clinch River between 1975 and 1981 ranged from <20 to 50 pCi kg^{-1}. The maximum Clinch River concentration was measured in 1977 about two miles below the confluence of Poplar Creek and the Clinch River.

- *Soil samples in the K-25 area were reported in the Annual Environmental Monitoring Reports (MMES 1993-1995) beginning in 1993.* The Np-237 concentrations ranged from less than the detection limit to 14 pCi kg^{-1}.

- *Np-237 concentrations in soil and sediments were measured as part of the East Fork Poplar Creek Remedial Investigation (EFPC RI) (1990 and 1991).* Np-237 concentrations in sediment and soil ranged from below the detection limit to 170 pCi kg^{-1}, with a mean of 18 pCi kg^{-1}. Data for individual samples are presented in the LMES OREIS database (LMES OREIS 1997).

- *Soil samples collected near Y-12 were reported in the Annual Monitoring reports beginning in 1993 (MMES 1993-1995).* These measurements range from below the detection limit to 11 pCi kg^{-1}.

Off-Site Soil/Sediment Concentration Used in the Screening Assessment

For the screening assessment of Np-237 releases from K-25, the highest sediment concentration (50 pCi kg^{-1}) reported in the Clinch River (MMES 1981) was used by the project team in the screening analysis for the soil/sediment pathways.

For Y-12, the highest sediment concentration (170 pCi kg^{-1}) reported in EFPC RI was used by the project team in the screening analysis for the soil/sediment pathways.

5.6.2.2.4 Neptunium-237 Concentrations in Food Items

No data were identified by the project team describing measured concentrations of Np-237 in fish, vegetables, meat, or milk.

Off-Site Fish, Vegetation, Meat, and Milk Concentrations Used in the Screening Assessment

For purposes of the screening analyses, concentrations of Np-237 in vegetation, meat, and milk were calculated using biotransfer factors that describe the transfer of Np-237 from other media, including air, water, and soil. The biotransfer factors used in this assessment are listed in Table 5-36. For purposes of the screening analyses, concentrations of Np-237 in fish were calculated using a bioconcentration factor (BCF) that describes the transfer of Np-237 from water to fish. The BCF used in this assessment is 30 (mg kg^{-1})/(mg L^{-1}) (NCRP 1996). Use of this BCF and the water concentration described in Section 5.6.2.2.2 (0.0204 pCi L^{-1}) predicts a fish concentration of 10 mg kg^{-1}.

Table 5-36: Biotransfer and Bioconcentration Factors for Np-237

Parameter	Value	Source
Soil to Vegetables (B_{veg})	0.02 (mg kg^{-1} wet)/(mg kg^{-1} dry)	NCRP 1996
Soil to Pasture ($B_{pasture}$)	0.1 (mg kg^{-1} dry)/(mg kg^{-1} dry)	NCRP 1996
Biotransfer to Milk (F_m)	0.00001 d L^{-1}	Ng et al. 1977
Biotransfer to Meat (F_f)	0.001 d kg^{-1}	NCRP 1996
Bioconcentration in Fish (BCF)	30 (mg kg^{-1})/(mg L^{-1})	NCRP 1996

5.6.3 Toxicity Assessment

The USEPA classifies all radionuclides as Group A carcinogens based on their property of emitting ionizing radiation and on the extensive weight of evidence provided by epidemiological studies of radiation-induced cancers in humans.

The International Commission on Radiological Protection (ICRP) recommended in 1990 that a probability of 5% per sievert (0.05% per rem) be used as an estimate of the probability of induced fatal cancer in populations of all ages. A smaller value of about 4% per sievert (0.04% per rem) is recommended for a working population aged 20-64 years. These estimates are primarily made for exposure to low dose, low dose rate, and low linear energy transfer (LET) radiation. A estimate of the probability of fatal cancer for other exposure conditions, 0.07% per rem, is recommended for "detriment" due to radiation exposure of the whole body at low doses. There are four main components of detriment considered by the ICRP: the risk of cancer in all relevant organs, expected years of life lost for induced cancer, morbidity resulting from induced non-fatal cancers, and risk of serious hereditary disease in all future generations descended from the exposed individual.

Neptunium-237 Dosimetric Concerns

All chemical forms of neptunium are relatively insoluble with regard to transfer through the gastrointestinal tract. Because Np-237 is primarily an alpha emitter, the major radiological hazard is with internalized deposition. The ICRP 30 published value of f_1=0.01 is believed to be an over-estimation of the solubility of neptunium. A more accurate value is f_1=0.001, as recommended by *Federal Guidance Report 11*. Lung clearance is Class W. The primary systemic transfer is to the bone (0.60) and the liver (0.15). The major dosimetric considerations of ingested Np-237 are the bone surfaces and liver; the doses per unit uptake of ingested Np-237 to bone surfaces and liver are 7.0×10^2 rem mCi^{-1} and 1.5×10^2 rem mCi^{-1}, respectively. For inhaled Np-237, the dose per unit uptake to bone surfaces and liver are 9.0×10^2 rem mCi^{-1} and 2.0×10^2 rem mCi^{-1}, respectively. The biological half-life of Np-237 is 100 years on bone surfaces and 40 years in the liver.

For radionuclides, radiation doses result either from the intake of quantities of the radionuclide into the body, or from being immersed in or in close proximity to the radionuclide in such a way that radiation that is emitted is absorbed in body tissue. In the former case, radiation committed effective dose equivalents, in Sieverts (Sv), are calculated by multiplying the quantity taken into the body (in becquerel, Bq) times a dose conversion factor. In the latter case, effective dose equivalent rates (e.g., Sv y^{-1}) are calculated by multiplying the radionuclide concentration in the contaminated medium (air, water, or surface soil) times a dose conversion factor. The dose conversion factors used for Np-237 in this screening assessment are presented in Table 5-37.

Table 5-37: Dose Conversion Factors for Np-237 Used in the Screening Analysis

Exposure Route	Dose Conversion Factor $(Sv\ m^{-3})/(Bq\ y^{-1})$
Inhalation	2.30×10^{-5}
Ingestion	1.10×10^{-7}
Air Immersion	2.96×10^{-7}
Water Immersion	6.52×10^{-10}
External Irradiation	1.59×10^{-10}

5.6.4 Exposure Assessment

Following the Task 7 Screening methodology for Level I and Level II screening (ChemRisk 1996), doses were calculated for exposure to Np-237 in air, surface water, and soil/sediment. Exposures to Np-237 through ingestion of fish, vegetables, meat, and milk were evaluated using equations (presented in Appendix B) and biotransfer factors (discussed in Section 5.6.2.2.4) that describe the uptake of Np-237 from air, surface water, and soil/sediment into these media. A summary of the concentrations for each medium that were used in the screening is presented in Table 5-38.

The radiation dose commitments for each pathway from Level I screenings are presented in Table 5-39. The total doses summed across all pathways for exposure to Np-237 released from K-25 and Y-12 were 2.3×10^{-6} Sv y^{-1} and 2.2×10^{-6} Sv y^{1}, respectively, for Level I. Contributions of each pathway to the total dose are presented in Appendix I.

Table 5-38: Summary of Environmental Concentrations of Np-237 used in Screening

Environmental Medium	Concentration	Source of Value	Typical Background
K-25			
Air	3.8×10^{-4} pCi m^{-3}	Estimated based on Np-237 levels in recycled uranium, an estimated release fraction, and dispersion modeling to Union/Lawnville	Not `available
Surface Water	5.1×10^{-4} pCi L^{-1}	Estimated from releases to water and Clinch River flow rates	Not available
Soil/Sediment	5.0×10^{1} pCi kg^{-1}	Highest sediment concentration in Clinch River (1977) below confluence with Poplar Creek	Not available
Fish	1.5×10^{-2} pCi kg^{-1}	Calculated based on the water concentration shown above and a BCF of 30	Not available
Y-12			
Air	6.6×10^{-7} pCi m^{-3}	Estimated based on Np-237 levels in recycled uranium, an estimated release fraction, and dispersion modeling to Scarboro	Not available
Surface Water	2.0×10^{-2} pCi L^{-1}	Estimated from releases to water and EFPC flow rates	Not available
Soil/Sediment	1.7×10^{2} pCi kg^{-1}	Highest sediment concentration in EFPC (1990-91)	Not available
Fish	6.0×10^{-1} pCi kg^{-1}	Calculated based on the water concentration shown above and a BCF of 30	Not available

Table 5-39: Np-237 Doses Calculated in the Level I Screen (Sv y^{-1})

	Total Dose
K-25	
Level I	2.3×10^{-6}
Y-12	
Level I	2.2×10^{-6}

5.6.5 Risk Characterization

Screening indices associated with screening level estimates of Np-237 dose commitments were evaluated for all pathways combined for releases from K-25 and Y-12. The results of the screening risk characterization are presented below, and compared to risk-based decision guides established by the Oak Ridge Health Agreement Steering Panel (ORHASP).

5.6.5.1 Cancer Screening Indices

Once intakes or exposures to Np-237 were translated into radiation dose commitments, estimates of excess lifetime risk of cancer or other health effects can be obtained by applying an appropriate dose-to-risk conversion factor. For the purpose of this screening, the dose-to-risk coefficient of 7.3% Sv^{-1} was used. Based on recommendations of the International Commission on Radiological Protection, this coefficient represents the probability of fatal cancer, non-fatal cancer, or severe hereditary effects in the whole population per unit radiation dose (ICRP 1992). The resulting estimate of excess annual risk of cancer was multiplied by the total number of years of potential exposure (1953-1995, 43 years) to produce a estimate of excess lifetime risk of cancer).

$$Cancer\ Screening\ Index = Committed\ Dose\ (Sv\ y^{-1}) \times Dose\ Conversion\ Factor\ (Sv^{-1}) \times 42\ y$$

5.6.5.1.1 Cancer Screening Indices Associated with Releases of Np-237 from K-25

At Level I, the cancer screening index for releases of Np-237 from K-25 was:

$$Cancer\ Screening\ Index = 2.3\times10^{-6}\ (Sv\ y^{-1}) \times 7.3\%\ (Sv^{-1}) \times 42 = 7.3\times10^{-6}$$

Since the Level I cancer screening index for releases from K-25 was below 1×10^{-4}, the decision guide established by ORHASP, a Refined Level I screen was not conducted.

5.6.5.1.2 Cancer Screening Indices Associated with Releases of Np-237 from Y-12

At Level I, the cancer screening index for releases of Np-237 from Y-12 was:

$$\mathit{Cancer\ Screening\ Index} = 2.2\times10^{-6}\ (Sv\ y^{-1}) \times 7.3\%\ (Sv^{-1}) \times 42\ y = 6.8\times10^{-6}$$

Since the Level I cancer screening index for releases from Y-12 was below 1×10^{-4}, the decision guide established by ORHASP, a Refined Level I screen was not conducted.

5.6.5.2 Comparison of Screening Indices to Decision Guides

The screening indices calculated using the Level I screening methodology for exposure to Np-237 are presented in Table 5-40, and compared to the risk-based decision guides established by ORHASP for screening of contaminants released from the ORR.

Table 5-40: Results of the Level I Screen for Np-237

	Cancer Screening Index for K-25 Releases	**Exceeds Decision Guide?** [a]	**Cancer Screening Index for Y-12 Releases**	**Exceeds Decision Guide?**[a]
Level I	7.3×10^{-6}	No	6.8×10^{-6}	No
Refined Level I	Not necessary	No	Not necessary	No

[a] For radionuclides, a lifetime excess cancer risk of 10^{-4} was established by ORHASP as a decision guide for evaluating the need for further study (ChemRisk 1996)

5.6.6 Conclusions

In this screening analysis, estimates of exposures of off-site residents to Np-237 in air and water released from the K-25 and Y-12 Plants led to screening indices that are below the decision guides in use on the project. It is important to note that this analysis is limited by the lack of information regarding environmental levels of Np-237 in soil and water and the inventory of Np-237 at either plant. In the future if such information becomes available, this analysis could be refined. However, it should be noted that, even in Refined Level I screening, the analysis maintains considerable conservatism. In the Refined Level I methodology used in this assessment, somewhat different (less conservative) exposure parameters were used than in the Level I methodology; however, the environmental concentrations and biotransfer factors remained the same. Both the biotransfer factors and the environmental concentrations add considerable conservatism to the analysis. Given the conservatism built into this analysis, the screening indicates that historical Np-237 releases do not warrant a high priority for further evaluation.

5.7 Nickel Releases from the Oak Ridge Reservation

In the Dose Reconstruction Feasibility Study quantitative screening evaluation, nickel was not identified as a "highest priority contaminant for further study" (ChemRisk 1993). In the Feasibility Study, nickel was evaluated as a noncarcinogen based on concentrations measured in soil and water downstream from the Oak Ridge Reservation (ORR). Because the USEPA has not identified nickel as a carcinogen via ingestion, exposure to nickel was evaluated by comparing doses to noncarcinogenic toxicity criteria. However, information gathered during Phase II indicates that nickel, historically used at the ORR in the production of barrier material in Buildings K-1037 and K-1041 at the K-25 Plant, may have been released to air from the K-25 Plant as a respirable dust. Since some forms of airborne nickel dusts (e.g., nickel refinery dust) have been identified by the USEPA as inhalation carcinogens, nickel was identified as warranting a reevaluation as part of the Task 7 review.

5.7.1 Nickel Use at K-25

Nickel was used in large quantities at the K-25 Plant in the production of barrier material for the gaseous diffusion process. The barrier manufacturing facilities included the main production facility in Building K-1037 and the Converter Retubing and Assembly Area (the "barrier maintenance shop") in Building K-1401. The Oak Ridge barrier manufacturing plant was the sole manufacturer of process barrier in the country.

Development of the barrier manufacturing program at Oak Ridge was initiated in April 1944, when the Carbide & Carbon Chemicals Company assumed overall management of the United States gaseous diffusion barrier development and production program. The Oak Ridge barrier plant was placed into operation in December 1947 (Handler 1991). The facility continued to operate until 1981, when the production equipment was shut down and placed on standby upon completion of the Cascade Improvement/ Upgrading Program (MMES 1992).

The K-1037 building housed equipment to produce barrier in multiple parallel production lines Production involved the continuous handling of ton quantities of nickel powder. This fine powder was widely dispersed throughout the production area (MMES 1992). The K-1037 Blend Tower was equipped with ventilation designed to exhaust dust escaping from the mechanical equipment. This tower was sized to exhaust a minimum of 150 ft^3 min^{-1} per cubic foot of opening (UCC 1971). K-1037-C housed a smelter that was provided to smelt "off-spec" scrap nickel barrier and produce ingots for recycle. The smelter "off gas" was discharged through large scrubber type vessels located on the east end of the second floor of the barrier production area (MMES 1992).

Between 1971 and 1981, barrier used in the gaseous diffusion process was replaced. During this Cascade Improvement/ Upgrading Program, production of barrier at the K-25 site was at its peak, and releases of nickel from the K-25 site were considered to be at their highest. Following shutdown of barrier production in 1981, ventilation ducting was isolated and sealed and partitioning provided to confine nickel powder to the standby production area (MMES 1992). This area is currently monitored by Site Facility Operations (MMES 1992).

The barrier manufacturing process is not discussed in this document because certain aspects remain classified.

5.7.2 Concentrations of Nickel in the Environment near the ORR

Nickel may be present in the environment from both natural and human-related sources. The following sections describe typical natural, or background, levels of nickel in the environment, and concentrations that have historically been measured in the environment near the ORR.

5.7.2.1 Background Levels

Nickel is found naturally in soils, and may be present in air from a number of natural sources including soil particulates, volcanoes, and forest fires. Several sources of data on background concentrations of nickel were identified by the project team. These include general background concentrations and near-site background concentrations measured in areas of Eastern Tennessee assumed to not be impacted by ORR activities.

Data on background concentrations of nickel were identified by the project team and are summarized briefly by media below.

- *Air* - Background concentrations of nickel in air in remote areas are reported to range from 0.00001 to 0.003 µg m^{-3}. In urban areas having no metallurgical industry, concentrations range from 0.003 to 0.03 µg m^{-3} (HSDB 1997). In general, concentrations of nickel in air are higher in heavily industrialized areas and very large cities than rural areas (ATSDR 1992).

- *Surface Water* - Typical background concentrations of nickel in surface water average between 15 and 20 µg L^{-1} (ATSDR 1992). Nickel was not detected in any of the surface water samples collected in Norris Reservoir during the Clinch River Remedial Investigation (detection limit 19 µg L^{-1}; Cook et al. 1992). Norris Reservoir was identified in the CRRI as the "reference location," indicating that it was considered unaffected by releases from the ORR.

- *Soil/Sediment* - Background concentrations of nickel in Tennessee soils are reported to range from 5 to 70 mg kg^{-1} (Dragun and Chiasson 1991). In 1973 and 1982, the Tennessee Valley Authority (TVA) measured concentrations of several metals, including nickel, in sediments just upstream of a number of TVA dams. Reservoirs evaluated included several on the Tennessee River upstream of the ORR (including Fort Loudoun Reservoir) and several on different river systems (including Douglas

Reservoir on the French Broad River and Wilson Reservoir on Fleet Hollow Embayment). Nickel concentrations in sediments of reservoirs assumed not to be impacted by ORR activities ranged from <2.3 to 46 mg kg^{-1} (dry weight) (TVA 1986).

5.7.2.2 Environmental Concentrations on or near the ORR

Beginning in the late 1940s, building air monitoring for nickel was conducted by Oak Ridge Gaseous Diffusion Plant (ORGDP) personnel in barrier manufacturing areas. The earliest environmental samples identified by the project team were collected in 1975, when routine monitoring for nickel in surface water and sediment in waterways upstream and downstream of K-25 was initiated. Available data describing nickel concentrations in air, surface water, soil, sediment, and other environmental media on or near the ORR are discussed in the following sections. In addition, determination of exposure point concentrations used in the screening assessment is described.

5.7.2.2.1 Nickel Concentrations in Air

No stack releases data for nickel were located by the project team. Joyner and Marshall (1975) describe a sampling program conducted in 1975 to evaluate the amount of nickel dust being exhausted through two stacks servicing the K-1037 Blend Tower. The goal of the program was to implement reductions in releases through changes in bag filter design and/or in blend tower operations. Samples were collected at a rate of three to four per week from each stack, over a 4- to 6-week period. In addition, periodic samples were collected on the inlet stream to the exhaust bag filter to determine filter efficiency. These data, however, were not located by the project team.

Sampling of workplace air was conducted routinely in various areas of the barrier plant (in buildings K-1037 and K-1401) from 1948 to 1963. For example, during the first quarter of 1948, 30 samples were collected in K-1037, two samples were collected in the K-1401 cleaning area, and two samples were collected in the K-1401 maintenance shop. Two samples collected in K-1037 had concentrations above 500 µg m^{-3}; the remaining samples were below this value (Ketcham 1948). Between 1948 and 1963, a total of 3044 air samples were collected and analyzed for nickel content (Godbold and Tompkins 1978).

A summary of air data collected in seven areas of the plant is presented by Godbold and Tompkins in their report describing a long-term mortality study conducted to evaluate occupational exposures of ORGDP personnel to airborne nickel (Godbold and Tompkins 1978). Median and maximum concentrations in "Manufacturing Area 2" (the area with the highest airborne concentrations) were 500 µg m^{-3} and 459,000 µg m^{-3}, respectively. The median of all samples was 130 µg m^{-3} (detection limit 100 µg m^{-3}). The routine sampling was discontinued in 1963 and thereafter performed on an "as-needed" basis only. Per Godbold and Tompkins, the results of these later samples were not retained.

Two studies of nickel concentrations in outside air on or near the K-25 site were undertaken by ORGDP staff beginning in the mid-1970s. From 1973 to 1980, data were collected weekly at four locations on the K-25 site, approximately north, south, east, and west of the center of the plant (Weber and White 1977). The sampler locations were as follows:

- "East", located approximately 300 feet east of the K-1037 building
- "West", located approximately 500 feet west of the K-31 building
- "North", located approximately 300 feet north of the K-25 building
- "South", located approximately 200 feet north of the K-1007 building

Individual sample results were tabulated by ORGDP staff (ORGDP 1981b). These data were obtained and statistically evaluated by the project team. The highest average concentrations were measured at the East sampling location between 1977 and 1979. Airborne nickel levels at the East station ranged from <0.0018 to 10 µg m^{-3} (Weber and White 1977).

The other sampling program consisted of collecting and analyzing a limited number of atmospheric samples from five different regions of East Tennessee, in several directions from K-25. Locations of sample collection were Claxton, North Knox, West Knox, Townsend, and Sugar Grove. Nickel concentrations at these five sites during December 1976 were obtained by the project team. Airborne nickel concentrations ranged from less than detectable (<0.004 µg m^{-3}) to 0.132 µg m^{-3}. The highest concentration (0.132 µg m^{-3}) was measured at Townsend (about 22 miles south-southeast of Knoxville, near Great Smoky Mountains National Park) on December 9-10. The average concentration measured at the Townsend station over the month of December was 0.023 µg m^{-3}.

Off-Site Air Concentration Used in the Screening Assessment

No data describing measured air concentrations of nickel at the location of the nearest downwind receptor (Union/Lawnville) were identified by the project team. Therefore, concentrations at the Union/Lawnville receptor were estimated based on air concentrations measured at the East sampler location, the location nearest the K-1037 building. It was assumed that the relationship between on-site nickel concentrations and concentrations at the receptor was the same as the relationship between the air concentration modeled to the center of the K-25 site, based on a unit release of 1 g s^{-1}, and the air concentration modeled to the Union/Lawnville receptor location, as presented in the Task 6 report.

In the Task 6 report, the modeled air concentration at the K-1037 building corresponding to a unit release from the center of the K-25 site was 1.1×10^{-6} sec m^{-3}. The corresponding modeled air concentration at the Union/Lawnville population (UTM-X 733000, UTM-Y 3976000) was 7.4×10^{-7} sec m^{-3}. Based on these two values, the ratio of downwind to on-site air concentrations was calculated as follows:

$$Ratio = \frac{modeled\ downwind\ (Union/Lawnville)}{modeled\ onsite\ (near\ the\ East\ sampler)} = \frac{7.4\times10^{-7}\ \sec\ m^{-3}}{1.1\times10^{-6}\ \sec\ m^{-3}} = 0.67$$

The air concentration of nickel at the Union/Lawnville population was calculated by multiplying the above ratio by the 95% upper confidence limit (UCL) of air concentrations measured at the East sampler during 1977 (2.3 µg m^{-3}), the year when the highest average concentration was measured:

$$Union/Lawnville\ air\ concentration = 2.3\ \mu g\ m^{-3} \times 0.67 = 1.5\ \mu g\ m^{-3}$$

This calculated air concentration was used in the screening dose calculations to evaluate exposure via the air pathways.

5.7.2.2.2 Nickel Concentrations in Surface Water

No data on nickel concentrations in liquid effluent from the K-25 Plant prior to the 1970s were identified by the project team. Beginning at least as early as 1970, concentrations of nickel were measured weekly in surface water runoff from the K-1037 area where it enters Poplar Creek at discharge point K-1700. Concentrations in the effluent between approximately 1970 and 1976 was reported to range from 0.05 mg L^{-1} to 1.8 mg L^{-1}, with an average of approximately 0.38 mg L^{-1} (UCC 1977a). During this same time period, concentrations downstream in Poplar Creek were reported to average about 0.03 mg L^{-1}, with similar concentrations reported in the Clinch River (UCC 1977a). Individual sample data were not located by the project team.

Results from routine analyses of nickel in surface water near K-25 are reported in the annual environmental monitoring reports beginning in 1975 (UCC 1976). Surface water grab samples were collected weekly at locations upstream and downstream of K-25 and composited monthly for analysis for a number of materials, including nickel. Sample locations included one location in EFPC (at the outlet of New Hope Pond), two locations in Poplar Creek (upstream of K-25 and near the confluence of Poplar Creek with the Clinch River), and four locations in the Clinch River (at Melton Hill Dam, at the ORGDP sanitary water intake, at the ORGDP recirculating water intake, and at Center's Ferry near Kingston). Statistics presented in the annual reports include the minimum, maximum, mean, standard deviation of sample concentrations, and the number of samples collected. Maximum concentrations in the Clinch River between 1975 and 1982 ranged from <0.010 to 0.30 mg L^{-1}, with the highest average concentrations (approximately 0.070 mg L^{-1}) measured in 1975.

In addition to this routine monitoring program, nickel concentrations were also measured in surface water during several special monitoring programs. These programs include:

- *The Tennessee Valley Authority (TVA) Instream Contaminant Study (1984)–* Although the focus of this program was evaluating the transport of mercury in surface water and sediment downstream of Y-12, samples were also analyzed for nickel and other trace metals. Surface water sample locations included one in EFPC at the outfall of New Hope Pond, one in Poplar Creek upstream of K-25, and two in the Clinch River both above and below the Poplar Creek/ Clinch River confluence (TVA 1985a). Nickel concentrations ranged from 2 to 21 μg L^{-1}. Data from individual samples are presented in the TVA reports.

- *The Clinch River Remedial Investigation (1989 and 1990)–* This program was initiated by DOE in 1989 to address the transport, fate, and distribution of waterborne contaminants released from the ORR to the Clinch and Tennessee Rivers (Cook et al. 1992). Surface water, sediment, and fish samples were collected from 10 reaches, six of which are potentially affected by releases from the ORR, and four which serve as reference or background areas. Sample locations included Poplar Creek and the Clinch River. Nickel was not detected in any surface water samples (detection limit 19 μg L^{-1}). Data for individual samples are presented in the LMES OREIS database (LMES OREIS 1997).

- *The East Fork Poplar Creek Floodplain Remedial Investigation (EFPC RI) (1990 and 1991)–* Surface water, sediment, and floodplain soil samples were collected at several locations in and along EFPC. Nickel was not detected in any surface water samples (detection limit 19 μg L^{-1}). Data for individual samples are presented in the LMES OREIS database (LMES OREIS 1997).

Off-Site Surface Water Concentration Used in the Screening Assessment

The 95% UCL on the highest mean surface water concentration reported for the Clinch River was used by the project team in the screening assessment for the surface water pathways. Data collected in the Clinch are assumed to reflect concentrations to which off-site individuals may have been exposed.

The highest reported surface water concentrations were measured as part of the routine surface water sampling presented in the annual environmental reports. The highest Clinch River mean, measured in 1975 at sampling site C-3 (at approximately CRM 14.5) was 0.070 mg L^{-1} (SD = 0.060, n = 12). The 95% UCL on this mean is 0.10 mg L^{-1}.

5.7.2.2.3 Nickel Concentrations in Soil/Sediment

Minimal data were identified by the project team describing nickel concentrations in surface soil near the K-25 site. Beginning in approximately 1971, surface soil samples were collected semiannually from several locations both near and distant to the K-1037 facility. Concentrations in surface soil within a 3,000 ft radius of K-1037 between approximately 1971 and 1976 are reported to range from 20 mg kg^{-1} to nearly 9,400 mg kg^{-1}, with an average of approximately 1,100 mg kg^{-1} (UCC 1977a). At more remote locations, such as Melton Hill Dam, the average concentration was reported to be about 45 mg kg^{-1}.

Surface sediment samples were collected semiannually at several locations beginning in the mid-1970s, including K-25 site holding ponds, Poplar Creek, and the Clinch River. Results from routine analyses of nickel in sediment near K-25 are reported in the annual environmental monitoring reports beginning in 1975 (UCC 1976). Samples were collected semiannually at up to 12 sites on Poplar Creek, both above and below K-25 plant discharge points, and up to two sites on the Clinch River. Summary data (i.e., annual averages) from this program are presented in the annual environmental monitoring reports. Average sediment concentrations in the Clinch River between 1975 and 1982 ranged from 14 to 325 mg kg^{-1}. The maximum Clinch River concentration was measured in 1976 just upstream of the confluence of Poplar Creek and the Clinch River.

In addition to the routine monitoring program, nickel concentrations were also measured in sediments/soils as part of several special monitoring programs. These programs include:

- *A survey of sediments in streams surrounding the K-25 Plant by ORGDP staff (1985)*– Surface sediment samples were collected by ORGDP staff at 180 locations in the Clinch River, Poplar Creek, EFPC, and tributaries to Poplar Creek, to identify locations where contaminants were entering surface water systems (Ashwood et al. 1986). Nickel concentrations were measured at some of these samples, including three locations in the Clinch River both upstream and downstream of the Poplar Creek/ Clinch River confluence. Data for individual samples are presented in the Ashwood et al. report. Concentrations in the Clinch River ranged from 2 to 24 mg kg^{-1}.

- *The Tennessee Valley Authority (TVA) Instream Contaminant Study (1984)*– In addition to surface water samples, sediment samples were also collected and analyzed for a variety of contaminants including nickel. Surface sediment sample locations included the length of EFPC, three locations in Poplar Creek, and five locations in the Clinch River (TVA 1985b,c). Concentrations ranged from 14 to 66 mg kg^{-1}. Data for individual samples are presented in the TVA reports.

- *The Clinch River Remedial Investigation (1989 and 1990)*– In addition to surface water (described in previous section) and fish samples, sediment samples were collected from the 10 reaches. Sample locations included Poplar Creek and the Clinch

River. The maximum sediment concentrations in the Clinch River was 58 mg kg^{-1}. Data for individual samples are presented in the LMES OREIS database (LMES OREIS 1997).

- *The East Fork Poplar Creek Floodplain Remedial Investigation (EFPC RI) (1990 and 1991)*– Sediment and floodplain soil samples were collected at several locations in and along EFPC. Maximum nickel concentrations in sediment and soil were 76 mg kg^{-1} and 174 mg kg^{-1}, respectively. Data for individual samples are presented in the LMES OREIS database (LMES OREIS 1997).

Off-Site Sediment Concentration Used in the Screening Assessment

The highest mean sediment concentration reported for the Clinch River was used by the project team in the screening analyses for the soil/sediment pathways. The highest reported concentrations were measured as part of the routine sediment sampling presented in the annual environmental reports. The highest mean, measured at sampling site CS-1 just upstream of the confluence with Poplar Creek in 1976, was 325 mg kg^{-1}. This concentration exceeded maximum surface soil concentrations measured in the East Fork Poplar Creek floodplain, or average concentrations reported for the Melton Hill Dam area.

5.7.2.2.4 Nickel Concentrations in Food Items

Nickel concentrations in fish in waterways near the ORR were measured during several of the same programs in which surface water and sediment/soil samples were collected (e.g., the TVA Instream Contaminant Study and the Clinch River Remedial Investigation). The earliest data on nickel concentrations in fish identified by the project team were collected by Loar et al. (1981a) in 1979. Identified data describing nickel concentrations in fish include the following:

- *A biological sampling program to evaluate the effects of ORNL operations on aquatic biota in the White Oak Creek watershed (1979)*– Nickel and other metals were measured in fish from a number of locations including Clinch River Mile (CRM) 19 and CRM 22 (confluence of the Clinch River and Poplar Creek is CRM 12) (Loar et al. 1981a). Summary data (i.e., mean concentrations) are presented in the Loar et al. report. Mean concentrations in Clinch River fish ranged from 0.49 to 1.54 mg kg^{-1}.

- *The Tennessee Valley Authority (TVA) Instream Contaminant Study (1984)*– Fish were collected from several locations in the Clinch River and Poplar Creek, including CRM 2, 6, and 11 and PCM 0.2 (TVA 1985d). Data for individual samples are given in the TVA reports. The maximum reported nickel concentration is 1 mg kg^{-1}.

- *A TVA fish tissue screening study of Eastern Tennessee reservoirs (1987)*– Fish were collected and analyzed for a variety of contaminants, including nickel, to assess the general level of contamination in Tennessee reservoirs (TVA 1989). Sample locations included CRM 20, in Watts Bar Reservoir, and CRM 24, in Melton Hill Reservoir. Nickel was not detected in any of the Clinch River samples (detection limit 1 mg kg^{-1}). Data from individual samples are presented in the TVA report.

- *The Clinch River Remedial Investigation (1989 and 1990)*– Fish were collected at a number of locations including CRM 0.5 and CRM 9.5 (Cook et al. 1992). Nickel was not detected in any of the samples (detection limit 0.25 mg kg^{-1}). Data for individual samples are presented in the LMES OREIS database (LMES OREIS 1997).

Nickel occurs in most food items and consequently food items typically comprise the highest level of nickel intake (ATSDR 1992). No data were identified by the project team describing measured concentrations of nickel in vegetables, meat, or milk, although UCC (1977a) reports that vegetation samples were collected concurrently with surface soil samples beginning in about 1971. Nickel concentrations in vegetation within 3,000 ft of the K-1037 facility were reported to range from 3 to 1,040 mg kg^{-1}, with an average of about 172 mg kg^{-1} (UCC 1977a). Concentrations in vegetation at Melton Hill Dam averaged about 6 mg kg^{-1} nickel (UCC 1977a). It is not reported whether these vegetation data are reported on a wet weight or a dry weight basis. However, it is assumed that they reflect the dry weight of vegetation since vegetation data for other contaminants reported in the annual environmental reports are reported on a dry weight basis. Individual sample data were not located by the project team.

Off-Site Fish, Vegetation, Meat, and Milk Concentrations Used in the Screening Assessment

For purposes of the screening analyses, concentrations of nickel in vegetation, meat, and milk were calculated using biotransfer factors that describe the transfer of nickel from other media, including air, water, and soil. The biotransfer factors used in this assessment are listed in Table 5-41. For purposes of the screening analyses, concentrations of nickel in fish were calculated using a bioconcentration factor (BCF) that describes the transfer of nickel from water to fish. The BCF used in this assessment is 100 (mg kg^{-1})/(mg L^{-1}) (USEPA 1986b). Use of this BCF and the water concentration described in Section 5.7.2.2.2 (0.10 mg L^{-1}) predicts a fish concentration of 10 mg kg^{-1}. This concentration exceeds nickel concentrations measured in fish near the ORR in the late 1970s and later.

Table 5-41: Biotransfer and Bioconcentration Factors for Nickel

Parameter	Value	Source
Soil to Vegetables (B_{veg})	0.05 (mg kg^{-1} wet)/(mg kg^{-1} dry)	NCRP 1996
Soil to Pasture ($B_{pasture}$)	1.0 (mg kg^{-1} dry)/(mg kg^{-1} dry)	NCRP 1996
Biotransfer to Milk (F_m)	0.001 d L^{-1}	Ng et al. 1977
Biotransfer to Meat (F_f)	0.005 d kg^{-1}	NCRP 1996
Bioconcentration in Fish (BCF)	100 (mg kg^{-1})/(mg L^{-1})	USEPA 1986b

Using, for example, the $B_{pasture}$ value of 1.0 (mg kg^{-1} dry)/(mg kg^{-1} dry) with the soil concentration described in Section 5.7.2.2.3 (325 mg kg^{-1}) results in a vegetation concentration of 325 mg kg^{-1} (dry). This concentration is conservative compared to the average concentrations of nickel in vegetation reported for the Melton Hill Dam area (UCC 1977a).

5.7.3 Toxicity Assessment

Nickel is considered to be noncarcinogenic through most routes of exposure, including ingestion of nickel in food products and/or soil and dermal contact. While inhalation of nickel refinery dust through occupational exposures has been associated with an increased incidence of lung and nose tumors, inhalation of nickel metal has not been clearly associated with respiratory cancer. The following sections describe data characterizing the toxicity of nickel through different routes of exposure, and summarize the toxicity criteria used to evaluate exposure to nickel in the screening analyses.

5.7.3.1 Inhalation Exposure

Occupational exposure studies have shown an association between inhalation of nickel refinery dust and an increased incidence of lung and nose tumors. Because the refinery dust is a mixture of many nickel forms or species, the carcinogenic component has not been identified with certainty. It is thought, however, that the component that produces the carcinogenicity is nickel subsulfide or nickel oxide. No association between occupational exposure to nickel metal and cancer has been observed (ATSDR 1992). For example, a long-term mortality study of workers occupationally exposed to metallic nickel at the Oak Ridge Gaseous Diffusion Plant between 1948 and 1973 was undertaken to determine whether mortality from respiratory cancer among workers occupationally exposed to airborne metallic nickel at the ORGDP differed from that of workers at the same plant with no record of occupational exposure to metallic nickel or any nickel compound (Godbold and Tompkins 1978). The data were reported to show no evidence of an increased risk of mortality due to respiratory cancer among nickel-exposed workers.

Several animal studies show an increased incidence of carcinomas in rats by inhalation and injection of nickel refinery dust (IRIS 1999). Based on evidence from worker studies and the supporting animal studies in animals, the USEPA derived a cancer unit risk value for inhalation of nickel refinery dust of 2.4×10^{-1} (mg $m^{-3})^{-1}$. This unit risk value was converted to a cancer slope factor of 8.4×10^{-1}(mg kg^{-1} $d^{-1})^{-1}$ by multiplying by a body weight of 70 kg and dividing by an inhalation rate of 20 m^3 d^{-1}. The USEPA has not established a RfD for noncarcinogenic effects of inhaled nickel refinery dust (USEPA 1999). In addition, the USEPA has not established any toxicity criteria for inhalation of nickel metal (neither a RfD nor a cancer slope factor) (USEPA 1999).

5.7.3.2 Oral (Ingestion) Exposure

Chronic ingestion studies in which nickel salts were administered to rats, mice, dogs, or monkeys in drinking water or the diet at dietary concentrations up to 1000 mg kg^{-1} did not produce evidence of a carcinogenic response (IRIS 1997, ACGIH 1996). No human data on the carcinogenicity of nickel following ingestion are available (ATSDR 1992).

Nickel and its inorganic compounds are not absorbed through unbroken skin in amounts sufficient to cause systemic intoxication (ACGIH 1996). However, they may cause contact dermatitis in sensitized individuals. Surveys of the general population indicate a 2.5% to 5% prevalence of nickel sensitization.

The hematological system (i.e., blood) has been shown to be a target for oral exposure to nickel. Rats administered nickel salts showed increased white blood cell and platelet counts. Dogs fed nickel salts for two years at very high concentrations in the diet had histological lesions in the bone marrow at 2500 mg kg^{-1}, but not at 1000 mg kg^{-1}. Decreased body weight gain was observed in rats fed nickel sulfate in the diet for two years at 2500 and 1000 mg kg^{-1} nickel (ATSDR 1992). No decrease in body weight gain was observed at the 100 mg kg^{-1} dose level. This 100 mg kg^{-1} no observed adverse effect level (NOAEL), equivalent to 5 mg kg^{-1} d^{-1}, was used as the basis for the USEPA's noncarcinogenic RfD for ingestion of nickel soluble salts of 2.0×10^{-2} mg kg^{-1} d^{-1}. The USEPA has not established a cancer slope factor for ingestion of nickel soluble salts. In addition, the USEPA has not established any toxicity criteria for ingestion of nickel refinery dusts (neither a RfD nor a cancer slope factor) (USEPA 1999).

5.7.3.3 Toxicity Criteria Used in the Screening Analyses

In this assessment, the USEPA cancer slope factor for inhalation of nickel refinery dusts was used to evaluate exposures to airborne nickel, and the USEPA reference dose for ingestion of nickel soluble salts was used to evaluate exposures to nickel through all other pathways. The toxicity criteria used in the screening analyses to evaluate exposures to nickel at Oak Ridge are presented in Table 5-42.

Table 5-42: Toxicity Criteria for Nickel Used in the Screening Analysis

Exposure Route	Toxicity Criteria/ Source	Value
Inhalation (Cancer)	USEPA Inhalation Slope Factor for nickel refinery dust (USEPA 1999)	8.4×10^{-1} (mg kg^{-1} d^{-1})$^{-1}$
Ingestion (Non-cancer)	USEPA Oral Reference Dose for nickel soluble salts (USEPA 1999)	2.0×10^{-2} mg kg^{-1} d^{-1}

5.7.4 Exposure Assessment

Following the Task 7 Screening methodology for Level I and Level II screening (ChemRisk 1996), doses were calculated for exposure to nickel in air, soil/sediment, and surface water. Exposures to nickel through ingestion of fish, vegetables, meat, and milk were evaluated using equations (presented in Appendix B) and biotransfer factors (discussed in Section 5.7.2.2.4) that describe the uptake of nickel from air, surface water, and soil/sediment into these media. A summary of the concentrations for each medium that were used in the screening is presented in Table 5-43.

Table 5-43: Summary of Environmental Concentrations of Nickel used in Screening

Environmental Medium	Concentration	Source of Value	Typical Background[1]
Air	0.0015 mg m^{-3}	Calculated from 95% UCL value for year of highest measured concentrations in on-site air samplers (1977) and air dispersion modeling to Union/Lawnville	0.00001 to 0.003 µg m^{-3} (remote); 0.003 to 0.03 µg m^{-3} (urban)
Surface Water	0.10 mg L^{-1}	95% UCL value for year of highest measured concentrations at Clinch River mile 14.5 (1975)	0.015 to 0.020 mg L^{-1}
Soil/Sediment	325 mg kg^{-1}	Highest average concentration from routine sediment sampling (Clinch River near mouth of Poplar Creek, 1976)	5 to 70 mg kg^{-1} (soil); <2.3 to 46 mg kg^{-1} (sediment)
Fish	10 mg kg^{-1}	Calculated based on the water concentration shown above and a BCF of 100	Not available

[1] For references, see Section 5.7.2.1.

Doses were calculated separately to support evaluation of potential excess cancer risks from inhalation of nickel and non-cancer health effects from ingestion of nickel. The total average daily doses of nickel calculated for the inhalation and ingestion (oral) exposure routes for releases from K-25 are summarized in Table 5-44. The doses calculated for individual pathways and the contribution of each pathway to the total dose are summarized in Appendix I.

Table 5-44: Nickel Doses Calculated in the Level I and Refined Level I Screening (mg kg^{-1} d^{-1})

	Lifetime Average Daily Inhalation Dose- (Carcinogen)	**Average Daily Oral Dose- (Noncarcinogen)**
Level I	$1.3{\times}10^{-4}$	$2.4{\times}10^{-1}$
Refined Level I	$7.3{\times}10^{-6}$	$1.5{\times}10^{-2}$

5.7.5 Risk Characterization

Cancer and non-cancer screening indices were calculated based on the screening estimates of lifetime average daily dose and average daily dose, respectively, for nickel released from K-25. The results of the screening risk characterization are presented below, and compared to risk-based decision guides established by the Oak Ridge Health Agreement Steering Panel (ORHASP).

5.7.5.1 Cancer Screening Indices Associated with Inhalation of Nickel

The following equation was used to calculate the cancer screening index for inhalation of nickel:

$$Cancer\ Screening\ Index = Lifetime\ Average\ Daily\ Dose\ (mg\ kg^{-1}\ d^{-1}) \times Slope\ factor\ (mg\ kg^{-1}\ d^{-1})^{-1}$$

At Level I, the cancer screening index for inhalation of nickel is:

$$Cancer\ Screening\ Index = 1.3 \times 10^{-4}\ (mg\ kg^{-1}\ d^{-1}) \times 0.84\ (mg\ kg^{-1}\ d^{-1})^{-1} = 1.1 \times 10^{-4}$$

Because the Level I cancer screening index for inhalation of nickel exceeded $1{\times}10^{-4}$, the decision guide established by ORHASP for evaluating the need for further study of carcinogens, a Refined Level I screening was conducted.

At Refined Level I, the cancer screening index for inhalation of nickel is:

$$Cancer\ Screening\ Index = 7.3 \times 10^{-6}\ (mg\ kg^{-1}\ d^{-1}) \times 0.84\ (mg\ kg^{-1}\ d^{-1})^{-1} = 6.1 \times 10^{-6}$$

5.7.5.2 Non-cancer Screening Indices Associated with Ingestion of Nickel

The following equation was used to calculate the non-cancer screening index for ingestion of nickel:

$$Noncancer\ Screening\ Index = \frac{Average\ Daily\ Dose\ (mg\ kg^{-1}\ d^{-1})}{Reference\ Dose\ (mg\ kg^{-1}\ d^{-1})}$$

At Level I, the non-cancer screening index for ingestion of nickel is:

$$Noncancer\ Screening\ Index = \frac{2.4 \times 10^{-1}\ mg\ kg^{-1}\ d^{-1}}{2.0 \times 10^{-2}\ mg\ kg^{-1}\ d^{-1}} = 12$$

Because the Level I non-cancer screening index for ingestion of nickel exceeded 1.0, the decision guide established by ORHASP for evaluating the need for further study of noncarcinogens, a Refined Level I screening was conducted.

At Refined Level I, the non-cancer screening index for ingestion of nickel is:

$$Noncancer\ Screening\ Index = \frac{1.5 \times 10^{-2}\ mg\ kg^{-1}\ d^{-1}}{2.0 \times 10^{-2}\ mg\ kg^{-1}\ d^{-1}} = 0.75$$

5.7.5.3 Comparison of Screening Indices to Decision Guides

The screening indices for nickel calculated using the Level I and II screening methodologies are presented in Table 5-45, and compared to the risk-based decision guides established by ORHASP for screening of contaminants released from the ORR.

Table 5-45: Results of the Level I and Refined Level I Screening for Nickel

	Cancer Screening Index	Exceeds Decision Guide? [a]	Non-cancer Screening Index	Exceeds Decision Guide? [b]
Level I	1.1×10^{-4}	Yes	12	Yes
Refined Level I	6.1×10^{-6}	No	0.75	No

[a] For carcinogenic chemicals, a lifetime excess cancer risk of 10^{-4} was established by ORHASP as a decision guide for evaluating the need for further study (ChemRisk 1996).

[b] For noncarcinogenic chemicals, a hazard index of 1.0 was established by ORHASP as a decision guide for evaluating the need for further study (ChemRisk 1996).

5.7.6 Conclusions

The Level I screening analysis of exposures of off-site residents to nickel released from the K-25 Plant yielded a screening index that is above the decision guide in use on the project. This indicates that nickel releases warrant further evaluation in any further assessment of ORR releases. The fact that the Refined Level I screening index falls below the decision guide indicates that this further evaluation does not warrant an immediate priority. It should be noted that, even in Refined Level I screening, the analysis maintains considerable conservatism. In the Refined Level I methodology used in this assessment, somewhat different (less conservative) exposure parameters were used than in the Level I methodology; however, the biotransfer factors and the environmental concentrations remained the same. Both the biotransfer factors and the environmental concentrations add considerable conservatism to the analysis. Given the conservatism built into this analysis, the screening indicates that historical nickel releases do not warrant a high priority for further evaluation.

5.8 Technetium-99 Releases from the Oak Ridge Reservation

In the Dose Reconstruction Feasibility Study quantitative screening evaluation, technetium-99 (Tc-99) was not identified as a high priority material for further study (ChemRisk 1993). However, information gathered in Phase II of the Oak Ridge Health Studies regarding the material balance of Tc-99 at K-25, as well as additional environmental data, suggested that a reevaluation of Tc-99 releases from the K-25 and Y-12 sites was warranted.

This section summarizes available information on past off-site releases of Tc-99 from the ORR and presents a conservative, screening-level evaluation of potential doses that could have been received by people living near the ORR. The purpose of the screening is to determine if releases of Tc-99 warrant a more detailed investigation.

5.8.1 Technetium-99 Use at the ORR

Tc-99 was introduced to the K-25 and Y-12 sites as a result of use of uranium that had been in nuclear reactors and then recycled within the nuclear weapons complex. Tc-99 is formed in nuclear reactors as a product of fission of uranium atoms and from neutron activation of stable molybdenum-98. Beginning in 1953, K-25 received recycled uranium from commercial reactors, the Savannah River and Hanford production reactors, and the Paducah Gaseous Diffusion Plant. The Y-12 Plant began receiving recycled uranium from Savannah River and Idaho Chemical Processing Plant in 1953 (Egli et al. 1985). Savannah River and Idaho processed the spent fuel from the reactors in order to recover and purify either plutonium or enriched uranium. The purification processes used were not perfect, and they yielded uranium that was contaminated with trace quantities of transuranic elements and fission products. The uranium was then either sent to K-25 for enrichment or to Y-12 for fabrication of weapon parts.

5.8.2 Concentrations of Technetium-99 in the Environment near the ORR

Tc-99 is present in the environment due to global fallout from nuclear weapons testing and as a result of nuclear fuel reprocessing. The following sections describe levels of Tc-99 in background locations and that have historically been measured in the environment near the ORR.

5.8.2.1 Background Levels

Information located by the project team on background concentrations of Tc-99 in the environment were limited to soil data. The estimated average concentration of Tc-99 in soil worldwide due to global fallout from nuclear weapons tests is 2.2 pCi kg^{-1} (Hoffman 1982). Remedial investigations of the Oak Ridge area did not include analyses of Tc-99 in water, soil, or sediment samples from reference locations thought to be unaffected by ORR releases.

5.8.2.2 Environmental Concentrations on or Near the ORR

Historical measurements of Tc-99 in the environment near the ORR are extremely limited. The earliest measurements reported in the site literature were measured in the mid-1970s, and these data are limited to concentrations in sediments surrounding the K-25 site. This section presents air source term calculations for K-25 and Y-12, and discusses environmental measurements of Tc-99 in surface water and soil /sediment.

5.8.2.2.1 Tc-99 Concentrations in Air

No historical stack monitoring or ambient air monitoring data for Tc-99 were located by the project team. Therefore, it was necessary to estimate source terms for K-25 and Y-12 based on knowledge of the technetium shipped to K-25 and knowledge of the amount of uranium recycle material sent to Y-12. The following sections describe the methodology used to estimate Tc-99 releases to air from the K-25 and Y-12 sites.

Air Releases of Tc-99 from K-25

The project team did not locate any information regarding airborne releases of Tc-99 prior to 1974. However, an estimate of the amount of Tc-99 received at K-25 from 1953 to 1977 was located in a draft material balance report for K-25 (ORGDP 1978). The purpose of the material balance report was to determine how much Tc-99 had entered the K-25 Plant, how much was released to the environment, and how much had accumulated at the plant (ORGDP 1978). The report's authors estimated the amount of Tc-99 that entered the site by determining the concentration of Tc-99 in the recycled uranium received at K-25 from various sources. The estimated quantities of Tc-99 received at K-25 from the report are presented in Table 5-46.

Table 5-46: Mass of Tc-99 Received at K-25 from all Sources

Source of Technetium	Mass of Technetium-99 Received (kg)
Paducah Gaseous Diffusion Plant	117
Savannah River Plant	56
Hanford	30
Commercial Reactors	<0.04
Total	**203**

The report notes that all recycled uranium from the Savannah River Plant and Hanford had been received by 1962. Further, the report provides a chronology of the masses of Tc-99 in the recycled uranium received at K-25 from the Paducah Gaseous Diffusion Plant. This chronology is presented in Table 5-47.

Table 5-47: Tc-99 Received at K-25 from Paducah Gaseous Diffusion Plant 1953-1977

Time Period	**Tc-99 Received from Paducah (kg)**
1953-1962	75.35
1963-1965	4.78
1966-1971	9.56
1972	2.31
1973	8.04
1974	13.60
1975	0.18
1976-1977	3.12
Total	**116.94**

The Tc-99 quantities were used by the authors of the material balance report to estimate total Tc-99 releases. The masses received from Savannah River and Hanford were divided equally among the years 1953 to 1962 (8.6 kg per year) and the masses from Paducah were divided evenly into their respective time periods (Table 5-47). To calculate releases of Tc-99 to air, the material balance report assumed that there were two release points for atmospheric releases: the K-1131 (UF_6 manufacturing facility) stack and the purge cascade vent.

Atmospheric releases from the Building K-1131 UF_6 manufacturing facility would have originated from the process used to convert the UO_3 received from Savannah River and Hanford to UF_6. This facility was closed in the early 1960s, and would not have contributed to releases beyond that time period. The material balance report assumed that the Oak Ridge fluorination facility functioned similarly to the Paducah facility, where an estimated 5% of the Tc-99 in the UO_3 was vented to the atmosphere during fluorination. A 5% release fraction applied to the 8.6 kg received each year yields an annual release of 0.43 kg (7.3 Ci) of Tc-99 from the K-1131 stack from 1953 to 1962.

For the purge cascade, the second source of airborne Tc-99 releases from K-25, the material balance report estimates airborne releases for 1953 to 1973 by averaging the purge cascade monitoring data for 1974 to 1976 (Table 5-48). The report states that the elevated releases for 1976 reflect adjustments and experimentation on the purge cascade vent by K-25 personnel. The average release of 2.5 Ci per year from the purge cascade over this three-year period was applied to earlier periods (1953 to 1973).

Table 5-48: Reported Releases of Tc-99 from the Purge Cascade

Year	Tc-99 Released (Ci)
1974	0.3
1975	0.3
1976	6.8
Average	**2.5**

In 1977, a scrubber was installed on the purge cascade vent. This scrubber removed most of the Tc-99 from the effluent stream. Airborne releases of Tc-99 declined considerably, to 2.0×10^{-6} Ci for 1977. In 1978, the plant began reporting annual releases of Tc-99 in the annual environmental monitoring reports. The 1991 annual report noted another source of airborne Tc-99 releases– the TSCA incinerator, which began operations that year. This report also noted minor releases of airborne Tc-99 from the K-1015 laundry area and the K-1420 cascade equipment disassembly area. For 1978 to 1995, this screening analysis uses release estimates reported in the annual environmental monitoring reports.

Air Releases of Technetium-99 from Y-12

No airborne effluent information for the Y-12 Plant was located by the project team. However, documentation on the total amounts of recycled uranium Y-12 received from the Idaho Chemical Processing Plant and the Savannah River Plant was located for each year from 1953 to 1984 (Table 5-49) (Egli et al. 1985).

Table 5-49: Y-12 Receipts of Uranium Recycle Material (kg U)

Year	Savannah River Plant	Idaho Chemical Processing Plant	Total
1953	0	101	101
1954	0	217	217
1955	3	828	831
1956	0	744	744
1957	201	797	998
1958	258	898	1156
1959	270	3741	4011
1960	6395	769	7164
1961	2305	0	2305
1962	2701	775	3476
1963	6461	0	6461
1964	2977	771	3748
1965	3546	425	3971
1966	3467	1408	4875
1967	2604	0	2604
1968	2097	394	2491
1969	4121	427	4548
1970	2045	108	2153
1971	3805	1660	5465
1972	4716	415	5131
1973	5051	563	5614
1974	4599	0	4599
1975	5110	1702	6812
1976	4320	195	4515
1977	4497	1333	5830
1978	2070	525	2595
1979	4591	535	5126
1980	1510	0	1510
1981	4918	905	5823
1982	5728	577	6305
1983	6682	1041	7723
1984	5776	2868	8644
Total	**102,824**	**24,722**	**127,546**

The yearly masses of uranium received were multiplied by the estimated Tc-99 concentration in the recycled uranium to arrive at an estimate of the total Tc-99 activity at the Y-12 site. Based on information in the K-25 material balance document (ORGDP 1978) the project team assumed a Tc-99 concentration of 7 ppm in the recycled uranium. The material balance document states Paducah personnel estimated government reactor recycled uranium at 7 ppm Tc-99, and that this estimate is consistent with K-25 data. The mass of Tc-99 received was then calculated using the following equation:

$$Tc\text{-}99\ (mg) = U\ (kg) \times Tc\text{-}99\ Concentration\ (mg\ kg^{-1})$$

The mass of Tc-99 received in the recycled uranium in 1953 would then be:

$$Tc\text{-}99\ (mg) = 101\ kg \times 7\ mg\ kg^{-1} = 707\ mg\ Tc\text{-}99$$

The activity of Tc-99 received was calculated by multiplying the mass of Tc-99 received by the specific activity of Tc-99 (1.7×10^{-2} Ci g^{-1}):

$$Tc\text{-}99\ (Ci) = (0.707\ g\ Tc\text{-}99) \times (1.7\times10^{-2}\ Ci\ g^{-1}) = 0.012\ Ci$$

The next step in determining the Tc-99 source term was to define the amount of Tc-99 released to the air. This was accomplished by calculating a release fraction based on the inventory differences for natural uranium at Y-12 reported in Owings (1995). Inventory difference values were once termed Material Unaccounted For (MUF). The calculated natural uranium release fraction based on inventory differences was 1.0×10^{-3} (or 0.1%). This value was multiplied by the Y-12 Tc-99 activity inventories to yield conservative annual airborne release estimates. The results are presented in Table 5-50.

Table 5-50: Estimated Tc-99 Releases from Y-12

Year	Estimated Tc-99 Release (Ci)
1953	1.2×10^{-5}
1954	2.6×10^{-5}
1955	9.9×10^{-5}
1956	8.9×10^{-5}
1957	1.2×10^{-4}
1958	1.4×10^{-4}
1959	4.8×10^{-4}
1960	8.5×10^{-4}
1961	2.7×10^{-4}
1962	4.1×10^{-4}
1963	7.7×10^{-4}
1964	4.5×10^{-4}
1965	4.7×10^{-4}
1966	5.8×10^{-4}
1967	3.1×10^{-4}
1968	3.0×10^{-4}
1969	5.4×10^{-4}
1970	2.6×10^{-4}
1971	6.5×10^{-4}
1972	6.1×10^{-4}
1973	6.7×10^{-4}
1974	5.5×10^{-4}
1975	8.2×10^{-4}
1976	5.4×10^{-4}
1977	6.9×10^{-4}
1978	3.1×10^{-4}
1979	6.1×10^{-4}
1980	1.8×10^{-4}
1981	6.9×10^{-4}
1982	7.5×10^{-4}
1983	9.2×10^{-4}
1984 through 1995	1.0×10^{-3} each year

Off-Site Air Concentrations Used in the Screening Assessment

This screening assessment evaluates the potential for health effects based on estimated air concentrations in the Union/Lawnville area for releases from K-25 and at Scarboro for releases from Y-12.

Estimated Tc-99 release rates (Ci y^{-1}) for each year were converted to release rates per second (Ci s^{-1}). For releases from K-25, each year's release rate (Ci s^{-1}) was multiplied by the χ/Q for Union/Lawnville of 7.4×10^{-7} s m^{-3}. This calculation yielded a Tc-99 air concentration at the Union/Lawnville location for each year from 1953 through 1995. The mean calculated Tc-99 air concentration for the 1953-1995 time period, 1.1×10^{-1} pCi m^{-3}, was used in the screening assessment for each year of release.

The Tc-99 release rates (Ci s^{-1}) were also multiplied by the χ/Q for Scarboro of 3×10^{-7} s m^{-3}. The mean calculated Tc-99 air concentration at Scarboro for the 1953-1995 time period, 5.9×10^{-6} pCi m^{-3}, was used in the screening assessment for all years of release.

5.8.2.2.2 Tc-99 Concentrations in Surface Water

No measurements of Tc-99 concentrations in liquid effluent from the K-25 or Y-12 Plants prior to the late 1980s were identified by the project team. Beginning in 1987, concentrations of Tc-99 were measured monthly in Poplar Creek around the K-25 site. Concentrations from 1987 to 1995 ranged from less than the limit of detection to 1,860 pCi L^{-1}. During this same time period, concentrations downstream in the Clinch River ranged from less than the limit of detection to 1,640 pCi L^{-1}. Results for individual samples were not located by the project team.

Beginning in 1991, concentrations of Tc-99 were measured monthly in EFPC at the junction of Bear Creek and Scarboro Roads. The concentrations ranged from less than background to 160 pCi L^{-1}. Individual sample results were not located.

In addition to routine monitoring, two special studies also measured Tc-99 concentrations in surface waters around the ORR. These studies are described below.

- *The Tennessee Valley Authority (TVA) Instream Contaminant Study (1984)*— the only surface water sample analyzed for Tc-99 as part of this study was located in Watts Bar Reservoir at Clinch River Mile 6.8. The concentration of Tc-99 in this sample was 0.73 pCi L^{-1} (TVA 1985a).

- *The Clinch River Remedial Investigation (1989 and 1990)*— Tc-99 concentrations in the Clinch River ranged from less than the limit of detection to 23 pCi L^{-1} (USDOE 1996). The maximum Clinch River concentration was measured in Lower Melton Hill Reservoir. The Poplar Creek concentrations ranged from less than the limit of detection to 32 pCi L^{-1}. The maximum Poplar Creek concentration was measured in Mitchell Branch, a stream that drains the eastern section of the K-25 plant and then flows into Poplar Creek.

Off-Site Water Concentration Used in the Screening Assessment

The highest surface water concentration reported for the Clinch River was 1,640 pCi L^{-1} in the 1992 annual monitoring report (MMES 1993). The measurement was recorded for a sample collected about 1 mile downstream from the K-25 plant. This value was used in the screening analysis for K-25 releases.

The highest surface water concentration reported for East Fork Poplar Creek was 160 pCi L^{-1} in the 1993 annual monitoring report (MMES 1993). The measurement was recorded for a sample collected in East Fork Poplar Creek near the junction of Bear Creek Road and Scarboro Road. This value was used in the screening analysis for Y-12 releases.

5.8.2.2.3 Tc-99 Concentrations in Soil/Sediment

Three sources of off-site soil and sediment data were located for Tc-99. Summary level data for the 1990s were located in the annual environmental surveillance reports (MMES 1993, 1994, 1995) and the Clinch River Remedial Investigation (Cook et al. 1993) . Raw data were located in a compilation of Clinch River and Poplar Creek bottom sediment measurements for the 1975 - 1981 time period (MMES 1981). A summary of the soil and sediment data follows.

Soil

- Six soil samples were collected at two locations along the K-25 site perimeter in 1978 and 1979. Measured Tc-99 concentrations ranged from 81 to 1700 pCi kg^{-1} (Hoffman 1982).

- Results of seven soil samples collected in the K-25 area were reported in the Annual Site Environmental Monitoring Reports in 1993 and 1994 (MMES 1994, LMES 1995). The Tc-99 concentrations ranged from less than background to 8200 pCi kg^{-1}.

Sediment

- Beginning in the mid-1970s, surface sediment samples were collected semiannually at several locations in Poplar Creek and the Clinch River. Sediment concentrations in the Clinch River between 1975 and 1981 ranged from <10,000 pCi kg^{-1} to 800,000 pCi kg^{-1} (dry weight). The maximum Clinch River concentration (5.6×10^{5} pCi kg^{-1}) was measured in November of 1977 about 1 mile below the confluence of Poplar Creek and the Clinch River (ORGDP 1981a).

- *TVA Instream Contaminant Study (1984)*— Sediment samples were collected and analyzed for a variety of contaminants including Tc-99. Surface sediment sample locations included EFPC at Miles 1.7 and 13.5, two locations on EFPC floodplain (Miles 1.7 and 13.5), three in Poplar Creek, and five in the Clinch River (Hoffman et al. 1984). Concentrations ranged from 405 to 6,500 pCi kg^{-1} (dry weight). Data for individual samples are presented in Hoffman et al. (1984).

- *The Clinch River Remedial Investigation (1989 and 1990)*— This program was initiated by DOE in 1989 to address the transport, fate, and distribution of waterborne contaminants released from the ORR to the Clinch and Tennessee Rivers (USDOE 1996). Surface water, sediment, and fish samples were collected from 10 reaches, six of which were potentially affected by releases from the ORR, and four of which served as reference or background areas. Sample locations included Poplar Creek and the Clinch River. Tc-99 concentrations in Poplar Creek sediment ranged from less than background[1] to 180,000 pCi kg^{-1} (dry weight.). Only one sample from the Clinch River was analyzed for Tc-99. This sample was collected 2.5 miles upstream of the Poplar Creek confluence with the Clinch River. The Tc-99 concentrations in the Clinch River sample was less than 1500 pCi kg^{-1}.

- *Annual Site Environmental Reports for 1993-1995*— Results of sediment sampling are reported for one Clinch River location downstream from DOE inputs, at one location on East Fork Poplar Creek downstream from the Y-12 plant, and at one location on Poplar Creek downstream from the K-25 plant. The Tc-99 concentrations in sediment samples collected at these sites ranged from 92 to 650 pCi kg^{-1}, 110 to 840 pCi kg^{-1}, 890 to 1,000 pCi kg^{-1}, for Clinch River, East Fork Poplar Creek, and Poplar Creek, respectively (all stated in terms of dry weights) (LMES 1996).

Off-Site Soil/Sediment Concentration Used in the Screening Assessment

The highest Tc-99 sediment concentration reported for the Clinch River, 5.6×10^5 pCi kg^{-1}, was used by the project team in the Level I K-25 screening analysis for the soil/sediment pathways. This concentration was measured as part of the routine sediment sampling program in the 1970s. The highest concentration

---

[1] The term "less than background" indicates that the count rate of Tc-99 radiations from the sample was less than the count rate from laboratory background. This often happens when measuring low levels of radioactivity, due to the random nature of radioactive decay. In some cases, negative concentrations were reported; they are not given in this report, as screening focuses on higher, positively detected values. When detection limits for measurement system are documented, they are presented.

from six soil samples taken along the K-25 site perimeter in 1978 and 1979, 1,700 pCi kg^{-1}, was used for the Refined Level I K-25 screening analysis.

The highest Tc-99 sediment concentration reported for East Fork Poplar Creek, 5.0×10^3 pCi kg^{-1}, was used by the project team in the Y-12 screening analysis for the soil/sediment exposure pathways. This concentration was measured as part of the TVA Instream Contaminant Study in 1984 (Hoffman et al. 1984).

5.8.2.2.4 Tc-99 Concentrations in Food Items

In the Tennessee Valley Authority (TVA) Instream Contaminant Study (1984), fish were collected from several sites in the Clinch River, Poplar Creek, and EFPC including at Clinch River Mile 23.5 (Melton Hill Dam), Poplar Creek Miles 0.2 and 13.8, and East Fork Poplar Creek Miles 1.7 and 13.8. Concentrations in fish tissue ranged from 0.079 to 1.4 pCi g^{-1}.

In the Clinch River Remedial Investigation (1990), fish were collected from several locations on the Clinch River and Poplar Creek, both upstream and downstream from the DOE sites. Maximum measured Tc-99 concentrations ranged from 0.04 to 0.1 pCi g^{-1} (Hoffman et al. 1991).

<u>Off-Site Fish Concentration Used in the Screening Assessment</u>

For the purposes of the screening analyses, concentrations of Tc-99 in vegetation, meat, and milk were calculated using biotransfer factors that describe the transfer of Tc-99 from other media, including air, water, and soil. The biotransfer factors used in this assessment are listed in Table 5-51. For purposes of the screening analyses, concentrations of Tc-99 in fish were calculated using a bioconcentration factor (BCF) that describes the transfer of Tc-99 from water to fish.

Table 5-51: Biotransfer and Bioconcentration Factors for Tc-99

Parameter	Value	Source
Soil to Vegetables (B_{veg})	5 (mg kg^{-1} wet)/(mg kg^{-1} dry)	NCRP 1996
Soil to Pasture ($B_{pasture}$)	40 (mg kg^{-1} dry)/(mg kg^{-1} dry)	NCRP 1996
Biotransfer to Milk (F_m)	0.001 d L^{-1}	NCRP 1996
Biotransfer to Meat (F_f)	0.0001 d kg^{-1}	NCRP 1996
Bioconcentration in Fish (BCF)	20 (mg kg^{-1})/(mg L^{-1})	NCRP 1996

5.8.3 Toxicity Assessment

The USEPA classifies all radionuclides as Group A carcinogens based on their property of emitting ionizing radiation and on the extensive weight of evidence provided by epidemiological studies of radiation-induced cancers in humans.

The International Commission on Radiological Protection (ICRP) recommended in 1990 that an estimate of 5% per sievert (0.05% per rem) be used for the probability of induced fatal cancer in populations of all ages. A smaller value of about 4% per sievert (0.04 per rem) is recommended for a working population of age 20 to 64 years. These estimates are primarily made for exposure to low dose, low dose rate, and low linear energy transfer (LET) radiation. The probability of fatal cancer for other exposure conditions (0.07% per rem) be used for "detriment" due to radiation exposure of the whole body at low doses. Their are four main components of detriment considered by the ICRP: the risk of cancer in all relevant organs, expected years of life lost for induced cancer, morbidity resulting from induced non-fatal cancers, and risk of serious hereditary disease in all future generations descended from the exposed individual.

Technetium-99 Dosimetric Concerns

Technetium is generally soluble and readily transfers into the bloodstream from the gastrointestinal tract. A value of 0.5 is recommended by the ICRP (1993) for the fraction of Tc-99 ingested in food that is transferred from the GI tract to the blood stream. ICRP Publication 71 (1995) also indicates that rapid uptake of inhaled technetium takes place.

For radionuclides, radiation doses result either from the intake of quantities of the radionuclide into the body or from being immersed in or in close proximity to the radionuclide in such a way that radiation that are emitted are absorbed in body tissue. In the first case, radiation committed effective dose equivalents, in sieverts (Sv), are calculated by multiplying the quantity taken into the body (in becquerel, Bq) times a dose conversion factor. In the latter case, effective dose equivalent rates (e.g., Sv y^{-1}) are calculated by multiplying the radionuclide concentration in the contaminated medium (air, water, or surface soil) times a dose conversion factor. The dose conversion factors used for Tc-99 in this screening are presented in Table 5-52.

5.8.4 Exposure Assessment

Following the Task 7 Screening methodology for Level I and Level II screening (ChemRisk 1996), doses were calculated for exposure to Tc-99 in air, soil/sediment, and surface water. Exposures to Tc-99 through ingestion of fish, vegetables, meat, and milk were evaluated using equations (presented in Appendix B) and biotransfer factors (discussed in Section 5.8.2.2.4) that describe the uptake of Tc-99 from air, surface water, and soil/sediment into these media. The exposure pathways that were included are listed in Table 5-53.

Table 5-52: Dose Conversion Factors for Tc-99

Exposure Route	Dose Conversion Factor
Inhalation	4.00×10^{-9} Sv Bq^{-1} [a]
Ingestion	6.4×10^{-10} Sv Bq^{-1} [b]
Air Immersion	5.11×10^{-11} Sv m^3 Bq^{-1} y^{-1} [c]
Water Immersion	9.90×10^{-14} Sv m^3 Bq^{-1} y^{-1} [c]
External Irradiation	2.11×10^{-14} Sv m^3 Bq^{-1} y^{-1} [c]

References: a: ICRP 1995 b: ICRP 1996 c: USEPA 1993

Table 5-53: Summary of Environmental Concentrations of Tc-99 used in Screening

Environmental Medium	Concentration	Source of Value	Typical Background[1]
K-25			
Air	1.1×10^{-1} pCi m^{-3}	Average of concentrations modeled to Union/Lawnville for 1953-1995	Not available
Surface Water	1.6×10^{3} pCi L^{-1}	Highest concentration measured in Clinch River from routine sampling (1992)	Not available
Soil/Sediment	5.6×10^{5} pCi kg^{-1} (Level I); 1.7×10^{3} (Refined Level I)	Level I: Highest sediment concentration in Clinch River (1970s); Refined Level I: Soil samples collected at K-25 perimeter in 1978 and 1979.	2.2 pCi kg^{-1} (soil worldwide)
Fish	32 nCi kg^{-1}	Calculated based on the water concentration above and a BCF of 20	Not available
Y-12			
Air	5.9×10^{-6} pCi m^{-3}	Average of concentrations modeled to Scarboro for 1953-1995	Not available
Surface Water	1.6×10^{2} pCi L^{-1}	Highest concentration measured in EFPC from routine sampling (1993)	Not available
Soil/Sediment	5.0×10^{3} pCi kg^{-1}	Highest EFPC sediment concentration (1984)	2.2 pCi kg^{-1} (soil worldwide)
Fish	3.2 nCi kg^{-1}	Calculated based on the water concentration above and a BCF of 20	Not available

[1] For references, see Section 5.8.2.1.

The radiation dose commitments for the Level I and Refined Level I screening assessments are presented in Tables 5-54. The total dose summed across all pathways for exposure to Tc-99 released from K-25 was 9.6×10^{-3} and 5.9×10^{-6} for Level I and Refined Level I screening, respectively. The total dose summed across all pathways for exposure to Tc-99 released from Y-12 was 8.6×10^{-5} and 9.1×10^{6} for the Level I and Refined Level I screening assessments, respectively. Contributions of each pathway to the total dose are presented in Appendix I.

Table 5-54: Tc-99 Doses Calculated in the Level I and Refined Level I Screening (Sv y^{-1})

	Total Dose
K-25	
Level I	9.6×10^{-3}
Refined Level I	5.9×10^{-6}
Y-12	
Level I	8.6×10^{-5}
Refined Level I	9.1×10^{-6}

5.8.5 Risk Characterization

Screening indices associated with screening level estimates of Tc-99 dose commitments were evaluated for all pathways combined for releases from K-25 and Y-12. The results of the screening risk characterization are presented below, and compared to risk-based decision guides established by the Oak Ridge Health Agreement Steering Panel (ORHASP).

5.8.5.1 Cancer Screening Indices

Once intakes or exposures to Tc-99 were translated into radiation dose commitments, estimates of excess lifetime risk of cancer or other health effects can be obtained by applying an appropriate dose-to-risk conversion factor. For the purpose of this screening, the dose-to-risk coefficient of 7.3% Sv^{-1} was used. Based on recommendations of the International Commission on Radiological Protection, this coefficient represents the probability of fatal cancer, non-fatal cancer, or severe hereditary effects in the whole population per unit radiation dose (ICRP 1992). The resulting estimate of excess annual risk of cancer was multiplied by the total number of years of potential exposure (1953-1995, 43 years) to produce a estimate of excess lifetime risk of cancer).

$$\textit{Cancer Screening Index} = \textit{Committed Dose}\ (Sv\ y^{-1}) \times \textit{Dose Conversion Factor}\ (Sv^{-1}) \times 42\ y$$

5.8.5.1.1 Cancer Screening Indices Associated with Releases of Tc-99 from K-25

At Level I, the cancer screening index for releases of Tc-99 from K-25 was:

$$Cancer\ Screening\ Index = 9.6\times10^{-3}\ (Sv\ y^{-1}) \times 7.3\%\ (Sv^{-1}) \times 42\ y = 3.0\times10^{-2}$$

At Refined Level I, the cancer screening index for releases of Tc-99 from K-25 was:

$$Cancer\ Screening\ Index = 5.9\times10^{-6}\ (Sv\ y^{-1}) \times 7.3\%\ (Sv^{-1}) \times 42 = 1.8\times10^{-5}$$

5.8.5.1.2 Cancer Screening Indices Associated with Releases of Tc-99 from Y-12

At Level I, the cancer screening index for releases of Tc-99 from Y-12 was:

$$Cancer\ Screening\ Index = 8.6\times10^{-5}\ (Sv\ y^{-1}) \times 7.3\%\ (Sv^{-1}) \times 42\ y = 2.7\times10^{-4}$$

At Refined Level I, the cancer screening index for releases of Tc-99 from Y-12 was:

$$Cancer\ Screening\ Index = 9.1\times10^{-6}\ (Sv\ y^{-1}) \times 7.3\%\ (Sv^{-1}) \times 42\ y = 2.8\times10^{-5}$$

5.8.5.2 Comparison of Screening Indices to Decision Guides

The screening indices calculated using the Level I and Refined Level I screening methodology for exposure to Tc-99 are presented in Table 5-55, and compared to the risk-based decision guides established by ORHASP for screening of contaminants released from the ORR.

Table 5-55: Results of the Level I and Refined Level I Screening for Tc-99

	Cancer Screening Index for K-25 Releases	Exceeds Decision Guide? [a]	Cancer Screening Index for Y-12 Releases	Exceeds Decision Guide?[a]
Level I	3.0×10^{-2}	Yes	2.7×10^{-4}	Yes
Refined Level I	1.8×10^{-5}	No	2.8×10^{-5}	No

[a] For radionuclides, a lifetime excess cancer risk of 10^{-4} was established by ORHASP as a decision guide for evaluating the need for further study (ChemRisk 1996).

5.8.6 Conclusions

The Level I screening of K-25 and Y-12 releases of Tc-99 yielded a screening index above the decision guide in use on this project. This indicates that further analysis of the Tc-99 releases and exposures is warranted. It is important to note that the exposure pathways that clearly dominated calculated total exposures in the Level I screen for K-25 are the pathways associated with contaminated soil. Because of the paucity of soil monitoring data for Tc-99, this analysis relied on sediment concentration data. The sediment concentrations used in the Level I screen are greater than the Tc-99 concentrations measured in soil around the ORR. Use of this sediment concentration lends considerable conservatism to this evaluation.

The fact that Refined Level I screening index is below the decision guide indicates that further evaluation of off-site exposures to Tc-99 does not warrant immediate priority.

This page intentionally left blank.

6.0 SUMMARY OF TASK 7 RESULTS

The results of the screening analyses of materials that were quantitatively evaluated are shown in Table 6-1. For each of the 13 assessments depicted in the table, the following information is provided:

- The identity of the contaminant and its source (K-25 Site or Y-12 Plant in most cases).

- The identity of the reference location for which concentrations, doses, and screening indices were estimated. These reference locations were selected as the areas where the highest off-site exposures likely occurred.

- The calculated cancer screening index for materials evaluated as carcinogens. These values were estimated by multiplying the total dose of a chemical by its cancer potency slope factor, or the radiation dose from a radionuclide times a risk factor of 7.3% Sv^{-1}.

  Results are presented for a Level I evaluation, and for a refined Level I analysis where applicable. Values are shown in bold when they exceed the decision guide in use on the project (that is, the screening index is 1×10^{-4} or greater).

- The non-cancer screening index for materials associated with toxic effects other than cancer. These values were in most cases calculated by dividing the dose of a chemical by its USEPA reference dose. Cases where different approached had to be used (for chemicals without established reference doses) are identified in the "Notes" column of the table.[1]

  In cases where doses were compared to reference doses for both inhalation and ingestion, the screening index that represents the largest fraction of (or multiple of) the applicable reference dose is provided. In each case described here, the highest screening indices resulted from comparing doses from ingestion to the oral reference dose.

  Results are presented for the Level I evaluation, and for a refined Level I analysis where applicable. Values are shown in bold when they exceed the decision guide in use on the project (that is, the screening index is 1 or greater).

- Notes are provided to indicate where non-standard approaches had to be used in an assessment. Notes are also provided, for non-carcinogens, to describe the relationship between the reference dose and a relevant toxicologic reference value, such as a NOAEL or LOAEL. In cases where screening indices indicate potential doses above the applicable reference dose, it is important to know how much separation there is between the reference dose and the NOAEL or LOAEL (that is, how much of a safety factor there is) in order to be able to evaluate the potential for health effects.

---

[1] Reference doses were derived by the project team for niobium (from an LD_{50} in mice), lithium (from the therapeutic dose of lithium carbonate), and copper and zirconium (from ACGIH Threshold Limit Values®, TLVs®).

Examination of the results in Table 6-1 shows that:

- For carcinogens, 3 of 10 analyses ended with the Level I screening (Np-237 from K-25, Np-237 from Y-12, and tritium from Y-12). In other words, the initial, most conservative screening calculations for these materials yielded results that were below the decision guide in use on the project.

- For the other seven assessments of carcinogens, refined screening was performed. Of these refined assessments, two yielded results that were still above the applicable decision guide (arsenic from K-25 and arsenic from Y-12). The other five were below the decision guide with refined screening (beryllium from Y-12, chromium from the ORR, nickel from K-25, technetium-99 from K-25 and technetium-99 from Y-12).

- For non-carcinogens, 1 of 8 analyses ended with the Level I screening (beryllium from Y-12). In other words, the initial, most conservative screening calculations for beryllium yielded results that were below the decision guide in use on the project.

- For the other seven assessments of non-carcinogens, refined screening was performed. Of these refined assessments, three yielded results that were still above the applicable decision guide (arsenic from K-25, arsenic from Y-12, and lead from the ORR). The other four were below the decision guide with refined screening (chromium from the ORR, copper from K-25, lithium from Y-12, and nickel from K-25).

Several materials were quantitatively evaluated in Task 7 that do not appear in Table 6-1. These are the materials that were evaluated using the threshold quantity approach, a method that does not yield numerical screening indices. Materials that fall in this category include tetramethyl-ammoniumborohydride (TMAB), niobium, and zirconium. Based on evaluation of the limited information available on these materials, it was determined that the quantities of each that were present at the Y-12 Plant were not likely great enough to have posed off-site health hazards.

Table 6-1: Summary of Task 7 Screening Results for Materials Evaluated Quantitatively[1]

Material	Cancer Screening Index	Non-cancer Screening Index	Notes
Arsenic from K-25 *exposure for Union/Lawnville*[2]	**Level I = 3.8×10^{-2}** **Refined Level I = 8.9×10^{-4}**	**Level I = 120** **Refined Level I = 13**	The NOAEL (from a human study) is a factor of 3 above the RfD. A non-cancer Screening Index above 3 could indicate exposures above the NOAEL.
Arsenic from Y-12 *exposure for Scarboro*	**Level I = 1.8×10^{-2}** **Refined Level I = 2.6×10^{-4}**	**Level I = 41** **Refined Level I = 4.0**	The NOAEL (from a human study) is a factor of 3 above the RfD. A non-cancer Screening Index above 3 could indicate exposures above the NOAEL.
Beryllium from Y-12 *exposure for Scarboro*	**Level I = 4.0×10^{-4}** Refined Level I = 1.3×10^{-5}	Level I = 0.066	The NOAEL (based on a rat study) is a factor of 100 above the RfD. A non-cancer Screening Index above 100 could indicate exposures above the NOAEL.
Chromium(VI) from the ORR *estimated based on Union/Lawnville air levels & EFPC/Poplar Creek water and soil/sediment data*	**Level I = 1.3×10^{-4}** Refined Level I = 1.0×10^{-5}	**Level I = 9.7** Refined Level I = 0.55	The NOAEL (based on a rat study) is a factor of 800 above the RfD. A non-cancer Screening Index above 800 could indicate exposures above the NOAEL.
Copper from K-25 *exposure for Union/Lawnville*		**Level I = 2.4** Refined Level I = 0.13	The LOAEL (from human studies of gastrointestinal effects) is a factor of 2 above the RfD. A non-cancer Screening Index above 2 could indicate exposures above the LOAEL.
Lead Releases from Y-12 *based on levels in EFPC water and soil/sediment, estimated average urban air levels prior to the 1970s*		**Level I = 5.2 to 6.7** **Refined Level I = 1.8 to 2.3**	These Screening Indices are based on the calculated range of blood lead levels divided by the CDC/USEPA action level of $10~\mu\text{g dL}^{-1}$. Non-cancer Screening Indices above 1 could indicate exposures above the CDC action level.
Lithium from Y-12 *exposure for Scarboro*		**Level I = 2.3** Refined Level I = 0.29	The derived RfD is a factor of 10 below the normal therapeutic dose. Lithium toxicity can occur at doses close to therapeutic levels. A non-cancer Screening Index above 10 could indicate exposures above the therapeutic dose.

Table 6-1: Summary of Task 7 Screening Results for Materials Evaluated Quantitatively[1]

Material	Cancer Screening Index	Non-cancer Screening Index	Notes
Neptunium-237 from K-25 *exposure for Union/Lawnville*	Level I = 7.3×10^{-6}		
Neptunium-237 from Y-12 *exposure for Scarboro*	Level I = 6.8×10^{-6}		
Nickel from K-25 *exposure for Union/Lawnville*	**Level I = 1.1×10^{-4}** Refined Level I = 6.1×10^{-6}	**Level I = 12** Refined Level I = 0.75	The NOAEL (from a rat study) is a factor of 250 above the RfD. A non-cancer Screening Index above 250 could indicate exposures above the NOAEL.
Technetium-99 from K-25 *exposure for Union/Lawnville*	**Level I = 3.0×10^{-2}** Refined Level I = 1.8×10^{-5}		
Technetium-99 from Y-12 *exposure for Scarboro*	**Level I = 2.7×10^{-4}** Refined Level I = 2.8×10^{-5}		
Tritium from Y-12 Heavy Water *exposure for Scarboro*	Level I = 1.6×10^{-6}		

[1] Several materials were quantitatively evaluated do not appear here. These materials (TMAB, niobium, and zirconium) were evaluated using the threshold quantity approach, a method that does not yield numerical screening indices. It was determined that the quantities of each that were present at the Y-12 Plant were not likely great enough to have posed off-site health hazards.

[2] The “reference locations” evaluated were selected as the areas where the highest off-site exposures likely occurred.

The "Refined Level I" screening evaluations described in this report were considerably less conservative than the Level I evaluations they followed. As described in Appendix C, for example, many of the exposure parameter values used in the dose and risk calculations are less conservative (more realistic or more typical) than the values of the same parameters used in Level I screening. A good example would be the assumed exposure duration for carcinogens, which is 50 years in Level I screening and 10 years in Refined Level I screening. The Refined Level I evaluations described in this report used the "Level II" exposure parameters from the standard Task 7 screening methodology.

While a general goal in refined screening is to reduce or eliminate sources of conservative bias, it is not always feasible or advisable to eliminate all conservative bias, or easily determined when a sufficient level of realism has been achieved. In the refined screening evaluations described in this report, some degree of conservatism was retained, particularly in the estimation of contaminant concentrations in environmental media of interest. One important reason for this is that there were very few measurements of the contaminants of concern made in the environment during the (pre-1970s) periods when levels of many contaminants in the environment were likely the highest. Measurements in process streams or effluents are even more rare. Because of the paucity of information for some vital components of the risk assessment process, some conservatism was retained in the estimation of exposure point concentrations for the Refined Level I assessments to ensure that exposures were not underestimated for significant portions of the potentially exposed populations. Because of this, the second-level assessments are called Refined Level I assessments rather than Refined Level I assessments.

This page intentionally left blank.

7.0 CONCLUSIONS

Based on the qualitative and quantitative screening performed under Task 7 of the Oak Ridge Dose Reconstruction project, it was possible to separate materials into classes based on their apparent importance in terms of potential off-site health hazards. This classification process was to a great degree dependent on the information that is available concerning past uses and releases of the materials of interest. In the course of Task 7 work, it was not possible to perform extensive directed searches for records relevant to each Task 7 material to the extent that was possible for the operations and contaminants studied in detail under Tasks 1, 2, 3, and 4. For some materials, very little historical information is available. As a result, it was necessary to make a significant number of conservative assumptions for some materials to ensure that potential doses were not underestimated. If, in the future, more extensive document searching is performed, some of the conclusions reached in the screening evaluations described herein might well change.

Based on the qualitative and quantitative screening performed by the Task 7 project team, five materials (or classes of materials) used at the K-25 Site and 14 materials (or classes of materials) used at the Y-12 Plant are judged to not warrant further study related to their potential for off-site health effects. These materials are identified in the second column of Table 7-1. The materials named to this category were placed there because either:

1) Quantitative screening of the most conservative nature (Level I screening) yielded screening indices that fell below the guides in use on the project;

2) Application of a threshold quantity approach demonstrated that not enough of the material was present to have posed an off-site health hazard; or

3) In qualitative evaluation of available information by project team members, it became obvious that quantities used, forms used, and/or manners of usage were such that off-site releases could not have been sufficient to have posed off-site health hazards.

Based on quantitative screening performed by the Task 7 team, three materials used at the K-25 Site, three materials (or classes of materials) used at the Y-12 Plant, and one material used at all sites were identified as potential candidates for further study. These materials are:

- At the K-25 Site: copper powder
  nickel
  technetium-99

- At the Y-12 Plant: beryllium compounds
  lithium compounds
  technetium-99

- From the ORR: Chromium(VI)

These materials were identified as potential candidates for further study because quantitative screening of the most conservative nature (Level I screening) yielded screening indices that exceeded the decision guide in use on the project. As shown in Table 7-1, some of these materials exceeded the decision guides as carcinogens, while others exceeded published or derived reference doses for materials that cause effects other than cancer. When less conservative, "refined" screening was performed for each of these materials, results in each case fell below the decision guides in use on the project.

Based on quantitative screening performed by the Task 7 team, one materials used at the K-25 Site and two used at the Y-12 Plant were identified as high priority candidates for further study. These materials are:

- From the K-25 Site: arsenic
- From the Y-12 Plant: arsenic
  lead

These materials were identified as high priority candidates for further study because less conservative, "refined" quantitative screening yielded screening indices that exceeded the decision guide in use on the project. As shown in Table 7-1, arsenic achieved this status as both a carcinogen and as a non-carcinogen, while lead achieved this status as a non-carcinogen.

For the non-carcinogenic contaminants with screening indices that exceeded 1 in refined screening, it is important to evaluate the relationship between the reference dose and toxicologic reference levels such as the NOAEL or LOAEL. The importance of a screening index above 1 varies from one material to the next, because the amount of separation between the reference dose and the dose at which health effects have been shown to occur varies significantly. For this project, the materials for which this type of evaluation is most critical are arsenic and lead. Following are summaries of the relationships between screening indices for these materials and applicable NOAELs or action levels.

- For arsenic, the NOAEL (from a human study) is a factor of three above the oral reference dose. Non-cancer screening indices above 3 could indicate that exposures above the NOAEL occurred. Because the screening indices from the refined assessments are 13 and 4.0 for K-25 and Y-12, respectively, it is possible that doses above the NOAEL were experienced. At the same time, it is impossible to say if health effects occurred or not.

- For lead, it has been reported that adverse health effects can occur in children at blood lead concentrations as low as 10 µg dL^{-1}; this is the action level set by CDC in 1991. Non-cancer screening indices for lead in Table 6-1 were based on the ranges of blood lead concentrations calculated with the IEUBK model, divided by the CDC action level of 10 µg dL^{-1}. Because the screening indices from the refined assessment for lead range between 1.8 and 2.3, it is possible that doses above action level were received by some children. It is less likely that any adults received doses that exceeded the OSHA standard of 40 µg dL^{-1}.

Some of the materials evaluated in this project have very limited toxicologic information available. For example, very little is known about potential effects from exposure to lithium at levels that can be expected to occur in the environment. Reference doses for niobium, lithium, copper, and zirconium are not available from the USEPA. For the purposes of this study, reference doses were derived from studies of lethal doses in mice (niobium), from ACGIH Threshold Limit Values® for the workplace (copper and zirconium), and from therapeutic doses used in humans (for lithium). If better toxicologic data become available for these materials, the analyses described herein would likely benefit from evaluation with that new information. This is particularly true for lithium, which was evaluated based on a "derived" reference dose equal to the lithium equivalent of one-tenth of the dose of lithium carbonate used in humans to control mania. Because lithium toxicity can occur at doses close to the therapeutic dose, the lack of information concerning effects of exposure to lithium at environmental levels is an important data gap.

Table 7-1: Categorization of Evaluated Materials Based on Screening Results

CONTAMINANT SOURCE	NOT CANDIDATES FOR FURTHER STUDY (Level I result less than the decision guide)	POTENTIAL CANDIDATES FOR FURTHER STUDY (Refined Level I result less than the decision guide)	HIGH PRIORITY FOR FURTHER STUDY (Refined Level I result greater than the decision guide)
THE K-25 SITE	Neptunium-237 (cancer) Evaluated qualitatively: carbon fibers, four-ring polyphenyl ether, glass fibers, Triplex coating	Copper powder (non-cancer) Nickel (cancer) Nickel (non-cancer) Technetium-99 (cancer)	Arsenic (cancer) Arsenic (non-cancer)
THE Y-12 PLANT	Beryllium Compounds[1] (non-cancer) Neptunium-237 (cancer) Niobium[2] (non-cancer) Tetramethylammoniumborohydride (TMAB) Tritium (cancer) Zirconium[3] (non-cancer) Evaluated qualitatively: boron carbide, boron nitride, rubidium nitrate, rubidium bromide, tellurium, titanium boride, yttrium boride, zirconium	Beryllium Compounds (cancer) Lithium Compounds[4] (non-cancer) Technetium-99 (cancer)	Arsenic (cancer) Arsenic (non-cancer) Lead (non-cancer)
THE OAK RIDGE RESERVATION *(ALL COMPLEXES)*		Chromium(VI) (cancer) Chromium(VI) (non-cancer)	

---

[1] Forms of beryllium used include beryllium hydride, beryllium deuteride, beryllium metal, beryllium oxide, niobium beryllide, tantalum beryllide.

[2] Niobium was used as part of “mulburry” alloy (depleted uranium, niobium, and zirconium) and “binary” alloy (depleted uranium and niobium).

[3] Zirconium was used as part of “mulburry” alloy (depleted uranium, niobium, and zirconium).

[4] Forms of lithium used include lithium chloride, lithium deuteride, lithium fluoride, lithium hydride, and lithium tetraborate.

8.0 REFERENCES

ACGIH 1996. American Conference of Governmental Industrial Hygienists, Inc. *Documentation of the Threshold Limit Values® and Biological Exposure Indices*. Ninth Edition. Cincinnati, Ohio.

Alloway 1990. Alloway, B.J. *Heavy Metals in Soil*. John Wiley and Sons, Inc. New York. 1993. p. 95-97.

Alpha-5 Chronology. Author, date unknown. *Chronology of the Alpha-5 Plant 1953-54*. (ChemRisk Repository No. 3265).

Anden et al. 1975. A.W. Anden, S.E. Lindberg, and L.C. Bate. Atmospheric Input and Geochemical Cycling of Selected Trace Elements in Walker Branch Watershed. ORNL-NSF-EATC-13.

Ashwood et al. 1986. T.L. Ashwood, C.R. Olsen, I.L. Larsen, and P.D. Lowry. *Sediment Contamination in Streams Surrounding the Oak Ridge Gaseous Diffusion Plant*. ORNL/TM-9791. (ChemRisk Repository No. 166).

ATSDR 1992. Agency for Toxic Substances and Disease Registry. *Toxicological Profile for Nickel: Update*. U.S. Department of Health and Human Services. February.

ATSDR 1993. *Toxicological Profile for Arsenic.* Prepared by Life Systems, Inc. under subcontract to Clement International Corporation, Contract No. 205-88-0608, for the U.S. Department of Health and Human Services, Public Health Service, Agency for Toxic Substances and Disease Registry. April 1993.

ATSDR. 1995. *Toxicological Profile for Asbestos (Update).* Agency for Toxic Substances and Disease Registry, Atlanta, GA and the U.S. Public Health & Human Services. August.

ATSDR 1996. Agency for Toxic Substances and Disease Registry. *Toxicological Profile for Copper: Update*. U.S. Department of Health and Human Services.

ATSDR 1997a. Agency for Toxic Substances and Disease Registry. *Beryllium Toxicological Profile* on CD-ROM. U.S. Public Health Service. Lewis Publishers New York.

ATSDR 1997b. Agency for Toxic Substances and Disease Registry. *Chromium Toxicolgical Profile* on CD-ROM. U.S. Public Health Service. Lewis Publishers New York.

ATSDR 1997c. Agency for Toxic Substances and Disease Registry. *Lead Toxicological Profile* on CD-ROM. U.S. Public Health Service. Lewis Publishers, New York.

Baes et al. 1984. C.F. Baes, R.D. Sharp, A.L. Sjoreen, and R.W. Shor. *A Review and Analysis of Parameters for Assessing Transport of Environmentally Released Radionuclides through Agriculture*. Oak Ridge National Laboratory. ORNL-5786. September.

Baylor 1997. Personal communications with Richard Baylor, Y-12 Classification Office, by Susan Flack of the project team on January 17 and July 15, 1997. Interview notes available.

Beeles 1997. Interview with Don Beeles, Chemical Engineer, and Bill Ross, Environmental Program Administrator, for the Bull Run Steam Plant by C.J. Lewis of the project team. May 9, 1997.

Beyer and Cromartie 1987. Beyer, W.N., and E.J. Cromartie. A survey of Pb, Cu, Zn, Cd, Cr, As, and Se in earthworms and soil from diverse sites. *Environ Monit Assess* 8: 27-36.

Bogard 1983. Bogard, J. Hazardous Material Classification of Tritium-Contaminated Deuterium. July 21, 1983. In: Trtium in D_2 Gas. Y/TS-1565/Del Rev.

Bolton 1971. N.E. Bolton. Report of Subcommittee #6. Oak Ridge National Laboratory Intra-laboratory Correspondence. July 8.

Byrnes 1947. Byrnes, JJ. *A History of the Process Utilities Department.* Process Utilities Department Carbide and Carbon Chemicals Corporation, Oak Ridge, Tennessee. KZ-2498. April 1947.

Cannon et al. 1983. Cannon, J.R., J.B. Saunders, and R.F. Toia. Isolation and preliminary toxicological evaluation of arsenobetaine-the water-soluble arsenical constituent from the hepatopancreas of the western rock lobster. *Sci Total Environ* 31: 181-185.

Case 1952. Case, JM, Watkins, ER. How Oak Ridge Machines Beryllium. *American Machinist*, December 22,1952. Pp. 93-97.

CDC 1991. Centers for Disease Control. *Preventing Lead Poisoning in Young Children*. Centers for Disease Control and Prevention, Atlanta, Georgia.

ChemRisk 1993. *Oak Ridge Health Studies Phase I Report, Volume II– Part B– Dose Reconstruction Feasibility Study. Tasks 3 & 4: Identification of Important Environmental Pathways for Materials Released from the Oak Ridge Reservation*. Prepared for the Tennessee Department of Health. September 1993.

ChemRisk 1996. *Oak Ridge Health Studies, Oak Ridge Dose Reconstruction Task 7 Report–Screening Method for the Oak Ridge Dose Reconstruction*. Prepared for the Tennessee Department of Health. April 1996.

ChemRisk 1997. *Task 6 Report. Uranium Releases from the Oak Ridge Reservation-A Review of the Quality of Historical Effluent Monitoring Data and a Screening Evaluation of Potential Off-Site Health Risks.* Prepared for the Tennessee Department of Health. February 1997.

Choat 1996. E.E. Choat. *Ventilation Systems of Y-12 Buildings 9201-4, 9201-5, 9204-4, 9201-2, 81-10 and steam plants.* Report Prepared for ChemRisk Under Contract to the Tennessee Department of Health. August 16, 1996. (ChemRisk Repository No. 3263).

Cook et al. 1992. R.B. Cook, S.M. Adams, J.J. Beauchamp, M.S. Bevelhimer, B.G. Blaylock, C.C. Brandt, C.J. Ford, M.L. Frank, M.J. Gentry, S.K. Holladay, L.A. Hook, D.A. Levine, R.C. Longman, C.W. McGinn, J.L. Skiles, G.W. Suter, and L.F. Williams. *Phase I Data Summary Report for the Clinch River Remedial Investigation: Health Risk and Ecological Risk Screening Assessment*. Environmental Sciences Division. Report ORNL/ER-155. December 1992. (ChemRisk Repository No. 1078).

Cromer 1969. S. Cromer. Letter to W. J. Wilcox, Jr. re: chromate contamination in Y-12 effluent. October 1.

Davidson et al. 1985. Davidson, C.I., W.D. Goold, T.P. Mathison et al. Airborne trace elements in Great Smoky Mountains, Olympic, and Glacier National Parks. *Environ Sci Technol* 19: 27-35.

DeMonbrum and Muenzer 1975. J.R. DeMonbrum and W.A. Muenzer. Chromate abatement in the Y-12 Plant's New Hope Pond. Oak Ridge Y-12 Plant. Y-1993. September 3.

Dragun and Chiasson 1991. J. Dragun and A. Chiasson. *Elements in North American Soils*. Hazardous Materials Control Resources Institute. Greenbelt, Maryland.

Dykstra 1970. Dykstra, J, Pashley, J.H., Shapiro, T., Richardson, E.W., Kite, H.T., Cooke, G.F., Seufzer, P.F. *Cooling Water Discharge Streams at Gaseous Diffusion Plants and Y-12.* Union Carbide Corporation, Nuclear Division. K-GD-279. Limited Distribution.

Eckel and Langley 1988. Eckel, W.P., and W.D. Langley. A background-based ranking technique for assessment of elemental enrichment in soils at hazardous waste sites. In: Superfund '88: Proceedings of the 9th National Conference, November 28-30, 1988, Washington, DC. Silver Spring, MD: The Hazardous Materials Control Research Institute, 282-286.

Egli et al. 1985. Egli, D. et al. *The Report of the Joint Task Force on Uranium Recycle Materials Processing*. DOE/OR-859, Oak Ridge Operations, U.S. Department of Energy, pp. 13, 40, 67. 1985. (ChemRisk Repository No. 2909).

FDA 1982-84. Food and Drug Administration. Arsenic intake from individual foods in market baskets 1-8. Unpublished data available in EPA/OPP files. 1982-1984.

Fee 1986. Fee, G. *Historic Chemical Release Report for the Y-12 Plant.* Prepared by Martin Marietta Energy Systems, Oak Ridge, Tennessee. Report No. A86X-0415-052. April 15.

Finley et al. 1996. Finley, B., Kerger, B., Paustenbach, D. Human ingestion of chromium (vi) in drinking water: Pharmacokinetics following repeated exposure. Submitted for publication.

Fowlkes et al. 1959. Fowlkes, C., Fletcher, M., Gamble, J. Summary of Recirculating Water Treatment Tests for March 1956 through December 1959. Union Carbide Corporation. (ChemRisk Repository No. 2486).

Franzblau and Lilis 1989. Franzblau, A., and R. Lilis. Acute arsenic intoxication from environmental arsenic exposure. *Arch Environ Health* 44: 385-390.

Gartrell et al. 1986. Gartrell, M.J., J.C. Craun, D.S. Podrebarac et al. Pesticides, selected elements, and other chemicals in adult total diet samples, October 1980-March 1982. *J Assoc Off Anal Chem* 69: 146-161.

Godbold and Tompkins 1978. J.H. Godbold, Jr. and E.A. Tompkins. *A Long-term Mortality Study of Workers Occupationally Exposed to Metallic Nickel at the Oak Ridge Gaseous Diffusion Plant*. Medical and Health Sciences Division, Oak Ridge Associated Universities, Oak Ridge, Tennessee. October.

Goddard et al. 1991.Goddard, P.L., A.J. Legeay, D.S. Pesce, and A.M. Stanley. “Site Description of Environmental Restoration Units at the Oak Ridge K-25 Site, Oak Ridge, Tennessee.” Martin Marietta Energy Systems. K/ER-47, ChemRisk Repository No. 242.

Goodpasture and Rogers 1986. Goodpasture, S.T. and J.G. Rogers. *Oak Ridge Gaseous Diffusion Plant Historical Chemical Release Report.* Prepared by Martin Marietta Systems, Inc., Oak Ridge Gaseous Diffusion Plant, Oak Ridge, Tennessee for U.S. Department of Energy. Report No. K/HS-106 under Contract No. DE-AC05-84OR21400. April 29.

Handler 1991. Memo from B.H. Handler to H. Groh with attachments: “Milestones in Gaseous Diffusion”, “Y-12 Plant Milestones”, and “Oak Ridge National Laboratory Milestones.” (ChemRisk Repository No. 255).

Headrick 1997. Headrick, L. Personal Communication January 1997.

Heit et al. 1984. Heit, M., C. Klusek, and J. Baron. Evidence of deposition of anthropogenic pollutants in remote Rocky Mountain lakes. *Water Air Soil Pollut* 22: 403- 416.

Hibbitts 1984-87. Hibbitts, H.W. Transmittal of environmental sampling data, 12-11-84 through 2-14-87. (ChemRisk Repository No. 536).

Hoffman et al. 1984. Hoffman, F.O., B.G. Blaylock, C.C. Travis, K.L. Daniels, E.L. Etnier, K.E. Cowser, and C.W. Weber. 1984. *Preliminary Screening of Contaminants in Sediments*. ORNL/TM-9370.

Hoffman 1982. F.O. Hoffman. Environmental Behavior of Technetium in Soil and Vegetation: Implications for Radiological Impact Assessment. ORNL -5856.

Hoffman et al. 1991. Hoffman, F.O., B.G. Blaylock, M.L. Frank, L.A. Hook, E.L. Ethier, and S.S. Talmage. *Preliminary Screening of Contaminants in the Off-Site Surface Water Environment Downstream of the U.S. DOE Oak Ridge Reservation.* ORNL/ER-9. (ChemRisk Repository No. 153).

HSDB 1999. Hazardous Substances Database. On-line search via TOXNET.

IAEA 1996. International Atomic Energy Agency. Draft Revision Annex III: Special Considerations for Assessment of Discharges of Tritium and Carbon-14. Safety Series No. 57. International Atomic Energy Agency.

IARC 1980. International Agency for Research on Cancer. Monographs on the evaluation of the carcinogenic risk of chemicals to humans. Vol. 23. Some metals and metallic compounds, p.39-141. Lyon, France.

ICRP 1990. International Commission on Radiological Protection. 1990. "Recommendations of the International Commission on Radiological Protection." ICRP Publication 60. Annals of the ICRP. Vol. 21, No. 1-3 Pergamon Press, Oxford. ISSN 0146-0653. November 1990.

ICRP 1992. International Commission on Radiological Protection. *Radiation Dose to Patients from Radiopharmaceuticals.* ICRP Publication 53. Oxford, England. (ChemRisk Repository No. 4013).

ICRP 1993. International Commission on Radiological Protection. *Age-dependent Doses to Members of the Public from Intake of Radionuclides: Part 2, Ingestion Dose Coefficients.* ICRP Publication 67. Oxford, England.

ICRP 1995. International Commission on Radiological Protection (ICRP). *Age-dependent Doses to Members of the Public from Intake of Radionuclides: Part 4 Inhalation Dose Coefficients.* ICRP Publication 71. Pergamon Press.

ICRP 1996. International Commission on Radiological Protection (ICRP). *Age-dependent Doses to Members of the Public from Intake of Radionuclides: Part 5 Compilation of Ingestion and Inhalation Dose Coefficients.* ICRP Publication 72. Pergamon Press.

IRIS 1997. Integrated Risk Information System. National Library of Medicine On-Line Data Base.

IRIS 1998. Integrated Risk Information System. National Library of Medicine On-Line Data Base.

Jallouk 1974. *Environmental Aspects of Cooling Tower Operation: Survey of the Emission, Transport, and Deposition of Drift from the K-31 and K-33 Cooling Towers at ORGDP*. Oak Ridge Gaseous Diffusion Plant. K-1859. February 1974.

Jongen et al. 1985. Jongen, W.M., J.M. Cardinaals, and P.M. Bos. Genotoxicity testing of arsenobetaine, the predominant form of arsenic in marine fishery products. *Food Chem Toxicol* 23: 669-673.

Joyner and Marshall 1975. J.D. Joyner and G.L. Marshall. *Laboratory Division Monthly Activities Report (U)– February 1975.* Oak Ridge Gaseous Diffusion Plant, Oak Ridge, Tennessee. K-TL-430. March.

Kaise et al. 1985. Kaise, T., S. Watanabe, and K. Itoh. The acute toxicity of arsenobetaine. *Chemosphere* 14: 1327-1332.

Kerger 1996. Kerger, B. Refined exposure assessment for ingestion of tapwater contaminated with hexavalent chromium: Consideration of exogenous and endogenous reducing agents. *Journal of Exposure Analysis and Environmental Epidemiology Vol. 6 No. 2 163-179.*

Ketcham 1948. N.H. Ketcham. Report of the Industrial Hygiene Field Investigations during the First and Second Quarters, 1948. Carbide and Carbon Chemicals Corporation, Medical Department, K-25 Plant. Report K-247, Parts 1 and 2. August 9, 1948. Sanitized version.

Kornegay et al. 1991. F. C. Kornegay, D.C. West, R.A. Evans, S.T. Goodpasture, M.F. Tardiff, and A.R. Wilson. *Oak Ridge Reservation Environmental Report for 1990; Volume 1: Narrative, Summary, and Conclusions.* Prepared by Oak Ridge National Laboratory, Environmental, Safety, and Health Compliance for U.S. Department of Energy, Oak Ridge, Tennessee. ES/ESH-18/V1. September.

Kornegay et al. 1992. F.C. Kornegay, D.C. West, R.A. Evans, M.F. Tardiff, F.D. Adams, and P.C. Mucke. *Oak Ridge Reservation Environmental Report for 1991; Volume 2: Data Presentation.* Prepared by Oak Ridge National Laboratory, Environmental, Safety, and Health Compliance for U.S. Department of Energy, Oak Ridge, Tennessee. ES/ESH-22/V2. October.

LaFrance 1955a. Letter from L. LaFrance, Industrial Hygienist, to D.A. Jennings regarding Stack Samples at Bldg. 9201-4. August 18, 1955.

LaFrance 1955b. Letter from L. LaFrance, Industrial Hygienist, to D.A. Jennings Regarding Stack Samples at Bldg. 9201-4. August 19, 1955.

LaFrance 1956. Letter from N.E. Bolton, Asst. Industrial Hygienist, to J.L. Williams regarding Air Analysis in Bldg. 9204-2. February 9, 1956.

Lay 1997. Lay, A. Personal Communication. September 12, 1997.

Layton 1987. Layton, DW, B.J. Mallon, D.H. Rosenblatt, M.J. Small. "Deriving Allowable Daily Intakes for Systemic Toxicants Lacking Chronic Toxicity Data." Regulatory Toxicology and Pharmacology 7:96-112.

LMES 1997. Lockheed Martin Energy Systems, Inc. Oak Ridge Environmental Information System (OREIS) environmental data collected during the offsite Clinch River Operable Unit project and the East Fork Poplar Creek Floodplain and Sewerline Beltway Remedial Investigation.

LMES 1996. Lockheed Martin Energy Systems. Environmental Monitoring and Surveillance on the Oak Ridge Reservation: 1995 Data. ES/ESH-71.

LMES 1997. Lockheed Martin Energy Systems, Inc. Oak Ridge Environmental Information System (OREIS) environmental data for lead collected during the East Fork Poplar Creek Floodplain and Sewerline Beltway Remedial Investigation.

Loar et al. 1981a. J.M. Loar, J.A. Solomon, and G.F. Cada. *Technical Background Information for the ORNL Environmental and Safety Report, Vol. 2: A Description of the Aquatic Ecology of White Oak Creek Watershed and the Clinch River below Melton Hill Dam*. Report ORNL/TM-7509/V2. October 1981. (ChemRisk Repository No. 154).

Loar et al. 1981b. Loar, J.M., F.A. Burkhart, G.F. Cada, J.W. Huckabee, J.T. Kitchings, K.D. Kumar, A.M. Sasson, J.A. Solomon, and J.D. Store. *Ecological Studies of the Biotic Communities in the Vicinity of the ORGDP*. ORNL/TM-6714. October. (ChemRisk Repository No. 250).

Lockheed-Martin 1996. Tritium Analysis on D_2 Gas. Y/TS-1567/Del Rev.

Marciante 1980. Memo from G. Marciante to W.S. Everett requesting beryllium emission data. January 9, 1980.

Marcus and Rispin 1988. Marcus, W.L., and A.S. Rispin. Threshold carcinogenicity using arsenic as an example. In: Cothern CR, Mehlman MA, Marcus WL, eds. *Risk Assessment and Risk Management of Industrial and Environmental Chemicals. Vol XV*. Princeton, NJ: Princeton Scientific Publishing Co., p.133-158.

Marks 1996. Marks' Standard Handbook for Mechanical Engineers. Avallone, E.A. and T. Baumeister III, eds. McGraw-Hill. New York. 1996.

MMES 1981. Martin Marietta Energy Systems. *Clinch River and Poplar Creek Bottom Sediments Data Routine Sampling Program 1975-1981*. (ChemRisk Repository No. 643).

MMES. 1984. *Unclassified Descriptions of Inactive Hazardous Waste Disposal Sites at the Oak Ridge Y-12 Plant.* Prepared by Martin Marietta Energy Systems, Inc. For U.S. Department of Energy, Oak Ridge, Tennessee. August.

MMES 1987. Martin Marietta Energy Systems. *Environmental Surveillance of the U.S. Department of Energy Oak Ridge Reservation and Surrounding Environs During 1986.* ES/ESH-1. ChemRisk Repository No. 1212.

MMES 1988. Martin Marietta Energy Systems. *Environmental Surveillance of the U.S. Department of Energy Oak Ridge Reservation and Surrounding Environs During 1987*. ES/ESH-4. ChemRisk Repository No. 370.

MMES 1988. Martin Marietta Energy Systems, Inc. *RCRA Facility Investigation Plan for the K-720 Ash Pile at the ORGDP*. K/HS-150. May.

MMES 1989. Martin Marietta Energy Systems. *Oak Ridge Reservation Environmental Report 1988.* ES/ESH-8. ChemRisk Repository No. 254.

MMES 1990. Martin Marietta Energy Systems. *Oak Ridge Reservation Environmental Report 1989.* ES/ESH-13. ChemRisk Repository No. 374.

MMES 1992. Martin Marietta Energy Systems, Inc. *Safety Analysis Report Update Program: Phase I Hazard Screening Analysis for Building K-1037 Barrier Process Plant*. Prepared by Plant Safety Evaluation Team, Oak Ridge K-25 Site. HS/K-1037/PK/48.0/R0. March.

MMES 1992. Martin Marietta Energy Systems. *Oak Ridge Reservation Environmental Report 1991.* ES/ESH-22. ChemRisk Repository No. 709.

MMES 1993. Martin Marietta Energy Systems. *Oak Ridge Reservation Environmental Report 1992.* ES/ESH-31.

MMES 1994. Martin Marietta Energy Systems. *Oak Ridge Reservation Annual Site Environmental Report 1993*. ES/ESH-47 & 48.

Martin Marietta 1995. Lockheed-Martin Energy Systems. “Compilation of Information about Tritium Use and Processing at Y-12.” Y/TS-1566. 1995.

Murray 1956. Letter from J.P. Murray, Y-12 Plant Manager, to R.C. Armstrong, USAEC Director of Production, regarding the Y-12 steam plant, 9401-3. January 27, 1956. (ChemRisk Repository No. 3267).

NAS 1977. National Academy of Sciences. Medical and biologic effects of environmental pollutants: Arsenic. Washington, DC.

NCRP 1996. National Council on Radiation Protection and Measurements. *Screening Models for Releases of Radionuclides to Atmosphere, Surface Water, and Ground.* NCRP Report No. 123.

Ng et al. 1977. Y.C. Ng, C.S. Colsher, D.J. Quinn, and S.E. Thompson. *Transfer Coefficients for the Prediction of the Dose to Man via the Forage-Cow-Milk Pathway from Radionuclides Released to the Biosphere*. Lawrence Livermore Laboratory. UCRL-51939.

Oakes et al. 1987. Oakes, T.W., C.W. Kimbrough, S.F. Huang, P.M. Pritz, C.S. Gist, S.T. Goodpasture, C.W. Weber, and F.M. O'Hara. *Environmental Surveillance of the U.S. Department of Energy Oak Ridge Reservation and Surrounding Environs During 1986.* Prepared by Martin Marietta Energy Systems, Inc., Environmental and Safety Activities for the U.S. Department of Energy, Oak Ridge, Tennessee. ES/ESH-1/V2. April.

ORGDP 1978. Oak Ridge Gaseous Diffusion Plant. *Draft Material Balance, ORGDP*. From Box 8-3-5, K-25 Site Records Center.

ORGDP 1981a. Oak Ridge Gaseous Diffusion Plant. *Clinch River and Poplar Creek Bottom Sediments Data routine Sampling Program 1975-1981*. ChemRisk Repository No. 643.

ORGDP 1981b. Oak Ridge Gaseous Diffusion Plant. Tabulation of Air Sampling Data for Copper.

ORGDP 1981c. Oak Ridge Gaseous Diffusion Plant. Tabulation of Air Sampling Data for Nickel.

ORNL 1995. Oak Ridge National Laboratory. Remedial Investigation/Feasibility Study for the Clinch River/ Poplar Creek operable unit. Volume 1. Main text. ORNL/ER-315/V1&D1.

OSHA 1978. Occupational Safety and Health Administration. 29 CFR 1910.1025.

Owings 1995. Owings, E. *Historical Review of Accountable Nuclear Materials at the Y-12 Plant (U).* Y/DG-256549 Rev. 2. July 28, 1995.

Page 1981. Page, G.W. Comparison of groundwater and surface water for patterns and levels of contamination by toxic substances. *Environ Sci Technol* 15: 1475-1481.

PDR 1995. Physicians Desk Reference, 49th Edition. Medical Economics Data Production Company, Montvale, NJ.

Peer 1993. PEER Consultants, P.C. Asbestos Inventory and Assessment for the U.S. Department of Energy for the City of Oak Ridge, Tennessee- Final Report. September.

Pesci 1995. N. Pesci. *Tabulation of Lifetime Coal Consumption.* Historical Investigation Special Report S/R 7. K-701 Power Station/ Boiler House.

Petito and Beck 1990. Petito, C.T. and Beck, B.D. Evaluation of evidence of nonlinearities in the dose-response curve for arsenic carcinogenesis. *Trace Substances in Environmental Health* 24: 143-176.

Ramsey 1959. Letter from M.E. Ramsey to W.H Jorday, T.A. Lincoln, and K.Z. Morgan answering a Request for Toxicological Information. Union Carbide Nuclear Company. January 29, 1959.

Richesin 1992. Richesin, M. Oxygen Stream, Memo to Johnny Skinner (Y-12 Plant), July 21, 1992. In: Tritium (from L. Headrick, 9204-2) Y/TS-1566.

Rogers et al. 1988a. Rogers, J.G., K.L. Daniels, S.T. Goodpasture, C.W. Kimbrough, and E.W. Whitfield. *Environmental Surveillance of the U.S. Department of Energy Oak Ridge Reservation and Surrounding Environs During 1987, Volume 1: Narrative, Summary, and Conclusions.* Prepared by Martin Marietta Energy Systems, Inc. for the U.S. Department of Energy, Oak Ridge, Tennessee. ES/ESH-4/V1. April.

Rogers et al. 1988b. Rogers, J.G., K.L. Daniels, S.T. Goodpasture, and C.W. Kimbrough. *Environmental Surveillance of the U.S. Department of Energy Oak Ridge Reservation and Surrounding Environs During 1987, Volume 2: Data Presentation.* Prepared by Martin Marietta Energy Systems, Inc. for the U.S. Department of Energy, Oak Ridge, Tennessee. ES/ESH-4/V2. April.

SAIC 1995. *Draft Phase 2 Remedial Investigation/Baseline Risk Assessment Report and Feasibility Study for the K-1070 C/D Classified Burial Ground at the Oak Ridge K-25 Site, Oak Ridge, Tennessee.* Prepared by Science Applications International Corporation and Jacobs Environmental Restoration Team for U.S. Department of Energy, Office of Environmental Restoration and Waste Management, Oak Ridge, Tennessee. DOE/OR/01-1297 & D1 &V1. June.

SAIC 1993. Science Applications International Corporation. *East Fork Poplar Creek-- Sewer Line Beltway Remedial Investigation Report*. Volumes I-IV. DOE/OR/02-1119&D1. April 1993. (ChemRisk Repository Nos. 1265-1268).

Schmitt 1970. Schmitt, C.R. *Survey of the Fly-Ash Disposal System at the Oak Ridge Y-12 Plant.* Y-1713. April 20.

Seiler et al. 1988. Seiler, H.G., Sigel, H., and Sigel. A. *Handbook on Toxicity of Inorganic Compounds.* Marcel Dekker, Inc., New York.

Smith et al. 1987. Smith, R.A., R.B. Alexander, and M.G. Wolman. Water-quality trends in the nation's rivers. *Science* 235: 1607-1615.

Thomason and Associates 1996. *Historical Building Assessment of the Oak Ridge Y-12 Plant - DRAFT*. May 1996. Y/TS-1525. (ChemRisk Repository No. 3267).

Tindell and Wood 1997. Tindell, T. and A. Wood. Interview with Terry Tindell and Al Wood. Y-12 Plant 9201-5 Building. January 16, 1997.

Tseng 1977. Tseng, W.P. Effects and dose-response relationships of skin cancer and blackfoot disease with arsenic. *Environ Health Perspect* 19: 109-119.

Tseng et al. 1968. Tseng, W.P., H.M. Chu, S.W. How et al. Prevalence of skin cancer in an endemic area of chronic arsenicism in Taiwan. *J Natl Cancer Inst* 40: 453- 463.

Turner et al. 1991. R.R. Turner, M.A. Bogle, L.L. Heidel and L.M. McCain. *Mercury in Ambient Air at the Oak Ridge Y-12 Plant July 1986 through December 1990.* August 1991. Y/TS-574. (ChemRisk Repository No. 396).

Turner et al. 1986. R.R. Turner, M.A. Bogle, M.A. Kane, T.M. Mercier. *Characterization of Y-12 Plant Coal Ash Discharge to McCoy Branch.* Y/TS-177. June. (ChemRisk Repository No. 375).
TVA 1979. Tennessee Valley Authority. *Status of the Nonradiological Water Quality and Nonfisheries Biological Communitees in the Clinch River prior to Construction of the Clinch River Breeder Reactor Plant, 1975-1978*. Division of Environmental Planning. Chattanooga, Tennessee. February 1979. (ChemRisk Repository No. 854).

TVA 1985a. Tennessee Valley Authority. *Instream Contaminant Study, Task 1: Water Sampling and Analysis*. Office of Natural Resources and Economic Development, Knoxville, Tennessee. April 1985. (ChemRisk Repository No. 840).

TVA 1985b. Tennessee Valley Authority. *Instream Contaminant Study, Task 2: Sediment Characterization, Volume I*. Office of Natural Resources and Economic Development, Knoxville, Tennessee. April 1985. (ChemRisk Repository No. 841).

TVA 1985c. Tennessee Valley Authority. *Instream Contaminant Study, Task 2: Sediment Characterization, Volume II*. Office of Natural Resources and Economic Development, Knoxville, Tennessee. January 1985. (ChemRisk Repository No. 842).

TVA 1985d. Tennessee Valley Authority. *Instream Contaminant Study, Task 4: Fish Sampling and Analysis*. Office of Natural Resources and Economic Development, Knoxville, Tennessee. April 1985. (ChemRisk Repository No. 844).

TVA 1986. Tennessee Valley Authority. *Heavy Metals and PCB Concentrations in Sediments from Selected TVA Reservoirs– 1982*. Division of Air and Water Resources. TVA/ONRED/ AWR 86/35. April 1986. (ChemRisk Repository No. 850).

TVA 1989. Tennessee Valley Authority. *Results of Fish Tissue Screening Studies from Sites in the Tennessee and Cumberland Rivers in 1987*. Office of Water Resources. Chattanooga, Tennessee. TVA/WR/AB– 89/5. May 1989. (ChemRisk Repository No. 846).

TVA 1990. Tennessee Valley Authority. *Results of Fish Tissue Screening Studies from Sites in the Tennessee and Cumberland Rivers in 1988*. Office of Water Resources. Chattanooga, Tennessee. TVA/WR/AB– 90/7. May 1989. (ChemRisk Repository No. 846).

UCC 1964. Union Carbide Corporation. Letters regarding USPHS water samples from the Clinch River. (ChemRisk Repository No. 2652).

UCC 1971. Union Carbide Corporation. "Design Criteria Production Support Facilities Special Facilities K-1037 Barrier Plant, Special Facility I-M-4 (U). Design Criteria K-1037-22, Issue No. 1." Prepared by Engineering Division, Oak Ridge Tennessee. April 5, 1971.

UCC 1973. Union Carbide Corporation. *Environmental Monitoring Report, United States Atomic Energy Commission, Oak Ridge Facilities. Calendar Year 1972.* Report UCC-ND-244. (ChemRisk Repository No. 943).

UCC 1974. Union Carbide Corporation. *Environmental Monitoring Report, United States Atomic Energy Commission, Oak Ridge Facilities. Calendar Year 1973.* Report UCC-ND-280. (ChemRisk Repository No. 944).

UCC 1975. Union Carbide Corporation. *Environmental Monitoring Report, United States Energy Research and Development Administration, Oak Ridge Facilities. Calendar Year 1974.* Report UCC-ND-302. (ChemRisk Repository No. 945).

UCC 1976. Union Carbide Corporation. *Environmental Monitoring Report, United States Energy Research and Development Administration, Oak Ridge Facilities. Calendar Year 1975.* Report Y/UB-4. (ChemRisk Repository No. 946).

UCC 1977a. Union Carbide Corporation. *Environmental Concentrations of Nickel Found Around the ORGDP.*

UCC 1977b. Union Carbide Corporation. *Environmental Monitoring Report, United States Energy Research and Development Administration, Oak Ridge Facilities. Calendar Year 1976.* Report Y/UB-6. (ChemRisk Repository No. 947).

UCC 1978. Union Carbide Corporation. *Environmental Monitoring Report, United States Department of Energy, Oak Ridge Facilities. Calendar Year 1977.* Report Y/UB-8. (ChemRisk Repository No. 948).

UCC 1979. Union Carbide Corporation. *Environmental Monitoring Report, United States Department of Energy, Oak Ridge Facilities. Calendar Year 1978.* Report Y/UB-10. (ChemRisk Repository No. 949).

UCC 1980. Union Carbide Corporation. *Environmental Monitoring Report, United States Department of Energy, Oak Ridge Facilities. Calendar Year 1979.* Report Y/UB-13. (ChemRisk Repository No. 950).

UCC 1981. Union Carbide Corporation. *Environmental Monitoring Report, United States Department of Energy, Oak Ridge Facilities. Calendar Year 1980*. Report Y/UB-15. (ChemRisk Repository No. 951).

UCC 1982. Union Carbide Corporation. *Environmental Monitoring Report, United States Department of Energy, Oak Ridge Facilities. Calendar Year 1981.* Report Y/UB-16. (ChemRisk Repository No. 952).

UCC 1983. Union Carbide Corporation. *Environmental Monitoring Report, United States Department of Energy, Oak Ridge Facilities. Calendar Year 1982.* Report Y/UB-18. (ChemRisk Repository No. 953).

UCCND 1950-1970. Union Carbide Corporation Nuclear Division. *Y-12 Plant Quarterly Reports.* Y-1002 through Y-1020, Y-1200 through Y-1220, Y-1421 through Y-1428, Y-1513, Y-1517, Y-1631, Y-1635, Y-1639.

UCCND 1956. Union Carbide Corporation Nuclear Division. *Y-12 Plant Quarterly Report for April-June 1956.* (ChemRisk Repository No. 3264).

UCCND 1957. Union Carbide Corporation Nuclear Division. *Alloy and Solvent Loss Study for Alpha-5*. October. Y/HG-437.

UCCND 1958. Union Carbide Corporation Nuclear Division. *A Preliminary Study of the Recovery of Lithium and Mercury Losses.* January. Y/HG-545/del rev.

UCCND 1959a. Union Carbide Corporation Nuclear Division. *Monthly Tabulations of Sump Losses*. April. Y/HG-0347/del rev.

UCCND 1959b. Union Carbide Corporation Nuclear Division. *A Brief Description of the Gaseous Diffusion Process.* KP-1753. October 23, 1959.

UCCND 1976. Union Carbide Corporation Nuclear Division. Memorandum from R.W. Levin to R.B. Farrar et al. Subject: Summary of October 11, 1976 Meeting for Review Environmental Monitoring Progress. Office of Health, Safety, and Environmental Affairs, Oak Ridge, Tennessee. November 29.

UCCND 1977. Union Carbide Corporation Nuclear Division. Memorandum from R.W. Levin to R.E. Barringer et al. Subject: December 13, 1976 Meeting for Review Environmental Monitoring Progress. Office of Health, Safety, and Environmental Affairs, Oak Ridge, Tennessee. January 4.

UCCND 1978. Union Carbide Corporation Nuclear Division. Memorandum from R.W. Levin to F.N. Bensey, Jr. et al. Subject: Meeting to Review Environmental Monitoring Progress. Office of Health, Safety, and Environmental Affairs, Oak Ridge, Tennessee. April 12.

UCCND 1983a. Union Carbide Corporation Nuclear Division. *Assessment of Coal Storage and Steam Management Plans.* Y/EN-4234. July 29. (ChemRisk Repository No. 910).

UCCND 1983b. Union Carbide Company Nuclear Division. The 1983 Mercury Task Force. *Mercury at Y-12: A Study of Mercury Use at the Y-12 Plant, Accountability, and Impacts on Y-12 Workers and the Environment- 1950-83*. August 18, 1983. Y/EX-21/del rev. (ChemRisk Repository No. 449).

UCNC 1957. Union Carbide Nuclear Company. *Separation of Lithium-6 and Lithium-7.* March 29, 1957. Y-F40-66/del rev. (ChemRisk Repository No. 3265).

Union Carbide 1962. *Tritium Analysis on Deuterium Gas 1959-1962.* Y/TS-1567/Del Rev. 1962.

Union Carbide 1969. *Tritium Analysis from 1969 to Fiscal Year 1987.* Y/TS-1568.

Unknown 1978. Unknown. *Material Balance ORGDP.* Released to ChemRisk by LMES Pp. 2-6. (ChemRisk Repository No. 3350).

USAEC 1965. U.S. Atomic Energy Commission. *Report of Investigating Committee- Loss of Special Nuclear Material at Y-12 Plant on January 15, 1965.* ORO 125208.

USDOE 1988. U.S. Department of Energy. *Historical Radionuclide Releases From Current DOE Oak Ridge Operations Office.* ORO-890, Appendix A Tables 11 and 12. 1988. ChemRisk Repository No. 3016.

USDOE 1995a. United States Department of Energy. *EM Fact Sheet: Chestnut Ridge Coal Ash Pond.* Summer.

USDOE 1995b. U.S. Department of Energy. *EM Fact Sheet: Powerhouse Implosion*. November.

USDOE 1995c. U.S. Department of Energy. *ORGDP Historical Inventory Differences for the Period 9/9/44 through 9/30/83.* Oak Ridge K-25 Site, K/EM-270. ChemRisk Repository No. 2366.

USDOE 1996. U.S. Department of Energy. *Remedial Investigation/ Feasibility Study of the Clinch River/ Poplar Creek Operable Unit Volumes 1 and 2*. DOE/OR/01-1393. ChemRisk Repository No. 2462.

USEPA 1974. U.S. Environmental Protection Agency. *Occurrence and Distribution of Potentially Volatile Trace Elements in Coal.* Office of Research and Development. EPA/650/2-74-054. July.

USEPA 1986a. U.S. Environmental Protection Agency. *Air Quality Criteria for Lead Volume I-IV.* Environmental Criteria and Assessment Office, Office of Research and Development, RTP, NC. EPA 600/8-83-028 a-d.

USEPA 1986b. U.S. Environmental Protection Agency. *Superfund Public Health Evaluation Manual*. Office of Emergency and Remedial Response, Washington D.C. EPA/540/1-86/060. October.

USEPA 1980. U.S. Environmental Protection Agency. *Ambient Water Quality Criteria for Arsenic.* Office of Water Regulations and Standards. EPA 440/5-80-021. Washington, DC.

USEPA 1982a. U.S. Environmental Protection Agency. Arsenic. In: *Intermedia Priority Pollutant Guidance Documents.* Office of Pesticides and Toxic Substances. Washington, DC.

USEPA 1982b. U.S. Environmental Protection Agency. *An Exposure and Risk Assessment for Arsenic.* Office of Water Regulations and Standards. EPA 440/4-85-005. Washington, DC.

USEPA 1984a. U.S. Environmental Protection Agency. *Health Assessment Document for Inorganic Arsenic. Final Report.* Environmental Criteria and Assessment Office, 2-1-3-22, 9-1-9-4. EPA 600/8-83-021F. Research Triangle Park, NC.

USEPA 1984b. U.S. Environmental Protection Agency. *Correlation of Coal Properties with Environmental Control Technology Needs for Sulfur and Trace Elements.* Office of Research and Development. EPA/600/7-84-066. June.

USEPA 1985. U.S. Environmental Protection Agency. *Ambient Water Quality Criteria for Arsenic.* EPA/440/5-84-033.

USEPA 1988. U.S. Environmental Protection Agency. *Special Report on Ingested Inorganic Arsenic: Skin Cancer; Nutritional Essentiality.* Risk Assessment Forum. EPA/625/3-87/013. Washington, DC.

USEPA 1989a. U.S. Environmental Protection Agency. *Risk Assessment Guidance for Superfund, Volume 1. Human Health Evaluation Manual (Part A),* interim final. Office of Emergency and Remedial Response, Washington, DC. EPA/540/1-89/002.

USEPA 1989b. U.S. Environmental Protection Agency. *Estimating Air Toxics Emissions from Coal and Oil Combustion Sources.* Office of Air Quality Planning and Standards. EPA/450/2-89-001. April.

USEPA 1992. U.S. Environmental Protection Agency. *Health Effects Assessment Summary Tables*. FY-1992 Annual. OHEA ECAO-CIN-821.

USEPA 1993. U.S. Environmental Protection Agency. *External Exposure to Radionuclides in Air, Water, and Soil.* Prepared by the Oak Ridge National Laboratory for the Office of Radiation and Indoor Air , Washington, D.C. Federal Guidance Report No. 12.

USEPA 1994a. U.S. Environmental Protection Agency. *Guidance Manual for the Integrated Exposure Uptake Biokinetic Model for Lead in Children (Version 0.99D*). Washington, DC.

USEPA 1994b. U.S. Environmental Protection Agency. *Guidance on Residential Lead-Based Paint, Lead-Contaminated Dust, and Lead-Contaminated Soil*. Washington, DC.

USEPA 1997. U.S. Environmental Protection Agency. Beryllium risk guidelines from the Integrated Risk Information System (IRIS). Available through the National Library of Medicine.

USEPA 1998. U.S. Environmental Protection Agency. *Region 9 Preliminary Remediation Goals.* Office of Waste Programs. May 1, 1998 update.

USEPA 1999. U.S. Environmental Protection Agency. On-line Integrated Risk Information System (IRIS).

USGS 1976. United States Geological Survey. *Collection, Chemical Analysis, and Evaluation of Coal Samples in 1975*. Open-file report 76-468.

Vahter et al. 1983. Vahter, M., Marafante, E., and Dencker, L. Metabolism of arsenobentaine in mice, rats and rabbits. *Sci Total Environ* 30: 197-211.

Walker and Blase 1960. Raymond L. Walker and Ernest F. Blase. *The Detection of a Lithium Separation Plant- A Completion Report*. June. ORNL CF/60-6-121 (ChemRisk Repository No. 2644).

Weber and White 1977. C.W. Weber and J.C. White. *Project Progress Report for January 1977– NIOSH Standards Evaluation of Nickel Powder*. Oak Ridge Gaseous Diffusion Plant, Oak Ridge, Tennessee. K/NI-1, Part 1. February.

Welch et al. 1988. Welch, A.H., M.S. Lico, and J.L. Hughes. Arsenic in groundwater of the western United States. *Ground Water* 26: 333-347.

Wilburn 1997. Wilburn, B. *Ridgelines*: Savings at steam plant are no hot air. April.

Wing 1980. *Submission of Beryllium Emission Rate Data*. Letter to H.E. Hodges from J.F. Wing, March 17, 1980.

Zimmerman 1969. B. Zimmerman. Letter to J.M. Googin re: Cascade uprating program process cooling water systems (ESO M-35446-03). May 19.

APPENDIX A

DETAILS OF THE STANDARD TASK 7 SCREENING METHOD

OAK RIDGE HEALTH STUDIES

OAK RIDGE DOSE RECONSTRUCTION

SCREENING METHOD
FOR THE
OAK RIDGE DOSE RECONSTRUCTION

April 1996

This document was prepared as a team effort by the following individuals:

Project Manager

Thomas E. Widner[1]

Task 7 Manager

Kathleen M. Thiessen[2]

Principal Authors

Kathleen M. Thiessen[2]
Jana S. Hammonds[2]
Cathy J. Lewis[2]
F. Owen Hoffman[2]
E. Ivan White[2]

---

[1] ChemRisk® Division, McLaren/Hart Environmental Services, Inc., 1135 Atlantic Avenue, Alameda, California 94501

[2] SENES Oak Ridge, Inc., 102 Donner Drive, Oak Ridge, Tennessee 37830

TABLE OF CONTENTS

1.0 INTRODUCTION

The purpose of screening in the Oak Ridge Dose Reconstruction is to permit attention and resources to be focused on the most important contaminants and pathways and to avoid dilution of resources by identifying situations that are obviously of only minor importance. This will be carried out by consideration of two goals:

(1) *Rapid identification of situations that have produced doses or health risks to exposed individuals or populations that are clearly below minimum levels of concern established by the Oak Ridge Health Agreement Steering Panel (ORHASP).* Detailed analysis of these contaminants and pathways will not be undertaken unless time and resources are available or unless there is sufficient public concern to warrant such studies. By postponing detailed analysis of contaminants that posed very low risks to exposed people, resources can be applied to those situations that are much more likely to have produced significant risk of adverse health effects in exposed people.

(2) *Rapid identification of any situations that are likely to have exceeded the established minimum exposure or risk levels of concern.* These situations warrant the highest priority for detailed investigation. By identification of such situations, resources can be focused on those situations in which people may have received significant exposures to past releases.

The results of the screening calculations will be compared to preselected, risk-based decision criteria (minimum levels of concern established by ORHASP). Use of screening criteria permits contaminants to be considered independently of each other, an important advantage for this study because the quality and quantity of information vary among contaminants, information will not be available for all contaminants at once, and new or improved information may become available for some contaminants. Designation of the priority of each contaminant will be made in terms of the risk of that contaminant to a specified target individual.

The values to be used as screening criteria (minimum levels of concern) are subject to the approval of the ORHASP; the present recommendations (described in detail in Section 2.4) are a lifetime risk of adverse health effects of 1×10^{-4} (1 in 10,000) for carcinogenic contaminants and noncarcinogens for which a dose-response function can be estimated and a Hazard Index of 1.0 for noncarcinogenic contaminants.

2.0 METHODOLOGY FOR SCREENING

In accordance with the goals described in Section 1, further screening of contaminants in the Oak Ridge Dose Reconstruction will be carried out using a two-level screening approach in which, for each contaminant, risks estimated using two alternative sets of assumptions for releases, environmental transport, exposures, and lifestyles are compared to appropriately selected, risk-based decision criteria. The target individual, assumptions, and parameter values for each level of screening will be chosen with the goal of that screening step clearly in mind. Contaminants for which the risks to a maximally exposed individual (Level I) are clearly below a specified minimum level of concern can be assigned a low priority for further study. Contaminants for which the risks to a more “typical” individual (Level II) are clearly above a specified level of concern can be assigned the highest priority for detailed study, with the remaining contaminants to be studied next. This screening approach is designed to make use of the best information available for each contaminant, even though the amount and quality of information may differ among contaminants. A flow chart describing the screening process is provided in Figure A-1.

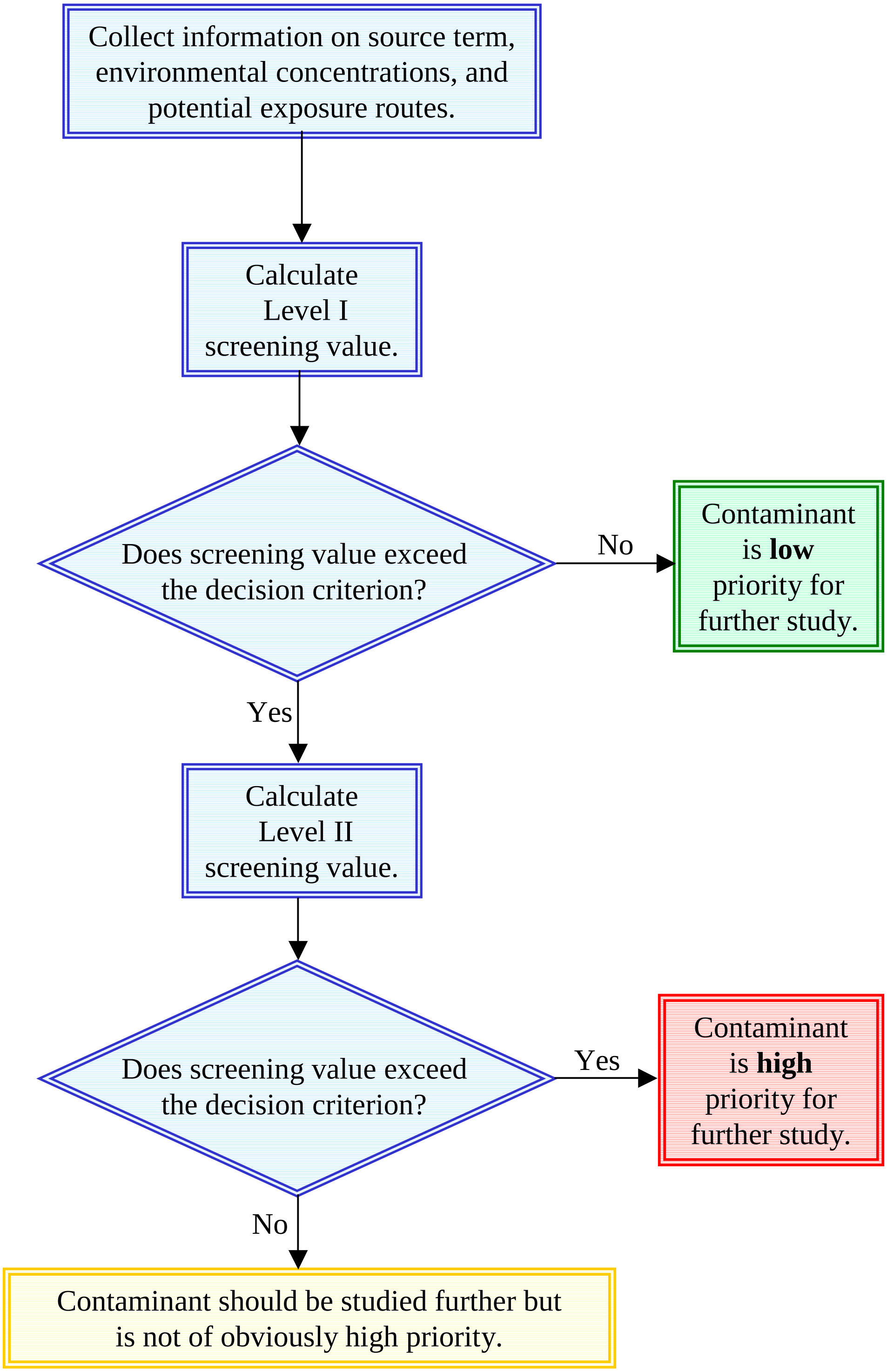


Figure A-1: Diagram of a screening process for determination of the priority of a contaminant for further study.

2.1 Level I Screening

First-level screening is intended to identify contaminants or pathways for which the dose or risk is clearly below a minimum level of concern defined by the decision criteria. In these situations, detailed study is not warranted because further investigation is expected to show that the risk to any actual individual would be much less than that calculated during the first level of screening. The target individual for first-level screening is the individual at highest potential risk from exposure to the contaminant or situation. This is generally a reference maximally exposed individual. Depending on the specific contaminant or exposure pathway being considered, other factors such as age, genetic sensitivity, or lifestyle (e.g., dietary habits) may be important considerations that will be used to modify the attributes of the reference individual. Parameter values (e.g., intake rates) that are not expected to lead to an underestimate of risk to any real person will be selected for Level I screening. The screening estimate of dose or risk to the reference most at-risk individual is expected to be a substantial overestimate for *most* members of the general population, but should not underestimate the dose or risk to *any* real person who was exposed.

2.2 Level II Screening

Second-level screening is intended to rapidly identify any situations which clearly warrant more detailed analysis. The calculations at this level of screening are designed to estimate the likely dose or risk to a more "typical" individual in the population of concern. Reasonable average or more typical values will be used for the source term, parameter values, and assumptions about human exposure pathways. The estimated dose or risk to the typical individual for screening Level II may still be an overestimate for a real member of the general public, but it is likely to be an underestimate for the individual who received the highest exposure. The Level II screening estimate will not necessarily be representative for either the "average" or "most likely" person in the population, as many more attributes of the release and exposure situation must be determined before these estimates can be made.

2.3 General Considerations

The Level I screening approach focuses on the dose or risk to a reference individual who should have received the highest exposure and thus will be most at-risk. Level II focuses on the dose or risk to a more typical individual located in the near vicinity of the Oak Ridge facilities. The target individual may be a child or an adult, depending on the contaminant in question. Our target individual will usually be an adult unless the duration of exposure is short and differences in dietary habits, metabolism, and body mass indicate that a child would be the most sensitive subgroup of the population. This would be the case for soil ingestion, milk ingestion, and exposures that occurred over a period of 5 years or less. For radiation exposures to the thyroid, for example, epidemiological evidence to date suggests that an individual must be below the age of 15 at the time of exposure in order to be considered at risk (Ron et al., 1995). Two target individuals, a child (Level I) and an adult (Level II), will be used for each screening level for noncarcinogenic chemicals. Appropriate exposure locations, exposure pathways, and parameter values will be selected for each target individual and screening level, depending on the source and nature of the release (e.g., which site, airborne vs. waterborne, etc.).

The screening calculations will use generic equations for calculation of dose and risk (Appendix B); the effects of radioactive decay and of exposure and release duration will be included. The calculations will include all pathways expected to be significant for the specific contaminant in question, based on the likely exposure routes, the potential for bioaccumulation in food chains, and the contaminant's toxicity for the exposure routes. Nonstandard equations for dose or risk (e.g., for tritium) and equations for pathways not included in Appendix B will be documented with the calculations for the relevant contaminants.

Release durations for carcinogens of 50 years and 10 years will be used for Level I and Level II, respectively, unless the actual release duration of the contaminant is known. Exposure durations of 50 years and 10 years for Level I and Level II, respectively, will be used for radionuclides and carcinogenic chemicals, for which the estimated risk of adverse health effects is proportional to the total intake over a lifetime, based on a linear nonthreshold model. The averaging time for carcinogenic chemicals is 70 years (for carcinogenic chemicals the risk is calculated in terms of the total mass-normalized intake averaged over the entire estimated lifetime to give an average mass-normalized daily intake over the lifetime; for radionuclides, the risk is calculated in terms of the total cumulative dose, and an averaging time is not needed).

For most noncarcinogens, estimation of the likelihood of an adverse health effect is based on a threshold in terms of intake rate (intake per day) normalized for body mass; by definition, the likelihood of an adverse health effect occurring at an average intake rate below the threshold value is zero. In the Task 7 screening approach, the normalized average intake rate over a specified period of time is compared to a Reference Dose (RfD; a normalized intake rate at which no adverse health effects are expected) derived for the appropriate period of time--the RfD by definition is expected to be at or below the threshold dose. The intake rate is averaged over the exposure duration, thus the exposure duration and averaging time are equal.

Depending on the data available for a given contaminant, RfDs may be derived for acute exposure (14 d or less), chronic exposure (1year or more, usually several years), or subchronic exposure (several months to 1 year). For the Oak Ridge situation, it is likely that some exposures over a 1-year period of time, or over a shorter period of time, may be substantially higher than the same individuals' average exposures over their entire lifetimes. For example, behavioral patterns may vary from the norm for occasional short periods of time (e.g., contrast the fish intake for an individual over a two-week fishing trip with the fish intake for the same individual over a whole year). Additionally, exposures from accident situations or releases that produced periods of peak concentrations for some months to a year would require a shorter averaging time.

For Level I screening, unless there is evidence that a shorter exposure or averaging time is appropriate for a given contaminant or exposure situation, an exposure duration and averaging time of 1 year will be used for noncarcinogenic contaminants, based on the conservative assumption that an individual's average intake rate for a single year could be very different from the individual's average intake rate over his entire lifetime.

Parameter values (including contaminant-specific transfer factors) will be based on historical knowledge of the Oak Ridge area (when possible), literature review, and professional judgment. In some cases, different values will be used in the Level I and Level II screening. Recommended default values for contaminant-independent parameters, with rationales for their selection, are provided in Appendix C; these default values are to be used in the absence of more detailed or site-specific information. Parameters for which the values depend on the specific contaminant or the site-specific situation are listed in Appendix C. In general, these parameters include toxicity values, radionuclide decay constants, and transfer factors. Contaminant-specific or site-specific parameter values will be documented with the calculations for the respective contaminants. Toxicity values will be based on the best information available, with consideration for the most important health effects for the specified target individuals.

Care will be taken in the Level I screening to avoid compounded conservatism leading to unrealistically extreme values for the risk posed by a contaminant. In general, the parameters to be varied between screening levels are those affecting the target individual, such as location, lifestyle (e.g., intake rates, time spent outdoors, etc.), and individual differences (e.g., age). Parameters such as the growing periods of vegetation will generally be kept the same for both levels, unless specific reasons exist to do otherwise; however, the fraction of a food type that is contaminated is an example of the parameter values which will change between screening levels, as these fractions are assumed to depend on the source of the person's food supply (e.g., home-grown vs. commercial). In most cases, contaminant-specific transfer factors and toxicity values will be kept constant for both levels of screening. Adjustment of these parameter values for Level II screening might be in order for two types of cases: (1) when the contaminant is known to be present in a chemical form that is less readily transferred than the form for which the transfer factor was derived, or (2) when a reference dose (RfD) is known to include a large safety factor. In either case, the values used will be documented with the rationales for their selection.

Source term information or environmental concentrations are essential for both levels of screening. At the very least, the source term or environmental concentrations must be bounded--that is, the upper bound of the releases must be determined. If the source term or environmental concentrations are not bounded, then only a Level II screen is possible. In other words, a Level I screen will not be possible if an upper bound for the release or environmental concentrations cannot be estimated; however, a Level II screen can be performed to determine whether, at the reported release amounts or environmental concentrations, a situation exists that clearly warrants more detailed investigation.

To the extent possible, preliminary bounded source term estimates will be made from available release information, and measured concentrations will be used to check these estimates. The bounded source term is used to estimate upper bound contaminant concentrations at the appropriate locations for Level I screening and typical concentrations at appropriate locations for Level II screening. For example, the nearest residence downwind or downstream of a release site is an appropriate location for Level I screening, while for Level II, the location of the nearest downwind or downstream population center might be more appropriate; contaminant concentrations would be lower for Level II screening due to dispersion or dilution of the contaminant.

2.4 Use of Decision Criteria

The following decision criteria will be used, subject to possible future revision by the ORHASP:

For radionuclides and carcinogenic chemicals, a decision criterion of 10^{-4} *lifetime risk of excess cancer incidence. For noncarcinogenic chemicals for which a dose-response function can be estimated, the decision criteria will be* 10^{-4} *lifetime risk of adverse health effects.*

A lifetime cancer risk of 10^{-4} (1 in 10,000, or less than 2 in one million per year) is below the limits of epidemiologic detection for all types of cancer. It is clearly below the lifetime risk level of 5×10^{-3} (based on a whole-body lifetime dose of 0.07 Sv) recommended for dose reconstruction by the National Research Council (1995). This level is also consistent with a level of negligible risk recommended by the International Atomic Energy Agency (1986) and the National Council for Radiation Protection and Measurements (1993). At Hanford, Washington, the Hanford Environmental Dose Reconstruction (HEDR) study used a thyroid dose of 1 rad to define the limits of the domain of the study (Shleien, 1992). This dose is consistent with a lifetime risk or radiation-induced thyroid cancer of approximately 10^{-4}. A risk level of 10^{-4} is also consistent with other decisions made in the Oak Ridge region, in connection with the evaluation of the need for environmental remediation (Levine et al., 1994; Jacobs, 1995). A value of 10^{-4} has also been recommended by EPA as being an acceptable level of risk for residual contamination at Superfund sites (Clay, 1991).

For noncarcinogenic chemicals, a decision criterion of a Hazard Index equal to 1.0.

A Hazard Index is defined as the dose of a contaminant divided by its Reference Dose (RfD). The RfD is defined as a dose of a chemical (but not necessarily the maximum dose) that is not expected to cause adverse health effects over a lifetime of daily exposure. A Hazard Index of 1.0 indicates that the dose equals the RfD. Hazard Indices below 1.0 should be equivalent to lifetime health risks that are clearly below a 10^{-4} level, if not equivalent to risks that are zero. A Hazard Index of 1.0 is consistent with other decisions made in the Oak Ridge area (Jacobs, 1995). Major issues associated with the calculation of the appropriate Hazard Index are the selection of the averaging time over which human intake rates are estimated and the influence of exposures from multiple sources and from background. A high background exposure could result in small incremental additional exposures leading to a Hazard Index that exceeds 1.0. Intake rates over short periods of time (e.g., 2 weeks vs. 1 year) may be higher than over longer periods. When acute or subchronic toxicity values are available, these will be used with corresponding adjustments in exposure durations and averaging times.

For Level I screening, the screening value (calculated estimate of risk to the most at-risk individual) is compared to the appropriate decision criterion as follows:

- If the screening value is clearly below the decision criterion, further study of the contaminant can be deferred until time and resources permit further study. The logic is as follows: If the maximally exposed reference individual is at low risk (i.e., the screening estimate of risk is below the decision criterion), then members of the general population will be at even less risk. Continued expenditure of time and resources on that contaminant is not justified as long as there are more important situations to be studied.

- If the screening value is above the decision criterion, the contaminant should be further evaluated through a second level of screening.

For Level II screening, the screening value (calculated estimate of risk to a more typical individual) is compared to the decision criterion as follows:

- If the screening value is above the decision criterion, the contaminant should be given a high priority for detailed study. It is likely that some people received exposures or doses high enough to warrant more detailed investigations.

- If the screening value is below the decision criterion, the contaminant is designated for further study in later phases of the project, after the highest priority contaminants are dealt with.

When sufficient information exists for a contaminant to permit a preliminary uncertainty analysis, the analysis will be carried out and the resulting confidence bounds compared to the decision criteria. In other words, if the uncertainty associated with estimating parameter values (e.g., releases, exposure durations, intakes) can be described as subjective probability distributions, the uncertainty in these parameter values can be propagated (e.g., with Monte Carlo techniques) to produce an estimate of the uncertainty associated with the calculated risk to the target individual (Apostoaei et al., 1995; Hammonds et al., 1994; IAEA, 1989; Morgan and Henrion, 1990; NCRP, in press). The calculated risk, with its uncertainty (expressed as 95% subjective confidence limits), is then compared to the appropriate decision criterion. If, for instance, the calculated 95% upper confidence limit on the risk to the most at-risk individual is below the Level I decision criterion, the contaminant is below a level of concern for further detailed study. However, if the confidence bounds overlap the decision criterion, the results of the uncertainty analysis can be used to identify the major contributors to the uncertainty in the risk (i.e., which uncertain parameters are most important), and these can be targeted for further study in an effort to reduce the uncertainty on the screening estimate of risk.

3.0 CONTAMINANTS TO BE SCREENED

Initial screening efforts will be focused on contaminants that were not evaluated during the Dose Reconstruction Feasibility Study or that the ORHASP has determined require further evaluation. Additional contaminants requiring screening may be identified during the process of systematic search and interviews in Task 5 of the Oak Ridge Dose Reconstruction.

3.1 Primary Contaminants to be Screened

The primary group of contaminants to be screened includes those listed in the Dose Reconstruction Contract and the Task 7 Plan:

Asbestos,
Arsenic,
Chromium VI,
Neptunium-237,
Plutonium,
Tritium from Y-12, and
Classified materials.

In addition, the ORHASP recently decided to retain lead on the list for further screening analysis.

3.2 Additional Contaminants Identified from Document Searches

It is expected that the systematic document search (Task 5) will identify contaminants not screened during the Dose Reconstruction Feasibility Study that might have caused an off-site health risk, or that additional information may become available to indicate that a contaminant should be re-evaluated (possibly beryllium, technetium, or protactinium). Information to be considered for these contaminants includes the following:

1. Was the substance actually used at an Oak Ridge site? When, and in what quantities?
2. Is the substance known to have been released on-site or off-site, or is there a reasonable likelihood that it might have been released off-site?
3. Can we bound the source term (the amount potentially released to the environment)?
4. What is the likely fate of the contaminant in the environment?
5. Is it possible or likely that human exposure occurred off-site (were there complete exposure pathways)?
6. Is the substance associated with adverse health effects in humans?

3.3 Requirements for Screening Calculations

Screening calculations will be carried out for all contaminants for which (a) calculations are warranted (i.e., off-site exposure was likely) and (b) sufficient information is available with which to perform calculations. Documentation for each contaminant will include the information and assumptions (including parameter values) used to perform screening calculations and an explanation of any decisions or recommendations made concerning the contaminant.

If insufficient information is available to permit screening calculations for a contaminant, this will also be documented. A summary of the information necessary to permit screening will be provided together with, as appropriate, an estimate of the release amount above which adverse health effects might be expected to occur.

4.0 REFERENCES

Apostoaei et al., 1995. Apostoaei, I., Hoffman, F.O., and Nair, S.K. Task 1: Screening Calculations to Estimate the Transfer Factor for ^{131}I from Air to Pasture to Milk of Cows and Goats. Prepared for the Oak Ridge Health Studies, Oak Ridge Dose Reconstruction. ChemRisk/McLaren Hart Environmental Services. June, 1995. (ChemRisk Repository # 4009)

ATSDR, 1992. Agency for Toxic Substances and Disease Registry (ATSDR). Public Health Assessment Guidance Manual. U.S. Department of Health and Human Services, Atlanta, Georgia. 1992. (ChemRisk Repository # 4002)

Burson and Profio, 1977. Burson, Z.G., and Profio, A.E. Structure shielding in reactor accidents. Health Physics **33**: 287-299. 1977. Cited in Snyder et al., 1994. (ChemRisk Repository # 3076)

Calabrese et al., 1990. Calabrese, E.J., Stanek, E.J., Gilbert, C.E., and Barnes, R.M. Preliminary adult soil ingestion estimates: Results of a pilot study. Reg. Tox. and Pharmac. **12**: 88-95. 1990. Cited in ATSDR, 1992. (ChemRisk Repository # 4002)

Clay, 1991. Clay, D.R. Role of the Baseline Risk Assessment in Superfund Remedy Selection Decisions. United States Environmental Protection Agency. Washington, DC. OSWER Directive 9355.0-30. PB91-921359. April 22, 1991. (ChemRisk Repository # 3099)

ChemRisk, 1993. ChemRisk. Tasks 3 & 4: Identification of Important Environmental Pathways for Materials Released from the Oak Ridge Reservation. Oak Ridge Health Studies, Phase I Report, Volume II - Part B. 1993. (ChemRisk Repository # 3084)

Cook et al., 1975. Cook, C.B., Eiler, D.A., and Forker, D.D. Beverage Consumption Patterns in New York State. J. Am. Diet. Assoc. 67(3):222-227. 1975. Cited in Rupp, 1980. (ChemRisk Repository # 4016)

Darwin, 1990. Darwin, R. Soil Ingestion by Dairy Cattle. Pacific Northwest Laboratories, Richland, Washington. PNL-SA-17918 HEDR. 1990. Cited in Snyder et al., 1994. (ChemRisk Repository # 3076)

Fields and Diamond, 1991. Fields, T., and Diamond, B. Human Health Evaluation Manual, Supplemental Guidance: "Standard Default Exposure Factors." Unites States Environmental Protection Agency. Washington, D.C. OSWER Directive 9285.6-03. March 25, 1991. (ChemRisk Repository # 3091)

Gilbert et al., 1995. Gilbert, R.O., Bittner, E.A., and Essington, E.H. On the Use of Uncertainty Analyses to Test Hypotheses Regarding Deterministic Model Predictions of Environmental Processes. J. Environ. Radioact. 27(3):231-260. 1995. (ChemRisk Repository # 4000)

Hammonds et al., 1994. Hammonds, J.S., Hoffman, F.O., and Bartell, S.M. An Introductory Guide to Uncertainty Analysis in Environmental and Health Risk Assessment. Oak Ridge National Laboratory. ES/ER/TM-35/R1. December, 1994. (ChemRisk Repository # 3090)

Honstead et al., 1971. Honstead, J.F., Beetle T.M., and Soldat, J.K. A Statistical Study of the Habits of Local Fishermen and Its Application to Evaluation of Environmental Dose. A Report to the Environmental Protection Agency by Battelle Pacific Northwest Laboratories, Richland, WA. 1971. Cited in Rupp et al., 1980. (ChemRisk Repository # 4017)

Humphrey, 1978. Humphrey H.E.B. Private Communication. 1978. Cited in Rupp et al. (1980). Cited in Rupp et al., 1980. (ChemRisk Repository # 4017)

Husted-Anderson, 1941. Husted-Anderson, S. Orienterende undersøgelser vedrørende losgaaende køers graesforbrug. Nord. Jordbrugsforkn: 101-114, as seen in Nutr. Abstr. Rev. 16:990. 1941. Cited in Koranda, 1965. (ChemRisk Repository # 4004)

IAEA, 1986. IAEA (International Atomic Energy Agency). Principles for Limiting Releases of Radioactive Effluents into the Environment. Safety Series No. 77. Vienna, Austria. July 1986. (ChemRisk Repository # 4003)

IAEA, 1989. IAEA. Evaluating the Reliability of Predictions Made Using Environmental Transfer Models. Safety Series No. 100. Vienna, Austria. 1989. (ChemRisk Repository # 2168)

IAEA, 1992. IAEA. Modelling of resuspension, seasonality and losses during food processing. First report of the VAMP Terrestrial Working Group. IAEA-TECDOC-647. Vienna, Austria. May 1992. (ChemRisk Repository # 4014)

IAEA, 1994. IAEA. Handbook of Parameter Values for the Prediction of Radionuclide Transfer in Temperate Environments. Technical Report Series No. 364. Vienna, Austria. 1994. (ChemRisk Repository # 4008)

IAEA, in press. IAEA. Draft: Modelling of radionuclide interception and loss processes in vegetation and of transfer in semi-natural ecosystems. Second report of the VAMP Terrestrial Working Group. IAEA TECDOC. Vienna, Austria. In press. (ChemRisk Repository # 2676)

IAEA, in preparation. IAEA. Draft of Revised Version of IAEA Safety Series 57. Vienna, Austria. In preparation. (ChemRisk Repository # 3081)

ICRP, 1975. ICRP (International Commission on Radiological Protection). Report of the Task Group on Reference Man. ICRP 23. New York, New York. 1975. (ChemRisk Repository # 4015)

ICRP, 1992. ICRP. Radiation Dose to Patients from Radiopharmaceuticals. ICRP 53. Oxford, England. 1994. (ChemRisk Repository # 4013)

Israeli and Nelson, 1992. Israeli, M., and Nelson, C.B. Distribution and expected time of residence for U.S. households. Risk Analysis **12**: 65-72. 1992. (ChemRisk Repository # 2675)

Jacobs, 1995. Jacobs Engineering Group. Record of Decision for Lower East Fork Poplar Creek. U.S. Department of Energy, Office of Environmental Restoration and Waste Management. DOE/OR/02-1370&D1. May 1995. (ChemRisk Repository # 4005)

Koranda, 1965. Koranda, J.J. Agricultural Factors Affecting the Daily Intake of Fresh Fallout by Dairy Cows. UCRL-12479, Biology and Medicine, UC-48. University of California, Lawrence Radiation Laboratory, Livermore, California. March 19, 1965. (ChemRisk Repository # 4004)

Lepow et al., 1975. Lepow, M.L., Bruckman, L., Gillette, M., Markowitz, S., Robino, R., and Kapish, J. Investigations onto sources of lead in the environment of urban children. Environmental Research **10**: 415-426. 1975. Cited in Chapter 7 of Paustenbach, 1989. (ChemRisk Repository # 3096)

Levine et al., 1994. Levine, D.A., Hargrove, W.W., Campbell, K.R., Wood, M.A., and Rash, C.D. Data Summary for the Near-Shore Sediment Characterization Task of the Clinch River Environmental Restoration Program. Environmental Sciences Division, Oak Ridge National Laboratory. ORNL/ER-264. October, 1994. (ChemRisk Repository # 3088)

Morgan and Henrion, 1990. Morgan, M. G. and Henrion, M. Uncertainty, A Guide to Dealing with Uncertainty in Quantitative Risk and Policy Analysis. Cambridge University Press, New York. 1990. (ChemRisk Repository # 2678)

National Research Council, 1995. National Research Council. Radiation Dose Reconstruction for Epidemiologic Uses. National Academy Press. Washington, DC. 1995. (ChemRisk Repository # 3077)

NCRP, 1989. NCRP (National Council on Radiation Protection and Measurements). Screening Techniques for Determining Compliance with Environmental Standards, Releases of Radionuclides to the Atmosphere, NCRP Commentary 3 (National Council on Radiation Protection, Bethesda, Maryland). 1989. (ChemRisk Repository # 3075)

NCRP, 1993. NCRP. Limitation of Exposure to Ionizing Radiation. NCRP Report No. 116. Bethesda, Maryland. March, 1993. (ChemRisk Repository # 3087)

NCRP, in press. NCRP. A Guide for Uncertainty Analysis in Dose and Risk Assessments Related to Environmental Contamination. Scientific Committee 64-17. NCRP Commentary No. 14. In press. (ChemRisk Repository # 3092)

Pao and Burk, 1975. Pao, E.M. and Burk, M.C. Portion Sizes and Days Intake of Selected Foods. U.S. Department of Agriculture, Agric. Res. Serv., Northeast. ARS-NE-67. December, 1975. Cited in Rupp, 1980. (ChemRisk Repository # 4016)

Paustenbach, 1989. Paustenbach, D.J. The Risk Assessment of Environmental and Human Health Hazards: A Textbook of Case Studies. New York, New York. 1989. (ChemRisk # 3096)

Pröhl et al., 1995. Pröhl, G., Hoffman, F.O., and Müller, H. Interception and post-deposition retention of radionuclides by vegetation and its importance for dose assessment. IAEA-SM-339/142. Vienna, Austria. 1995. (ChemRisk Repository # 4006)

Roed, 1990. Roed, J. Deposition and Removal of Radioactive Substances in an Urban Area. Final Report of the NKA Project AKTU-245. Ris∅ National Laboratory. October, 1990. (ChemRisk Repository # 4011)

Ron et al., 1995. Ron, E., Lubin, J.H., Shore, R.E., Mabuchi, K., Modan, B., Pottern, L.M., Schneider, A.B., Tucker, M.A., and Boice, J.D., Jr. Thyroid Cancer after Exposure to External Radiation: A Pooled Analysis of Seven Studies. Radiation Research 141:259-277. 1995. (ChemRisk Repository # 3070)

Rupp, 1980. Rupp, E.M. Age dependent values of dietary intake for assessing human exposures to environmental pollutants. Health Physics **39**: 151-163. 1980. (ChemRisk Repository # 4016)

Rupp et al., 1980. Rupp, E.M., Miller, F.L., and Baes, C.F., III. Some results of recent surveys of fish and shellfish consumption by age and region of U.S. residents. Health Physics **39**: 164-175. 1980. (ChemRisk Repository # 4017)

Sedman, 1989. Sedman, R. Development of applied action levels for soil contact: A scenario for the exposure of humans to soils in a residential setting. Environmental Health Perspectives **79**: 291-313. 1989. Cited in ATSDR, 1992. (ChemRisk Repository # 4002)

Shleien, 1992. Shleien, B. Scoping Document for Determination of Temporal and Geographic Domains for the HEDR Project. Hanford Technical Steering Panel. Summer, 1992. (ChemRisk Repository # 3098)

Snyder et al., 1994. Snyder, S.F., Farris, W.T., Napier, B.A., Ikenberry, T.A., and Gilbert, R.O. Parameters Used in the Environmental Pathways and Radiological Dose Modules (DESCARTES, CIDER, and CRD Codes) of the Hanford Environmental Dose Reconstruction Integrated Codes (HEDRIC). Battelle Pacific Northwest Laboratories, Richland, Washington. PNWD-2023 HEDR Rev. 1. 1994. (ChemRisk Repository # 3076)

Thiessen et al., 1995. Thiessen, K.M., Hoffman, F.O., Hammonds, J.S., and White, E.I. Task 7: A Review of the Preliminary Screening Analysis Carried out during the Oak Ridge Dose Reconstruction Feasibility Study. Prepared for the Oak Ridge Health Studies, Oak Ridge Dose Reconstruction. ChemRisk/McLaren Hart Environmental Services. August 11, 1995. (ChemRisk Repository # 2677)

Till and Meyer, 1983. Till, J.E., and Meyer, H.R. (eds.) Radiological Assessment: A Textbook on Environmental Dose Analysis. Washington, D.C. U.S. Nuclear Regulatory Commission. 1983. (ChemRisk Repository # 3079)

USEPA, 1985. USEPA (United States Environmental Protection Agency). Development of Statistical Distribution of Ranges of Standard Factors Used in Exposure Assessments. Washington, D.C. EPA No. 600/8-85-010. 1985. Cited in USEPA, 1989. (ChemRisk Repository # 3089)

USEPA, 1989. USEPA. Exposure Factors Handbook. Washington, D.C. EPA 600-8-89-043. 1989. (ChemRisk Repository # 3089)

Whicker and Kirchner, 1987. Whicker, F.W., and Kirchner, T.B. PATHWAY: A Dynamic Food-chain Model to Predict Radionuclide Ingestion after Fallout Deposition. Health Physics 52(6):717-737. June 1987. (ChemRisk Repository # 3095)

Zach and Mayoh, 1984. Zach, R., and Mayoh, K.R. Soil Ingestion by Cattle: A Neglected Pathway. Health Physics 46(2):426-431. 1984. (ChemRisk Repository # 3094)

APPENDIX B

EXPOSURE PATHWAY EQUATIONS

APPENDIX B: EQUATIONS TO BE USED IN SCREENING CALCULATIONS

Generic equations to be used for the screening calculations of doses and risks are provided in this Appendix. The exposure pathways represented by the equations (listed in Table A-1) were selected to represent the potentially dominant routes of exposure for the Oak Ridge situation and to be adequate surrogates for pathways that are not explicitly included. For instance, we have not included equations for ingestion of beef or milk from cows that inhaled contaminated air; however, since direct deposition of contaminants to pasture vegetation and to soil on which the pasture vegetation is grown is the dominant pathway for contamination of beef and milk due to airborne contaminants, use of the air-to-vegetation pathways for cattle is expected to provide an adequate surrogate for the direct inhalation pathway for cattle. Likewise, the equations for dose or risk from ingestion of contaminated beef are intended as a surrogate for all meat consumption, including game. We expect that most pathways not explicitly included here (e.g., ingestion of water plants) will be implicitly included in the exposure pathways that we have described. For special situations (e.g., accidents, or indication that an unusual exposure pathway occurred for a contaminant), we will document the equations, assumptions, and calculations used for each relevant contaminant.

The equations provided in the document all start with an environmental concentration of a contaminant (e.g., air, water, soil, fish). In general, for the screening carried out in Task 7, we will start our analyses with environmental data when adequate data are available. When adequate data are not available, environmental concentrations at the relevant locations will be estimated using appropriate dispersion models. For atmospheric releases, the models will be based on those used for Tasks 1, 2, and 6; for waterborne releases, the models will be based on those used for Tasks 2, 3, and 4. Documentation of the source term estimates and/or environmental concentrations used will be made for each contaminant screened.

The equations for radionuclides are provided in terms of excess lifetime risk of cancer (symbolized LR). For chemical contaminants, the equations are provided in terms of the mass-normalized intake rate (I; mg kg^{-1} d^{-1}). For carcinogenic chemicals, the excess lifetime risk of cancer is obtained by multiplying the intake rate by a slope factor ($LR = I * SF$, where the units of SF are risk per mg kg^{-1} d^{-1}). For noncarcinogenic chemicals, the hazard quotient (HQ) is obtained by dividing the intake rate by a reference dose ($HQ = I \div RfD$, where the units of RfD are mg kg^{-1} d^{-1}).

For special situations, the actual equations used will be documented with the discussion of that situation. An example of a special situation that is expected is the analysis of tritium, for which a different set of calculations in terms of total dose (rather than doses by pathways) is generally used (Till and Meyer, 1983; ChemRisk, 1993; Thiessen et al., 1995). Another type of special case is the evaluation of chemical contaminants for which a reference concentration (RfC; mg m^{-3}) rather than a reference dose is available. In this case, the hazard quotient for inhalation exposure is obtained by dividing the calculated contaminant concentration in air by the reference concentration ($HQ = C_{air} \div RfC$).

Table B-1. Exposure pathways for which equations are provided.

Exposure Pathway	Page number	
	Radionuclides	Chemicals
Atmospheric Releases		
Inhalation	B – 5	B – 15
External exposure (air deposition to ground)	B – 5	NA [a]
Ingestion of vegetables	B – 6	B – 15
Ingestion of beef or milk	B – 7	B – 17
Ingestion of beef	B – 8	B – 18
Ingestion of milk	B – 9	B – 18
Aquatic Releases		
Water ingestion	B – 9	B – 19
Fish ingestion	B– 10	B – 19
Soil or Sediment Contamination		
External exposure (ground)	B – 10	NA [a]
Soil/sediment ingestion	B – 11	B – 20
Soil to beef	B – 11	B – 20
Soil to dairy cattle (milk)	B – 12	B – 21
Soil to vegetables (ingestion)	B – 13	B – 22
Soil to pasture to beef or milk (ingestion)	B – 13	B – 23
Ingestion of beef	B – 14	B – 23
Ingestion of milk	B – 14	B – 24

[a] These pathways are not applicable for chemical (nonradioactive) contaminants.

Radionuclides

Atmospheric Releases:

Inhalation:

$$LR = C_{air} * U_{air} * F_{occ} * EF * ED * SF$$

where

LR = excess lifetime risk (risk),

C_{air} = concentration of radionuclide in air (Bq m^{-3})[*] ,

U_{air} = average inhalation of contaminated air (m^3 d^{-1}),

F_{occ} = occupancy factor, fraction of time in the contaminated area (unitless),

EF = exposure frequency (d yr^{-1}),

ED = exposure duration (yr),

and SF = slope factor (risk Bq^{-1}).

External Exposure (air deposition to ground):

$$LR = C_{air} * V_d * \left[\frac{1 - e^{-l_R T_r}}{l_R} \right] * \left[F_{out} + (F_{in} * F_s) \right] * ED * SF$$

where

LR = excess lifetime risk (risk),

C_{air} = concentration of radionuclide in air (Bq m^{-3}),

V_d = deposition velocity (m d^{-1}),

λ_R = radioactive decay constant (d^{-1}),

T_r = release duration (d),

F_{out} = fraction of time outside (unitless),

F_{in} = fraction of time inside (unitless),

F_s = indoor shielding factor (unitless),

ED = exposure duration (yr),

and SF = external slope factor [(risk yr^{-1}) per (Bq m^{-2})].

---

[*] Regarding units with negative exponents: 1 Bq m^{-3} is equal to 1 Becquerel per cubic meter, etc.

Ingestion of Vegetables:

$$C_{veg(soil)} = B_{v(veg)} * C_{air} * V_d * \left[\frac{1 - e^{-(\lambda_R + \lambda_L)T_r}}{\rho(\lambda_R + \lambda_L)} \right]$$

where

$C_{veg(soil)}$ = concentration of radionuclide in vegetables from root uptake (Bq kg^{-1} wet),

$B_{v(veg)}$ = soil-to-plant transfer factor, wet weight (unitless),

C_{air} = concentration of radionuclide in air (Bq m^{-3}),

V_d = deposition velocity (m d^{-1}),

λ_L = soil leaching constant (d^{-1}),

λ_R = radioactive decay constant (d^{-1}),

T_r = release duration (d),

and ρ = effective surface soil density for crops other than pasture (kg m^{-2} dry).

$$C_{veg(air)} = C_{air} * V_d * \alpha_{veg} * F_w * \left[\frac{1 - e^{-(k_w + \lambda_R)T_{g(veg)}}}{k_w + \lambda_R} \right]$$

where

$C_{veg(air)}$ = concentration of radionuclide in vegetables due to direct deposition from air (Bq kg^{-1} wet),

C_{air} = concentration of radionuclide in air (Bq m^{-3}),

V_d = deposition velocity (m d^{-1}),

F_w = fraction of contamination remaining after washing (unitless),

α_{veg} = mass interception factor for vegetables (m^2 kg^{-1}),

k_w = weathering constant (d^{-1}),

λ_R = radioactive decay constant (d^{-1}),

and $T_{g(veg)}$ = growing period for vegetables (d).

$$C_{veg(total)} = C_{veg(soil)} + C_{veg(air)}$$

where

$C_{veg(total)}$ = total concentration of radionuclide in vegetables (Bq kg^{-1} wet),
$C_{veg(soil)}$ = concentration of radionuclide in vegetables from root uptake (Bq kg^{-1} wet),
and $C_{veg(air)}$ = concentration of radionuclide in vegetables due to direct deposition from air (Bq kg^{-1} wet).

$$LR = C_{veg(total)} * F_{cv} * U_{veg} * EF * ED * SF$$

where

LR = excess lifetime risk (risk),
$C_{veg(total)}$ = total concentration of radionuclide in vegetables (Bq kg^{-1} wet),
F_{cv} = fraction of contaminated vegetables ingested (unitless),
U_{veg} = ingestion rate of vegetables by humans, wet weight (kg d^{-1}),
EF = exposure frequency (d yr^{-1}),
ED = exposure duration (yr),
and SF = oral slope factor (risk Bq^{-1}).

Ingestion of Beef or Milk:

Pasture Concentration**:**

$$C_{past(soil)} = B_{v(past)} * C_{air} * V_d * \left[\frac{1 - e^{-(l_R + l_L)T_r}}{r_{past}(l_R + l_L)} \right]$$

where

$C_{past(soil)}$ = concentration of radionuclide in pasture from root uptake (Bq kg^{-1} dry),
$B_{v(past)}$ = soil-to-plant transfer factor, dry weight (unitless),
C_{air} = concentration of radionuclide in air (Bq m^{-3}),
V_d = deposition velocity (m d^{-1}),

λ_L = soil leaching constant (d^{-1}),

λ_R = radioactive decay constant (d^{-1}),

T_r = release duration (d),

and ρ_{past} = effective surface soil density for pasture (kg m^{-2} dry).

$$C_{past(air)} = C_{air} * V_d * \alpha_{past} * \left[\frac{1 - e^{-(k_w + \lambda_R) T_{g(past)}}}{k_w + \lambda_R} \right]$$

where

$C_{past(air)}$ = concentration of radionuclide in pasture due to direct deposition from air (Bq kg^{-1} dry),

C_{air} = concentration of radionuclide in air (Bq m^{-3}),

V_d = deposition velocity (m d^{-1}),

α_{past} = mass interception factor for pasture (m^2 kg^{-1}),

k_w = weathering constant (d^{-1}),

λ_R = radioactive decay constant (d^{-1}),

and $T_{g(past)}$ = growing period for pasture (d).

$$C_{past(total)} = C_{past(soil)} + C_{past(air)}$$

where

$C_{past(total)}$ = total concentration of radionuclide in pasture (Bq kg^{-1} dry),

$C_{past(soil)}$ = concentration of radionuclide in pasture from root uptake (Bq kg^{-1} dry),

and $C_{past(air)}$ = concentration of radionuclide in pasture due to direct deposition from air (Bq kg^{-1} dry).

Ingestion of Beef:

$$LR = C_{past(total)} * F_{pb} * Q_{feed,\,b} * F_f * U_{meat} * F_{cb} * EF * ED * SF$$

where

LR = excess lifetime risk (risk),

$C_{past(total)}$ = total concentration of radionuclide in pasture (Bq kg^{-1} dry),

F_{pb} = fraction of feed that is contaminated pasture (unitless),

$Q_{feed,\,b}$ = ingestion rate of feed by beef cattle (kg d^{-1} dry),

F_f = feed-to-meat transfer factor (Bq kg^{-1} per Bq d^{-1}),

U_{meat} = ingestion rate of meat by humans (kg d^{-1}),
F_{cb} = fraction of meat that is contaminated (unitless),
EF = exposure frequency (d yr^{-1}),
ED = exposure duration (yr),
and SF = oral slope factor (risk Bq^{-1}).

Ingestion of Milk:

$$LR = C_{past\,(total)} \; * \; F_{pd} \; * \; Q_{feed,\,d} \; * \; F_m \; * \; e^{-\lambda_R T_{cons}} \; * \; U_{milk} \; * \; F_{cm} \; * \; EF \; * \; ED \; * \; SF$$

where
LR = excess lifetime risk (risk),
$C_{past(total)}$ = total concentration of radionuclide in pasture (Bq kg^{-1} dry),
F_{pd} = fraction of feed that is contaminated pasture (unitless),
$Q_{feed,\,d}$ = ingestion rate of feed by dairy cattle (kg d^{-1} dry),
F_m = feed-to-milk transfer factor (Bq L^{-1} per Bq d^{-1}),
λ_R = radioactive decay constant (d^{-1}),
T_{cons} = time between milking and consumption (d),
U_{milk} = ingestion rate of milk by humans (L d^{-1}),
F_{cm} = fraction of milk that is contaminated (unitless),
EF = exposure frequency (d yr^{-1}),
ED = exposure duration (yr),
and SF = oral slope factor (risk Bq^{-1}).

Aquatic Releases:

Water Ingestion:

$$LR = C_{water} \; * \; U_{water} \; * \; F_{cw} \; * \; EF \; * \; ED \; * \; SF$$

where
LR = excess lifetime risk (risk),
C_{water} = concentration of radionuclide in water (Bq L^{-1}),
U_{water} = ingestion rate of water by humans (L d^{-1}),
F_{cw} = fraction of water that is contaminated (unitless),
EF = exposure frequency (d yr^{-1}),

ED = exposure duration (yr),

and SF = oral slope factor (risk Bq^{-1}).

Fish Ingestion:

$$C_{fish} = C_{water} * BCF$$

where

C_{fish} = concentration of radionuclide in fish ($Bq\ kg^{-1}$),

C_{water} = concentration of radionuclide in water ($Bq\ L^{-1}$),

and BCF = bioconcentration factor for fish [($Bq\ kg^{-1}$) per ($Bq\ L^{-1}$)].

$$LR = C_{fish} * U_{fish} * F_{cf} * EF * ED * SF$$

where

LR = excess lifetime risk (risk),

C_{fish} = concentration of radionuclide in fish ($Bq\ kg^{-1}$),

U_{fish} = ingestion rate of fish by humans ($kg\ d^{-1}$),

F_{cf} = fraction of fish that is contaminated (unitless),

EF = exposure frequency ($d\ yr^{-1}$),

ED = exposure duration (yr),

and SF = oral slope factor (risk Bq^{-1}).

Soil or Sediment Contamination:

External Exposure (ground):

$$LR = C_{soil} * \left[\frac{1 - e^{-\lambda_R ED}}{\lambda_R} \right] * \left[F_{out} + (F_{in} * F_s) \right] * SF$$

where

LR = excess lifetime risk (risk),

C_{soil} = concentration of radionuclide in soil or sediment ($Bq\ kg^{-1}$), ($Bq\ m^{-2}$), or ($Bq\ m^{-3}$),

λ_R = radioactive decay constant (yr^{-1}), ED = exposure duration (yr),

F_{out} = fraction of time outside (unitless),

F_{in} = fraction of time inside (unitless),

F_s = indoor shielding factor (unitless)

and SF = external slope factor [(risk yr^{-1}) per (Bq kg^{-1})], [(risk yr^{-1}) per (Bq m^{-2})], or [(risk yr^{-1}) per (Bq m^{-3})].

Soil/Sediment Ingestion:

$$LR = C_{soil/sed(surf)} * U_{soil/sed(surf)} * F_{sc} * EF * ED * SF$$

where

LR = excess lifetime risk (risk),

$C_{soil/sed(surf)}$ = concentration in surface soil or sediment (Bq kg^{-1}),

$U_{soil/sed(surf)}$ = ingestion rate of soil or sediment by humans (kg d^{-1}),

F_{sc} = fraction of contaminated soil or sediment (unitless),

EF = exposure frequency (d yr^{-1}),

ED = exposure duration (yr),

and SF = oral slope factor (risk Bq^{-1}).

Soil to beef:

$$C_{beef(soil)} = C_{soil(surf)} * Q_{soil} * F_f$$

where

$C_{beef(soil)}$ = concentration of radionuclide in beef or game from ingestion of soil (Bq kg^{-1}),

$C_{soil(surf)}$ = concentration of radionuclide in surface soil (Bq kg^{-1}),

Q_{soil} = ingestion rate of contaminated soil (kg d^{-1}),

and F_f = feed-to-meat transfer factor (Bq kg^{-1} per Bq d^{-1}).

$$LR = C_{beef(soil)} * U_{meat} * F_{cb} * EF * ED * SF$$

where

LR = excess lifetime risk (risk),

$C_{beef(soil)}$ = concentration of radionuclide in beef or game from ingestion of soil (Bq kg^{-1}),

U_{meat} = ingestion rate of meat by humans (kg d^{-1}),

F_{cb} = fraction of meat that is contaminated (unitless),

EF = exposure frequency (d yr^{-1}),

ED = exposure duration (yr),

and SF = oral slope factor (risk Bq^{-1}).

Soil to Dairy Cattle (milk):

$$C_{milk(soil)} = C_{soil(surf)} * Q_{soil} * F_m$$

where

$C_{milk(soil)}$ = concentration of radionuclide in milk from ingestion of soil (Bq L^{-1}),

$C_{soil(surf)}$ = concentration of radionuclide in surface soil (Bq kg^{-1}),

Q_{soil} = ingestion rate of contaminated soil (kg d^{-1}),

and F_m = feed-to-milk transfer factor (Bq L^{-1} per Bq d^{-1}).

$$LR = C_{milk(soil)} * U_{milk} * F_{cm} * EF * ED * SF$$

where

LR = excess lifetime risk (risk),

$C_{milk(soil)}$ = concentration of radionuclide in milk from consumption of soil (Bq L^{-1}),

U_{milk} = ingestion rate of milk by humans (L d^{-1}),

F_{cm} = fraction of milk that is contaminated (unitless),

EF = exposure frequency (d yr^{-1}),

ED = exposure duration (yr),

and SF = oral slope factor (risk Bq^{-1}).

Soil to Vegetables (ingestion):

$$C_{veg(soil)} = C_{soil} * B_{v(veg)}$$

where

$C_{veg(soil)}$ = concentration of radionuclide in vegetables from root uptake, wet weight (Bq kg^{-1}),

C_{soil} = concentration of radionuclide in bulk soil (Bq kg^{-1}), and

$B_{v(veg)}$ = soil-to-plant transfer factor, wet weight (unitless).

$$LR = C_{veg(soil)} * U_{veg} * F_{cv} * EF * ED * SF$$

where

LR = excess lifetime risk (risk),

$C_{veg(soil)}$ = concentration of radionuclide in vegetables from root uptake (Bq kg^{-1} wet),

U_{veg} = ingestion rate of vegetables by humans, wet weight (kg d^{-1}),

F_{cv} = fraction of contaminated vegetables ingested (unitless),

EF = exposure frequency (d yr^{-1}),

ED = exposure duration (yr),

and SF = oral slope factor (risk Bq^{-1}).

Soil to Pasture to Beef or Milk (ingestion):

Pasture Contamination:

$$C_{past(soil)} = C_{soil} * B_{v(past)}$$

where

$C_{past(soil)}$ = concentration of radionuclide in pasture from root uptake, (Bq kg^{-1} dry),

C_{soil} = concentration of radionuclide in bulk soil (Bq kg^{-1}),

and $B_{v(past)}$ = soil-to-plant transfer factor, dry weight (unitless).

Ingestion of Beef**:**

$$C_{beef(past)} = C_{past(soil)} * Q_{feed,\,b} * F_f * F_{pb}$$

where

$C_{beef(past)}$ = concentration of radionuclide in beef (Bq kg^{-1}),

$C_{past(soil)}$ = concentration of radionuclide in pasture from root uptake (Bq kg^{-1} dry),

$Q_{feed,\,b}$ = ingestion rate of feed by beef cattle, dry weight (kg d^{-1}),

F_f = feed-to-meat transfer factor (Bq kg^{-1} per Bq d^{-1}),

and F_{pb} = fraction of feed that is contaminated pasture (unitless).

$$LR = C_{beef(past)} * U_{meat} * F_{cb} * EF * ED * SF$$

where

LR = excess lifetime risk (risk),

$C_{beef(past)}$ = concentration of radionuclide in beef (Bq kg^{-1}),

U_{meat} = ingestion rate of meat by humans (kg d^{-1}),

F_{cb} = fraction of meat that is contaminated (unitless),

EF = exposure frequency (d yr^{-1}),

ED = exposure duration (yr),

and SF = oral slope factor (risk Bq^{-1}).

Ingestion of Milk**:**

$$C_{milk(past)} = C_{past(soil)} * Q_{feed,\,d} * F_m * F_{pd}$$

where

$C_{milk(past)}$ = concentration of radionuclide in milk from ingestion of pasture (Bq L^{-1}),

$C_{past(soil)}$ = concentration of radionuclide in pasture from root uptake (Bq kg^{-1} dry),

$Q_{feed,\,d}$ = ingestion rate of feed by dairy cattle (kg d^{-1} dry),

F_m = feed-to-milk transfer factor (Bq L^{-1} per Bq d^{-1}),

and F_{pd} = fraction of feed that is contaminated pasture (unitless).

$$LR = C_{milk(past)} * U_{milk} * F_{cm} * EF * ED * SF$$

where

LR = excess lifetime risk (risk),

$C_{milk(past)}$ = concentration of radionuclide in milk from pasture (Bq L^{-1}),

U_{milk} = ingestion rate of milk by humans (L d^{-1}),

F_{cm} = fraction of milk that is contaminated (unitless),

EF = exposure frequency (d yr^{-1}),

ED = exposure duration (yr),

and SF = oral slope factor (risk Bq^{-1}).

Chemicals

Atmospheric Releases:

Inhalation:

$$\dot{I} = \frac{C_{air} * U_{air} * F_{occ} * EF * ED}{BW * AT}$$

where

I = mass-normalized intake rate (mg kg^{-1} d^{-1}),

C_{air} = concentration of contaminant in air (mg m^{-3}),

U_{air} = average inhalation of contaminated air (m^3 d^{-1}),

F_{occ} = occupancy factor, fraction of time in the contaminated area (unitless),

EF = exposure frequency (d yr^{-1}),

ED = exposure duration (yr),

BW = body weight (kg),

and AT = averaging time (d).

Ingestion of Vegetables:

$$C_{veg(soil)} = B_{v(veg)} * C_{air} * V_d * \left\lfloor \frac{1 - e^{-l_L T_r}}{r\, l_L} \right\rfloor$$

where

$C_{veg(soil)}$ = concentration of contaminant in vegetables from root uptake (mg kg^{-1} wet),

$B_{v(veg)}$ = soil-to-plant transfer factor, wet weight (unitless),

C_{air} = concentration of contaminant in air (mg m^{-3}),

V_d = deposition velocity (m d^{-1}),

λ_L = soil leaching constant (d^{-1}),

T_r = release duration (d),

and ρ = effective surface soil density for crops other than pasture (kg m^{-2} dry).

$$C_{veg(air)} = C_{air} * V_d * a_{veg} * \left[\frac{1-e^{-k_w T_{g(veg)}}}{k_w} \right] * F_w$$

where

$C_{veg(air)}$ = concentration of contaminant in vegetables due to direct deposition from air (mg kg^{-1} wet),

C_{air} = concentration of contaminant in air (mg m^{-3}),

V_d = deposition velocity (m d^{-1}),

α_{veg} = mass interception factor (m^2 kg^{-1}).

k_w = weathering constant (d^{-1}).

$T_{g(veg)}$ = growing period for vegetables (d),

and F_w = fraction of contamination remaining after washing (unitless)

$$C_{veg(total)} = C_{veg(soil)} + C_{veg(air)}$$

where

$C_{veg(total)}$ = total concentration of contaminant in vegetables (mg kg^{-1} wet),

$C_{veg(soil)}$ = concentration of contaminant in vegetables from root uptake (mg kg^{-1} wet),

and $C_{veg(air)}$ = concentration of contaminant in vegetables due to direct deposition from air (mg kg^{-1} wet).

$$\dot{I} = \frac{C_{veg(total)} * F_{cv} * U_{veg} * EF * ED}{BW * AT}$$

where

I = mass-normalized intake rate (mg kg^{-1} d^{-1}),

$C_{veg(total)}$ = total concentration of contaminant in vegetables (mg kg^{-1} wet),

F_{cv} = fraction of contaminated vegetables (unitless),

U_{veg} = ingestion rate of vegetables by humans (kg d^{-1} wet),

EF = exposure frequency (d yr^{-1}),

ED = exposure duration (yr),

BW = body weight (kg),

and AT = averaging time (d).

Ingestion of Beef or Milk:

Pasture Concentration:

$$C_{past(soil)} = B_{v(past)} * C_{air} * V_d * \left[\frac{1 - e^{-\lambda_L T_r}}{\rho_{past} * \lambda_L} \right]$$

where

$C_{past(soil)}$ = concentration of contaminant in pasture from root uptake (mg kg^{-1} dry),

$B_{v(past)}$ = soil-to-plant transfer factor, dry weight (unitless),

C_{air} = concentration of contaminant in air (mg m^{-3}),

V_d = deposition velocity (m d^{-1}),

λ_L = soil leaching constant (d^{-1}),

T_r = release duration (d),

and ρ_{past} = effective surface soil density for pasture (kg m^{-2} dry).

$$C_{past(air)} = C_{air} * V_d * a_{past} * \left[\frac{1 - e^{-k_w T_{g(past)}}}{k_w} \right]$$

where

$C_{past(air)}$ = concentration of contaminant in pasture due to direct deposition from air (mg kg^{-1} dry),

C_{air} = concentration of contaminant in air (mg m^{-3}),

V_d = deposition velocity (m d^{-1}),

α_{past} = mass interception factor for pasture (m^2 kg^{-1}),

k_w = weathering constant (d^{-1}),

and $T_{g(past)}$ = growing period for pasture (d).

$$C_{past(total)} = C_{past(soil)} + C_{past(air)}$$

where

$C_{past(total)}$ = total concentration of contaminant in pasture (mg kg^{-1} dry),

$C_{past(soil)}$ = concentration of contaminant in pasture from root uptake (mg kg^{-1} dry),

and $C_{past(air)}$ = concentration of contaminant in pasture due to direct deposition from air (mg kg^{-1} dry).

Ingestion of Beef:

$$\dot{I} = \frac{C_{past(total)} * F_{pb} * Q_{feed,b} * F_f * U_{meat} * F_{cb} * EF * ED}{BW * AT}$$

where

I = mass-normalized intake rate (mg $kg^{-1}d^{-1}$),

$C_{past(total)}$ = total concentration of contaminant in pasture (mg kg^{-1} dry),

F_{pb} = fraction of feed that is contaminated pasture (unitless),

$Q_{feed,b}$ = ingestion rate of feed by beef cattle (kg d^{-1} dry),

F_f = feed-to-meat transfer factor (mg kg^{-1} per mg d^{-1}),

U_{meat} = ingestion rate of meat by humans (kg d^{-1}),

F_{cb} = fraction of meat that is contaminated (unitless),

EF = exposure frequency (d yr^{-1}),

ED = exposure duration (yr),

BW = body weight (kg),

and AT = averaging time (d).

Ingestion of Milk:

$$\dot{I} = \frac{C_{past(total)} * F_{pd} * Q_{feed,d} * F_m * U_{milk} * F_{cm} * EF * ED}{BW * AT}$$

where

I = mass-normalized intake rate (mg kg^{-1}d^{-1}),

$C_{past(total)}$ = total concentration of contaminant in pasture (mg kg^{-1} dry),

F_{pd} = fraction of feed that is contaminated pasture (unitless),

$Q_{feed,d}$ = ingestion rate of feed by dairy cattle (kg d^{-1} dry),

F_m = feed-to-milk transfer factor (mg L^{-1} per mg d^{-1}),

U_{milk} = ingestion rate of milk by humans (L d^{-1}),

F_{cm} = fraction of milk that is contaminated (unitless),

EF = exposure frequency (d yr^{-1}),

ED = exposure duration (yr),

BW = body weight (kg),

and AT = averaging time (d).

Aquatic Releases:

Water Ingestion:

$$\dot{I} = \frac{C_{water} * U_{water} * F_{cw} * EF * ED}{BW * AT}$$

where

I = mass-normalized intake rate (mg kg^{-1} d^{-1}),

C_{water} = concentration of contaminant in water (mg L^{-1}),

U_{water} = ingestion rate of water by humans (L d^{-1}),

F_{cw} = fraction of water that is contaminated (unitless),

EF = exposure frequency (d yr^{-1}),

ED = exposure duration (yr),

BW = body weight (kg),

and AT = averaging time (d).

Fish Ingestion:

$$C_{fish} = C_{water} * BCF$$

where

C_{fish} = concentration of contaminant in fish (mg kg^{-1}),

C_{water} = concentration of contaminant in water (mg L^{-1}), and BCF = bioconcentration factor for fish [(mg kg^{-1}) per (mg L^{-1})].

$$\dot{I} = \frac{C_{fish} * U_{fish} * F_{cf} * EF * ED}{BW * AT}$$

where,

I = mass-normalized intake rate (mg kg^{-1} d^{-1}),

C_{fish} = concentration of contaminant in fish (mg kg^{-1}),

U_{fish} = ingestion rate of fish by humans(kg d^{-1}),

F_{cf} = fraction of fish that is contaminated (unitless),

EF = exposure frequency (d yr^{-1}),

ED = exposure duration (yr),

BW = body weight (kg),

and AT = averaging time (d).

<u>Soil or Sediment Contamination:</u>

Soil/Sediment Ingestion:

$$\dot{I} = \frac{C_{soil/sed(surf)} * U_{soil/sed(surf)} * F_{sc} * EF * ED}{BW * AT}$$

where

I = mass-normalized intake rate (mg kg^{-1} d^{-1}),

$C_{soil/sed(surf)}$ = concentration of contaminant in soil or sediment (mg kg^{-1}),

$U_{soil/sed(surf)}$ = ingestion rate of soil or sediment by humans (kg d^{-1}),

F_{sc} = fraction of contaminated soil or sediment (unitless),

EF = exposure frequency (d yr^{-1}),

ED = exposure duration (d),

BW = body weight (kg),

and AT = averaging time (d).

Soil to beef:

$$C_{beef(soil)} = C_{soil(surf)} * Q_{soil} * F_f * F_{sol}$$

where

$C_{beef(soil)}$ = concentration of contaminant in beef or game from ingestion of soil (mg kg^{-1}),

$C_{soil(surf)}$ = concentration of contaminant in surface soil (mg kg^{-1}),

Q_{soil} = ingestion rate of contaminated soil (kg d^{-1}),

F_f = feed-to-meat transfer factor [(mg kg^{-1}) per (mg d^{-1})],

and F_{sol} = solubility (bioavailability) fraction (unitless).

$$\dot{I} = \frac{C_{beef(soil)} * U_{meat} * F_{cb} * EF * ED}{BW * AT}$$

where

I = mass-normalized intake rate (mg kg^{-1} d^{-1}),

$C_{beef(soil)}$ = concentration of contaminant in beef or game from ingestion of soil (mg kg^{-1}),

U_{meat} = ingestion rate of meat by humans (kg d^{-1}),

F_{cb} = fraction of meat that is contaminated (unitless),

EF = exposure frequency (d yr^{-1}),

ED = exposure duration (yr),

BW = body weight (kg),

and AT = averaging time (d).

Soil to Dairy Cattle (milk):

$$C_{milk(soil)} = C_{soil(surf)} * Q_{soil} * F_m * F_{sol}$$

where

$C_{milk(soil)}$ = concentration of contaminant in milk from ingestion of soil (mg L^{-1}),

$C_{soil(surf)}$ = concentration of contaminant in surface soil (mg kg^{-1}),

Q_{soil} = ingestion rate of contaminated soil (kg d^{-1}),

F_m = feed-to-milk transfer factor (mg L^{-1}) per (mg d^{-1}),

and F_{sol} = solubility (bioavailability) fraction (unitless).

$$\dot{I} = \frac{C_{milk(soil)} * U_{milk} * F_{cm} * EF * ED}{BW * AT}$$

where

I = mass-normalized intake rate (mg kg^{-1} d^{-1}),

$C_{milk(soil)}$ = concentration of contaminant in milk from ingestion of soil (mg L^{-1}),

U_{milk} = ingestion rate of milk by humans (L d^{-1}),

F_{cm} = fraction of milk that is contaminated (unitless),

EF = exposure frequency (d yr^{-1}),

ED = exposure duration (yr),

BW = body weight (kg),

and AT = averaging time (d).

Soil to Vegetables (ingestion):

$$C_{veg(soil)} = C_{soil} * B_{v(veg)}$$

where

$C_{veg(soil)}$ = concentration of contaminant in vegetables from root uptake (mg kg^{-1} wet),

C_{soil} = concentration of contaminant in bulk soil (mg kg^{-1}),

and $B_{v(veg)}$ = soil-to-plant transfer factor, wet weight (unitless).

$$\dot{I} = \left[\frac{C_{veg(soil)} * U_{veg} * F_{cv} * EF * ED}{BW * AT} \right]$$

where

I = mass-normalized intake rate (mg kg^{-1} d^{-1}),

$C_{veg(soil)}$ = concentration of contaminant in vegetables from root uptake (mg kg^{-1} wet),

U_{veg} = ingestion rate of vegetables by humans (kg d^{-1} wet),

F_{cv} = fraction of contaminated vegetables (unitless),

EF = exposure frequency (d yr^{-1}),

ED = exposure duration (yr),

BW = body weight (kg),

and AT = averaging time (d).

Soil to Pasture to Beef or Milk (ingestion):

Pasture Contamination:

$$C_{past(soil)} = C_{soil} * B_{v(past)}$$

where

$C_{past(soil)}$ = concentration of contaminant in pasture from root uptake (mg kg^{-1} dry),

C_{soil} = concentration of contaminant in bulk soil (mg kg^{-1}),

and $B_{v(past)}$ = soil-to-plant transfer factor, dry weight (unitless).

Ingestion of Beef:

$$C_{beef(past)} = C_{past(soil)} * Q_{feed,b} * F_f * F_{pb}$$

where

$C_{beef(past)}$ = concentration of contaminant in beef (mg kg^{-1}),

$C_{past(soil)}$ = concentration of contaminant in pasture from root uptake (mg kg^{-1} dry),

$Q_{feed,b}$ = ingestion rate of feed by beef cattle (kg d^{-1} dry),

F_f = feed-to-meat transfer factor [(mg kg^{-1}) per (mg d^{-1})],

and F_{pb} = fraction of feed that is contaminated pasture (unitless).

$$\dot{I} = \frac{C_{beef(past)} * U_{meat} * F_{cb} * EF * ED}{BW * AT}$$

where

I = mass-normalized intake rate (mg kg^{-1} d^{-1}),

$C_{beef(past)}$ = concentration of contaminant in beef (mg kg^{-1}),

U_{meat} = ingestion rate of meat by humans (kg d^{-1}),

F_{cb} = fraction of meat that is contaminated (unitless),

EF = exposure frequency (d yr^{-1}),

ED = exposure duration (yr),

BW = body weight (kg),

and AT = averaging time (d).

Ingestion of Milk:

$$C_{milk(past)} = C_{past(soil)} * Q_{feed,d} * F_m * F_{pd}$$

where

$C_{milk(past)}$ = concentration of contaminant in milk from ingestion of pasture (mg L^{-1}),

$C_{past(soil)}$ = concentration of contaminant in pasture from root uptake (mg kg^{-1} dry),

$Q_{feed,d}$ = ingestion rate of feed by dairy cattle (kg d^{-1} dry),

F_m = feed-to-milk transfer factor [(mg L^{-1}) per (mg d^{-1})],

and F_{pd} = fraction of feed that is contaminated pasture (unitless).

$$\dot{I} = \frac{C_{milk(past)} * U_{milk} * F_{cm} * EF * ED}{BW * AT}$$

where

I = mass-normalized intake rate (mg kg^{-1} d^{-1}),

$C_{milk(past)}$ = concentration of contaminant in milk from ingestion of contaminated pasture (mg L^{-1}),

U_{milk} = ingestion rate of milk by humans (L d^{-1}),

F_{cm} = fraction of milk that is contaminated (unitless),

EF = exposure frequency (d yr^{-1}),

ED = exposure duration (yr),

BW = body weight (kg),

and AT = averaging time (d).

APPENDIX C

EXPOSURE PARAMETER ASSUMPTIONS

APPENDIX C: EXPOSURE PARAMETERS

Parameter values for Level I and Level II screening are selected based on the goals of the respective levels. Level I screening is intended to estimate the dose or risk to a "most at-risk" reference individual, while Level II screening is intended to estimate the risk to a more typically exposed individual (not necessarily the arithmetical average person or the least exposed person in the population). For example, for Level I screening of terrestrial food chain pathways, the target individual is assumed to derive most of his food (milk, meat, produce) from home-grown sources at a contaminated location, while the Level II target individual consumes a much smaller fraction of food produced at contaminated locations. Similarly, for external exposure pathways, the Level I target individual is assumed to spend a large fraction of time outdoors at a contaminated location, while the Level II target individual is assumed to spend somewhat more time indoors or away from the site.

Recommended default values for contaminant-independent parameters are provided in Table B-1, together with the rationales for their selection. These default values are to be used in the absence of more detailed or site-specific information. Parameters for which the values may depend on the specific contaminant (e.g., radionuclide, element, or chemical form of a contaminant) or the site-specific situation are listed in Table C-2. These include parameters such as toxicity values, radionuclide decay constants, transfer factors, and the release duration. The values for these parameters have not been included in this report. Evaluation of the contaminant-specific or site-specific parameter values will occur during the processes of screening for the individual contaminants, and the documentation of the selection of these values will be included with the screening calculations performed for each contaminant. In general, when site-specific data are not readily available, we expect to take Level I parameter values from sources such as the screening models developed by the International Atomic Energy Agency and the National Council on Radiation Protection and Measurements (for transfer factors) and documents prepared by the U.S. Environmental Protection Agency (for toxicity values). Release durations, in conjunction with source term information, will be obtained from site-specific documentation and the use of professional judgment to bound true but imperfectly known values.

Table C-1. Default Values for Contaminant-Independent Parameters.

Parameter	Symbol	Level I Value	Level II Value	Rationale
Exposure duration for carcinogenic chemicals (yr)	ED	50	10	The Level I value (50) is equivalent to the number of years the Oak Ridge National Laboratory has been in operation. The Level II value (10) is the 50th percentile for a farm residence (Israeli and Nelson, 1992). The national median time (50th percentile) at one residence is 9.1 years (ATSDR, 1992).
Exposure duration for noncarcinogenic chemicals (yr)	ED	1	1	This value is based on a subchronic exposure. For noncarcinogenic chemicals, the concern lies in exceeding a threshold for a toxic effect. Unusual habits in a given year or over a shorter period of time could result in this threshold being exceeded. If there is concern for potential acute exposure (e.g., a short-term accidental release), then this parameter will be adjusted on a case-by-case basis.
Exposure frequency (d yr^{-1})	EF	365	350	The values are based on the assumptions of no vacation (365) and a two week vacation away from the area of interest (350). The latter value is recommended by the USEPA (Fields and Diamond, 1991).
Averaging time for carcinogenic chemicals (days)	AT	25,550	25,550	This value, equivalent to 70 years, is based on the average lifetime of an individual (Fields and Diamond, 1991; USEPA, 1989).
Averaging time for noncarcinogenic chemicals (days)	AT	365	365	This value should equal the exposure duration (ED, above) of one year for noncarcinogenic chemicals (Fields and Diamond, 1991). If the exposure duration is adjusted, this parameter value will be adjusted also.
Average adult body weight (kg)	BW	70	70	ICRP (1975) and the USEPA (1985; cited in USEPA, 1989) consider this value to be representative of an average adult male.

Table C-1. Cont.

Parameter	Symbol	Level I Value	Level II Value	Rationale
Average child body weight (kg)	BW	16	20	The Level I value (16) is the 50th percentile of the average body weight of children aged 1-6 years old (ATSDR, 1992). The Level II value (20) is representative of an average child (approximately 6 years old, regardless of sex) (ICRP, 1975; 1992).
Average daily consumption of meat by adults (kg d^{-1})	U_{meat}	0.3	0.1	The Level I value (0.3) is based on the average total intake of meat (0.258 kg d^{-1}) for adults (Rupp, 1980). The number should be representative of a person who raised most of his own meat. The Level II value (0.1) is based on the average total intake of beef (0.086 kg d^{-1}) for adults (Rupp, 1980).
Average daily consumption of meat by children (kg d^{-1})	U_{meat}	0.15	0.03	The Level I value (0.15) is the average consumption of all meat, poultry, and fish for children aged 6-8 years old (ATSDR, 1992) and is the average intake of all meat for children aged 1-11 years old (Rupp, 1980). The Level II value (0.03) is the average intake of beef per day in urban areas for 6-8 year olds (ATSDR, 1992) and is the average intake of beef per day for children aged 1-11 years old (Rupp, 1980).
Fraction of meat consumed that is contaminated (unitless)	F_{cb}	0.8	0.3	These values are based on the assumption that meat was obtained from several sources, as opposed to a single source.
Average daily consumption of milk by adults (L d^{-1})	U_{milk}	1.0	0.3	The Level I value (1.0) is exceeded by fewer than 2.6% of adults aged 20-54 years old (Pao and Burk, 1975; cited in Rupp, 1980). Ninety percent or more are below 0.971 L d^{-1}. The Level II value (0.3) is approximately equal to the average for a male between 30 and 60 years of age (Pao and Burk, 1975; cited in Rupp, 1980).

Table C-1. Cont.

Parameter	Symbol	Level I Value	Level II Value	Rationale
Average daily consumption of milk by children (L d^{-1})	U_{milk}	1.0	0.5	The Level I value (1.0) is consistent with the value reported in ICRP (1975). In Pao and Burk (1975; cited in Rupp, 1980), a range of 0.971 to 1.33 L d^{-1} is exceeded by fewer than 2.5% of children aged 3-11 years old. Ninety percent or more are below 0.971 L d^{-1}. The Level II value (0.5) is approximately equal to the 50th percentile for children aged 3-11 years old (Pao and Burk, 1975; cited in Rupp, 1980).
Fraction of milk consumed that is contaminated (unitless)	F_{cm}	1.0	0.5	For Level I the value (1.0) was based on the assumption that the maximally exposed individual obtained his milk from a backyard cow. For Level II, it was assumed that an individual would get one-half of his milk from contaminated local sources.
Delay time between milking and consumption of milk (d)	T_{cons}	1.0	4.0	The Level I value (1.0) is the midpoint of the range (0 to 2 days) for backyard cows and is the minimum value of the range (1 to 4 days) for commercial cows as reported by Snyder et al. (1994). The Level II value (4.0) is the maximum of the range for commercial cows as reported by Snyder et al. (1994).
Average daily consumption of vegetables by adults, wet weight (kg d^{-1})	U_{veg}	0.5	0.2	The Level I value (0.5) is based on average total intake of all fresh produce by adults (including leafy vegetables, deep yellow vegetables, legumes, other vegetables, citrus including tomatoes, other fruit, and potatoes) of approximately 0.48 kg d^{-1} in 1955 and 0.44 kg d^{-1} in 1965 (Rupp, 1980). The Level II value (0.2) is the average intake of vegetables for adults reported by Rupp (1980), Fields and Diamond (1991), and ATSDR (1992).
Average daily consumption of vegetables by children, wet weight (kg d^{-1})	U_{veg}	0.4	0.1	The Level I value (0.4) is based on the average total intake of all fresh produce (including leafy vegetables, deep yellow vegetables, legumes, other vegetables, citrus including tomatoes, other fruit, and potatoes) by children aged 1-11 years of approximately 0.37 kg d^{-1} in 1955 and 0.34 kg d^{-1} in 1965 (Rupp, 1980). The Level II value (0.1) is based on the average value (0.107 kg d^{-1}) for the consumption of vegetables by children aged 1-11 years (Rupp, 1980).

Table C-1. Cont.

Parameter	Symbol	Level I Value	Level II Value	Rationale
Fraction of vegetables consumed that is contaminated (unitless)	F_{cv}	0.6	0.2	The values are based on the assumption that not all vegetables consumed are homegrown. For Level I, 60% of the vegetables consumed were assumed to have been contaminated. For Level II, 20% of the vegetables were assumed to have been contaminated.
Fraction of contamination remaining on vegetables after washing (unitless)	F_w	0.7	0.2	The Level I value (0.7) is consistent with the upper bounds of the ranges provided by the IAEA (1992; 1994) for removal of ^{90}Sr, ^{137}Cs, ^{131}I, and ^{106}Ru from spinach by washing and blanching. The Level II value (0.2) is consistent with the lower bounds or midpoints of the ranges provided by the IAEA (1992; 1994) for removal of ^{137}Cs, ^{131}I, and ^{106}Ru from lettuce and/or removal of inedible parts.
Average daily consumption of water by adults (L d^{-1})	U_{water}	2.2	1.4	The Level I value (2.2) is the upper bound of total fluid intake excluding milk for adults (USEPA, 1989; ICRP, 1975). The Level II value (1.4) is the average daily intake of fluid excluding milk for adults (Cook et al., 1975; cited in Rupp, 1980).
Average daily consumption of water by children (L d^{-1})	U_{water}	1.3	1.0	The Level I value (1.3) is the upper bound of total fluid intake excluding milk for children aged 12-17 (Cook et al., 1975; cited in Rupp, 1980). The Level II value (1.0) is the average intake of total fluids excluding milk for children aged 12-17 (Cook et al., 1975; cited in Rupp, 1980); ATSDR (1992) gives 1.0 as the average intake for children, age unspecified.
Fraction of water consumed by humans that is contaminated (unitless)	F_{cw}	0.5	0.2	The values are based on the assumption that tap water is not the only source of fluid intake.

Table C-1. Cont.

Parameter	**Symbol**	**Level I Value**	**Level II Value**	**Rationale**
Average daily consumption of fish by adults, noncarcinogens (kg d^{-1})	U_{fish}	0.09	0.04	For analysis of exposure to noncarcinogens, the fish intake will be examined over a shorter time period than for carcinogens. For short time periods, it is possible to have a much higher fish consumption rate than if averaged over a longer time frame. The Level I value (0.09) is the maximum consumption (33 kg yr^{-1}) for an adult fisherman eating freshwater sportfish from the Columbia River (Honstead et al., 1971; cited in Rupp et al., 1980). The Level II value (0.04) is the 50th percentile of the range of fish consumption reported for Lake Michigan fishermen (Humphrey, 1978; cited in Rupp et al., 1980).
Average daily consumption of fish by children, noncarcinogens (kg d^{-1})	U_{fish}	0.045	0.01	The Level I value (0.045) is one-half of the value used for adults, based on an adult fisherman eating freshwater sportfish from the Columbia River. The Level II value (0.01) is the per capita average for children and adults (5 kg yr^{-1}) reported by Rupp et al. (1980).
Average daily consumption of fish by adults, carcinogens (kg d^{-1})	U_{fish}	0.03	0.01	For analysis of exposure to carcinogens, the fish intake will be examined over a longer time period; therefore, a lower fish consumption rate is appropriate. The Level I value (0.03) is the most frequently recorded consumption rate of freshwater fish in the Lake Michigan Survey (10 to 15 kg yr^{-1}) (Humphrey, 1978; cited in Rupp et al., 1980). The Level II value (0.01) is the per capita average value for children and adults (5 kg yr^{-1}) reported by Rupp et al. (1980) and the average intake for fish and shellfish for the total population (ATSDR, 1992).
Fraction of fish consumed that is contaminated (unitless)	F_{cf}	0.8	0.2	The Level I value (0.8) is based on the assumption that the maximally exposed individual is an avid fisherman and that the fisherman obtained the majority of his fish from a contaminated source. The Level II value (0.2) is based on the assumption that an average individual would obtain only 20% of his fish from a contaminated source.
Weathering rate for vegetation (d^{-1})	k_w	0.05	0.05	This is the recommended screening value reported by IAEA (in preparation), based on a wide survey of the literature performed by Pröhl et al. (1995) and IAEA (in press).

Table C-1. Cont.

Parameter	**Symbol**	**Level I Value**	**Level II Value**	**Rationale**
Growing period for pasture grass (d)	$T_{g(past)}$	30	30	This is the recommended screening value for the growing period of pasture grass given by the IAEA (in preparation).
Growing period for leafy vegetables (d)	$T_{g(veg)}$	60	60	This is the recommended screening value for the growing period of food crops given by the IAEA (in preparation).
Daily ingestion of feed by beef cattle, dry weight (kg d^{-1})	$Q_{feed, b}$	10	7.2	The Level I value (10) is the upper bound of the range reported by the IAEA (1994) for the dry matter intake of beef cattle. The Level II value (7.2) is the expected value for the ingestion of dry matter for beef cattle weighing 500 kg (IAEA, 1994).
Fraction of feed ingested by beef cattle that is from contaminated pasture (unitless)	F_{pb}	1	0.4	The Level I value (1) is based on a worst-case scenario, where all of the cow's food was obtained from a contaminated pasture. The Level II value (0.4) is based on the assumption that supplemental feed was used and only 40% of the total feed came from contaminated pasture.
Daily ingestion of feed by dairy cattle, dry weight (kg d^{-1})	$Q_{feed, d}$	16	9.1	According to Husted-Anderson (1941), dairy cattle consumed 11-17.8 kg d^{-1} of dry matter in a closely managed feeding system. However, for Level I it is assumed that the milk was obtained from backyard cattle. "These animals typically forage on semi-wild vegetation and not much effort is made to improve the quality of pasture unless other grazing stock require it" (Koranda, 1965). Given the poorer economic conditions in the area during the 1940s to 1960s, improvements to the grazing pasture would have been unlikely. Therefore, the Level I value (16) was chosen to be consistent with the upper bound estimate reported by Koranda (1965) for cattle raised in an unmanaged feeding regime. For Level II, the value is the mean estimate of 9.1 kg d^{-1} reported by Koranda (1965).

Table C-1. Cont.

Parameter	Symbol	Level I Value	Level II Value	Rationale
Fraction of feed ingested by dairy cattle that is from contaminated pasture (unitless)	F_{pd}	0.75	0.3	The Level I value (0.75) was assumed to be the worst-case scenario, where the cow obtains 75% of its food from contaminated pasture. The Level II value (0.3) assumes that the cow receives only 30% of its food from the contaminated pasture. Unlike beef cows, dairy cows are almost always provided with some supplemental feed.
Pasture soil bulk density, dry weight (kg m^{-2})	ρ_{past}	130	130	The value of 130 kg m^{-2} (based on a soil density of 1.3 g cm^{-3}) is the recommended screening value for the effective surface soil density for pasture at a depth of 0-10 cm (IAEA, in preparation).
Soil bulk density for crops other than pasture, dry weight (kg m^{-2})	ρ	260	260	The value of 260 kg m^{-2} (based on a soil density of 1.3 g cm^{-3}) is the recommended screening value for the effective surface soil density for all other crops at a depth of 0-20 cm (IAEA, in preparation).
Quantity of air inhaled per day by adults (m^3 d^{-1})	U_{air}	20	20	This value is the upper bound for housewives, retired employees, unemployed workers, service workers, and household workers as reported by the USEPA (1985; cited in Fields and Diamond, 1991) and is consistent with the average total quantity of air breathed per day for men and women (23 and 21, respectively) working light activity (8 hr d^{-1}), conducting nonoccupational activities (8 hr d^{-1}), and resting (8 hr d^{-1}) (ICRP, 1975).
Quantity of air inhaled per day by children (m^3 d^{-1})	U_{air}	12	12	This value is within the range of resting and light activity breathing rates for a 10-year old child as reported by the ICRP (1992).
Fraction of time that person is exposed to contaminated air (unitless)	F_{occ}	0.8	0.4	The Level I value (0.8) is based on the worst-case scenario of a farmer who is away from his property no more than 5 hr d^{-1}. The Level II value (0.4) is based on the assumption that an individual is in the area an average of 10 hr d^{-1}.

Table C-1. Cont.

Parameter	**Symbol**	**Level I Value**	**Level II Value**	**Rationale**
Fraction of day when individual is exposed, ground exposure (unitless)	$F_{t,\,ext}$	1.0	0.3	The Level I value (1) is based on the assumption that the individual lives on contaminated soil, works outside, and also receives an exposure while indoors. The Level II value (0.3) is based on the assumption that the individual is exposed only 30% of the time; the value is representative of a person present in the area about 8 hr d^{-1}.
Indoor/outdoor reduction (shielding) factor (unitless)	F_s	0.5	0.3	The Level I value (0.5) is the upper-bound estimate for 1-2 story wood frame houses reported by Roed (1990) and is consistent with the range of 0.05 to 0.65 for wood frame houses reported by Burson and Profio (1977; cited in Snyder et al., 1994). The Level II value (0.3) is the upper bound for structures composed of block and brick as reported by Burson and Profio (1977; cited in Snyder et al., 1994) and is consistent with the range of 0.05 to 0.4 for 1-2 story brick/block houses reported by Roed (1990).
Average daily ingestion of soil by adults (kg d^{-1})	U_{soil}	1.0×10^{-4}	5.0×10^{-5}	The Level I value (1.0×10^{-4}) is the reasonable maximum exposure for apartment dwellers, typical homeowners, office workers, teachers, and professionals (non-contact intensive) reported in Sedman (1989; cited in ATSDR, 1992). The Level II value (5.0×10^{-5}) is the central tendency for non-contact intensive persons (Calabrese et al., 1990; cited in ATSDR, 1992).
Average daily ingestion of soil by children (kg d^{-1})	U_{soil}	2.5×10^{-4}	7.5×10^{-5}	The Level I value (2.5×10^{-4}) is the upper bound of the range reported by Lepow et al. (1975; cited in Paustenbach, 1989) for the ingestion of lead-contaminated soils by children, age unspecified. The Level II value (7.5×10^{-5}) is the midpoint of the range provided by Paustenbach (1989) for a reasonable average daily intake of soil by toddlers (ages 2-4).
Fraction of soil ingested that is contaminated (unitless)	F_{sc}	0.7	0.25	The Level I value (0.7) is based on the assumption that a child lives near a contaminated playground or an adult lives in or near a contaminated area. The Level II value (0.25) is based on the assumption that the individual is exposed to contaminated soil 25% of the time or 6 hr d^{-1}.

Table C-1. Cont.

Parameter	Symbol	Level I Value	Level II Value	Rationale
Daily ingestion rate of contaminated soil by beef and dairy cattle (kg d^{-1})	Q_{soil}	0.5	0.25	The Level I value (0.5) is consistent with the central value reported by Darwin (1990; cited in Snyder et al., 1994). Zach and Mayoh (1984) and Whicker and Kirchner (1987) recommend the use of 0.5 kg d^{-1} in food-chain models. The Level II value (0.25) is the geometric mean based on measured weights of soil in cattle GI tracts (Gilbert et al., 1995).
Total deposition velocity (m d^{-1})	$V_{d(total)}$	1000	500	The Level I value (1000 m d^{-1}) is the screening value recommended by the IAEA (in preparation) and NCRP (1989) for deposition of aerosols and reactive gases. The Level I value was divided by two to obtain a less conservative value for Level II.
Mass interception factor for vegetables (m^2 kg^{-1}, wet weight)	α_{veg}	0.3	0.3	This is the screening value recommended by the IAEA (in preparation).
Mass interception factor for pasture (m^2 kg^{-1}, dry weight)	α_{past}	3.0	3.0	This is the screening value recommended by the IAEA (in preparation).
Soil leaching constant, anionic substances (d^{-1})	$\lambda_{s,\ anionic}$	1.4×10^{-3}	--	The Level I value (1.4×10^{-3}) is the screening value recommended by the IAEA (in preparation) for anionic substances. The Level II value will be evaluated on a case-by-case basis.
Soil leaching constant, non-anionic substances (d^{-1})	$\lambda_{s,\ other}$	0.0	--	The Level I value (0.0), based on the assumption that the contaminant will persist indefinitely in the environment, is the screening value recommended by the IAEA (in preparation) for nonanionic substances. The Level II value will be evaluated on a case-by-case basis.

Table C-2. Examples of Contaminant- or Site-Specific Parameters[a].

Parameter	Definition (units)
λ_R	radioactive decay constant (yr^{-1} or d^{-1})
$B_{v(veg)}$	soil-to-plant transfer factor for vegetables, wet weight (unitless)
$B_{v(past)}$	soil-to-plant transfer factor for pasture, dry weight (unitless)
F_f	feed-to-meat transfer factor ($d\ kg^{-1}$, $mg\ kg^{-1}$ per $mg\ d^{-1}$, or $Bq\ kg^{-1}$ per $Bq\ d^{-1}$)
F_m	feed-to-milk transfer factor ($d\ L^{-1}$, $mg\ L^{-1}$ per $mg\ d^{-1}$, or $Bq\ L^{-1}$ per $Bq\ d^{-1}$)
BCF	bioconcentration factor/bioaccumulation factor ($Bq\ kg^{-1}$ per $Bq\ L^{-1}$ or $mg\ kg^{-1}$ per $mg\ L^{-1}$)
F_{sol}	solubility (bioavailability) fraction (unitless); this is equal to 1 unless the form of the contaminant is considerably different from the form for which F_f was derived
T_r	release duration for air-to-soil-deposition (yr)
RfD	oral reference dose ($mg\ kg^{-1}\ d^{-1}$)
SF	slope factor (risk Bq^{-1}, risk yr^{-1} per $Bq\ kg^{-1}$, risk yr^{-1} per $Bq\ m^{-3}$, risk yr^{-1} per $Bq\ m^{-2}$, or risk per $mg\ kg^{-1}\ d^{-1}$)

[a] Values for these parameters will be selected during the screening process for each contaminant. Documentation for the selection of the values will be included with the screening calculations for each contaminant.

APPENDIX D

SUMMARY OF ASBESTOS WASTE DISPOSAL INFORMATION

Table D-1: Summary of Asbestos Disposal at the ORR

Year	Disposal Site	Type of waste	Quantity	Reference
1950-76	K-1070-B	asbestos-containing classified equipment from K-25	NR	TSD 1977; Goddard et al. 1991
1978-84	Y-12 Asbestos Disposal Pits	asbestos and asbestos-containing materials from Y-12	354,500 kg	MMES 1984
1979	Y-12 Burial Grounds	asbestos from K-25	13,368 ft^2	Goodpasture and Rogers 1986
1980	Y-12 Burial Grounds	asbestos from K-25	11,778 ft^2	Goodpasture and Rogers 1986
1981	Y-12 Burial Grounds	asbestos from K-25	24,987 ft^2	Goodpasture and Rogers 1986
1982	Y-12 Burial Grounds	asbestos from K-25	5,275 ft^2	Goodpasture and Rogers 1986
1983	Y-12 Burial Grounds	asbestos from K-25	27,020 ft^2	Goodpasture and Rogers 1986
1984	Y-12 Burial Grounds	asbestos from K-25	82,569 ft^2	Goodpasture and Rogers 1986
#1984	ORNL Burial Site	asbestos from Y-12	NR	MMES 1984
1985	Y-12 Burial Grounds	asbestos from K-25	28,820 ft^2	Goodpasture and Rogers 1986
1986	Y-12 Burial Grounds	asbestos from Y-12	172,000 kg	Oakes et al. 1987
1986	Y-12 Sanitary Landfill II	asbestos from Y-12	139,000 kg	Oakes et al. 1987
1987	ORNL SWSA-6	radiological asbestos waste from ORNL	575 kg	Rogers et al. 1988 a,b
1987	Y-12 Sanitary Landfill	nonradiological asbestos waste from ORNL, K-25	33,518 kg, 31 m^3	Rogers et al. 1988 a,b
1987	NR	asbestos/beryllium oxide material from Y-12	3,700,000 kg	Rogers et al. 1988 a
1987	NR	uranium-contaminated asbestos /beryllium oxide from Y-12	270,000 kg	Rogers et al. 1988 a
#1989	K-1070-C/D	asbestos from K-25	NR	SAIC 1995

Year	Disposal Site	Type of waste	Quantity	Reference
1990	NR	radiological asbestos/beryllium waste from K-25	75 m^3	Kornegay et al. 1991
1990	Y-12 Landfill	nonradiological asbestos/ beryllium waste from K-25	1332 m^3	Kornegay et al. 1991
1990	ORNL SWSA-6	radiological asbestos waste from ORNL	6600 kg	Kornegay et al. 1991
1990	Y-12 Landfill	nonradiological asbestos waste from ORNL	15,237 kg	Kornegay et al. 1991
1990	NR	radiological asbestos waste from Y-12	84,638 kg	Kornegay et al. 1991
1990	NR	nonradiological asbestos waste from Y-12	217,000 kg	Kornegay et al. 1991
1991	Y-12 Landfill	nonradiological asbestos from K-25	88 m^3	Kornegay et al. 1992
1991	Y-12 Landfill	asbestos from ORNL	19,229 kg	Kornegay et al. 1992
1991	K-770 Scrap Metal Yard	asbestos-covered lead pipe from K-25	2318 kg	Kornegay et al. 1992

NR = Not reported.

APPENDIX E

SUMMARY OF ORIGEN2.1 CALCULATIONS FOR THE TASK 7 PLUTONIUM ASSESSMENT

SRA
4939 Lower Roswell Road, Suite 106
Marietta, GA 30068
Phone: (770) 509-7606
FAX: (770) 509-7507
Internet: sra@crl.com

May 1, 1997

TO:	Tom Widner ChemRisk - Alameda
FROM:	R. E. Burns, Jr. Shonka Research Associates, Inc.
SUBJECT:	Plutonium ratios for pilot plant slugs for Task 7
MEMO NO:	REB.008 C97
FINAL [**X**]	DRAFT [ ]

Distribution:

References:

Memo:

Mass ratios for the isotopes of plutonium present in X-slugs irradiated in the Clinton pile were computed using the ORIGEN2.1 computer code with the CANDUNAU cross-section and fission product yield library. Ratios were computed for irradiation times of 50, 100 and 150 days. Results are given for the time of pile shutdown following irradiation and for 100 days thereafter in 10 day increments.

The ORIGEN2.1 code was used to irradiate 258 X-slugs for durations of 50, 100 and 150 days at a flux of 9.704×10^{11} neutrons cm^{-2} second^{-1}. 258 slugs are the equivalent of one-third ton of uranium, which was the nominal daily throughput for the pilot plant in 1944. Assuming a total mass of uranium in the Clinton pile at that time of 42.7 tons (33,000 slugs), a throughput of one-third ton per day equates to a mean residence time in the pile of 128 days. The three irradiation times were chosen to sufficiently bound this value.

From review of the known power levels for the pile during this period, a nominal power level of 3000 kW was chosen. The conversion from this power level to flux was

$$3000 \text{ kW} \times (3.102 \times 10^{5} \text{ neutrons cm}^{-2} \text{ watt}^{-1} \text{ second}^{-1}) \times 3.16 \times 0.66 \times 0.5 = 9.704 \times 10^{11}$$

where 3.102×10^{5} neutrons cm^{-2} watt^{-1} second^{-1} is the conversion from total power to peak thermal flux in a pile experimental hole;

3.16 is the conversion from thermal flux at ambient temperature to total flux for the CANDUNAU model (1000° C);
0.66 is the ratio of flux in fuel to that in the experimental hole; and
0.5 is the ratio of average to peak flux for the Clinton pile.

Results from the three ORIGEN2.1 runs are shown in Tables 1 through 3. These are in units of total grams for the 258 slugs.

Table 1 Plutonium content (grams) versus decay time for 258 X-slugs irradiated for 50 days

Nuclide	Discharge	10 days	20 days	30 days	40 days	50 days	60 days	70 days	80 days	90 days	100 days
PU236	3.043E-13	3.224E-13	3.202E-13	3.181E-13	3.160E-13	3.139E-13	3.118E-13	3.097E-13	3.077E-13	3.057E-13	3.036E-13
PU237	2.110E-12	1.813E-12	1.557E-12	1.338E-12	1.149E-12	9.870E-13	8.478E-13	7.282E-13	6.255E-13	5.373E-13	4.616E-13
PU238	2.842E-07	3.258E-07	3.273E-07	3.273E-07	3.272E-07	3.271E-07	3.271E-07	3.270E-07	3.269E-07	3.268E-07	3.268E-07
PU239	1.497E+00	1.601E+00	1.606E+00	1.607E+00	1.607E+00	1.607E+00	1.607E+00	1.607E+00	1.607E+00	1.607E+00	1.607E+00
PU240	4.365E-04	4.365E-04	4.365E-04	4.365E-04	4.365E-04	4.365E-04	4.365E-04	4.365E-04	4.365E-04	4.365E-04	4.365E-04
PU241	1.184E-07	1.182E-07	1.181E-07	1.179E-07	1.178E-07	1.176E-07	1.175E-07	1.173E-07	1.172E-07	1.170E-07	1.169E-07
PU242	1.555E-11	1.555E-11	1.555E-11	1.555E-11	1.555E-11	1.555E-11	1.555E-11	1.555E-11	1.555E-11	1.555E-11	1.555E-11
PU243	9.896E-18	2.609E-32	0.000E+00	0.000E+00	0.000E+00	0.000E+00	0.000E+00	0.000E+00	0.000E+00	0.000E+00	0.000E+00
PU244	3.477E-22	3.477E-22	3.477E-22	3.477E-22	3.477E-22	3.477E-22	3.477E-22	3.477E-22	3.477E-22	3.477E-22	3.477E-22
PU245	5.564E-29	8.503E-36	1.373E-42	0.000E+00	0.000E+00	0.000E+00	0.000E+00	0.000E+00	0.000E+00	0.000E+00	0.000E+00
PU246	3.653E-35	3.653E-35	3.653E-35	3.653E-35	3.653E-35	3.653E-35	3.653E-35	3.653E-35	3.653E-35	3.653E-35	3.653E-35

Table 2 Plutonium content (grams) versus decay time for 258 X-slugs irradiated for 100 days

Nuclide	Discharge	10 days	20 days	30 days	40 days	50 days	60 days	70 days	80 days	90 days	100 days
PU236	1.472E-12	1.507E-12	1.497E-12	1.487E-12	1.477E-12	1.467E-12	1.457E-12	1.448E-12	1.438E-12	1.429E-12	1.419E-12
PU237	7.308E-12	6.277E-12	5.392E-12	4.632E-12	3.979E-12	3.418E-12	2.936E-12	2.522E-12	2.166E-12	1.861E-12	1.598E-12
PU238	1.472E-06	1.569E-06	1.572E-06	1.572E-06	1.572E-06	1.571E-06	1.571E-06	1.571E-06	1.570E-06	1.570E-06	1.570E-06
PU239	3.100E+00	3.204E+00	3.210E+00	3.210E+00	3.210E+00	3.210E+00	3.210E+00	3.210E+00	3.210E+00	3.210E+00	3.210E+00
PU240	1.847E-03	1.847E-03	1.847E-03	1.847E-03	1.847E-03	1.847E-03	1.847E-03	1.847E-03	1.847E-03	1.847E-03	1.847E-03
PU241	1.024E-06	1.023E-06	1.021E-06	1.020E-06	1.019E-06	1.017E-06	1.016E-06	1.015E-06	1.013E-06	1.012E-06	1.011E-06
PU242	2.749E-10	2.749E-10	2.749E-10	2.749E-10	2.749E-10	2.749E-10	2.749E-10	2.749E-10	2.749E-10	2.749E-10	2.749E-10
PU243	1.750E-16	4.613E-31	0.000E+00	0.000E+00	0.000E+00	0.000E+00	0.000E+00	0.000E+00	0.000E+00	0.000E+00	0.000E+00
PU244	1.529E-20	1.529E-20	1.529E-20	1.529E-20	1.529E-20	1.529E-20	1.529E-20	1.529E-20	1.529E-20	1.529E-20	1.529E-20
PU245	2.446E-27	3.738E-34	5.699E-41	0.000E+00	0.000E+00	0.000E+00	0.000E+00	0.000E+00	0.000E+00	0.000E+00	0.000E+00
PU246	7.360E-33	7.360E-33	7.360E-33	7.360E-33	7.360E-33	7.360E-33	7.360E-33	7.360E-33	7.360E-33	7.360E-33	7.360E-33

Table 3 Plutonium content (grams) versus decay time for 258 X-slugs irradiated for 150 days

Nuclide	Discharge	10 days	20 days	30 days	40 days	50 days	60 days	70 days	80 days	90 days	100 days
PU236	3.511E-12	3.557E-12	3.534E-12	3.510E-12	3.487E-12	3.464E-12	3.441E-12	3.418E-12	3.396E-12	3.373E-12	3.351E-12
PU237	1.395E-11	1.198E-11	1.029E-11	8.839E-12	7.593E-12	6.522E-12	5.602E-12	4.812E-12	4.134E-12	3.551E-12	3.050E-12
PU238	3.613E-06	3.766E-06	3.771E-06	3.770E-06	3.769E-06	3.768E-06	3.767E-06	3.767E-06	3.766E-06	3.765E-06	3.764E-06
PU239	4.701E+00	4.805E+00	4.810E+00	4.811E+00	4.811E+00	4.811E+00	4.811E+00	4.811E+00	4.811E+00	4.811E+00	4.811E+00
PU240	4.234E-03	4.234E-03	4.234E-03	4.234E-03	4.234E-03	4.234E-03	4.234E-03	4.234E-03	4.234E-03	4.234E-03	4.234E-03
PU241	3.545E-06	3.540E-06	3.535E-06	3.531E-06	3.526E-06	3.521E-06	3.517E-06	3.512E-06	3.507E-06	3.503E-06	3.498E-06
PU242	1.440E-09	1.440E-09	1.440E-09	1.440E-09	1.440E-09	1.440E-09	1.440E-09	1.440E-09	1.440E-09	1.440E-09	1.440E-09
PU243	9.167E-16	2.417E-30	0.000E+00	0.000E+00	0.000E+00	0.000E+00	0.000E+00	0.000E+00	0.000E+00	0.000E+00	0.000E+00
PU244	1.239E-19	1.239E-19	1.239E-19	1.239E-19	1.239E-19	1.239E-19	1.239E-19	1.239E-19	1.239E-19	1.239E-19	1.239E-19
PU245	1.983E-26	3.031E-33	4.631E-40	0.000E+00	0.000E+00	0.000E+00	0.000E+00	0.000E+00	0.000E+00	0.000E+00	0.000E+00
PU246	1.030E-31	1.030E-31	1.030E-31	1.030E-31	1.030E-31	1.030E-31	1.030E-31	1.030E-31	1.030E-31	1.030E-31	1.030E-31

Tables 4 through 6 show the data from above for each decay duration as fractions of the total plutonium content for each isotope.

Table 4 Fractions of the total plutonium content as a function of decay time for each isotope for slugs irradiated 50 days

Nuclide	Discharge	10 days	20 days	30 days	40 days	50 days	60 days	70 days	80 days	90 days	100 days
PU236	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000
PU237	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000
PU238	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000
PU239	0.9997	0.9997	0.9997	0.9997	0.9997	0.9997	0.9997	0.9997	0.9997	0.9997	0.9997
PU240	0.0003	0.0003	0.0003	0.0003	0.0003	0.0003	0.0003	0.0003	0.0003	0.0003	0.0003
PU241	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000
PU242	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000
PU243	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000
PU244	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000
PU245	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000
PU246	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000

Table 5 Fractions of the total plutonium content as a function of decay time for each isotope for slugs irradiated 100 days

Nuclide	Discharge	10 days	20 days	30 days	40 days	50 days	60 days	70 days	80 days	90 days	100 days
PU236	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000
PU237	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000
PU238	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000
PU239	0.9994	0.9994	0.9994	0.9994	0.9994	0.9994	0.9994	0.9994	0.9994	0.9994	0.9994
PU240	0.0006	0.0006	0.0006	0.0006	0.0006	0.0006	0.0006	0.0006	0.0006	0.0006	0.0006
PU241	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000
PU242	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000
PU243	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000
PU244	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000
PU245	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000
PU246	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000

Table 6 Fractions of the total plutonium content as a function of decay time for each isotope for slugs irradiated 150 days

Nuclide	Discharge	10 days	20 days	30 days	40 days	50 days	60 days	70 days	80 days	90 days	100 days
PU236	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000
PU237	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000
PU238	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000
PU239	0.9991	0.9991	0.9991	0.9991	0.9991	0.9991	0.9991	0.9991	0.9991	0.9991	0.9991
PU240	0.0009	0.0009	0.0009	0.0009	0.0009	0.0009	0.0009	0.0009	0.0009	0.0009	0.0009
PU241	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000
PU242	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000
PU243	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000
PU244	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000
PU245	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000
PU246	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000

The data in Tables 4 through 6 show that all of the plutonium present in slugs processed at the pilot plant in 1944 was ^{239}Pu for all intents and purposes.

APPENDIX F

THRESHOLD QUANTITY
SCREENING METHODOLOGY

APPENDIX F

THRESHOLD QUANTITY SCREENING METHODOLOGY

F.1 Introduction

In the Dose Reconstruction Feasibility Study, there were a number of materials whose mere presence at the Oak Ridge facilities were classified, as a result several materials were not evaluated for exposure to the off-site population. These materials present a challenge in the conduct of an open, public study. In order to screen these materials in a manner that is consistent with the spirit of the study the following screening process will be used to evaluate the materials using quantitative toxicity criteria and annual usage rates or inventory information. This method will predict whether a sufficient quantity of the classified material was likely present at the Oak Ridge Reservation (ORR) to pose a potential off-site health hazard. This is accomplished by determining the threshold quantity of a chemical that can be present at the site without posing an off-site health hazard based on conservatively designed exposure scenarios. Two exposure scenarios, inhalation of vapors or particulates and ingestion of drinking water, are included in the screening procedure.

F.2 Inhalation Exposure

The tmaximum allowable quantity of chemical that can be present at the ORR without posing a potential inhalation hazard can be determined in three steps, each of which is described below. The first step is to calculate the maximum annual average air concentration of a chemical (mg/ m^3) that can be inhaled by an off-site individual without adverse health effects. This calculation is based on the maximum allowable lifetime average daily dose as defined by a certain level of risk (carcinogens) or exposure (non-carcinogens).

The maximum allowable lifetime average daily dose for a carcinogen is derived from the carcinogenic potency slope factor (SF) and a defined level of risk. By using the 95 percent upper confidence limit, this estimate of carcinogenic response is conservative in that it usually over-estimates the actual risk posed by the chemical. For the purposes of this screening procedure, an excess cancer risk of one in one million (1 x 10^{-6}) over a lifetime was used as the cut-off point. Using an excess cancer risk of 1 x 10^{-6} and assuming a SF of 10 $(\text{mg/kg-day})^{-1}$, the maximum allowable lifetime average daily dose of a carcinogen can be calculated as follows:

$$Maximum\ Allowable\ Lifetime\ Average\ Daily\ Dose = \frac{1 \times 10^{-6}}{10\ (mg/kg{-}day)^{-1}}$$

For non-carcinogens, the maximum allowable lifetime average daily dose is simply equal to the reference dose (RfD). When a chemical does not have a RfD, a “derived RfD” can be estimated by multiplying the oral LD_{50} (mg/kg) of the chemical by a factor of 1×10^{-5} (Layton 1987).

$$= 1 \times 10^{-7} \; mg/kg\text{-}day$$

In a few instances where an oral LD_{50} cannot be obtained for a chemical, the lowest toxic dose (TD_{LO}) is used instead. If a TD_{LO} is based on animal data, the “derived RfD” is estimated by multiplying the TD_{LO} (mg/kg) by a factor of 1×10^{-5}. However, if the TD_{LO} is based on human data, the “derived RfD” is estimated by multiplying the TD_{LO} (mg/kg) by a factor of 1×10^{-4}. This is because a safety factor of 10 is usually used to compensate for the potential difference in sensitivity between laboratory animals and humans.

The maximum allowable annual average air concentration can then be determined using the following equation:

$$Dose = \frac{[Air] \times BR}{BW} \qquad \text{or} \qquad [Air] = \frac{Dose \times BW}{BR}$$

Where:

Dose = Maximum allowable lifetime average daily dose (as described above; mg/kg-day)

[Air] = Maximum allowable annual average air concentration (mg/m^3)

BR = Breathing rate (m^3/day)

BW = Body weight (kg)

Assuming an allowable lifetime average daily dose of 1×10^{-7} mg/kg-day (see above), an average adult body weight of 70 kg, and an adult daily breathing rate of 20 m^3/day (U.S. EPA, 1990a), the allowable annual air concentration equals:

$$[Air] = \frac{1 \times 10^{-7} \; mg/kg\text{-}day \times 70 \; kg}{20 \; m^3/day}$$

$$= 3.5 \times 10^{-7}\ mg/m^3$$

the second step in defining an “allowable quantity” is to relate the maximum allowable air concentration to an emission rate. For the purposes of this screening procedure the results of the ISCST3 air dispersion modeling for Task 6 were utilized. The Task 6 modeling, used ISCST3 to estimate annual-average concentrations based on unit release rates. The modeling was conducted using unit emission rates of 1 g/sec or 1 Ci/sec, from 15 different sources at Y-12. In using the ISCST3 model, several conservative (i.e. assumption likely to overestimate contaminant concentrations) input parameters and assumptions were used regarding the emission condition. It was also assumed that the receptor was located approximately 8 km downvalley from the Y-12 plant, 24 hours per day, 365 days per year. For releases from the 9212-2 stack, based on a unit emission rate of 1 g/sec (31,536 kg/year), the ISCST3 model predicted an annual average air concentration of 0.0709 μg/m^3. Thus, an emission rate of 31,536 kg/yr from the 9212-2 stack is required to produce an annual average air concentration of 0.0709μg/m^3, 8,000 meters from the emission source.

The relationship between the allowable annual average air concentration and the corresponding emission rate can be used to determine the emission rate for any chemical based on its SF or RfD. For example, using the previous example of a SF equal to 10 (mg/kg/day)$^{-1}$, the maximum allowable annual average air concentration was shown to be 3.5×10^{-7} mg/m^3. The corresponding emission rate can be determined as follows:

$$\frac{31536\ kg/yr}{0.0709\ \mu g/m^3} = \frac{Emission\ Rate}{3.5 \times 10^{-7}\ mg/m^3 \times 1000\ \mu g/mg}$$

or

$$Emission\ Rate = \frac{31536\ kg/yr \times 3.5 \times 10^{-7}\ mg/m^3 \times 1000\ \mu g/mg}{0.0709\ \mu g/m^3}$$

$$= 160\ kg/yr$$

By simple proportion, if the SF of a carcinogen is decreased by a factor of 10, the corresponding "allowable" emission rate would be increased by a factor of 10. Thus, a carcinogen with a slope factor of 1 (mg/kg-day)$^{-1}$ would have a maximum allowable dose equal to 1×10^{6} mg/kg-day, a maximum allowable air concentration equal to 3.5×10^{-6} mg/m^3 and a corresponding emission rate of 1600 kg/yr. This relationship can be extended to non-carcinogens as well. For a chemical with a RfD of 1×10^{-4} mg/kg-day, the maximum allowable air concentration would be 3.5×10^{-4} mg/m^3, and a corresponding allowable emission rate of 160,000 kg/yr.

The third and final step is to relate the emission rate of a chemical to the annual quantity of the material used at ORR. Annual use quantities will be gathered through plant records or professional judgement of knowledgeable plant personnel. Once an annual quantity is established, an estimate of the amount of the material that could have been released off-site will be made. This estimate will be based on physical state and means of storage and use of the material.

The allowable inventory quantity of a chemical will be calculated from the allowable emission rate as follows:

$$\mathit{Allowable\ annual\ use}\ (kg) = \frac{\mathit{Allowable\ emission\ rate}\ (kg/yr)}{\mathit{Fraction\ of\ material\ released\ off\text{-}site}}$$

F.3 Drinking Water Exposure

This screening analysis will assume conservatively that East Fork Poplar Creek was a source of drinking water for the residents of Oak Ridge. It was assumed that a chemical from the Y-12 plant was released directly to East Fork Poplar Creek. For releases from K-25 the analysis will assume that the Clinch River was a source of drinking water for the people of Kingston and that a chemical from the K-25 plant was released directly to the Clinch River. The following discussion is based on releases from Y-12, a similar methodology will be used for releases from K-25.

The maximum allowable quantity of a chemical that can be present at the Y-12 plant without posing a potential drinking water ingestion hazard is determined in a similar manner as described above for the inhalation scenario. First, the maximum allowable dose is calculated from a defined level of risk (carcinogens) or exposure (non-carcinogens). Second, the corresponding drinking water concentration is determined using conservative exposure assumptions. Lastly, the allowable drinking water concentration is related to the maximum allowable inventory quantity. The following example illustrates each of the three steps.

As described above, the maximum allowable lifetime average daily dose for a carcinogen with a slope factor of 10 is equal to 1×10^{-7} mg/kg-day. The corresponding drinking water concentration can be determined using the following equation:

$$Dose = \frac{[Water] \times BR}{BW} \qquad \text{or} \qquad [Water] = \frac{Dose \times BW}{IR}$$

Where:

Dose = Maximum allowable lifetime average daily dose (mg/kg-day)
[Water] = Annual average drinking water concentration (mg/L)
IR = Water ingestion rate (L/day)
BW = Body weight (kg)

Assuming an allowable lifetime average daily dose of 1 × 10^{-7} mg/kg-day, an average adult body weight of 70 kg, and a water ingestion rate of 2 L/day (USEPA 1990a), the allowable annual average drinking water concentration is:

$$[Water] = \frac{1\times10^{-7}\ mg/kg-day \times 70\ kg}{1\ L/day}$$

$$= 7.0 \times 10^{-6}\ mg/L$$

As before, if the SF is decreased by a factor of 10, the corresponding allowable annual average drinking water concentration would be increased by a factor of 10. This relationship also applies to non-carcinogens. For a chemical with an RfD of 1 × 10^{-4} mg/kg-day, the allowable annual average drinking water concentration would be 7.0 × 10^{-3} mg/L.

The next step is to relate the allowable annual average drinking water concentration to the amount of chemical that must be released to East Fork Poplar Creek. According to research conducted for Task 2 of the Oak Ridge Dose Reconstruction, the average flow rate of East Fork Poplar Creek ranged between 8 and 11 million gallons per day. For this analysis the smaller value of 8 million gallons per day will be used. This is a conservative assumption, since for a given quantity of release a smaller volume of water would result in a higher water concentration. A volume of 8 million gallons per (30 million liters per day) would equal 1.1 × 10^{10} liters per year. If it is assumed that the annual quantity of a chemical released is fully mixed into East Fork Poplar Creek at one time, an upper limit on the annual average drinking water concentration can be determined from:

$$[Water] = \frac{Released\ Quantity}{Annual\ Volume\ of\ EFPC}$$

This results in the following estimate of the release quantity that will keep annual average drinking water concentration below the allowable level:

$$Released\ Quantity = [Water] \times Volume\ of\ EFPC$$

Assuming an allowable annual average drinking water concentration of 7.0 × 10^{-3} mg/L (see above), the corresponding released quantity equals:

$$Released\ Quantity = 7.0 \times 10^{-3}\ mg/L \times 1.1 \times 10^{10}\ L/yr$$

$$= 7.7 \times 10^{7}\ mg/yr\ \text{or}\ 77\ kg/yr$$

The final step is to relate the released quantity of a chemical to its inventory quantity. As described for the inhalation scenario, annual usage quantities will be gathered through plant records or professional judgement of knowledgeable plant personnel. Once an annual quantity is established, an estimate of the amount of the material that could have been released off-site will be made. This estimate will be based on physical state and means of storage and use of the material.

References

ChemRisk, 1991. “Task 2 Report Selection of the Chemicals and Radionuclides of Concern.” Prepared for the Rocky Flats Dose Reconstruction, Colorado Department of Health. June 1991.

Layton, 1987. Layton, D.W., B.J. Mallon, D.H. Rosenblatt, M.J. Small. “Deriving Allowable Daily Intakes for Systemic Toxicants Lacking Chronic Toxicity Data.” Regulatory Toxicology and Pharmacology 7: 96-112.

APPENDIX G

CALCULATION OF LEAD UPTAKE USING THE IEUBK MODEL

APPENDIX G

CALCULATION OF LEAD UPTAKE USING THE IEUBK MODEL

The following sections describe the assumptions used in the IEUBK model to calculate uptake of lead from air, drinking water/ milk, meat, vegetables, and fish.

G.1 Uptake of Lead from Air

Consistent with the Task 7 screening methodology, uptake of lead from air through inhalation was calculated assuming contributions from the following pathways:

- Air to Humans (direct inhalation)
- Soil to Air to Humans (inhalation of resuspended dust)

The IEUBK model and Task 7 screening default parameters for the air pathway, and the values selected for use in this assessment, are presented in Table G-1. The rationale for selection of the parameters used in this assessment follows.

Table G-1: Model Parameters for Calculating Lead Uptake from Air

IEUBK Model Parameter	IEUBK Default	Task 7 Screening Default		Lead Screening Assessment Value	
		Level I	Level II	Level I	Refined Level I
Air Concentration (mg m^{-3})	0.0001	NA	NA	0.0012	0.0012
Time Outdoors (h d^{-1})	1-4 (a)	19	19	19	19
Indoor Air Conc. (% of Outdoor)	30%	0%	0%	0%	0%
Inhalation Rate (m^3 d^{-1})	2-7 (b)	12	12	2-7 (b)	2-7 (b)
Lung Absorption	32%	100%	100%	32%	32%

NA Not applicable.

a Age-dependent. Ranges from 1 h d^{-1} for children age 0-1 y to 4 h d^{-1} for children age 6-7 y.

b Age-dependent. Ranges from 2 m^3 d^{-1} for children age 0-1 y to 7 m^3 d^{-1} for children age 6-7 y.

The IEUBK model estimates uptake of lead from air based on the 24-hour time-weighted average (TWA) air concentration for children in seven age groups ranging from 6 months to 7 years. Set to default, the model assumes that an individual is exposed to lead through inhalation for 24 hours per day, that an individual spends 20 hours per day indoors, and that the indoor air lead concentration is 30% of the outdoor air lead concentration. The model provides age-specific inhalation rates and assumes that 32% of the lead that is inhaled is absorbed into the bloodstream where it can contribute to the blood lead concentration. By contrast, the Task 7 default methodology assumes that a child is away from home and not exposed to the contaminant of concern for 5 hours per day, and that there is no difference between indoor and outdoor air concentrations. To be consistent with the Task 7 methodology, the project team assumed that an individual is exposed to lead for 19 hours per day and that the indoor and outdoor air concentrations are the same (effectively assuming that the individual is "outdoors" 19 hours per day and for the rest of the day receives no exposure).

The IEUBK model provides age-specific inhalation rates for a 6 month to 7 year old child ranging from 2 to 7 $m^3\ d^{-1}$. The Task 7 screening methodology default child inhalation rate is 12 $m^3\ d^{-1}$, based on the inhalation rate of a 10 year old child. Since the IEUBK model is designed to predict age-specific blood lead levels in children ages 6 months to 7 years, inhalation rates for these age groups were used.

The Task 7 default methodology does not include an absorption factor for inhalation (essentially assuming that 100% of an inhaled contaminant is absorbed). However, since the IEUBK model is designed to predict how much lead is absorbed into the blood rather than how much is taken into the body, the IEUBK default absorption factor of 32% was used in this assessment.

G.2 Uptake of Lead from Soil/Dust

The IEUBK model and Task 7 screening default parameters for the soil/dust ingestion pathways, and the values selected for use in this assessment, are presented in Table G-2. The rationale for selection of the parameters used in this assessment follows.

Table G-2: Model Parameters for Calculating Lead Uptake from Soil/Dust

IEUBK Model Parameter	IEUBK Default	Task 7 Screening Default		Lead Screening Assessment Value	
		Level I	Level II	Level I	Refined Level I
Soil Concentration (mg kg^{-1})	200	NA	NA	200	200
Dust Concentration (mg kg^{-1})	200	NA	NA	200	200
Soil/ Dust Ingestion Weighting Factor (percent soil)	45%	ND	ND	45%	45%
Soil/Dust Ingestion Rate (g d^{-1})	0.085-0.135 (a)	0.18	0.18	0.085-0.135 (a)	0.085-0.135 (a)
GI Absorption (%)	30%	100%	100%	30%	30%

NA Not applicable

ND Not determined

a Age-dependent. Ranges from a minimum of 0.085 g d^{-1} for children age 0-1 y and age 6-7 y to a maximum of 0.135 g d^{-1} for children age 1-4 y.

The IEUBK model assumes lead uptake from ingestion of both soil and dust. The model allows input of different lead concentrations for soil and dust; for the purpose of this assessment, the same concentration (200 mg kg^{-1}, based on data from the EFPC Floodplain RI) was used for both. The IEUBK model provides age-specific soil/dust ingestion rates for children ages 6 months to 7 years ranging from 0.085 to 0.135 g d^{-1}. By contrast, the Task 7 default screening methodology assumes a child soil ingestion rate of 0.25 g d^{-1} and that 70% of soil ingested is contaminated (equivalent to ingestion of 0.18 g d^{-1} contaminated soil). Since the IEUBK model is designed to predict age-specific blood lead concentrations in children ages 6 months to 7 years, soil/dust ingestion rates for these age groups were used.

The IEUBK model assumes, at default, that the dust ingestion rate is 45% of the soil ingestion rate. Since, in this assessment, it was assumed that there was no difference between lead concentrations in soil and dust, this fraction was not adjusted.

The IEUBK model assumes that 30% of the lead that is ingested in soil/dust is absorbed into the bloodstream where it can contribute to the blood lead concentration. The Task 7 default methodology does not include a gastrointestinal absorption factor for ingestion of lead in soil (essentially assuming that 100% of an ingested contaminant is absorbed). However, since the IEUBK model is designed to predict how much lead is absorbed into the blood rather than how much is taken into the body, the IEUBK default absorption factor of 30% was used in this assessment.

G.3 Uptake of Lead from Drinking Water/ Milk

Set to default, the IEUBK model calculates lead uptake from ingestion of tap water but does not consider uptake from contaminated milk. For purposes of this evaluation, it was assumed that total fluid ingestion was comprised of 57% water and 43% milk, consistent with Task 7 Level I methodology assumptions for intake of water and milk by children. Further, per the Task 7 methodology, 50% of drinking water and 100% of milk consumed was assumed to be contaminated. Therefore, the average lead concentration in the total fluid intake was calculated as follows:

$$\text{Avg. fluid conc. } (mg\ L^{\&1})\ '\ 0.57\ (0.5\ (\ \text{Contam. water conc.}\ \%\ 0.5\ (\ \text{Uncontam. water conc.})\ \%\ 0.43\ (\ \text{Milk conc.}$$

As described in Section 4.2.2.2.2, the concentration of lead in contaminated water was assumed to be 0.2 mg L^{-1}. The background concentration of lead in uncontaminated drinking water was assumed to be 0.004 mg L^{-1} (IEUBK default). Consistent with the Task 7 screening methodology, the concentration of lead in milk from backyard cows was calculated by summing the contributions from the following pathways:

C Air to Dairy Cattle to Milk
C Air to Pasture to Dairy Cattle to Milk
C Water to Dairy Cattle to Milk
C Soil to Dairy Cattle to Milk
C Soil to Pasture to Dairy Cattle to Milk

Lead concentrations in milk were calculated using Task 7 screening methodology default parameters (ChemRisk 1996), exposure point concentrations for lead in air, water, and soil described in Section 4.2.2, and the biotransfer-to-milk factor described in Section 4.2.2.2.5. The equations and parameters used to calculate lead concentrations in milk are summarized in the attached spreadsheet (Table G-5). Because the Task 7 screening methodology presents different values for some parameters used to estimate milk concentrations (e.g., ingestion rate of soil by cattle, ingestion rate of feed by cattle, fraction of feed that is contaminated) for the Level I and Refined Level I screens, different milk concentrations were estimated for the two levels. Based on these calculations, the concentrations of lead in milk for the Level I and Refined Level I screen were estimated to be 0.22 mg L^{-1} and 0.040 mg L^{-1}, respectively. The concentrations of lead in total fluids for the Level I and Refined Level I screens were estimated to be 0.16 mg L^{-1} and 0.071 mg L^{-1}, respectively.

The IEUBK model default parameters for the fluid ingestion pathway, and the values selected for use in this assessment, are presented in Table G-3. The rationale for selection of the parameters used in this assessment follows.

Table G-3: Model Parameters for Evaluating Lead Uptake from Water/Milk

IEUBK Model Parameter	IEUBK Default	Task 7 Screening Default		Lead Screening Assessment Value	
		Level I	Level II	Level I	Refined Level I
Fluid Concentration (mg L^{-1})	0.004	NA	NA	0.16	0.071
Total Fluid Ingestion Rate (L d^{-1})	0.20-0.59 (a)	2.3	1.5	1.2-1.6 (b)	0.7-1.1 (c)
GI Absorption (%)	50%	100%	100%	50%	50%

NA = Not applicable.

a Age-dependent, tap water ingestion only. Ranges from 0.20 L d^{-1} for children age 0-1 y to 0.59 L d^{-1} for children age 6-7 y.

b Age-dependent, water and milk ingestion. Ranges from 1.2 L d^{-1} for children age 0-1 y to 1.6 L d^{-1} for children age 6-7 y.

c Age-dependent, water and milk ingestion. Ranges from 0.7 L d^{-1} for children age 0-1 y to 1.1 L d^{-1} for children age 6-7 y.

At default, the IEUBK model assumes lead uptake from ingestion of tap water only. For purposes of this assessment, the lead concentration in total fluid intake accounted for ingestion of milk as well. The IEUBK model provides age-specific tap water ingestion rates for children ages 6 months to 7 years ranging from 0.20 to 0.59 L d^{-1}. The Task 7 Level I screening methodology assumes a child water ingestion rate of 1.3 L d^{-1}, based on water ingestion by children approximately 12 to 17 years old, and a child milk ingestion rate of 1.0 L d^{-1}. Since the IEUBK model is designed to predict age-specific blood lead levels in children ages 6 months to 7 years, for purposes of this assessment, the age-specific total fluid ingestion rate for the Level I Screen was assumed to be equal to the IEUBK default age-specific tap water ingestion rate (range 0.20 to 0.59 L d^{-1}) plus the Task 7 Level I screening methodology child milk ingestion rate (1.0 L d^{-1}). The age-specific total fluid ingestion rate for the Refined Level I Screen was assumed to be equal to the

IEUBK default age-specific tap water ingestion rate (range 0.20 to 0.59 L d^{-1}) plus the standard Task 7 Level II screening methodology child milk ingestion rate (0.5 L d^{-1}).

The IEUBK model assumes that 50% of the lead that is ingested in water is absorbed into the bloodstream where it can contribute to the blood lead concentration. The Task 7 default methodology does not include a gastrointestinal absorption factor for ingestion of lead in water or milk (essentially assuming that 100% of an ingested contaminant is absorbed). However, since the IEUBK model is designed to predict how much lead is absorbed into the blood rather than how much is taken into the body, the IEUBK default absorption factor of 50% was used in this assessment.

G.4 Uptake of Lead from Diet

Set to default, the IEUBK model calculates lead uptake from ingestion of "baseline" diet. The model does not allow the user to adjust the intake rates of various dietary components, such as vegetables, meat, or milk, and does not indicate what the assumed intake rates are. In its "alternate dietary intake" menu, however, the model does allow the user to specify the percentage of each food category that is comprised of home-grown or home-caught foods and to input lead concentrations in locally grown/caught fruits, vegetables, fish, and game.

For purposes of this evaluation, Task 7 screening methodology assumptions were used estimate the percentage of each food category that is comprised of home-grown or home-caught foods, and to calculate lead concentrations in home-grown or caught vegetables, fish, and meat.

G.4.1 Lead Concentration in Fruit/Vegetables

In its "alternate dietary intake" menu, the IEUBK model allows the user to input lead concentrations in homegrown fruits and homegrown vegetables. Per the Task 7 screening methodology, for Level I, the "vegetable" ingestion rate was assumed to include both fruits and vegetables and for Refined Level I, the "vegetable" ingestion rate was assumed to include only vegetables. Therefore, for the Level I IEUBK model calculations, the calculated concentration in "vegetables" was input into both the fruit and vegetable concentration fields. For Refined Level I, the calculated vegetable concentration was input into the vegetable concentration field only.

Consistent with the Task 7 screening methodology, concentrations of lead in vegetables from home gardens were calculated by summing the contributions from the following pathways:

- Air to Vegetables
- Soil to Vegetables

Lead concentrations in vegetables were calculated using Task 7 screening methodology default parameters (ChemRisk 1996), exposure point concentrations for lead in air and soil described in Section 4.2.2, and the deposition and plant uptake from soil factors described in Section 4.2.2.2.5. The equations and parameters used to calculate lead concentrations in vegetables are summarized in the attached spreadsheet (Table G-5). Because the Task 7 screening methodology presents different values for some parameters used to estimate vegetable concentrations (e.g., deposition velocity, fraction of contaminant remaining after washing) for the Level I and Refined Level I screens, different vegetable concentrations were estimated for the two levels. Based on these assumptions, the calculated lead concentrations in homegrown vegetables for the Level I and Refined Level I screening were 6.6 mg kg^{-1} and 2.5 mg kg^{-1}, respectively. These concentrations represent the concentration of lead in backyard vegetables only, *not* the average of lead concentrations in all consumed vegetables (e.g., backyard + store-bought vegetables).

G.4.2 Lead Concentration in Meat

In its "alternate dietary intake menu", the IEUBK model allows the user to input lead concentrations in locally caught "game animals from hunting." For purposes of the screening assessment, uptake from backyard beef cattle or other locally-raised domestic meat was evaluated by inputting calculated concentrations in meat in the "game animals from hunting" field.

Consistent with the Task 7 screening methodology, the concentration of lead in meat from backyard beef cattle or other locally-raised domestic meat was calculated by summing the contributions from the following pathways:

- Air to Livestock/Game to Meat
- Air to Pasture to Livestock/Game to Meat
- Water to Livestock/Game to Meat
- Soil to Livestock/Game to Meat
- Soil to Pasture to Livestock/Game to Meat

Lead concentrations in meat were calculated using Task 7 screening methodology default parameters (ChemRisk 1996), exposure point concentrations for lead in air and soil described in Section 4.2.2, and the deposition, pasture uptake from soil, and biotransfer factors described in Section 4.2.2.2.5. The equations and parameters used to calculate lead concentrations in meat are summarized in the attached spreadsheet (Table G-5). Because the Task 7 screening methodology presents different values for some parameters used to estimate meat concentrations (e.g., ingestion rate of soil by cattle, ingestion rate of feed by cattle, fraction of feed that is contaminated) for the Level I and Refined Level I screens, different vegetable concentrations were estimated for the two levels. Based on these assumptions, the calculated lead concentrations in meat from backyard beef cattle or other locally-raised domestic meat for the Level I and Refined Level I screen were 0.23 mg kg^{-1} and 0.050 mg kg^{-1}, respectively. These concentrations represent the concentration of lead in backyard/ locally-raised domestic meat only, *not* the average of lead concentrations in all consumed meat (e.g., backyard + store bought meat).

G.4.3 Lead Concentration in Fish

Consistent with the Task 7 screening methodology, the concentration of lead in locally-caught fish through uptake from water was calculated. Lead concentrations in fish were calculated using the exposure point concentration for lead in water described in Section 4.2.2.2.2 and the fish bioconcentration factor described in Section 4.2.2.2.5. Based on these assumptions, the calculated lead concentration in fish for the Level I and Refined Level I screen was 9.8 mg kg^{-1}. This concentration represents the concentration of lead in locally-caught fish only, *not* the average of lead concentrations in all consumed fish (e.g., locally-caught + fish caught out of the area).

G.4.4 IEUBK Model Input Parameters

The IEUBK model and Task 7 screening default parameters for the diet pathway, and the values selected for use in this assessment, are presented in Table G-4. The rationale for selection of the parameters used in this assessment (other than concentration) follows.

Table G-4: Model Parameters for Calculating Lead Uptake from Diet

IEUBK Model Parameter	**IEUBK Default**	**Task 7 Screening Default**		**Lead Screening Assessment Value**	
		Level I	**Level II**	**Level I**	**Refined Level I**
Baseline Diet Intake (µg d^{-1})	5.53-7.00 (a)	NA	NA	5.53-7.00 (a)	5.53-7.00 (b)
Homegrown Fruits					
Concentration (mg kg^{-1})	0	NA	NA	6.6	0
% of all fruits	0	60%	0%	60%	0%
Homegrown Vegetables					
Concentration (mg kg^{-1})	0	NA	NA	6.6	2.5
% of all vegetables	0	60%	20%	60%	20%
Game Animals from Hunting					
Concentration (mg kg^{-1})	0	NA	NA	0.23	0.050
% of all meats	0	60%	23%	60%	23%

IEUBK Model Parameter	**IEUBK Default**	**Task 7 Screening Default**		**Lead Screening Assessment Value**	
		Level I	**Level II**	**Level I**	**Refined Level I**
Fish from Recreational Fishing					
Concentration (mg kg^{-1})	0	NA	NA	9.8	9.8
% of all meats	0	20%	5%	20%	5%
GI Absorption (%)	50%	100%	100%	50%	50%

NA Not applicable.

Consistent with Task 7 screening methodology parameters for these pathways, it was assumed that 60% of all fruits/vegetables consumed are contaminated for the Level I screen and 20% are contaminated for the Refined Level I screen. For the meat ingestion pathway, it was assumed that 75% of all meat consumed is beef/game (calculated by dividing the Level I/ Refined Level I beef ingestion rates by the beef + fish ingestion rates) and that 25% is fish. Consistent with Task 7 screening methodology parameters, it was assumed that 80% of all beef/game consumed is contaminated for the Level I screen and 30% is contaminated for the Refined Level I screen. Thus, the resulting “% of all meats” input into the IEUBK model for the “Game Animals from Hunting” pathway was 60% for the Level I screen and 23% for the Refined Level I screen. For the fish ingestion pathways, it was assumed that 80% of all fish consumed is contaminated for the Level I screen and 20% is contaminated for the Refined Level I screen. Thus, the resulting “% of all meats” input into the IEUBK model for the “Fish from Fishing” pathway was 20% for the Level I screen and 5% for the Refined Level I screen.

The IEUBK model assumes that 50% of the lead that is ingested in the diet is absorbed into the bloodstream where it can contribute to the blood lead concentration. The Task 7 default methodology does not include a gastrointestinal absorption factor for ingestion of vegetables or meat (essentially assuming that 100% of an ingested contaminant is absorbed). However, since the IEUBK model is designed to predict how much lead is absorbed into the blood, the IEUBK default absorption factor of 50% was used in this assessment.

Table G-5: Modeling of Lead Transfer from Soil/Air/Water at Y-12 to Meat, Milk, Vegetables, and Fish

Parameters

		Units	Level I Value	Level II Value	
Cair	Concentration of lead in air	mg/m3	0.0012	0.0012	
Csoil	Concentration of lead in soil	mg/kg	200	200	
Cwater	Concentration of lead in surface water	mg/L	0.2	0.2	
Bveg	Soil to vegetable transfer factor	(mg/kg-plant (wet)/mg/kg-soil (dry))	0.009	0.009	0.045 from Baes 1984 * 0.2 dry/wet wt conv. factor
Bpast	Soil to pasture transfer factor	(mg/kg-plant (dry)/mg/kg-soil (dry))	0.045	0.045	Baes 1984
Fm	Cattle intake-to-milk biotransfer factor	(mg/L)/(mg/d)	0.00025	0.00025	Baes 1984; Ng 1977
Ff	Cattle intake-to-meat biotransfer factor	(mg/kg)/(mg/d)	0.0003	0.0003	Baes 1984
BCF	Bioconcentration factor for fish	(mg/kg)/(mg/L)	49	49	USEPA1986
Qair(b)	Inhalation rate of air by beef cattle	m3/d	122	122	Task 7 Screening Methodology
Qair(d)	Inhalation rate of air by dairy cattle	m3/d	150	150	Task 7 Screening Methodology
Qsoil	Ingestion rate of soil by cattle	kg/d	0.5	0.25	Task 7 Screening Methodology
Qwater(b)	Ingestion rate of water by beef cattle	L/d	44	44	Task 7 Screening Methodology
Qwater(d)	Ingestion rate of water by dairy cattle	L/d	48	48	Task 7 Screening Methodology
Qfeed(b)	Ingestion rate of feed by beef cattle	kg(dry)/d	10	7.2	Task 7 Screening Methodology
Fpb	Fraction of feed ingested by beef cattle that is contaminated pasture	unitless	1	0.4	Task 7 Screening Methodology
Qfeed(d)	Ingestion rate of feed by dairy cattle	kg(dry)/d	16	9.1	Task 7 Screening Methodology
Fpd	Fraction of feed ingested by dairy cattle that is contaminated pasture	unitless	0.75	0.3	Task 7 Screening Methodology
Vd	Deposition velocity	m/d	1000	500	Task 7 Screening Methodology
alpha(past)	Mass interception factor for pasture	m2/kg-dry	3	3	Task 7 Screening Methodology
alpha(veg)	Mass interception factor for vegetables	m2/kg-wet	0.3	0.3	Task 7 Screening Methodology
kw	Weathering rate constant	d-1	0.05	0.05	Task 7 Screening Methodology
Tg(veg)	Growth period or exposure period for vegetables	d	60	60	Task 7 Screening Methodology
Tg(past)	Growth period or exposure period for pasture	d	30	30	Task 7 Screening Methodology
Umeat	Ingestion rate of meat by children	kg/d	0.15	0.03	Task 7 Screening Methodology
Fcb	Fraction of meat that is contaminated	unitless	0.8	0.3	Task 7 Screening Methodology
Umilk	Ingestion rate of milk by children	L/d	1	0.5	Task 7 Screening Methodology
Fcm	Fraction of milk that is contaminated	unitless	1	0.5	Task 7 Screening Methodology
Uveg	Ingestion rate of vegetables by children	kg/d (wet)	0.4	0.1	Task 7 Screening Methodology
Fcv	Fraction of vegetables that is contaminated	unitless	0.6	0.2	Task 7 Screening Methodology
Ufish	Ingestion rate of fish by children	kg/d	0.045	0.01	Task 7 Screening Methodology
Fcf	Fraction of fish consumed that is contaminated	unitless	0.8	0.2	Task 7 Screening Methodology

Equations

Concentration in Meat

*C(meat)air=Cair * Qair(b) * Ff*

Cair	Concentration of lead in air	mg/m3	0.0012	0.0012
Qair(b)	Inhalation rate of air by beef cattle	m3/d	122	122
Ff	Cattle intake-to-meat biotransfer factor	(mg/kg)/(mg/d)	0.0003	0.0003
C(meat)air		***mg/kg***	**0.000044**	**0.000044**

*C(meat)soil=Csoil * Qsoil(b) * Ff*

Csoil	Concentration of lead in soil	mg/kg	200	200
Qsoil	Ingestion rate of soil by cattle	kg/d	0.5	0.25
Ff	Cattle intake-to-meat biotransfer factor	(mg/kg)/(mg/d)	0.0003	0.0003
C(meat)soil		***mg/kg***	**0.030**	**0.015**

*C(meat)water=Cwater * Qwater(b) * Ff*

Cwater	Concentration of lead in surface water	mg/L	0.2	0.2
Qwater(b)	Ingestion rate of water by beef cattle	L/d	44	44
Ff	Cattle intake-to-meat biotransfer factor	(mg/kg)/(mg/d)	0.0003	0.0003
C(meat)water		***mg/kg***	**0.0026**	**0.0026**

*C(meat)soil-pasture= Csoil * Bpast * Qfeed(b) * Ff * Fpb*

Csoil	Concentration of lead in soil	mg/kg	200	200
Bpast	Soil to pasture transfer factor	(mg/kg-plant (dry)/mg/kg-soil (dry))	0.045	0.045
Qfeed(b)	Ingestion rate of feed by beef cattle	kg(dry)/d	10	7.2
Ff	Cattle intake-to-meat biotransfer factor	(mg/kg)/(mg/d)	0.0003	0.0003
Fpb	Fraction of feed ingested by beef cattle that is contaminated pasture	unitless	1	0.4
C(meat)soil-pasture		***mg/kg***	**0.027**	**0.0078**

*C(meat)air-pasture= Cair * Vd * alpha(past) * [(1-e^(-kwTg))/kw] * Qfeed(b) * Ff * fpb*

Cair	Concentration of lead in air	mg/m3	0.0012	0.0012
Vd	Deposition velocity	m/d	1000	500
alpha(past)	Mass interception factor for pasture	m2/kg-dry	3	3
kw	Weathering rate constant	d-1	0.05	0.05
Tg(past)	Growth period or exposure period for pasture	d	30	30
Qfeed(b)	Ingestion rate of feed by beef cattle	kg(dry)/d	10	7.2
Ff	Cattle intake-to-meat biotransfer factor	(mg/kg)/(mg/d)	0.0003	0.0003
Fpb	Fraction of feed ingested by beef cattle that is contaminated pasture	unitless	1	0.4
C(meat)air-pasture		***mg/kg***	**0.17**	**0.024**

Cmeat=C(meat)air + C(meat)soil + C(meat)water + C(meat)soil-pasture + C(meat)air-pasture

Cmeat		***mg/kg***	**0.23**	**0.05**

*Intake(Pb-meat, contaminated)= Cmeat * Umeat*

Cmeat	Concentration of lead in contaminated meat	mg/kg	0.23	0.050
Umeat	Ingestion rate of meat by children	kg/d	0.15	0.03
Fcb	Fraction of meat that is contaminated	unitless	0.8	0.3
Intake(Pb-meat, contaminated)		***mg/d***	***0.027***	***0.00045***

Concentration in Milk

*C(milk)air=Cair * Qair(d) * Fm*

Cair	Concentration of lead in air	mg/m3	0.0012	0.0012
Qair(d)	Inhalation rate of air by dairy cattle	m3/d	150	150
Fm	Cattle intake-to-milk biotransfer factor	(mg/L)/(mg/d)	0.00025	0.00025
C(milk)air		***mg/L***	**0.000045**	**0.000045**

*C(milk)soil=Csoil * Qsoil(d) * Fm*

Csoil	Concentration of lead in soil	mg/kg	200	200
Qsoil	Ingestion rate of soil by cattle	kg/d	0.5	0.25
Fm	Cattle intake-to-milk biotransfer factor	(mg/L)/(mg/d)	0.00025	0.00025
C(milk)soil		***mg/L***	**0.025**	**0.013**

*C(milk)water=Cwater * Qwater(d) * Fm*

Cwater	Concentration of lead in surface water	mg/L	0.2	0.2
Qwater(d)	Ingestion rate of water by dairy cattle	L/d	48	48
Fm	Cattle intake-to-milk biotransfer factor	(mg/L)/(mg/d)	0.00025	0.00025
C(milk)water		***mg/L***	**0.0024**	**0.0024**

*C(milk)soil-pasture= Csoil * Bpast * Qfeed(d) * Fm * Fpd*

Csoil	Concentration of lead in soil	mg/kg	200	200
Bpast	Soil to pasture transfer factor	(mg/kg-plant (dry)/mg/kg-soil (dry))	0.045	0.045
Qfeed(d)	Ingestion rate of feed by dairy cattle	kg(dry)/d	16	9.1
Fm	Cattle intake-to-milk biotransfer factor	(mg/L)/(mg/d)	0.00025	0.00025
Fpd	Fraction of feed ingested by dairy cattle that is contaminated pasture	unitless	0.75	0.3
C(milk)soil-pasture		***mg/L***	**0.027**	**0.0061**

*C(milk)air-pasture= Cair * Vd * alpha(past) * [(1-e^(-kwTg))/kw] * Qfeed(d) * Fm * fpd*

Cair	Concentration of lead in air	mg/m3	0.0012	0.0012
Vd	Deposition velocity	m/d	1000	500
alpha(past)	Mass interception factor for pasture	m2/kg-dry	3	3
kw	Weathering rate constant	d-1	0.05	0.05
Tg(past)	Growth period or exposure period for pasture	d	30	30
Qfeed(d)	Ingestion rate of feed by dairy cattle	kg(dry)/d	16	9.1
Fm	Cattle intake-to-milk biotransfer factor	(mg/L)/(mg/d)	0.00025	0.00025
Fpd	Fraction of feed ingested by dairy cattle that is contaminated pasture	unitless	0.75	0.3
C(milk)air-pasture		***mg/L***	**0.17**	**0.019**

Cmilk=C(milk)air + C(milk)soil + C(milk)water + C(milk)soil-pasture + C(milk)air-pasture

Cmilk		***mg/L***	**0.22**	**0.040**

*Intake(Pb-milk, contaminated)= Cmilk * Umilk * Fcm*

Cmilk	Concentration of lead in contaminated milk	mg/L	0.22	0.040
Umilk	Ingestion rate of milk by children	L/d	1	0.5
Fcm	Fraction of milk that is contaminated	unitless	1	0.5
Intake(Pb-milk, contaminated)		***mg/d***	***0.22***	***0.010***

Concentration in Vegetables

*C(veg)soil = Bveg * Csoil*

Bveg	Soil to vegetable transfer factor	(mg/kg-plant (wet)/mg/kg-soil (dry))	0.009	0.009
Csoil	Concentration of lead in soil	mg/kg	200	200
C(veg)soil			**1.8**	**1.8**

*C(veg)air= Cair * Vd * alpha(veg) * [(1-e^(-kwTg))/kw] * Fw*

Cair	Concentration of lead in air	mg/m3	0.0012	0.0012
Vd	Deposition velocity	m/d	1000	500
alpha(veg)	Mass interception factor for vegetables	m2/kg-wet	0.3	0.3
kw	Weathering rate constant	d-1	0.05	0.05
Tg(veg)	Growth period or exposure period for vegetables	d	60	60
Fw	Fraction of contaminant remaining after washing	unitless	0.7	0.2
C(veg)air		***mg/kg***	**4.8**	**0.68**

Cveg = C(veg)soil + C(veg)air		***mg/kg***	**6.6**	**2.5**

*Intake(Pb-veg, contaminated)= Cveg * Uveg * Fcv*

Cveg	Concentration of lead in contaminated vegetables	mg/kg	6.6	2.5
Uveg	Ingestion rate of vegetables by children	kg/d (wet)	0.4	0.1
Fcv	Fraction of vegetables that is contaminated	unitless	0.6	0.2
Intake(Pb-veg, contaminated)		***mg/d***	***1.6***	***0.050***

Concentration in Fish

*C(fish) = Cwater * BCF*

Cwater	Concentration of lead in surface water	mg/L	0.2	0.2
BCF	Bioconcentration factor for fish	(mg/kg)/(mg/L)	49	49
C(fish)		***mg/kg***	***9.8***	***9.8***

*Intake(Pb-fish, contaminated) - Cfish * Ufish * Fcf*

Cfish	Concentration of lead in contaminated fish	mg/kg	9.8	9.8
Ufish	Ingestion rate of fish by children	kg/d	0.045	0.01
Fcf	Fraction of fish consumed that is contaminated	unitless	0.8	0.2
Intake(Pb-fish, contaminated)		***mg/d***	***0.35***	***0.020***

APPENDIX H

INTERNATIONAL ATOMIC ENERGY AGENCY (IAEA) TRITIUM DOSE ASSESSMENT METHOD

APPENDIX H

INTERNATIONAL ATOMIC ENERGY AGENCY (IAEA) TRITIUM DOSE ASSESSMENT METHOD

Since tritium can be incorporated into a great variety of different chemical compounds within the human body, the radiological dose from releases of these radionuclides is best assessed using models that employ a specific activity approach. These models are based on the assumption that a steady-state equilibrium has been attained between the environment and the exposed individuals so that the ration between the radionuclide and its stable counterpart is fixed. Such models are considered to give conservative dose estimates when an individual is assumed to be in complete equilibrium with maximum levels of environmental specific activity of tritium. However, more advanced models have been developed to assess exposure under non-equilibrium conditions and to include the special behavior of tritium in organic compounds. They indicate that taking account of incorporation into organic compounds can lead to estimated doses that are higher than those obtained using the specific activity model. For simplicity, and taking account of other conservatisms inherent in the dose calculation, the specific activity approach is adopted here. More detailed models may be required if estimated doses are within a factor of 10 of the required limit or constraint.

The specific activity model for tritium assumes that the nuclide is transferred through the environment and incorporated into the organism through its association with water molecules and that the concentration of tritium in humans is derived from equilibrium concentration of tritium in:

(a) The water vapor of the atmosphere receiving the airborne effluents, and

(b) The water of aquatic environments receiving liquid effluents:

$$D_T^{\max} = [(C_A)_m^{\max} \times (f_A)_m + (C_W)_n^{\max} \times (f_W)_n]\, g$$

Where:

$D_t^{\max}$	is the dose rate (Sv y^{-1}) for tritium to the total body of a maximally exposed individual;
g	is the dose rate conversion factor (Sv y^{-1} per Bq L^{-1} of human body water content);
$(C_A)_m^{\max}$	is the steady state concentration of tritium in atmospheric water vapor (Bq L^{-1}) at location *m* from airborne releases (this is the atmospheric concentration assumed to contribute most significantly to the dose received by the potential maximally exposed individual).

The concentration of tritium atmospheric water vapor at location *m* is calculated as:

$$(C_A)_m^{\max} = \frac{(X)_m^{\max}}{(H)_m}$$

Where:	X_m^{max}	is the concentration of tritium in air at location *m* (Bq m^{-3}) resulting from a release to the atmosphere;
	$(H)_m$	is the absolute humidity of the atmosphere, assumed as a default value to be 6×10^{-3} L m^{-3} of air or 6×10^{-3} kg m^{-3};
	$(f_A)_m$	is the fraction of total water intake for a potentially exposed person that is derived from atmospheric water vapor at location *m* (this fraction includes water absorbed through the skin and through inhalation, as well as water vapor included in the formation of rain at location *m* that is incorporated into foods and drinking water and other liquids produced at location *m* and subsequently consumed by the individual);
	$(C_W)_n^{max}$	is the steady state concentration of tritium in water at location *n* resulting from releases to the aquatic environment (Bq L^{-1}) (this is the concentration in the aquatic environment assumed to contribute most significantly to the dose received by the potential maximally exposed individual).
	$(f_W)_n$	is the fraction of the total water intake of the potentially exposed person that is derived from water at location *n* that has been contaminated with aquatic discharges of tritiated water (this fraction includes the consumption of water in foods that have been irrigated with location *n* water as well as drinking water and other water-based beverages derived from this location).

For the parameters $(f_A)_m$ and $(f_W)_n$, default values of 1.0 were used. The value of 1.0 results in a sufficiently continuous estimate of dose rates, given accurate estimates of the maximum specific activity of tritium at locations *m* and *n*.

The error introduced by assuming that the concentration of tritium in the body including its organic molecules equals that of water can be neglected in these calculations. Doses to the body can be obtained by multiplying the activity levels by the appropriate dose rate factor *g*. This relates the dose rate for the total body of an exposed individual to the concentration of tritium per liter of water in the body. For tritium the dose-rate factor is 2.6×10^{-8} Sv y^{-1} per Bq L^{-1}.

APPENDIX I

DOSE SUMMARY TABLES FOR MATERIALS SCREENED USING THE STANDARD TASK 7 METHOD

DOSE SUMMARY TABLE - LEVEL I SCREEN *ARSENIC* *K-25*

	DOSE-Car mg / (kg-d)	% of Total	DOSE-Non-C mg / (kg-d)	% of Total
INHALATION PATHWAYS				
Air to Humans (Inhalation)	9.62E-06	97.1%	1.35E-05	97.1%
Soil (Particulates) to Air to Humans (Inhalation)	2.92E-07	2.9%	4.09E-07	2.9%
TOTAL INHALATION DOSE	**9.91E-06**		**1.39E-05**	
INGESTION PATHWAYS				
Air to Livestock/Game (Beef) to Humans (ingestion)	7.04E-07	0.0%	9.86E-07	0.0%
Air to Dairy Cattle (Milk) to Humans (ingestion)	1.80E-08	0.0%	2.53E-08	0.0%
Air to Vegetables to Humans (Ingestion)	1.44E-03	5.7%	2.02E-03	5.7%
Air to Pasture to Livestock/Game (Beef) to Humans (Ingestion)	2.69E-03	10.6%	3.77E-03	10.6%
Air to Pasture to Dairy Cattle (Milk) to Humans (Ingestion)	6.73E-05	0.3%	9.42E-05	0.3%
Water to Humans (Incidental Ingestion)	8.98E-07	0.0%	1.26E-06	0.0%
Water to Livestock/Game (Beef) to Humans (Ingestion)	4.31E-05	0.2%	6.03E-05	0.2%
Water to Dairy Cattle (Milk) to Humans (Ingestion)	9.80E-07	0.0%	1.37E-06	0.0%
Water to Fish to Humans (Ingestion)	8.16E-04	3.2%	1.14E-03	3.2%
Soil to Humans (Ingestion)	3.93E-05	0.2%	5.50E-05	0.2%
Soil to Livestock/Game (Beef) to Humans (Ingestion)	1.35E-03	5.3%	1.89E-03	5.3%
Soil to Dairy Cattle (Milk) to Humans	2.81E-05	0.1%	3.93E-05	0.1%
Soil to Vegetables to Humans (Ingestion)	1.35E-02	52.9%	1.89E-02	52.9%
Soil to Pasture to Livestock/Game (Beef) to Humans (Ingestion)	5.39E-03	21.2%	7.54E-03	21.2%
Soil to Pasture to Dairy Cattle (Milk) to Humans (Ingestion)	1.35E-04	0.5%	1.89E-04	0.5%
TOTAL INGESTION DOSE	**2.55E-02**		**3.57E-02**	

DOSE SUMMARY TABLE - REFINED LEVEL I SCREEN ARSENIC K-25

	DOSE-Car mg / (kg-d)	**% of Total**	**DOSE-Non-C** mg / (kg-d)	**% of Total**
INHALATION PATHWAYS				
Air to Humans (Inhalation)	1.85E-06	97.1%	1.29E-05	97.1%
Soil (Particulates) to Air to Humans (Inhalation)	5.60E-08	2.9%	3.92E-07	2.9%
TOTAL INHALATION DOSE	**1.90E-06**		**1.33E-05**	
INGESTION PATHWAYS				
Air to Livestock/Game (Beef) to Humans (ingestion)	1.69E-08	0.0%	1.18E-07	0.0%
Air to Dairy Cattle (Milk) to Humans (ingestion)	5.20E-10	0.0%	3.64E-09	0.0%
Air to Vegetables to Humans (Ingestion)	5.27E-06	0.9%	3.69E-05	0.9%
Air to Pasture to Livestock/Game (Beef) to Humans (Ingestion)	9.30E-06	1.6%	6.51E-05	1.6%
Air to Pasture to Dairy Cattle (Milk) to Humans (Ingestion)	2.20E-07	0.0%	1.54E-06	0.0%
Water to Humans (Incidental Ingestion)	1.72E-07	0.0%	1.21E-06	0.0%
Water to Livestock/Game (Beef) to Humans (Ingestion)	1.03E-06	0.2%	7.23E-06	0.2%
Water to Dairy Cattle (Milk) to Humans (Ingestion)	2.82E-08	0.0%	1.97E-07	0.0%
Water to Fish to Humans (Ingestion)	1.57E-04	27.3%	1.10E-03	27.3%
Soil to Humans (Ingestion)	1.35E-06	0.2%	9.42E-06	0.2%
Soil to Livestock/Game (Beef) to Humans (Ingestion)	1.61E-05	2.8%	1.13E-04	2.8%
Soil to Dairy Cattle (Milk) to Humans	4.04E-07	0.1%	2.83E-06	0.1%
Soil to Vegetables to Humans (Ingestion)	3.44E-04	60.1%	2.41E-03	60.1%
Soil to Pasture to Livestock/Game (Beef) to Humans (Ingestion)	3.72E-05	6.5%	2.60E-04	6.5%
Soil to Pasture to Dairy Cattle (Milk) to Humans (Ingestion)	8.82E-07	0.2%	6.17E-06	0.2%
TOTAL INGESTION DOSE	**5.73E-04**		**4.01E-03**	

DOSE SUMMARY TABLE - LEVEL I SCREEN ARSENIC Y-12

	DOSE-Car mg / (kg-d)	% of Total	DOSE-Non-C mg / (kg-d)	% of Total
INHALATION PATHWAYS				
Air to Humans (Inhalation)	1.35E-06	92.0%	1.89E-06	92.0%
Soil (Particulates) to Air to Humans (Inhalation)	1.17E-07	8.0%	1.63E-07	8.0%
TOTAL INHALATION DOSE	**1.47E-06**		**2.05E-06**	
INGESTION PATHWAYS				
Air to Livestock/Game (Beef) to Humans (ingestion)	9.87E-08	0.0%	9.86E-07	0.0%
Air to Dairy Cattle (Milk) to Humans (ingestion)	1.80E-08	0.0%	1.38E-07	0.0%
Air to Vegetables to Humans (Ingestion)	1.44E-03	11.6%	3.54E-09	0.0%
Air to Pasture to Livestock/Game (Beef) to Humans (Ingestion)	2.69E-03	21.7%	2.83E-04	2.3%
Air to Pasture to Dairy Cattle (Milk) to Humans (Ingestion)	6.73E-05	0.5%	5.28E-04	4.3%
Water to Humans (Incidental Ingestion)	6.73E-09	0.0%	9.43E-09	0.0%
Water to Livestock/Game (Beef) to Humans (Ingestion)	5.17E-06	0.0%	7.24E-06	0.1%
Water to Dairy Cattle (Milk) to Humans (Ingestion)	1.18E-07	0.0%	1.65E-07	0.0%
Water to Fish to Humans (Ingestion)	3.92E-05	0.3%	5.49E-05	0.4%
Soil to Humans (Ingestion)	1.57E-05	0.1%	2.20E-05	0.2%
Soil to Livestock/Game (Beef) to Humans (Ingestion)	5.39E-04	4.3%	7.54E-04	6.1%
Soil to Dairy Cattle (Milk) to Humans	1.12E-05	0.1%	1.57E-05	0.1%
Soil to Vegetables to Humans (Ingestion)	5.39E-03	43.4%	7.54E-03	61.3%
Soil to Pasture to Livestock/Game (Beef) to Humans (Ingestion)	2.16E-03	17.4%	3.02E-03	24.5%
Soil to Pasture to Dairy Cattle (Milk) to Humans (Ingestion)	5.39E-05	0.4%	7.54E-05	0.6%
TOTAL INGESTION DOSE	**1.24E-02**		**1.23E-02**	

DOSE SUMMARY TABLE - REFINED LEVEL I SCREEN *ARSENIC* *Y-12*

	DOSE-Car mg / (kg-d)	% of Total	DOSE-Non-C mg / (kg-d)	% of Total
INHALATION PATHWAYS				
Air to Humans (Inhalation)	7.51E-08	91.8%	5.26E-07	91.8%
Soil (Particulates) to Air to Humans (Inhalation)	6.72E-09	8.2%	4.70E-08	8.2%
TOTAL INHALATION DOSE	**8.19E-08**		**5.73E-07**	
INGESTION PATHWAYS				
Air to Livestock/Game (Beef) to Humans (ingestion)	2.29E-09	0.0%	1.60E-08	0.0%
Air to Dairy Cattle (Milk) to Humans (ingestion)	7.05E-11	0.0%	4.93E-10	0.0%
Air to Vegetables to Humans (Ingestion)	7.14E-07	0.4%	5.00E-06	0.4%
Air to Pasture to Livestock/Game (Beef) to Humans (Ingestion)	1.26E-06	0.7%	8.83E-06	0.7%
Air to Pasture to Dairy Cattle (Milk) to Humans (Ingestion)	2.99E-08	0.0%	2.09E-07	0.0%
Water to Humans (Incidental Ingestion)	1.29E-09	0.0%	9.04E-09	0.0%
Water to Livestock/Game (Beef) to Humans (Ingestion)	1.24E-07	0.1%	8.68E-07	0.1%
Water to Dairy Cattle (Milk) to Humans (Ingestion)	3.38E-09	0.0%	2.37E-08	0.0%
Water to Fish to Humans (Ingestion)	7.51E-06	4.4%	5.26E-05	4.4%
Soil to Humans (Ingestion)	5.38E-07	0.3%	3.77E-06	0.3%
Soil to Livestock/Game (Beef) to Humans (Ingestion)	6.46E-06	3.8%	4.52E-05	3.8%
Soil to Dairy Cattle (Milk) to Humans	1.61E-07	0.1%	1.13E-06	0.1%
Soil to Vegetables to Humans (Ingestion)	1.38E-04	81.1%	9.64E-04	81.1%
Soil to Pasture to Livestock/Game (Beef) to Humans (Ingestion)	1.49E-05	8.8%	1.04E-04	8.8%
Soil to Pasture to Dairy Cattle (Milk) to Humans (Ingestion)	3.53E-07	0.2%	2.47E-06	0.2%
TOTAL INGESTION DOSE	**1.70E-04**		**1.19E-03**	

DOSE SUMMARY TABLE - LEVEL I SCREEN *BERYLLIUM* *Y-12*

	DOSE-Car mg / (kg-d)	% of Total	DOSE-Non-C mg / (kg-d)	% of Total
INHALATION PATHWAYS				
Air to Humans (Inhalation)	3.86E-12	0.0%	5.41E-12	0.0%
Soil (Particulates) to Air to Humans (Inhalation)	1.43E-08	100.0%	2.01E-08	100.0%
TOTAL INHALATION DOSE	**1.43E-08**		**2.01E-08**	
INGESTION PATHWAYS				
Air to Livestock/Game (Beef) to Humans (ingestion)	7.07E-14	0.0%	9.89E-14	0.0%
Air to Dairy Cattle (Milk) to Humans (ingestion)	1.45E-16	0.0%	2.03E-16	0.0%
Air to Vegetables to Humans (Ingestion)	5.78E-10	0.0%	8.09E-10	0.0%
Air to Pasture to Livestock/Game (Beef) to Humans (Ingestion)	2.70E-10	0.0%	3.78E-10	0.0%
Air to Pasture to Dairy Cattle (Milk) to Humans (Ingestion)	5.40E-13	0.0%	7.56E-13	0.0%
Water to Humans (Incidental Ingestion)	5.61E-09	0.0%	7.86E-09	0.0%
Water to Livestock/Game (Beef) to Humans (Ingestion)	1.08E-06	1.1%	1.51E-06	1.1%
Water to Dairy Cattle (Milk) to Humans (Ingestion)	1.96E-09	0.0%	2.74E-09	0.0%
Water to Fish to Humans (Ingestion)	8.16E-06	8.7%	1.14E-05	8.7%
Soil to Humans (Ingestion)	1.93E-06	2.1%	2.70E-06	2.1%
Soil to Livestock/Game (Beef) to Humans (Ingestion)	1.65E-05	17.6%	2.31E-05	17.6%
Soil to Dairy Cattle (Milk) to Humans	2.76E-08	0.0%	3.86E-08	0.0%
Soil to Vegetables to Humans (Ingestion)	3.31E-05	35.2%	4.63E-05	35.2%
Soil to Pasture to Livestock/Game (Beef) to Humans (Ingestion)	3.31E-05	35.2%	4.63E-05	35.2%
Soil to Pasture to Dairy Cattle (Milk) to Humans (Ingestion)	6.61E-08	0.1%	9.26E-08	0.1%
TOTAL INGESTION DOSE	**9.39E-05**		**1.31E-04**	

DOSE SUMMARY TABLE - REFINED LEVEL I SCREEN BERYLLIUM Y-12

	DOSE-Car mg / (kg-d)	% of Total	DOSE-Non-C mg / (kg-d)	% of Total
INHALATION PATHWAYS				
Air to Humans (Inhalation)	2.22E-13	0.0%	1.56E-12	0.0%
Soil (Particulates) to Air to Humans (Inhalation)	8.24E-10	100.0%	5.77E-09	100.0%
TOTAL INHALATION DOSE	**8.24E-10**		**5.77E-09**	
INGESTION PATHWAYS				
Air to Livestock/Game (Beef) to Humans (ingestion)	1.69E-15	0.0%	1.19E-14	0.0%
Air to Dairy Cattle (Milk) to Humans (ingestion)	4.17E-18	0.0%	2.92E-17	0.0%
Air to Vegetables to Humans (Ingestion)	2.11E-12	0.0%	1.48E-11	0.0%
Air to Pasture to Livestock/Game (Beef) to Humans (Ingestion)	9.32E-13	0.0%	6.52E-12	0.0%
Air to Pasture to Dairy Cattle (Milk) to Humans (Ingestion)	1.77E-15	0.0%	1.24E-14	0.0%
Water to Humans (Incidental Ingestion)	1.08E-09	0.0%	7.53E-09	0.0%
Water to Livestock/Game (Beef) to Humans (Ingestion)	2.58E-08	0.9%	1.81E-07	0.9%
Water to Dairy Cattle (Milk) to Humans (Ingestion)	5.64E-11	0.0%	3.95E-10	0.0%
Water to Fish to Humans (Ingestion)	1.57E-06	53.4%	1.10E-05	53.4%
Soil to Humans (Ingestion)	6.60E-08	2.3%	4.62E-07	2.3%
Soil to Livestock/Game (Beef) to Humans (Ingestion)	1.98E-07	6.8%	1.39E-06	6.8%
Soil to Dairy Cattle (Milk) to Humans	3.96E-10	0.0%	2.77E-09	0.0%
Soil to Vegetables to Humans (Ingestion)	8.45E-07	28.8%	5.92E-06	28.8%
Soil to Pasture to Livestock/Game (Beef) to Humans (Ingestion)	2.28E-07	7.8%	1.60E-06	7.8%
Soil to Pasture to Dairy Cattle (Milk) to Humans (Ingestion)	4.33E-10	0.0%	3.03E-09	0.0%
TOTAL INGESTION DOSE	**2.93E-06**		**2.05E-05**	

DOSE SUMMARY TABLE - LEVEL I SCREEN *CHROMIUM(VI)* *from the ORR*

	DOSE-Car mg / (kg-d)	% of Total	DOSE-Non-C mg / (kg-d)	% of Total
INHALATION PATHWAYS				
Air to Humans (Inhalation)	2.86E-06	92.1%	5.71E-06	92.1%
Soil (Particulates) to Air to Humans (Inhalation)	2.45E-07	7.9%	4.90E-07	7.9%
TOTAL INHALATION DOSE	**3.10E-06**		**6.20E-06**	
INGESTION PATHWAYS				
Air to Livestock/Game (Beef) to Humans (ingestion)	3.14E-07	0.0%	6.27E-07	0.0%
Air to Dairy Cattle (Milk) to Humans (ingestion)	1.07E-07	0.0%	2.14E-07	0.0%
Air to Vegetables to Humans (Ingestion)	4.28E-04	3.0%	8.55E-04	3.0%
Air to Pasture to Livestock/Game (Beef) to Humans (Ingestion)	1.20E-03	8.3%	2.40E-03	8.3%
Air to Pasture to Dairy Cattle (Milk) to Humans (Ingestion)	4.00E-04	2.8%	7.99E-04	2.8%
Water to Humans (Incidental Ingestion)	7.86E-06	0.1%	1.57E-05	0.1%
Water to Livestock/Game (Beef) to Humans (Ingestion)	5.66E-04	3.9%	1.13E-03	3.9%
Water to Dairy Cattle (Milk) to Humans (Ingestion)	1.71E-04	1.2%	3.43E-04	1.2%
Water to Fish to Humans (Ingestion)	3.57E-03	24.7%	7.14E-03	24.7%
Soil to Humans (Ingestion)	3.30E-05	0.2%	6.60E-05	0.2%
Soil to Livestock/Game (Beef) to Humans (Ingestion)	1.70E-03	11.7%	3.39E-03	11.7%
Soil to Dairy Cattle (Milk) to Humans	4.71E-04	3.3%	9.43E-04	3.3%
Soil to Vegetables to Humans (Ingestion)	1.41E-03	9.8%	2.83E-03	9.8%
Soil to Pasture to Livestock/Game (Beef) to Humans (Ingestion)	3.39E-03	23.4%	6.79E-03	23.4%
Soil to Pasture to Dairy Cattle (Milk) to Humans (Ingestion)	1.13E-03	7.8%	2.26E-03	7.8%
TOTAL INGESTION DOSE	**1.45E-02**		**2.90E-02**	

DOSE SUMMARY TABLE - REFINED LEVEL I SCREEN *CHROMIUM(VI)* *from the ORR*

	DOSE-Car mg / (kg-d)	% of Total	DOSE-Non-C mg / (kg-d)	% of Total
INHALATION PATHWAYS				
Air to Humans (Inhalation)	2.35E-07	98.6%	1.64E-06	98.6%
Soil (Particulates) to Air to Humans (Inhalation)	3.36E-09	1.4%	2.35E-08	1.4%
TOTAL INHALATION DOSE	**2.38E-07**		**1.67E-06**	
INGESTION PATHWAYS				
Air to Livestock/Game (Beef) to Humans (ingestion)	1.07E-08	0.0%	7.52E-08	0.0%
Air to Dairy Cattle (Milk) to Humans (ingestion)	4.40E-09	0.0%	3.08E-08	0.0%
Air to Vegetables to Humans (Ingestion)	2.23E-06	1.0%	1.56E-05	1.0%
Air to Pasture to Livestock/Game (Beef) to Humans (Ingestion)	5.91E-06	2.5%	4.14E-05	2.5%
Air to Pasture to Dairy Cattle (Milk) to Humans (Ingestion)	1.87E-06	0.8%	1.31E-05	0.8%
Water to Humans (Incidental Ingestion)	4.31E-07	0.2%	3.01E-06	0.2%
Water to Livestock/Game (Beef) to Humans (Ingestion)	3.87E-06	1.7%	2.71E-05	1.7%
Water to Dairy Cattle (Milk) to Humans (Ingestion)	1.41E-06	0.6%	9.86E-06	0.6%
Water to Fish to Humans (Ingestion)	1.96E-04	83.6%	1.37E-03	83.6%
Soil to Humans (Ingestion)	2.69E-07	0.1%	1.88E-06	0.1%
Soil to Livestock/Game (Beef) to Humans (Ingestion)	4.84E-06	2.1%	3.39E-05	2.1%
Soil to Dairy Cattle (Milk) to Humans	1.61E-06	0.7%	1.13E-05	0.7%
Soil to Vegetables to Humans (Ingestion)	8.61E-06	3.7%	6.03E-05	3.7%
Soil to Pasture to Livestock/Game (Beef) to Humans (Ingestion)	5.58E-06	2.4%	3.91E-05	2.4%
Soil to Pasture to Dairy Cattle (Milk) to Humans (Ingestion)	1.76E-06	0.8%	1.23E-05	0.8%
TOTAL INGESTION DOSE	**2.34E-04**		**1.64E-03**	

DOSE SUMMARY TABLE - LEVEL I SCREEN *COPPER* *K-25*

	DOSE-Car mg / (kg-d)	% of Total	DOSE-Non-C mg / (kg-d)	% of Total
INHALATION PATHWAYS				
Air to Humans (Inhalation)	4.23E-05	99.4%	8.46E-05	99.4%
Soil (Particulates) to Air to Humans (Inhalation)	2.41E-07	0.6%	4.83E-07	0.6%
TOTAL INHALATION DOSE	**4.25E-05**		**8.51E-05**	
INGESTION PATHWAYS				
Air to Livestock/Game (Beef) to Humans (ingestion)	1.55E-06	0.0%	3.10E-06	0.0%
Air to Dairy Cattle (Milk) to Humans (ingestion)	1.59E-06	0.0%	3.17E-06	0.0%
Air to Vegetables to Humans (Ingestion)	6.33E-03	14.2%	1.27E-02	14.2%
Air to Pasture to Livestock/Game (Beef) to Humans (Ingestion)	5.91E-03	13.2%	1.18E-02	13.2%
Air to Pasture to Dairy Cattle (Milk) to Humans (Ingestion)	5.91E-03	13.2%	1.18E-02	13.2%
Water to Humans (Incidental Ingestion)	2.61E-06	0.0%	5.22E-06	0.0%
Water to Livestock/Game (Beef) to Humans (Ingestion)	3.13E-05	0.1%	6.26E-05	0.1%
Water to Dairy Cattle (Milk) to Humans (Ingestion)	2.85E-05	0.1%	5.69E-05	0.1%
Water to Fish to Humans (Ingestion)	5.93E-04	1.3%	1.19E-03	1.3%
Soil to Humans (Ingestion)	3.25E-05	0.1%	6.50E-05	0.1%
Soil to Livestock/Game (Beef) to Humans (Ingestion)	5.57E-04	1.2%	1.11E-03	1.2%
Soil to Dairy Cattle (Milk) to Humans	4.64E-04	1.0%	9.29E-04	1.0%
Soil to Vegetables to Humans (Ingestion)	6.96E-03	15.6%	1.39E-02	15.6%
Soil to Pasture to Livestock/Game (Beef) to Humans (Ingestion)	8.91E-03	20.0%	1.78E-02	20.0%
Soil to Pasture to Dairy Cattle (Milk) to Humans (Ingestion)	8.91E-03	20.0%	1.78E-02	20.0%
TOTAL INGESTION DOSE	**4.47E-02**		**8.93E-02**	

DOSE SUMMARY TABLE - REFINED LEVEL I SCREEN *COPPER* *K-25*

	DOSE-Car mg / (kg-d)	% of Total	DOSE-Non-C mg / (kg-d)	% of Total
INHALATION PATHWAYS				
Air to Humans (Inhalation)	3.48E-06	99.4%	2.43E-05	99.4%
Soil (Particulates) to Air to Humans (Inhalation)	1.98E-08	0.6%	1.39E-07	0.6%
TOTAL INHALATION DOSE	**3.50E-06**		**2.45E-05**	
INGESTION PATHWAYS				
Air to Livestock/Game (Beef) to Humans (ingestion)	5.30E-08	0.0%	3.71E-07	0.0%
Air to Dairy Cattle (Milk) to Humans (ingestion)	6.52E-08	0.0%	4.56E-07	0.0%
Air to Vegetables to Humans (Ingestion)	3.30E-05	4.7%	2.31E-04	4.7%
Air to Pasture to Livestock/Game (Beef) to Humans (Ingestion)	2.92E-05	4.2%	2.04E-04	4.2%
Air to Pasture to Dairy Cattle (Milk) to Humans (Ingestion)	2.76E-05	3.9%	1.93E-04	3.9%
Water to Humans (Incidental Ingestion)	7.15E-07	0.1%	5.00E-06	0.1%
Water to Livestock/Game (Beef) to Humans (Ingestion)	1.07E-06	0.2%	7.50E-06	0.2%
Water to Dairy Cattle (Milk) to Humans (Ingestion)	1.17E-06	0.2%	8.19E-06	0.2%
Water to Fish to Humans (Ingestion)	1.62E-04	23.1%	1.14E-03	23.1%
Soil to Humans (Ingestion)	1.59E-06	0.2%	1.11E-05	0.2%
Soil to Livestock/Game (Beef) to Humans (Ingestion)	9.54E-06	1.4%	6.68E-05	1.4%
Soil to Dairy Cattle (Milk) to Humans	9.54E-06	1.4%	6.68E-05	1.4%
Soil to Vegetables to Humans (Ingestion)	2.54E-04	36.3%	1.78E-03	36.3%
Soil to Pasture to Livestock/Game (Beef) to Humans (Ingestion)	8.79E-05	12.5%	6.15E-04	12.5%
Soil to Pasture to Dairy Cattle (Milk) to Humans (Ingestion)	8.33E-05	11.9%	5.83E-04	11.9%
TOTAL INGESTION DOSE	**7.02E-04**		**4.91E-03**	

DOSE SUMMARY TABLE - LEVEL I SCREEN *LITHIUM* *Y-12*

	DOSE-Car mg / (kg-d)	% of Total	DOSE-Non-C mg / (kg-d)	% of Total
INHALATION PATHWAYS				
Air to Humans (Inhalation)	1.38E-05	98.3%	1.94E-05	98.3%
Soil (Particulates) to Air to Humans (Inhalation)	2.36E-07	1.7%	3.30E-07	1.7%
TOTAL INHALATION DOSE	**1.41E-05**		**1.97E-05**	
INGESTION PATHWAYS				
Air to Livestock/Game (Beef) to Humans (ingestion)	1.01E-06	0.0%	1.42E-06	0.0%
Air to Dairy Cattle (Milk) to Humans (ingestion)	1.30E-05	0.0%	1.82E-05	0.0%
Air to Vegetables to Humans (Ingestion)	2.07E-03	0.4%	2.90E-03	0.4%
Air to Pasture to Livestock/Game (Beef) to Humans (Ingestion)	3.87E-03	0.7%	5.42E-03	0.7%
Air to Pasture to Dairy Cattle (Milk) to Humans (Ingestion)	4.84E-02	8.8%	6.77E-02	8.8%
Water to Humans (Incidental Ingestion)	4.77E-05	0.0%	6.68E-05	0.0%
Water to Livestock/Game (Beef) to Humans (Ingestion)	3.66E-02	6.7%	5.13E-02	6.7%
Water to Dairy Cattle (Milk) to Humans (Ingestion)	4.16E-01	75.7%	5.83E-01	75.7%
Water to Fish to Humans (Ingestion)	6.94E-04	0.1%	9.71E-04	0.1%
Soil to Humans (Ingestion)	3.17E-05	0.0%	4.44E-05	0.0%
Soil to Livestock/Game (Beef) to Humans (Ingestion)	1.09E-03	0.2%	1.52E-03	0.2%
Soil to Dairy Cattle (Milk) to Humans	1.13E-02	2.1%	1.59E-02	2.1%
Soil to Vegetables to Humans (Ingestion)	1.36E-04	0.0%	1.90E-04	0.0%
Soil to Pasture to Livestock/Game (Beef) to Humans (Ingestion)	2.17E-03	0.4%	3.04E-03	0.4%
Soil to Pasture to Dairy Cattle (Milk) to Humans (Ingestion)	2.72E-02	4.9%	3.81E-02	4.9%
TOTAL INGESTION DOSE	**5.50E-01**		**7.70E-01**	

DOSE SUMMARY TABLE - REFINED LEVEL I SCREEN *LITHIUM* *Y-12*

	DOSE-Car mg / (kg-d)	**% of Total**	**DOSE-Non-C** mg / (kg-d)	**% of Total**
INHALATION PATHWAYS				
Air to Humans (Inhalation)	7.98E-07	98.3%	5.59E-06	98.3%
Soil (Particulates) to Air to Humans (Inhalation)	1.36E-08	1.7%	9.49E-08	1.7%
TOTAL INHALATION DOSE	**8.12E-07**		**5.68E-06**	
INGESTION PATHWAYS				
Air to Livestock/Game (Beef) to Humans (ingestion)	2.44E-08	0.0%	1.70E-07	0.0%
Air to Dairy Cattle (Milk) to Humans (ingestion)	3.74E-07	0.0%	2.62E-06	0.0%
Air to Vegetables to Humans (Ingestion)	7.59E-06	0.1%	5.31E-05	0.1%
Air to Pasture to Livestock/Game (Beef) to Humans (Ingestion)	1.34E-05	0.1%	9.38E-05	0.1%
Air to Pasture to Dairy Cattle (Milk) to Humans (Ingestion)	1.59E-04	1.1%	1.11E-03	1.1%
Water to Humans (Incidental Ingestion)	1.46E-04	1.1%	1.02E-03	1.1%
Water to Livestock/Game (Beef) to Humans (Ingestion)	8.78E-04	6.3%	6.15E-03	6.3%
Water to Dairy Cattle (Milk) to Humans (Ingestion)	1.20E-02	86.2%	8.38E-02	86.2%
Water to Fish to Humans (Ingestion)	3.33E-04	2.4%	2.33E-03	2.4%
Soil to Humans (Ingestion)	1.09E-06	0.0%	7.60E-06	0.0%
Soil to Livestock/Game (Beef) to Humans (Ingestion)	1.30E-05	0.1%	9.12E-05	0.1%
Soil to Dairy Cattle (Milk) to Humans	1.63E-04	1.2%	1.14E-03	1.2%
Soil to Vegetables to Humans (Ingestion)	3.48E-06	0.0%	2.43E-05	0.0%
Soil to Pasture to Livestock/Game (Beef) to Humans (Ingestion)	1.50E-05	0.1%	1.05E-04	0.1%
Soil to Pasture to Dairy Cattle (Milk) to Humans (Ingestion)	1.78E-04	1.3%	1.25E-03	1.3%
TOTAL INGESTION DOSE	**1.39E-02**		**9.72E-02**	

DOSE SUMMARY TABLE - ADULT   Neptunium-237   K-25   Union/Lawnville   LEVEL I

INITIAL CONTAMINANT CONCENTRATIONS IN MEDIA

C(air)	Concentration of contaminant in air	3.82E-04	pCi/m3
C(sed)	Concentration of contaminant in sediment	Not Used	pCi/kg
C(water)	Concentration of contaminant in water	5.12E-04	pCi/L
C(soil)	Concentration of contaminant in soil	5.00E+01	pCi/kg
C(fish)	Concentration of contaminant in fish	Not Used	pCi/kg

AIR PATHWAYS	Sv/year	*Percent this Pathway Represents of:* Air Pathway	All Pathways
Air to Humans (Inhalation of Atm. Air Concentration)	9.50E-07	56%	41%
Air to Humans (Immersion-External)	1.68E-12	0%	0%
Air to Livestock/Game (Beef) to Humans (ingestion)	1.66E-11	0%	0%
Air to Dairy Cattle (Milk) to Humans (ingestion)	8.52E-13	0%	0%
Air to Vegetables to Humans (Ingestion)	6.80E-07	40%	29%
Air to Pasture to Livestock/Game (Beef) to Humans (Ingestion)	6.36E-08	4%	3%
Air to Pasture to Dairy Cattle (Milk) to Humans (Ingestion)	3.18E-09	0%	0%

WATER PATHWAYS	Sv/year	*Percent this Pathway Represents of:* Water Pathway	All Pathways
Water to Humans (Incidental Ingestion from Clinch)	3.35E-12	1%	0%
Water to Livestock/Game (Beef) to Humans (Ingestion)	8.04E-12	3%	0%
Water to Dairy Cattle (Milk) to Humans (Ingestion)	3.65E-13	0%	0%
Water to Fish to Humans (Ingestion)	2.28E-10	95%	0%
Water to Humans (Ext by Water Immersion)-Clinch	4.52E-14	0%	0%

SOIL PATHWAYS	Sv/year	*Percent this Pathway Represents of:* Soil Pathway	All Pathways
Soil (Particulates) to Air to Humans (Inhalation)	8.08E-09	1%	0%
Soil to Humans (Ingestion)	5.20E-09	1%	0%
Soil to Livestock/Game (Beef) to Humans (Ingestion)	8.92E-09	1%	0%
Soil to Dairy Cattle (Milk) to Humans	3.72E-10	0%	0%
Soil to Vegetables to Humans (Ingestion)	4.46E-07	70%	19%
Soil to Pasture to Livestock/Game (Beef) to Humans (Ingestion)	1.78E-08	3%	1%
Soil to Pasture to Dairy Cattle (Milk) to Humans (Ingestion)	8.92E-10	0%	0%
Soil to Humans (Ground Exposure)	1.47E-07	23%	6%

DOSE SUMMED ACROSS ALL PATHWAYS	2.33E-06	Sv/Year
ANNUAL RISK (Risk Coeff. 0.073 Sv^{-1})	1.70E-07	RISK
TOTAL RISK (1953-1995)	7.32E-06	RISK

K-25   Union/Lawnville   LEVEL I

DOSE SUMMARY TABLE - ADULT Neptunium-237 Y-12 Scarboro LEVEL I

INITIAL CONTAMINANT CONCENTRATIONS IN MEDIA

C(air)	Concentration of contaminant in air	6.61E-07	pCi/m3
C(sed)	Concentration of contaminant in sediment	Not Used	pCi/kg
C(water)	Concentration of contaminant in water	2.04E-02	pCi/L
C(soil)	Concentration of contaminant in soil	1.70E+02	pCi/kg
C(fish)	Concentration of contaminant in fish	Not Used	pCi/kg

AIR PATHWAYS	Sv/year	*Percent this Pathway Represents of:*	
		Air Pathway	All Pathways
Air to Humans (Inhalation of Atm. Air Concentration)	1.64E-09	56%	0%
Air to Humans (Immersion-External)	3.92E-14	0%	0%
Air to Livestock/Game (Beef) to Humans (ingestion)	2.88E-14	0%	0%
Air to Dairy Cattle (Milk) to Humans (ingestion)	1.47E-15	0%	0%
Air to Vegetables to Humans (Ingestion)	1.18E-09	40%	0%
Air to Pasture to Livestock/Game (Beef) to Humans (Ingestion)	1.10E-10	4%	0%
Air to Pasture to Dairy Cattle (Milk) to Humans (Ingestion)	5.50E-12	0%	0%

WATER PATHWAYS	Sv/year	*Percent this Pathway Represents of:*	
		Water Pathway	All Pathways
Water to Humans (Incidental Ingestion from EFPC)	8.36E-12	0%	0%
Water to Livestock/Game (Beef) to Humans (Ingestion)	3.21E-10	8%	0%
Water to Dairy Cattle (Milk) to Humans (Ingestion)	1.46E-11	0%	0%
Water to Fish to Humans (Ingestion)	3.65E-09	91%	0%
Water to Humans (Ext by Water Immersion)-EFPC	1.13E-13	0%	0%

SOIL PATHWAYS	Sv/year	*Percent this Pathway Represents of:*	
		Soil Pathway	All Pathways
Soil (Particulates) to Air to Humans (Inhalation)	2.75E-08	1%	1%
Soil to Humans (Ingestion)	1.77E-08	1%	1%
Soil to Livestock/Game (Beef) to Humans (Ingestion)	3.03E-08	1%	1%
Soil to Dairy Cattle (Milk) to Humans	1.26E-09	0%	0%
Soil to Vegetables to Humans (Ingestion)	1.52E-06	70%	70%
Soil to Pasture to Livestock/Game (Beef) to Humans (Ingestion)	6.07E-08	3%	3%
Soil to Pasture to Dairy Cattle (Milk) to Humans (Ingestion)	3.03E-09	0%	0%
Soil to Humans (Ground Exposure)	4.99E-07	23%	23%

DOSE SUMMED ACROSS ALL PATHWAYS	2.16E-06	Sv/Year
ANNUAL RISK (Risk Coeff. 0.073 Sv^{-1})	1.58E-07	RISK
TOTAL RISK (1953-1995)	6.79E-06	RISK

Y-12 Scarboro LEVEL I

DOSE SUMMARY TABLE - LEVEL I SCREEN NICKEL K-25

	DOSE-Car mg / (kg-d)	% of Total	DOSE-Non-C mg / (kg-d)	% of Total
INHALATION PATHWAYS				
Air to Humans (Inhalation)	1.26E-04	98.6%	1.76E-04	98.6%
Soil (Particulates) to Air to Humans (Inhalation)	1.72E-06	1.4%	2.41E-06	1.4%
TOTAL INHALATION DOSE	**1.27E-04**		**1.78E-04**	
INGESTION PATHWAYS				
Air to Livestock/Game (Beef) to Humans (ingestion)	2.30E-06	0.0%	3.22E-06	0.0%
Air to Dairy Cattle (Milk) to Humans (ingestion)	2.36E-06	0.0%	3.30E-06	0.0%
Air to Vegetables to Humans (Ingestion)	1.88E-02	11.0%	2.63E-02	11.0%
Air to Pasture to Livestock/Game (Beef) to Humans (Ingestion)	8.79E-03	5.1%	1.23E-02	5.1%
Air to Pasture to Dairy Cattle (Milk) to Humans (Ingestion)	8.79E-03	5.1%	1.23E-02	5.1%
Water to Humans (Incidental Ingestion)	4.49E-06	0.0%	6.29E-06	0.0%
Water to Livestock/Game (Beef) to Humans (Ingestion)	5.39E-05	0.0%	7.54E-05	0.0%
Water to Dairy Cattle (Milk) to Humans (Ingestion)	4.90E-05	0.0%	6.86E-05	0.0%
Water to Fish to Humans (Ingestion)	1.02E-03	0.6%	1.43E-03	0.6%
Soil to Humans (Ingestion)	2.32E-04	0.1%	3.25E-04	0.1%
Soil to Livestock/Game (Beef) to Humans (Ingestion)	1.99E-03	1.2%	2.79E-03	1.2%
Soil to Dairy Cattle (Milk) to Humans	1.66E-03	1.0%	2.32E-03	1.0%
Soil to Vegetables to Humans (Ingestion)	4.97E-02	29.1%	6.96E-02	29.1%
Soil to Pasture to Livestock/Game (Beef) to Humans (Ingestion)	3.98E-02	23.3%	5.57E-02	23.3%
Soil to Pasture to Dairy Cattle (Milk) to Humans (Ingestion)	3.98E-02	23.3%	5.57E-02	23.3%
TOTAL INGESTION DOSE	**1.71E-01**		**2.39E-01**	

DOSE SUMMARY TABLE - REFINED LEVEL I SCREEN NICKEL K-25

	DOSE-Car mg / (kg-d)	% of Total	DOSE-Non-C mg / (kg-d)	% of Total
INHALATION PATHWAYS				
Air to Humans (Inhalation)	7.23E-06	98.6%	5.06E-05	98.6%
Soil (Particulates) to Air to Humans (Inhalation)	9.92E-08	1.4%	6.95E-07	1.4%
TOTAL INHALATION DOSE	**7.33E-06**		**5.13E-05**	
INGESTION PATHWAYS				
Air to Livestock/Game (Beef) to Humans (ingestion)	5.52E-08	0.0%	3.86E-07	0.0%
Air to Dairy Cattle (Milk) to Humans (ingestion)	6.78E-08	0.0%	4.75E-07	0.0%
Air to Vegetables to Humans (Ingestion)	6.87E-05	3.1%	4.81E-04	3.1%
Air to Pasture to Livestock/Game (Beef) to Humans (Ingestion)	3.03E-05	1.4%	2.12E-04	1.4%
Air to Pasture to Dairy Cattle (Milk) to Humans (Ingestion)	2.88E-05	1.3%	2.01E-04	1.3%
Water to Humans (Incidental Ingestion)	8.61E-07	0.0%	6.03E-06	0.0%
Water to Livestock/Game (Beef) to Humans (Ingestion)	1.29E-06	0.1%	9.04E-06	0.1%
Water to Dairy Cattle (Milk) to Humans (Ingestion)	1.41E-06	0.1%	9.86E-06	0.1%
Water to Fish to Humans (Ingestion)	1.96E-04	8.9%	1.37E-03	8.9%
Soil to Humans (Ingestion)	7.95E-06	0.4%	5.57E-05	0.4%
Soil to Livestock/Game (Beef) to Humans (Ingestion)	2.39E-05	1.1%	1.67E-04	1.1%
Soil to Dairy Cattle (Milk) to Humans	2.39E-05	1.1%	1.67E-04	1.1%
Soil to Vegetables to Humans (Ingestion)	1.27E-03	58.1%	8.90E-03	58.1%
Soil to Pasture to Livestock/Game (Beef) to Humans (Ingestion)	2.75E-04	12.5%	1.92E-03	12.5%
Soil to Pasture to Dairy Cattle (Milk) to Humans (Ingestion)	2.60E-04	11.9%	1.82E-03	11.9%
TOTAL INGESTION DOSE	**2.19E-03**		**1.53E-02**	

DOSE SUMMARY TABLE - ADULT **Technetium-99** *K-25* **Union/Lawnville** *LEVEL I*

INITIAL CONTAMINANT CONCENTRATIONS IN MEDIA

C(air)	Concentration of contaminant in air	1.09E-01	pCi/m3
C(sed)	Concentration of contaminant in sediment	Not Used	pCi/kg
C(water)	Concentration of contaminant in water	1.64E+03	pCi/L
C(soil)	Concentration of contaminant in soil	5.58E+05	pCi/kg
C(fish)	Concentration of contaminant in fish	Not Used	pCi/kg

AIR PATHWAYS	Sv/year	Percent this Pathway Represents of: Air Pathway	All Pathways
Air to Humans (Inhalation of Atm. Air Concentration)	4.72E-08	3%	0%
Air to Humans (Immersion-External)	8.26E-14	0%	0%
Air to Livestock/Game (Beef) to Humans (ingestion)	2.76E-12	0%	0%
Air to Dairy Cattle (Milk) to Humans (ingestion)	1.42E-10	0%	0%
Air to Vegetables to Humans (Ingestion)	1.13E-06	66%	0%
Air to Pasture to Livestock/Game (Beef) to Humans (Ingestion)	1.06E-08	1%	0%
Air to Pasture to Dairy Cattle (Milk) to Humans (Ingestion)	5.28E-07	31%	0%

WATER PATHWAYS	Sv/year	Percent this Pathway Represents of: Water Pathway	All Pathways
Water to Humans (Incidental Ingestion from Clinch)	6.24E-08	2%	0%
Water to Livestock/Game (Beef) to Humans (Ingestion)	1.50E-08	0%	0%
Water to Dairy Cattle (Milk) to Humans (Ingestion)	6.81E-07	19%	0%
Water to Fish to Humans (Ingestion)	2.84E-06	79%	0%
Water to Humans (Ext by Water Immersion)-Clinch	2.20E-11	0%	0%

SOIL PATHWAYS	Sv/year	Percent this Pathway Represents of: Soil Pathway	All Pathways
Soil (Particulates) to Air to Humans (Inhalation)	1.57E-08	0%	0%
Soil to Humans (Ingestion)	3.38E-07	0%	0%
Soil to Livestock/Game (Beef) to Humans (Ingestion)	5.79E-08	0%	0%
Soil to Dairy Cattle (Milk) to Humans	2.41E-06	0%	0%
Soil to Vegetables to Humans (Ingestion)	7.24E-03	75%	75%
Soil to Pasture to Livestock/Game (Beef) to Humans (Ingestion)	4.63E-05	0%	0%
Soil to Pasture to Dairy Cattle (Milk) to Humans (Ingestion)	2.32E-03	24%	24%
Soil to Humans (Ground Exposure)	2.18E-07	0%	0%

DOSE SUMMED ACROSS ALL PATHWAYS	9.61E-03	Sv/Year
ANNUAL RISK (Risk Coeff. 0.073 Sv^{-1})	7.02E-04	RISK
TOTAL RISK (1953-1995)	2.95E-02	RISK

K-25 ***Union/Lawnville*** ***LEVEL I***

DOSE SUMMARY TABLE - ADULT *Technetium-99* *K-25* *Union/Lawnville* *REFINED LEVEL I*

INITIAL CONTAMINANT CONCENTRATIONS IN MEDIA

C(air)	Concentration of contaminant in air	1.09E-01	pCi/m3
C(sed)	Concentration of contaminant in sediment	Not Used	pCi/kg
C(water)	Concentration of contaminant in water	1.60E+03	pCi/L
C(soil)	Concentration of contaminant in soil	1.70E+03	pCi/kg
C(fish)	Concentration of contaminant in fish	Not Used	pCi/kg

AIR PATHWAYS

	Sv/year	*Percent this Pathway Represents to:*	
		Air Pathway	All Pathways
Air to Humans (Inhalation of Atm. Air Concentration)	1.36E-08	32%	0%
Air to Humans (Immersion-External)	2.47E-14	0%	0%
Air to Livestock/Game (Beef) to Humans (ingestion)	3.31E-13	0%	0%
Air to Dairy Cattle (Milk) to Humans (ingestion)	2.03E-11	0%	0%
Air to Vegetables to Humans (Ingestion)	2.06E-08	48%	0%
Air to Pasture to Livestock/Game (Beef) to Humans (Ingestion)	1.82E-10	0%	0%
Air to Pasture to Dairy Cattle (Milk) to Humans (Ingestion)	8.63E-09	20%	0%

WATER PATHWAYS

	Sv/year	*Percent this Pathway Represents to:*	
		Water Pathway	All Pathways
Water to Humans (Incidental Ingestion from Clinch)	5.84E-08	2%	1%
Water to Livestock/Game (Beef) to Humans (Ingestion)	1.75E-09	0%	0%
Water to Dairy Cattle (Milk) to Humans (Ingestion)	9.56E-08	3%	2%
Water to Fish to Humans (Ingestion)	2.65E-06	94%	45%
Water to Humans (Ext by Water Immersion)-Clinch	2.06E-11	0%	0%

SOIL PATHWAYS

	Sv/year	*Percent this Pathway Represents to:*	
		Soil Pathway	All Pathways
Soil (Particulates) to Air to Humans (Inhalation)	1.38E-11	0%	0%
Soil to Humans (Ingestion)	1.76E-10	0%	0%
Soil to Livestock/Game (Beef) to Humans (Ingestion)	1.06E-11	0%	0%
Soil to Dairy Cattle (Milk) to Humans	5.29E-10	0%	0%
Soil to Vegetables to Humans (Ingestion)	2.82E-06	92%	48%
Soil to Pasture to Livestock/Game (Beef) to Humans (Ingestion)	4.87E-09	0%	0%
Soil to Pasture to Dairy Cattle (Milk) to Humans (Ingestion)	2.31E-07	8%	4%
Soil to Humans (Ground Exposure)	1.20E-10	0%	0%

DOSE SUMMED ACROSS ALL PATHWAYS	5.91E-06	Sv/Year
ANNUAL RISK (Risk Coeff. 0.073 Sv^{-1})	4.32E-07	RISK
TOTAL RISK (1953-1995)	1.81E-05	RISK

K-25 ***Union/Lawnville*** ***Refined Level I***

DOSE SUMMARY TABLE - ADULT Technetium-99 Y-12 Scarboro LEVEL I

INITIAL CONTAMINANT CONCENTRATIONS IN MEDIA

C(air)	Concentration of contaminant in air	5.87E-06	pCi/m3
C(sed)	Concentration of contaminant in sediment	Not Used	pCi/kg
C(water)	Concentration of contaminant in water	1.60E+02	pCi/L
C(soil)	Concentration of contaminant in soil	5.00E+03	pCi/kg
C(fish)	Concentration of contaminant in fish	Not Used	pCi/kg

AIR PATHWAYS

AIR PATHWAYS	Sv/year	*Percent this Pathway Represents of:* Air Pathways	All Pathways
Air to Humans (Inhalation of Atm. Air Concentration)	2.54E-12	3%	0%
Air to Humans (Immersion-External)	4.44E-18	0%	0%
Air to Livestock/Game (Beef) to Humans (ingestion)	1.49E-16	0%	0%
Air to Dairy Cattle (Milk) to Humans (ingestion)	7.62E-15	0%	0%
Air to Vegetables to Humans (Ingestion)	6.08E-11	66%	0%
Air to Pasture to Livestock/Game (Beef) to Humans (Ingestion)	5.68E-13	1%	0%
Air to Pasture to Dairy Cattle (Milk) to Humans (Ingestion)	2.84E-11	31%	0%

WATER PATHWAYS

WATER PATHWAYS	Sv/year	*Percent this Pathway Represents of:* Water Pathways	All Pathways
Water to Humans (Incidental Ingestion from EFPC)	3.81E-10	0%	0%
Water to Livestock/Game (Beef) to Humans (Ingestion)	1.46E-09	1%	0%
Water to Dairy Cattle (Milk) to Humans (Ingestion)	6.64E-08	37%	0%
Water to Fish to Humans (Ingestion)	1.11E-07	62%	0%
Water to Humans (Ext by Water Immersion)-EFPC	1.34E-13	0%	0%

SOIL PATHWAYS

SOIL PATHWAYS	Sv/year	*Percent this Pathway Represents of:* Soil Pathways	All Pathways
Soil (Particulates) to Air to Humans (Inhalation)	1.41E-10	0%	0%
Soil to Humans (Ingestion)	3.03E-09	0%	0%
Soil to Livestock/Game (Beef) to Humans (Ingestion)	5.19E-10	0%	0%
Soil to Dairy Cattle (Milk) to Humans	2.16E-08	0%	0%
Soil to Vegetables to Humans (Ingestion)	6.49E-05	75%	75%
Soil to Pasture to Livestock/Game (Beef) to Humans (Ingestion)	4.15E-07	0%	0%
Soil to Pasture to Dairy Cattle (Milk) to Humans (Ingestion)	2.08E-05	24%	24%
Soil to Humans (Ground Exposure)	1.95E-09	0%	0%

DOSE SUMMED ACROSS ALL PATHWAYS	8.63E-05	Sv/Year
ANNUAL RISK (Risk Coeff. 0.073 Sv $^{-1}$)	6.30E-06	RISK
TOTAL RISK (1953-1995)	2.71E-04	RISK

Y-12 Scarboro LEVEL I

DOSE SUMMARY TABLE - ADULT Technetium-99 Y-12 Scarboro REFINED LEVEL I

INITIAL CONTAMINANT CONCENTRATIONS IN MEDIA

C(air)	Concentration of contaminant in air	5.87E-06	pCi/m3
C(sed)	Concentration of contaminant in sediment	Not Used	pCi/kg
C(water)	Concentration of contaminant in water	1.60E+02	pCi/L
C(soil)	Concentration of contaminant in soil	5.00E+03	pCi/kg
C(fish)	Concentration of contaminant in fish	Not Used	pCi/kg

AIR PATHWAYS

AIR PATHWAYS	Sv/year	*Percent this Pathway Represents of:* Air Pathway	All Pathways
Air to Humans (Inhalation of Atm. Air Concentration)	7.30E-13	32%	0%
Air to Humans (Immersion-External)	1.33E-18	0%	0%
Air to Livestock/Game (Beef) to Humans (ingestion)	1.78E-17	0%	0%
Air to Dairy Cattle (Milk) to Humans (ingestion)	1.10E-15	0%	0%
Air to Vegetables to Humans (Ingestion)	1.11E-12	48%	0%
Air to Pasture to Livestock/Game (Beef) to Humans (Ingestion)	9.80E-15	0%	0%
Air to Pasture to Dairy Cattle (Milk) to Humans (Ingestion)	4.65E-13	20%	0%

WATER PATHWAYS

WATER PATHWAYS	Sv/year	*Percent this Pathway Represents of:* Water Pathway	All Pathways
Water to Humans (Incidental Ingestion from EFPC)	3.65E-10	0%	0%
Water to Livestock/Game (Beef) to Humans (Ingestion)	1.75E-10	0%	0%
Water to Dairy Cattle (Milk) to Humans (Ingestion)	9.56E-09	8%	0%
Water to Fish to Humans (Ingestion)	1.06E-07	91%	1%
Water to Humans (Ext by Water Immersion)-EFPC	1.29E-13	0%	0%

SOIL PATHWAYS

SOIL PATHWAYS	Sv/year	*Percent this Pathway Represents of:* Soil Pathway	All Pathways
Soil (Particulates) to Air to Humans (Inhalation)	4.04E-11	0%	0%
Soil to Humans (Ingestion)	5.19E-10	0%	0%
Soil to Livestock/Game (Beef) to Humans (Ingestion)	3.11E-11	0%	0%
Soil to Dairy Cattle (Milk) to Humans	1.56E-09	0%	0%
Soil to Vegetables to Humans (Ingestion)	8.30E-06	92%	91%
Soil to Pasture to Livestock/Game (Beef) to Humans (Ingestion)	1.43E-08	0%	0%
Soil to Pasture to Dairy Cattle (Milk) to Humans (Ingestion)	6.79E-07	8%	7%
Soil to Humans (Ground Exposure)	3.52E-10	0%	0%

DOSE SUMMED ACROSS ALL PATHWAYS	9.11E-06	Sv/Year
ANNUAL RISK (Risk Coeff. 0.073 Sv^{-1})	6.65E-07	RISK
TOTAL RISK (1953-1995)	2.79E-05	RISK

Y-12 Scarboro REFINED LEVEL I

APPENDIX J

K-25 NP-237 AIR AND WATER RELEASE ESTIMATES

Table J-1: Estimated Releases of Np-237 to Air from K-25

Assumptions:

PGDP Product Properties

Period	U (kg *10^6)
1953 to 1962	33.05
1963 to 1965	7.24
1966 to 1971	14.48
1972	3.45
1973	1.96
1974	2.23
1975	0.18
1976 to 1977	4.73

Government Reactors

Note: assumptions were 5 g Tc per ton of UO3, about 7 ppm U basis

	kg Tc	
Savannah R.	56	Note: Per Tc Material Balance report, all of the UO3 material had been received from Govt Reactors by 1962
Hanford	30	
	kg x 10^6 U	
Savannah	8.00	shipped for recycle (1953-1962, assuming Tc conc of 7 mg/kg U)
Hanford	4.29	shipped for recycle (1953-1962, assuming Tc conc of 7 mg/kg U)

Year	U (kg x 10^6)	Activity Np-237 (Ci)	Release Fraction	Np-237 Release (mCi)
1953	4.53	408.4	0.000271	110.695
1954	4.53	408.4	0.000119	48.424
1955	4.53	408.4	0.000123	50.304
1956	4.53	408.4	0.000058	23.622
1957	4.53	408.4	0.000059	24.139
1958	4.53	408.4	0.000330	134.856
1959	4.53	408.4	0.000095	38.629
1960	4.53	408.4	0.000177	72.297
1961	4.53	408.4	0.000131	53.584
1962	4.53	408.4	0.000033	13.489
1963	2.41	217.4	0.000227	49.394
1964	2.41	217.4	0.000011	2.309
1965	2.41	217.4	0.000060	12.958
1966	2.41	217.4	0.000008	1.659
1967	2.41	217.4	0.000007	1.555
1968	2.41	217.4	0.000010	2.203
1969	2.41	217.4	0.000013	2.913
1970	2.41	217.4	0.000011	2.283
1971	2.41	217.4	0.000016	3.406
1972	3.45	310.8	0.000012	3.851
1973	1.96	176.6	0.000037	6.488
1974	2.23	200.9	0.000068	13.636
1975	0.18	16.2	0.000050	0.806
1976	2.365	213.1	0.000011	2.376
1977	2.365	213.1	0.000007	1.451
1978	2.365	213.1	0.000007	1.451
1979	2.365	213.1	0.000007	1.451
1980	2.365	213.1	0.000007	1.451
1981	2.365	213.1	0.000007	1.451
1982	2.365	213.1	0.000007	1.451
1983	2.365	213.1	0.000007	1.451
1984	2.365	213.1	0.000007	1.451
1985	2.365	213.1	0.000007	1.451
1986	2.365	213.1	0.000007	1.451
1987	2.365	213.1	0.000007	1.451
1988	2.365	213.1	0.000007	1.451
1989	2.365	213.1	0.000007	1.451
1990	2.365	213.1	0.000007	1.451
1991	2.365	213.1	0.000007	1.451
1992	2.365	213.1	0.000007	1.451
1993	2.365	213.1	0.000007	1.451
1994	2.365	213.1	0.000007	1.451
1995	2.365	213.1	0.000007	1.451

Table J-2: Total Off-Area Uranium Receipts at K-25, Estimated Releases, and Corresponding Release Fractions

Year	U (kg x 10^6)	U (kg) Atm. Release	Release Fraction
1953	4.75	1287.2	0.000271
1954	5.37	637.0	0.000119
1955	4.85	597.2	0.000123
1956	7.93	458.3	0.000058
1957	8.53	504.0	0.000059
1958	8.21	2711.0	0.000330
1959	7.79	737.0	0.000095
1960	6.85	1211.8	0.000177
1961	7.00	919.0	0.000131
1962	5.62	185.8	0.000033
1963	4.42	1005.0	0.000227
1964	5.82	61.9	0.000011
1965	7.66	456.6	0.000060
1966	7.55	57.6	0.000008
1967	7.98	57.1	0.000007
1968	5.81	58.9	0.000010
1969	5.01	67.1	0.000013
1970	6.00	63.1	0.000011
1971	6.92	108.4	0.000016
1972	6.64	82.3	0.000012
1973	7.74	284.5	0.000037
1974	9.16	622.0	0.000068
1975	7.47	371.0	0.000050
1976	10.26	114.4	0.000011
1977	10.98	74.7	0.000007

Table J-3: K-25 Np-237 Release Estimates

Year	Air Release (mCi)	Water Release (mCi)
1953	110	2.2
1954	48	2.2
1955	50	2.2
1956	24	2.2
1957	24	2.2
1958	140	2.2
1959	39	2.2
1960	72	2.2
1961	54	2.2
1962	13	2.2
1963	49	2.2
1964	2.3	2.2
1965	13	2.2
1966	1.7	2.2
1967	1.6	2.2
1968	2.2	2.2
1969	2.9	2.2
1970	2.3	2.2
1971	3.4	2.2
1972	3.9	2.2
1973	6.5	4.5
1974	14	1.1
1975	0.81	1.1
1976	2.4	0.56
1977	1.5	1.7
1978	1.5	1.7
1979	1.5	1.5
1980	1.5	1.4
1981	1.5	2.1
1982	1.5	1.9
1983	1.5	0.4
1984	1.5	2.2
1985	1.5	2.2
1986	1.5	2.2
1987	1.5	2.2
1988	1.5	2.2
1989	1.5	2.2
1990	1.5	2.2
1991	1.5	2.2
1992	1.5	2.2
1993	1.5	2.2
1994	1.5	2.2
1995	1.5	2.2
TOTAL (mCi)	**710**	**88**

APPENDIX K

Y-12 URANIUM RECYCLE RECEIPTS AND NP-237 RELEASE ESTIMATES

Table K-1: Reported Y-12 Receipts of Uranium Recycle Material

Year	SRP (kg U)	ICPP (kg U)	Total (kg U)
1953	0	101	101
1954	0	217	217
1955	3	828	831
1956	0	744	744
1957	201	797	998
1958	258	898	1156
1959	270	3741	4011
1960	6395	769	7164
1961	2305	0	2305
1962	2701	775	3476
1963	6461	0	6461
1964	2977	771	3748
1965	3546	425	3971
1966	3467	1408	4875
1967	2604	0	2604
1968	2097	394	2491
1969	4121	427	4548
1970	2045	108	2153
1971	3805	1660	5465
1972	4716	415	5131
1973	5051	563	5614
1974	4599	0	4599
1975	5110	1702	6812
1976	4320	195	4515
1977	4497	1333	5830
1978	2070	525	2595
1979	4591	535	5126
1980	1510	0	1510
1981	4918	905	5823
1982	5728	577	6305
1983	6682	1041	7723
1984	5776	2868	8644
Total	**102824**	**24722**	**127546**

Egli 1985. "The Report of the Joint Task Force on Uranium Recycle Materials Processing." DOE/OR-859. p. 43.

SRP=Savannah River Plant
ICPP=Idaho Chemical Processing Plant

Table K-2: Estimated Y-12 Plant Np-237 Releases

Year	Air Release (µCi)	Water Release (µCi)
1953	2.3	6.8
1954	4.9	15
1955	19	56
1956	17	50
1957	22	67
1958	26	78
1959	90	270
1960	160	480
1961	52	160
1962	78	230
1963	150	440
1964	84	250
1965	89	270
1966	110	330
1967	59	180
1968	56	170
1969	100	310
1970	48	150
1971	120	370
1972	120	350
1973	130	380
1974	100	310
1975	150	460
1976	100	310
1977	130	390
1978	58	180
1979	120	350
1980	34	100
1981	130	390
1982	140	430
1983	170	520
1984	190	580
1985	10	100
1986	10	100
1987	10	100
1988	10	100
1989	10	100
1990	10	100
1991	10	100
1992	10	100
1993	10	100
1994	10	100
1995	10	100
TOTAL (µCi)	**3000**	**9700**

APPENDIX L

EQUATIONS FOR EXPOSURE PATHWAYS NOT INCLUDED IN THE TASK 7 SCREENING METHODOLOGY REPORT

Air → Livestock/Game (Beef) → Humans (Ingestion)

Equation #1

$$C_{beef(air)} = C_{air}\ Q_{air(b)}\ F_f$$

where:

$C_{beef(air)}$ = Equilibrium concentration of contaminant in beef due to inhalation, mg/kg or pCi/kg;

C_{air} = Average concentration of contaminant in air, mg/m^3 or pCi/m^3;

$Q_{air(b)}$ = Daily inhalation rate of beef cattle, m^3/day; and

F_f = Biotransfer factor from cattle intake to meat concentration, (mg/kg)/(mg/day) or (pCi/kg)/(pCi/day).

Equation #2

$$I_{beef(air)} = \frac{C_{beef(air)}\ U_{beef}}{BW}\ f_{cb} \qquad \text{chemicals}$$

$$I_{beef(air)} = C_{beef(air)}\ U_{beef}\ f_{cb} \qquad \text{radionuclides}$$

where:

$I_{beef(air)}$ = Daily intake of contaminant due to beef ingestion (air pathway), mg/kg-day or pCi/day;

$C_{beef(air)}$= Equilibrium concentration of contaminant in beef due to inhalation, mg/kg or pCi/kg;

U_{beef} = Average daily consumption of beef, kg/day;

BW = Average body weight, kg; and

f_{cb} = Fraction of beef consumed that is contaminated, dimensionless.

Air → Dairy Cattle (Milk) → Humans (Ingestion)

Equation #1

$$C_{milk(air)} = C_{air}\ Q_{air(d)}\ F_m$$

where:

$C_{milk(air)}$ = Equilibrium concentration of contaminant in milk due to inhalation, mg/L or pCi/L;

C_{air} = Average concentration of contaminant in air, mg/m^3 or pCi/m^3;

$Q_{air(d)}$ = Daily inhalation rate of dairy cattle, m^3/day; and

F_m = Biotransfer factor from cattle intake to milk concentration, (mg/L)/ (mg/day) or (pCi/L)/(pCi/day).

Equation #2

$$I_{milk(air)} = \frac{C_{milk(air)}\ U_{milk}}{BW}\ f_{cm} \qquad \text{chemicals}$$

$$I_{milk(air)} = C_{milk(air)}\ U_{milk}\ f_{cm} \qquad \text{radionuclides}$$

where:

$I_{milk(air)}$ = Daily intake of contaminant due to milk ingestion (air pathway), mg/kg-day or pCi/day;

$C_{milk(air)}$ = Equilibrium concentration of contaminant in milk due to inhalation, mg/L or pCi/L;

U_{milk} = Average daily consumption of milk, L/day;

BW = Average body weight, kg; and

f_{cm} = Fraction of milk consumed that is contaminated, dimensionless.

Water → Livestock/Game (Beef) → Humans (Ingestion)

Equation #1

$$C_{beef(water)} = C_{water}\ Q_{water(b)}\ F_f\ f_{cw}$$

where:

$C_{beef(water)}$ = Equilibrium concentration of contaminant in beef due to drinking contaminated water, mg/kg or pCi/kg;

C_{water} = Average concentration of contaminant in water, mg/L or pCi/L;

$Q_{water(b)}$ = Daily intake of water by beef cattle, L/day;

F_f = Biotransfer factor from cattle intake to meat concentration, (mg/kg)/(mg/day) or (pCi/kg)/(pCi/day); and

f_{cw} = Fraction of water obtained from a contaminated source, dimensionless.

Equation #2

$$I_{beef(water)} = C_{beef(water)}\ U_{beef}\ f_{cb} \qquad \text{radionuclides}$$

$$I_{beef(water)} = \frac{C_{beef(water)} U_{beef}}{BW}\ f_{cb} \qquad \text{chemicals}$$

where:

$I_{beef(water)}$ = Daily intake of contaminant due to beef ingestion (water pathway), mg/kg-day or pCi/day;

$C_{beef(water)}$ = Equilibrium concentration of contaminant in beef due to water, mg/kg or pCi/kg;

U_{beef} = Average daily consumption of beef, kg/day;

BW = Average body weight, kg; and

f_{cb} = Fraction of beef consumed that is contaminated

Water → Dairy Cattle (Milk) → Humans (Ingestion)

Equation #1

$$C_{milk(water)} = C_{water}\ Q_{water(d)}\ F_m\ f_{cw}$$

where:

$C_{milk(water)}$ = Equilibrium concentration of contaminant in milk due to drinking contaminated water, mg/L or pCi/L;

C_{water} = Average concentration of contaminant in water, mg/L or pCi/L;

$Q_{water(d)}$ = Daily intake of water by dairy cattle, L/day;

F_m = Biotransfer factor from cattle intake to milk concentration, (mg/L)/(mg/day) or (pCi/L)/(pCi/day); and

f_{cw} = Fraction of water obtained from a contaminated source, dimensionless.

Equation #2

$$I_{milk(water)} = \frac{C_{milk(water)} U_{milk}}{BW}\ f_{cm} \qquad \text{chemicals}$$

$$I_{milk(water)} = C_{milk(water)}\ U_{milk}\ f_{cm} \qquad \text{radionuclides}$$

where:

$I_{milk(water)}$ = Daily intake of contaminant due to milk ingestion (water pathway), mg/kg-day or pCi/day;

$C_{milk(water)}$ = Equilibrium concentration of contaminant in milk due to water, mg/L or pCi/L;

U_{milk} = Average daily consumption of milk, L/day;

BW = Average body weight, kg; and

f_{cm} = Fraction of milk consumed that is contaminated.

Soil → Air → Humans (Inhalation)

Equation #1

$$C_{(air)resus} = A\ M\ F\ Cf_1$$

where:

$C_{(air)resus}$ = Average concentration of contaminant in air due to resuspension, mg/m^3 or pCi/m^3;

A = Equilibrium concentration of contaminant on surface soil, mg/kg or pCi/kg;

M = Mass loading of particles in ambient air, mg/m^3;

F = Enhancement factor, dimensionless; and

Cf_1 = Conversion factor, kg/mg.

Equation #2

$$Intake_{(air)resus} = \frac{C_{(air)resus}\ U_{air}\ f_t\ f_s}{BW} \qquad \text{chemicals}$$

$$Intake_{(air)resus} = C_{(air)resus}\ U_{air}\ f_t\ f_s \qquad \text{radionuclides}$$

where:

$Intake_{(air)resus}$ = Daily intake of contaminant due to inhalation of resuspended particulates, mg/kg-day or pCi/day;

$C_{(air)resus}$ = Average concentration of resuspended contaminant in air, mg/m^3 or pCi/m^3;

U_{air} = Average volume of air inhaled per day, m^3/day;

f_t = Fraction of time that a person is exposed, dimensionless;

f_s = Indoor/outdoor shielding factor, dimensionless; and

BW = Average body weight, kg.

References

NCRP, 1991. National Council on Radiation Protection and Measurements (NCRP). *Screening Models for Releases of Radionuclides to Air, Surface Water and Ground Water. Description of Models.* Draft Document.

USEPA, 1979. U.S. Environmental Protection Agency. *AIRDOS-EPA: A Computerized Methodology for Estimating Environmental Concentrations and Dose to Man from Airborne Releases of Radionuclides.* Prepared by the Oak Ridge National Laboratory for the Office of Radiation Programs, Washington, D.C. EPA 520/1-79-009.

USEPA, 1988. U.S. Environmental Protection Agency. *Limiting Values of Radionuclide Intake and Air Concentration and Dose Conversion Factors for Inhalation, Submersion, and Ingestion.* Prepared by the Oak Ridge National Laboratoryand the Office of Radiation Programs, Washington, D.C. Federal Guidance Report No. 11: EPA-520/1-88-020.

USEPA, 1989. U.S. Environmental Protection Agency. *Risk Assessment Guidance for Superfund, Volume 1, Human Health Evaluation Manual (Part A)—Interim Final.* Office of Emergency and Remedial Response, Washington, D.C. EPA/540/1-89/002. December 1989.

USEPA, 1993. U.S. Environmental Protection Agency. *External Exposure to Radionuclides in Air, Water, and Soil.* Prepared by the Oak Ridge National Laboratory for the Office of Radiation and Indoor Air , Washington, D.C. Federal Guidance Report No. 12.

KEY TECHNICAL REPORTS OF THE OAK RIDGE DOSE RECONSTRUCTION PROJECT

• Volume 1 •
Iodine-131 Releases from Radioactive Lanthanum Processing at the X-10 Site
in Oak Ridge, Tennessee (1944-1956)– an Assessment of Quantities Released,
Off-Site Radiation Doses, and Potential Excess Risks of Thyroid Cancer
The report of project Task 1

• Volume 1A •
Appendices to the Iodine-131 Report

• Volume 2 •
Mercury Releases from Lithium Enrichment at the Oak Ridge Y-12 Plant–
a Reconstruction of Historical Releases and Off-Site Doses and Health Risks
The report of project Task 2

• Volume 2A •
Appendices to the Mercury Report

• Volume 3 •
PCBs in the Environment near the Oak Ridge Reservation–
a Reconstruction of Historical Doses and Health Risks
The report of project Task 3

• Volume 4 •
Radionuclide Releases to the Clinch River from White Oak Creek on the
Oak Ridge Reservation– an Assessment of Historical Quantities Released,
Off-Site Radiation Doses, and Health Risks
The report of project Task 4

• Volume 4A •
Appendices to the White Oak Creek Report

• Volume 5 •
Uranium Releases from the Oak Ridge Reservation–
a Review of the Quality of Historical Effluent Monitoring Data
and a Screening Evaluation of Potential Off-Site Exposures
The report of project Task 6

• Volume 6 •
Screening-Level Evaluation of Additional Potential Materials of Concern
The report of project Task 7

• Volume 7 •
Oak Ridge Dose Reconstruction Project Summary Report